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Assessment of the effects of the 1997/98 forest fires and anthropogenic deforestation on
the vegetation and Coleoptera found within the Tropical Peat Swamp forest habitat of
Central Kalimantan, Indonesia

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*‘Die uebergrosse Mannishfaltigkeit der bluehtenreichen Waldflora verbiet die
Frageworaus diese Walder bestehen’.*

‘The excessive diversity of the flora and its richness in flowering plants forbid one to ask
‘What is the composition of these forests?’

Humboldt (quoted by Kurz, 1878)

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Abstract

This study recorded and compared primary floristic stands and Coleoptera populations of the three main habitats, that occurred in within the tropical peat swamp of Central Kalimantan, that of the Mixed Peat Swamp Forest, Low Pole Peat Swamp forest and Tall Pole Peat Swamp Forest. A further study was carried out to investigate, analyse and determine the effects of several forms of perturbations on one of the most utilised Peat Swamp forest habitat's - Mixed Peat Swamp Forest. The perturbations studied were classified into two distinct groups; natural disturbance through forest fires and disturbance as a direct result of human intervention, namely by selective and illegal logging.

Tree density in each of the three forest types was determined to be dramatically reduced by each of the forms of disturbance sampled. In comparison to the primary forest MSF, almost 75% of tree density was lost in the burnt area, whilst the least level of tree density loss and therefore damage, was found in the selectively logged area where a loss of 40% occurred. The abundance of dominant primary tree species was also established to decline with disturbance, as did the general diversity of tree species found in a given area. Vegetative regeneration - at varying degrees was present in all of the disturbed sites. However, repeated surveys of the area are vital to establish the viability and species assemblages of these new emergent saplings.

There was a significant difference in the Coleopteran beetles species found in the baited pitfall sites within each area. The disturbed sites - in particular the illegally logged forest area, showed a high invertebrate diversity. However abundance was lost with only 309 individuals collected in the illegally logged sample site, a drop of over 60% in the total number of individuals collected when compared with the primary Mixed Peat Swamp forest area.

The long-term prognosis of the Tropical Peat Swamp forest area studied, if the deforestation, fragmentation and therefore the destruction of the primary forest continues at this unabated pace, does not currently seem to be a healthy one. The potential loss of biodiversity is great and although the results of this investigation highlighted that the small-scale illegal removal of timber trees has a markedly reduced effect on the immediate forest, than both burnt and selectively logged tree removal. But if unmanaged

this form of logging will continue to increase and its negative effects on the forest will escalate.

Introduction

Scientific understanding of the complexity of tropical rain forest ecosystems is still highly undeveloped (Tang 1987; Krieger 1991). One such tropical forest ecosystem is to be found in Indonesia, and is thought to be one of the most species rich terrestrial systems in the world (Soepadmo 1995). Myers (1990) identified the Indonesian tropical rainforests as an ecological hotspot and in particular the extremely diverse lowland tropical rainforest. However, this is an ecosystem that is under increasing threat from deforestation.

Haila and Levins (1992) stated there are no absolute standards, to distinguish between human-induced and natural processes, but for the purpose of this investigation logging was considered to be a human-induced process of deforestation and fire a natural event, that is stochastic in nature. Therefore it can be conjectured, that the current, prevalent source of forest disturbance in Indonesia is spurred by human influence, and perpetuated by present commercial logging and agricultural practices. These methods of forest removal have led to a drastic reduction in primary rainforest and as a direct result, the forest fragmentation of pristine forested areas has increased rapidly and the formation of less productive secondary growth forest has become endemic to Indonesia (Kartawinata *et al* 1989).

The rate of deforestation for timber extraction is rising rapidly in Indonesia, even more so with a recent extraction target of 57 million m³year⁻¹ being set by the Government (Tatawi and Mori 2000). Originally 61% of Indonesia forest that had been set-aside for production forest, but by 1985 over 51% of the allotted forest area had already been logged (Gillis 1988).

There are 2 major forms of tree removal in commercial logging practices; selective logging; where the commercially viable trees only are targeted for removal, and clear fell removal; through which the majority of the vegetation in the concession area is removed (MacKinnon *et al* 1996). The mechanised selective logging technique has been subject to quite intensive studies (e.g. Uuttera *et al* 2000; Cannon *et al* 1994; Woods 1989). All of which found that mechanised selective logging resulted in a high mortality rate in the remnant trees excluded from removal, which in turn reduced the diversity of an area and increased the vulnerability of the habitat to further disturbance.

Concerns for the fragile nature of tropical peat swamp forests as an ecosystem continues to mount (Radjagukguk 1997). In response to these concerns, the ecology of the tropical peat swamp of Central Kalimantan, has been the subject of numerous recent studies (Rieley *et al* 1997; Shepherd *et al* 1997). Tropical peatlands and therefore tropical peat swamp forests are a vital carbon store containing up to a third of the carbon stored globally by peatlands (Page 1997). Therefore any form of disturbance to this ecosystem will have known detrimental effects to the environment, on a global scale. In one of the latest studies Pinard and Cooper (2000), computer simulated a dipterocarp forest post-logging and determined, though slightly extraneously that these area had still not regained pre-logging carbon storage level after 5000 years. Though at present, knowledge concerning the actual structure of this lowland dipterocarp forest - when it has been subjected to any form of disturbance is limited (Tatawi and Mori 2000)

The effects of illegal logging has until recently, not been fully addressed by researchers as separate logging technique, most probably due to the small and erratic scale of tree removal that occurs. However, incidents of this form of timber extraction have grown more frequent, particularly in areas of pristine forest that have become accessible due to the close proximity of approved logging concession areas. This increased access has been facilitated by the communications (logging skids, roads and railways) established, used and consequently deserted by selectively logging concessions (*per obs*). Okimori and Matius (2000) studied the effects of similar small-scale forest removal, through the traditional slash and burn technique. It was concluded that a minimum of 70 years was required from cessation of cultivation, before the commercially viable and predominant dipterocarp species such as *Dipterocarpus borneensis* returned to dominance.

Tropical forest fires have been inferred to be the only primary disturbance mechanism - other than logging, to trigger secondary succession in primary forest (Woods 1989). Previously tropical rainforests were thought to have been able to withstand fires, due to the presence of a static high level of ambient moisture (Mutch 1970; Whitmore 1983). Recently however, the elevated rate of deforestation and secondary growth has converted fire-resistant forest into a fire-prone ecosystem (Cochrane and Schulze 1998), through the presence of combustible vegetation such as vines and grasses (Wirawan 1984; Tagawa *et al* 1988). The first large-scale forest fires in Indonesia occurred in 1983, where over 4.6

million hectares were destroyed in Borneo (Beaman *et al* 1985), a large percentage of the forest lost was secondary growth forest (Woods 1989).

The forest fires in Kalimantan during 1997/98, have been attributed to a combination of the cyclic southern oscillation weather front of El Niño 'ENSO' (Cane 1983; Gurhardja *et al* 2000) and deforestation. The forest fires that occurred were classified into 3 distinct forms, surface fires, underground fires and crown fires (Mori 2000). The peat levels within the study area had been found to reach depths of up to 10m (Page *et al* 1999), therefore the fires that occurred were specifically underground in nature. Thus, the fires were intense and slow burning, that gave off little surface heat but large dense, particulate laden clouds of smoke (Yeager 1998). The total area in Indonesia to have been damaged by the fires that burned in 1997-1998, was estimated to be over 5 million hectares (Schwiethelm 1998).

Previous forestry studies have compared remnant primary forest patches, and/or large-scale production forest with pristine primary forest. Thus, deriving key patterns and highlighting vital features, of which a certain degree of loss would be detrimental to many endemic species of flora and fauna, such as the orangutan (*Pongo pygmaeus*) (MacKinnon 1974; Kuuluvainen 1988; Huth *et al* 1997; Uttera 2000). An example of this detrimental effect's can be seen in forested areas that have been subjected to the clear fell logging technique or those areas that have been repeatedly burnt. The degradation of previously forested areas reaches an exhaustion point, where forest regeneration - even through pioneer growth, has ceased. An area in this state will often become a vegetation monostand with the rapid growth of the invasive, aggressive pioneers species *Imperata cylindrica*, which forms the grasslands that are currently sweeping through historically forested areas of Kalimantan.

The vegetation study carried out at Setia Alam field station, Central Kalimantan, focused on forest structure, with particular reference to canopy structure and light availability (Hallé 1995; Torquebiau 1986). These measurements have a direct bearing on sapling growth and therefore regeneration capabilities of a given area (Yahata 2000; Brown 1993; Kennedy and Swaine 1992; Whitmore 1975). However, this is one of only a myriad of factors, that tree growth and survival depends on (Turner 1990), though for this survey it was adopted as the most quantifiable factor to act as an indicator for viability.

To assist in accurate comparisons, control areas of pristine peat swamp forest - or near as possible, in the 3 of the catena of peat swamp forest vegetation zones were surveyed. The

study of primary forests also allows an insight into the habitat boundaries that occur within this particular biogeographic region.

Vegetation and Coleoptera studies have been commonplace in Western Europe (Niemela 1996; Eyre and Luff 1990), as a means of disturbance analysis and establishing the biodiversity of an area. Therefore acting as another form bio-indicator, to ascertain the viability of an area (Speight 1986; Holloway 1987). Numerous investigations in tropical rain forests have been carried out using canopy dwelling Coleopteran species as disturbance indicators (Didham *et al* 1998; Stork 1987; Davis 1998). Didham *et al* (1998) found that 47% of the 30 abundant species found in a given area, were significantly affected by forest fragmentation through deforestation. Combined surveys using the methodology employed in this research project, involving Coleoptera are rare in dipterocarp lowland forest (Makihara *et al* 2000; Davis 1998), particularly when using in pitfall trapping method in a peat swamp forest.

This paper is an analysis of the forest structure and distribution of Coleoptera with a main focus on the disturbance that has occurred within the mixed swamp forest habitat system in the Sungai Sebangau catchment area of Central Kalimantan. So that in turn, a fuller understanding of the heterogeneity of the peat swamp forest habitat may be gained and the future implications of disturbance to this rare forest type and it's dependent faunal population can be determined

Method

Study site conversely

The study site was situated within 6500 km² of tropical Peat Swamp forest, located in the upper Sungai Sebangau catchment area Central Kalimantan (Island of Borneo). It is a unique tropical peat swamp forest ecosystems due to the absence of *Shorea albida*, the dominant dipterocarp tree species found in other peat swamp forests (Rieley *et al* 1997). The peat depth within the study area has been found to increase with distance from the River Sebangau, and measured at a depth ranging from 1m to 10m+ (Rieley *et al* 1997).

There are approximately nine vegetation sequences in the cantena of tropical Peat Swamp forest habitats; Riverine Forest, Mixed Swamp Forest, Transition Forest, Low Pole Forest, Tall Pole Interior Forest, Granite Intrusion Forest and very Low Pole/Low canopy forest (Shepherd *et al* 1997). The 6 survey stands chosen for this study were located in 3 of these forest types (Shepherd *et al* 1997; Rieley *et al* 1997);

The Tall Pole forest (TPPSF) is situated 12 to 25 km from the Sebangau River and is located at the highest elevation level of the catchment area. It covers approx. 400 km² and the peat depth was found to be in excess of 12 m (Rieley *et al* 1997). In a natural pristine state, the canopy reaches height of up to 45 m with a multi-stratified continuous canopy, resulting in low incident light at the ground level. It has been found to have 3 distinct sub-canopy layers and emergent trees height has been recorded at the maximum 45m. Previous surveys have found that the dominant trees in the upper canopy are; *Agathis dammara*, *C. lowii*, *Palaquium spp* and the middle canopy houses mainly *Calophyllum fragrans*, *Eugenia spp*, and the lower canopy *Myrtica lowiana*, and *Litsea spp* (Shepherd *et al* 1997).

The Low Pole forest (LPPSF) type is situated 6 to 11 km from the Sungai Sebangau and covers 2200 km² with the peat depth reaching up to 10m (Rieley *et al* 1997). It is a semi-riparian habitat with persistent ground water pools all year round, thus the majority of vegetation growth is restricted to upraised hummocks. The canopy reaches a height of approx. 20m, and is generally open with high light penetration to ground level. At the shrub layer there is a very dense growth of *Pandanus spp* and *Freycintia spp*. The dominant tree species are *Garcinis cuneifoli* and *Combretocarpus rotundatus*, there have been very few commercial tree species found within this habitat (Shepherd *et al* 1997).

The Mixed Peat Swamp forest (MPSF) is situated closest to the Sebangau river, at a distance of 1 to 3 km and a peat depth up to 6m deep (Rieley *et al* 1997). It also covers the largest area of the research station - at 3900km². Mixed Peat Swamp forest is subject to seasonal flooding and subsequent desiccation in the dry season. The canopy structure is stratified and individual emergent trees have been found to reach a height of up to 35m. Upper canopy species include *Aglaia rubiginosa*, *Dipteocarpus coriaceus*; the lower canopy features saplings of the dominant trees and also *Ixora spp*, *Diospyros evena*, *D.siamang* and *Litseas paludosa*. The shrub layer consists of *Pandanus spp* and a high number of saplings. Numerous commercial trees are also present (Shepherd *et al* 1997).

Sample stands were to include data from primary Low Pole, Tall Pole and Mixed Peat Swamp forests. However, the lack of availability of a pristine Tall Pole Peat Swamp forest sample, a stand with an unknown ratio of selectively logged/primary forest was used. Thus a representative sample of each of the peat swamp forest types at a primary level was surveyed.

It was determined that the Mixed and Tall Pole Peat Swamp forest, had been subjected to the greatest variety and levels of disturbance, mainly due to the presence of commercially important tree species. Due to the increased accessibility of a larger sample area and a pristine primary forest sample, the Mixed Peat Swamp forest type was chosen to represent the disturbed forest types - burnt, selective and illegal logged, for this study (see Table 1)

Four of the six transects were located in the Mixed Peat Swamp forest. The first transect was located in a stand that had been subjected to mechanical selective/polycyclic concession logging during a period of 1987- 1991. The second transect was situated in an area that had been intensively illegal logging during 1997/8. The third transect was located in an area where a minimum of 10 hectares of the forest had been subjected to fire damage during 1997/98 and the fourth was situated within a pristine primary Mixed Peat Swamp forest stand.

Therefore the transects were number as follows;

1. Selectively logged Mixed Peat Swamp Forest
2. Illegally logged Mixed Peat Swamp Forest
3. Burnt Mixed Peat Swamp Forest
4. Primary Mixed Peat Swamp Forest
5. Tall Pole Peat Swamp Forest
6. Low Pole Peat Swamp Forest

The primary Tall Pole, primary Low Pole, selectively and illegally logged Mixed Peat Swamp forest study sites, were located within the Natural Laboratory for the Study of Peat Swamp Forest (NAMTROP). This is a 500 km² area granted low level protection status by the Indonesian Ministry of Forestry (PHPHA) and the Provincial Government of Central Kalimantan (Rieley *et al* 1997) and contains the research station of Setia Alam.

The remaining 2 study sites (burnt and primary Mixed Swamp forest) were located in the PLG Mega-Rice projects area. Over the last two years this area had been subjected to some channel/canal construction within the interior, which resulted in a reduction of ground water level and therefore a degree of desiccation at the forest edge. In response to this the survey transect was located 75 m in from any possible edge effect.

The rainfall range for this area was approximately 2400mm to 3600mm, though particularly erratic annual weather data has been recorded in the last 2 years due to the El Niño and El Niña weather fronts. Annual mean air temperature had previously been recorded at 25.5°C within the forest (Rieley *et al* 1997). Sampling for this research occurred during the drier season from June, until mid of October 1999.

Vegetation sampling protocol

At each sampling site a belt transect 200m in length, was used; this methodology of sampling has been developed as an effective means of estimating both plant and animal populations (Gillison and Brewer 1985). The transect was then sub-divided into 5 separate subplots of 30 x 30m, with 10 m spacing between each plot to give a rectangular grid system (see fig. 1). A GPS reading was taken at the start point of each transect, to allow for future G.I.S studies.

Vegetation survey

Within each subplot, the tree vegetation was recorded at several structural and population levels. The first level sampled was that of adults trees; an adult tree for this investigation, was classified as an individual with a dbh >7 cm (Page *et al* 1999). Adult trees within each sub-plot were identified to local name and where possible genus/species, this was carried out for all of the samples sites, with the exception of the illegally logged Mixed Swamp forest type.

Structurally adult trees were measured for diameter breast height (dbh) at 1.3 m, basal circumference/area and height (using a clinometer). Tree morphology was assessed using 2 keys created concerning canopy shape and tree architecture (after Torquebiau 1986 and Hallé 1995). These were recorded to qualitatively assess tree morphology and tree health, if a tree had no canopy and no shoots, it was considered to be dead and that was also recorded. It was also noted if an individual tree trunk was slanted, twisted, buttressing or had a combination of one or more of these growth forms.

Other vegetation data recorded in each 30 x 30m plot included a total count of all viable seedlings/saplings. Under the guidelines for foresters, the maximum accepted height for saplings is 2.7m (Whitmore 1984). However, for this investigation the reduced climax height for Peat Swamp forest tree species was taken into consideration and the classification of a sapling was established as those with a dbh <7cm, a height of >1m was also used as a characteristic to reinforce the viability of an individual.

At five separate points in each plot (see fig. 1), the percentage of canopy cover was gauged at four separate stratification levels; ground canopy (0-2m), lower canopy (2-5m), mid-canopy (5-15m) and upper canopy (>15m). As this measure was estimated, consideration was given to ensure that the same researcher carried out the estimation in each 6 transects, at 12 p.m. (sun at highest point) for each transect, thus reducing the variation in sun inclination and human error. The 10 central trees of each sub-plot were recorded and the distance of each tree from the centre were measured, the distance of the furthest tree was used to calculate tree density (TD) ha⁻¹.

The adult trees (both alive and standing dead) in the burnt areas, were studied and for fire scars/scorching, if such damage was present, the height of the scars from ground level were measured (Yeager 1998). In the primary Low Pole forest and Tall Pole forest only 3 out of the 5 subplots for each transect were sampled in its entirety. However, the distance of the 10 central trees from the centre point, and the total number of saplings present in each plot were measured in all 5 of the subplots.

Invertebrate sampling

The use of Coleoptera as the indicator for this investigation, were chosen in line with Debinski and Brussard's (1992) eight criteria for choosing an indicator species. They were chosen specifically for their ability in indicating ecological and environmental changes. The pit fall trapping method was used to collect Coleoptera species, as when it

is placed in two or more ground cover habitat types - such as those used for this survey, if a species is repeatedly caught in those areas, the inverse pattern attained shows the true distribution of the species involved (Greenslade 1964).

A pilot study was carried out to determine the duration, size and shape of pitfall traps that were to be used. Using a species accumulation curve, it was determined that 3 sampling days using a baited pitfall trap, allowed the collection of a clear representative sample for each subplot. A dark coloured bowl measuring a depth of 7cm depth and a diameter of 20cm provided adequate trappability (Melbourne 1999). Each trap was filled with to a depth of 3 cm with a solution of water, salt and detergent (D.Mann *pers comm.*).

In the Mixed Swamp forest types pitfall traps were baited with carrion (chicken), placed in a sealed, perforated plastic cup and suspended over each trap. The traps were laid in 3m x 5m parallel lines within 30 x 30m subplots (see Fig. 1). Due to adverse field conditions the sample grids located in the Low Pole Peat Swamp forest and the Tall Pole Peat Swamp forest, were laid with only unbaited traps for two days. Each line started 5m within the plot and was laid at 5m intervals until 5 meters before the end of the plots, therefore 15 pitfall traps/subplot and a total of 75 traps/ transect (see Fig 1). The pitfall traps collections were emptied on a daily basis to minimise the level of predation by necrophytic fauna such as ants and to increase preservation quality.

Method - Statistical analysis

Vegetation

The SPSS statistical package version 6.0 was used to calculate variance analysis (one way ANOVA) with Tukeys test. The analysis of variance was carried out at 3 Levels (see Table 2), to provide an in-depth and clarified view of the sample set. Each of the individual transects were subjected to correlation co-efficient analysis to mark any possible interaction between the variables measured within a transect.

Discriminant analysis was also carried out using the SPSS package, on two separate levels (see Table 2), with an aim to create a standardised set of variables that would typify each of the forest types and disturbance levels surveyed. This was done on 2 scales; the first was on an alpha (local) habitat scale that dealt separately with primary forest and then the disturbed Mixed Peat Swamp forest samples. The second scale was at a Gamma (regional) scale for the Tropical Peat Swamp Forest Ecosystem Central Kalimantan as a whole, therefore all 6 areas sampled were analysed.

The biomass for each of the 6 transects was estimated at (Mg Ha^{-1}), using the formula for this moist tropical zone (Whittaker *et al* 1999; Brown and Iverson 1992)

$$\text{Biomass (Mg Ha}^{-1}\text{)} = 1.276 + 0.034 (\text{dbh}^2 \times \text{height})$$

The tree density (TD) was also determined using the formula (*after* Marsden 1998) using;

$$\text{TD} = \frac{100,000}{\pi d^2}$$

Where d = the single greatest distance of the ten central trees from the plot centre point.

Insects

The Coleoptera collected in each of the forest types were converted in to an abundance scale based upon the DAFOR scale, where;

- 1** = Rare (0-19%),
- 2** = Occasional (20-39%),
- 3** = Frequent (40-59%),
- 4** = Abundant (60-79%),
- 5** = Dominant (80-100%)

The samples collected in the carrion baited pitfall traps were manipulated further by converting the species collected in each transect into a diversity indices using the Shannon-Weaver Diversity Index (H), where

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

Results

a) Tree physiology

Overall the burnt Mixed Peat Swamp Forest (MPSF) forest type was found to have the largest dbh mean ($\bar{x} = 20.76 \pm 11.03$) of all the forest types sampled (see Table 5). The analysis of variance for dbh measurements was tested at several levels using one way ANOVA. The first level was focused at the mixed peat swamp forest (MPSF) level, where all four transects were compared. The second at the primary forests level, using Primary Tall Pole Peat Swamp Forest (TPPSF), Primary Low Pole Peat Swamp Forest (LPPSF) and Primary (MSPF). At level three the sample set as a whole was analysed (Selectively logged MPSF, Illegally logged MPSF, Burnt MPSF, Primary MPSF, Primary TPPSF, Primary LPPSF).

The variance at all 3 levels was found to be statistically significant ($p < 0.001$). At the first level (MSPF only), Tukeys test ($T = 6.34$; d.f = 3,1426; $p < 0.05$) found that statistically significant variance of dbh occurred only between the primary forest and the burnt MPSF type (with the dbh of the 2 logged MPSF). There was no significant dbh variation between selectively logged MPSF and Illegally logged MPSF. When subjected to ANOVA at the second level (that of the all primary forest), significant variance of dbh in Tukeys test ($T = 6.29$; d.f = 2,1106; $p < 0.05$) was found to have occurred only in the LPPSF, in comparison to the primary MPSF and TPPSF. Therefore no significant difference between the primary MPSF and the TPPSF was determined. At the third level (all transects), dbh variance was found to be significant ($F = 11.14$; d.f = 5,1963; $p < 0.001$). However, a significant level of variance did not occur between the primary TPPSF and the primary MPSF ($T = 6.34$; d.f = 5,1963; $p < 0.05$). When all 6 transect were included at level 3, the dbh variance between the primary MSPF, illegal and selective logged MSPF forest transects is no longer statistically significant.

This variance was reflected in the basal area, at level 1 (MPSF only), basal area variance was significant ($F = 12.79$; d.f = 3,1426; $p < 0.001$). With Tukeys test highlighted that selectively logged MPSF and the primary MPSF, were the only transects that did not have significant levels of variance ($T = 329.64$; $p < 0.001$) when compared. At level 2 level basal area variance was found to be significant ($F = 7.83$; d.f = 2,1106; $p < 0.001$), only occurred in the primary LPPSF ($T = 356.12$; $p < 0.05$). When the data set was looked at as a whole (level 3), basal area variance for all of the transects was found to be statistically significant ($F = 83.96$; d.f = 5,1963; $p < 0.001$).

Transect 3 (burnt MPSF) was determined to have the greatest mean for basal area ($x=571.74 \pm 688.68$). Of the 6 forest types sampled, the lowest basal area was shown to be in transect 6 (LPPSF) ($x=283.28 \pm 334.64$), closely followed by the illegally logged MPSF ($x=287.19 \pm 309.47$). The greatest basal area was found to occur in transect 3, the burnt MPSF forest type (see Table 5).

Transect 5 (primary TPPSF) showed the tallest mean tree height ($x = 19.57 \pm 5.87$) and the lowest tree mean height was found in the LPPSF transect (15.42 ± 4.38) (see Table 5). ANOVA at level one (MPSF only), however tree height was determined to be statistically significant ($f=37.32$; $d.f = 3,1426$; $p<0.001$). Tukeys test indicated that the only variance levels of height that were not significant at ($T=>3.822$, $p=<0.05$) were the variance between primary MPSF and the burnt MPSF. Level 2 ANOVA for height was found to be significant ($F=48.10$; $d.f = 2,1106$; $p=<0.001$), with the primary LPPSF in Tukeys test shown as being the only forest type to have a statistically high level of variance ($T=>3.75$; $p = 0.05$) of height. At level 3, variance again was found to be statistically significant ($f=22.94$; $d.f = 5,1963$, $P=<0.001$). Tukeys test determined that all the transects had a significant level of variance ($T=>3.79$; $d.f = 5,1963$; $p=<0.001$), except the TPPSF, which did not have high level of height variance when compared with the burnt MPSF and primary MPSF types.

b) Tree morphology

The mean canopy cover (see table fig. 2) at ground level (0-2m) was absent in the primary MSF and primary TPPSF. The densest percentage of canopy cover at this height occurred in the illegally logged forest ($x=11.6 \pm 19.88\%$). At the lower canopy level (2-5m), the transect forest type determined to have the most dense coverage was the selectively logged MPSF ($x=47.6 \pm 26.23\%$), and the most sparse mean canopy cover recorded at was in the burnt MPSF ($x=9.2 \pm 18.69\%$). The burnt MPSF continued to have the lowest mean canopy cover at both the middle ($x=5.60 \pm 5.18$) and upper levels ($x=1.8 \pm 5.18$). At the middle canopy layer (10-15m), the greatest mean density was recorded in the primary LPPSF ($x=71.60 \pm 23.57\%$), and at the upper canopy level (15m+), the most dense mean canopy cover was recorded in the TPPSF type ($x=68.40 \pm 27.68\%$).

The largest percentage of gaps in the canopy - recorded as distance to the next canopy, was found to be in the burnt MPSF, 66% of distances were shown to be greater than 0 m

therefore canopies not overlapping. The mean size for each gap in the burnt forest type was calculated to be $x = 2.41\text{m} \pm 2.72$. The transect with the greatest percentage of canopy overlap ($< 0\text{m}$) was the TPPSF at 84%, the remaining gaps that did occur in the TPPSF canopy had a mean size of $x=0.57 \pm 1.49$ m. In transect 6 (primary LPPSF), a significantly weak correlation between height and distance to the next canopy was found ($r=0.1531$; $p<0.05$).

The median canopy shape for 4 (TPPST, LPPSF, Primary MPSF and Selectively logged MPSF) of the 6 transects (see table 6) was shape vi (see appendix i). The other 2 transects the illegally logged MPSF recorded a canopy shape of shape v and the burnt transect recorded a median result of 0, therefore no canopy was present. The most frequent median recorded for canopy architecture was architectural shape ii (see table 5). Transect 1 (Selectively logged MPSF), Transect 2 (illegally MPSF), and Transect 6 (primary LPPSF), had a significantly ($p<0.001$) weak correlation that occurred between tree architecture and canopy shape ($r=0.2843$; $r=0.2662$; $r=0.284$) respectively. A weak relationship was also found in Transect 1 (Selectively logged MPSF) ($r=0.2117$; $p<0.001$) and Transect 2 (Illegally logged) ($r=0.2325$; $p<0.001$), between tree architecture and dbh.

The transect with the highest percentage of trees that displayed some form of tree trunk growth (slanted, twisted and buttressed) was the primary TPPSF, totalling 51% of the tree sampled. A significantly weak negative correlation was also determined for trunk growth and height ($r=-0.2069$; $p<0.001$) in Transect 1 (selectively logged MPSF), this weak negative correlation was found again in Transect 3 (burnt MPSF) ($r=-0.2988$; $p<0.001$). A positive weak correlation was found in Transect 5 (primary TPPSF), between trunk growth and tree species ($r=0.2864$). The only transects that did not display a significant level of variance of tree trunk growth was the selective logged MPSF with the primary TPPSF ($f=47.01$; $d.f=5,1963$; $p<0.001$), where all seven trunk growth forms were found (see Appendix ii).

c) Forest composition

Tree species data was not available for transect 2 (illegally logged MPSF), but the species data for the remaining 5 transects was recorded and analysed. A total of 35 separate tree morpho-species (Dyak names – see appendix ii) were found in the 5 transects. The primary MPSF transect had the greatest total amount of tree species found (see Table 3,

Appendix iii) in a transect, with 27 species ($x=20 \pm 1.48 \text{ plot}^{-1}$), the next most varied species count occurred in the selectively logged MPSF ($x=15 \pm 4 \text{ plot}^{-1}$). The least diverse transect, in so far as species presence recorded, was the burnt MPSF, only 16 species ($x=9.0 \pm 4 \text{ plot}^{-1}$) in total were recorded.

The largest percentage of a single species to be found in each plot was for Transect 1 (selectively logged MPSF), Species 17 (Kayu Sepat) at 21%. Transect 3 (burnt MPSF), Species 1 (Banitan) at 47%, Transect 4 (primary MPSF) Species 19 (Mahalis) at 9%, Transect 5 (primary TPPSF) Species 7 (Hangkang) at 44% and lastly in Transect 6 (primary LPPSF) Species 31 (Tarentang) was the most dominant at 18%. Banitan (Species 1), was found to occur both in the burnt and primary MPSF with the same number of 46 individuals, but this fell to just 2 individuals in the selectively logged forest. Species 31 (Tarentang), populations found in the primary TPPSF dropped from 42 to 18 and 15 respectively in the selectively logged forest and in the burnt MPSF. The highest number of Ramin trees (Species 27) was recorded in the selectively logged forest. Species only found in the primary MPSF, that were not found in the disturbed forms of MPSF were Hangtangen (Species 6), Rasak (Species 28) and Galam tikus (Species 9).

Tree species was found to have a significantly ($p<0.001$) weak correlation with dbh and basal area for all 5 transects/forest types. Analysis of variance at level one (MPSF only) found a high level of variance ($f=14.91$; $d.f=3,1246$; $p<0.001$), Tukeys test revealed that this was so for all of the MPSF transects except in between the primary MPSF and the selectively logged MPSF ($T=1.83$; $p<0.05$). At Level 2 (all primary), ANOVA was determined to be significant ($f=10.93$; $d.f=2,1106$; $p<0.001$), but further test found that there was no significant variation between the species found in primary LPPSF and primary TPPSF, ($t=>8.35$; $p<0.05$), thus similar species composition can be found. This similarity continued through to level 3 (All transects), ($f = >240.47, t=0.7418; p<0.05$).

The primary LPPSF had the lowest mean biomass of $117.67 \text{ Mg Ha}^{-1}$ (see table 4, graph 3), and was also found to have the lowest tree density of $374.82 \text{ tree ha}^{-1}$. The highest mean tree density ha^{-1} was found to occur in the primary TPPSF were 1435.97 ha^{-1} , the biomass for this transect was calculated to be $213.77 \text{ Mg ha}^{-1}$. The primary MPSF transect was found to have the largest mean number of saplings per plots ($x=2417 \pm 478.95 \text{ plot}^{-1}$), this forest type also had the highest number of adult trees recorded 570 in a transect ($x=114.6 \pm 11.8 \text{ plot}^{-1}$). When a correlation co-efficient was carried out to test the

relationship between sapling numbers and tree density, the only strongly positive relationship was found in the Primary MPSF ($r=0.8928$; $p<0.05$).

Tree mortality was found to be the highest in the burnt MPSF type, with only 37.2% of the standing trees within this forest type being classified as alive, thus giving a transect mean of dead trees for the transect as $x=62.8\% \pm 0.49$. Also in the burnt MPSF transect, a significant modest correlation ($p<0.001$) between scar height and basal area ($r=0.4806$) and scar height and dbh ($r=0.4638$) was shown to occur.

d) Grouping Analysis

Discriminant stepwise analysis was carried out at two levels (see table 2), to see that whether it would be possible for future researchers to be able to predict the forest type that their sample set belonged to, by solely using the variables e.g. dbh, sampled for this investigation. The first level analysed all of the transects as a whole data set. It determined that the main discriminating within group correlating factors were; Tree species (-0.70655) and canopy shape (0.42888), this was followed by trunk growth (0.71062) and tree architecture (0.37665). The least discriminating variables were distance to the next tree canopy (0.6613) and basal area (0.35351). The overall success rate of predicting group membership for each of the transect was 50.53 %.

The most correctly classified of the groupings at this level occurred in the illegally logged MPSF forest type (Transect 2) at 92.5%, the second was the burnt MPSF at 62.8%. The least correctly classified transect was the primary mixed swamp forest where a mere 22.6% success rate was determined. The most likely grouping for the primary mixed swamp variables to be allocated to, was that of the LPPSF at 26.8%. The remaining 51% of the primary MPSF variables, were allocated evenly to the other transects - except the burnt MPSF.

Discriminant analysis was then used on a second more complex level, that of the plots themselves thus giving a total of 30 plots to be grouped and allocated to the 6 forest types. The overall success rate of predicting the correct grouping for each plot was extremely low at only 16%. Plots 1 to 5 (Selectively logged MPSF), where $x=10.68\%$ were accurately predicted to the correct plots, with $x=14.6\%$ more likely to be allocated to the primary LPPSF type plots. For plots 5 to 10 (Illegally logged MPSF), a higher mean percentage of $x=27.9\%$ was found to have been allocated to the correct plots and

x=26.48% to at least the correct forest type of MPSF. Plots 11-15 (burnt MPSF), showed an accuracy of 25.3% in allocation to the correct plot, plot 11 exhibited the highest correct grouping out of all the 30 plots analysed, with an accuracy percentage of 63%.

Plots 15-20 (Primary MPSF), had the least accurate groupings allocated with only x=5.74% being allocated to the correct plot, the most likely allocations for this forest type plots were to the primary LPPSF at a mean allocation percent of x=15.94%. The primary TPPSF plots 21-25 also showed x=20.82% allocation success to the correct plots followed by a 25.17% success rate of being allocated in the next step to the correct forest type. Lastly plots 25-30 (primary LPPSF), had a mean success rate x=23.58% at being allocated to the correct grouping and 20.1% of being allocated to the correct forest type, the next most likely forest type that these plots would have allocated to was the primary MPSF type.

Coleoptera abundance

The insect abundance data scale (see Table 7) showed that the primary MPSF had the highest number of 727 individuals sampled, with the highest abundance scale rating of 4, for Species 5 (Scarabaeidae family). This species was also present in the other 3 MPSF transects, though at more reduced levels. The illegally logged MPSF and the burnt MPSF transects for this Scarabaeidae species did have very similar sample numbers. Certain species e.g. Species 27 (Histerida) and Species 31 (Curculionidae), only occurred in both the selectively logged MPSF and the primary MPSF, indicating a possible similarity of some habitat factor or niche requirement.

There were certain species (Appendix iii), found only to have occurred in the disturbed forest types; Species 25 (Carabidae), Species 33 (Staphylinidae), Species 73 (Chrysomelidae) and Species 88 (Hydrophilidae). 17 species of beetles were found to be specialist only to transect 1 (selectively logged MPSF). These 17 species were identified to 11 families, of which 3 belonged to the Chrysomelidae family and 2 species each to the Curculionidae and Elateridae families. 3 families Lycidae, Attelabidae, and Noteridae displayed a high degree of habitat preference, as they were not found in any of the other 5 forest types.

22 Coleoptera species (Appendix iii) were found to only occur in transect 2 (illegally logged MPSF). These were found to be in 14 families (1 unknown), 4 of which were representative of the Staphylinidae family, and from each of the following families, with

2 species occurring in each; Anthicidae, Coccinellidae, Curculionidae and Carabidae. Of the remaining species, 3 were identified to families that were only found in this forest habitat type, these were Anthribidae, Colydiidae, Leodidae and also Anthridae. 14 specialist species were found to be specific to transect 3 (burnt MPSF); these belonged to 6 families (1 unknown). 5 of the species collected belonged to the Chrysomelidae family, and 2 to the Carabidae family. No clear and singular habitat preference for any individual family was found for this forest type.

Transect 4 (Primary MPSF) had 18 species of Coleoptera that were exclusive to this forest type, these consisted of 11 families (1 unknown) with the most species (4), occurring in the Curculionidae family with 3 more species belonging to the Staphylinidae family. 3 families; Scaptiidae, Anaspidae and Ceratocanthidae, displayed a habitat specificity for primary forest. Transect 5 (primary TPPSF), had 19 species present, belonging to 5 families (2 unknown). 6 of the species identified belonged to the Staphylinidae family. Transect 6 (LPPSF) which is a semi-riparian forest was found to have only one Coleoptera species, Species 28, which is of the Curculionidae family and also found to occur in a reduced frequency only in the selectively logged MPSF habitat

The diversity of the beetle species was explored by converting the species found in each plot within a transect into the Shannon-Weaver (H) Diversity Index (see Figure 4). Due to the low species count - due to the different sampling criteria that occurred in Transect 5 (primary TPPSF) and Transect 6 (primary LPPSF), the index was not calculated for these samples. The greatest overall diversity was shown to be in plot 3 of Transect 2 (illegally logged MPSF), where $H = 2.65$. The least overall diverse plot occurred within Transect 4 (primary MPSF) plot 2, where $H = 1.05$. In Transect 1 (Selectively logged MPSF), the greatest diversity plot^{-1} was found in plot 2 ($H = 2.44$) and the lowest in plot 1. The diversity on transect 3 (burnt MPSF) was found to reach $H = 2.03$.

Discussion

The vast amount of quantifiable data generated by this investigation has highlighted the exceptional complexity of tropical forest ecosystems. However it has been the aim of the investigator to try to simplify this myriad of factors, into distinct characteristics for each of the forest types observed, through such methods as discriminate analysis.. The Tropical Peat Swamp Forests of Central Kalimantan are under increasing pressure from human disturbance. The response of the Peat Swamp Forest to such pressures can clearly be seen to be significant when the two logged (MPSF) types and the burnt MPSF are taken in comparison with the primary forests (MPSF, LPPSF, TPPSF).

Vegetation

Numerous studies have been carried out into the successional patterns of tropical forest following disturbance, with particular reference to the dipterocarp forest of Borneo (Fox 1976; Whitmore 1975; Whitmore 1984; Woods 1989; Pinard and Cropper 2000). Whitmore (1984) expanded the simple model of forest growth from gap, building and mature phase to a more generalised pattern. It is this pattern that, can be determined to be most relevant to this investigation. If a gap is small then shade tolerant seedlings (particularly climax tree species) already present in the forest structure will tend to dominate the gap. If the gaps are larger, shade tolerant tree species mortality is increased or their growth is suppressed by fast growing species that have a light affinity and germinate rapidly in response to the high light levels present. These light tolerant pioneer species grow at an accelerated rate, ultimately closing the gap to form a new, lower canopy. Over a longer period of time more shade tolerant trees replace those of the lower pioneer canopy, therefore reforming the original tall canopy. This characterised pattern is reflected by the physiology and composition of the disturbed forests of this study, each will be dealt with in turn.

Prior to widespread mechanised logging pioneer species, possibly Marutan (Species 22), Kayu sepat (Species 17) and Bintangur (Species 4) in this investigation, were uncommon in the forests of Borneo (Whitmore 1978; Corner 1988). Once an area had become opened up through deforestation, these pioneer species had a propensity to form monostands (Fox 1978), such as Marutan in the selectively logged MPSF sample and possibly Hangkapas in the primary TPPSF transect. Dipterocarpacea are a group that have been found to be particularly dominant in terms of basal area (Richards 1996), thus Transect 5 - primary

TPPSF with the higher results of basal area, could be seen as being a more Dipterocarpacea dominant stand than the other forest types. In the MPSF habitat stand, tree species found to be intolerant to deforestation were Hangtangen, Mahalilis, Rasak and Glam tikus. Species found to increase in numbers in disturbed areas were Hangkapas, Gerronggang, Gemur and Ramin.

Ramin is an important timber cash crop, though it has very recently been classified as an appendix 3 species in the CITES agreement. Gemur bark is sold commercially, therefore the high presence of this species in the selectively logged area may be a direct result of forest management by the timber concessionaire. The scarcity of some species in the primary forest areas highlights the high diversity and low population density found and required by most pristine old growth tropical rainforest. A survey in Los Tuxtlas (Mexico) found more than 50% of tree species found, had a frequency of 1 (Alvarez-Buylla 1996), therefore a larger area would ideally need to be surveyed to represent this area of complete diversity of the tree species found in the tropical peat swamp forest.

The primary MPSF was found to have a higher sapling population than the disturbed forest types, it could be conjectured that this was due to the presence of saplings of shade tolerant dominant species. The opening up of the canopy by tree removal would have resulted in an increase the level of ground light reaching the forest floor; this would have dramatically increased the mortality of the shade tolerant saplings. This can be supported by Coley and Barone (1996), who stated the structure of primary forest has been characterised as having a large amount of small trees in the under storey creating a reversed J-shaped distribution. Fox (1976) postured that the seedlings and saplings of upper canopy species which survive logging and competition from secondary tree species, are actually more important than post logging germinants as a source of replacement individuals within the upper canopy. Although the saplings in this study were not identified to species level, their presence in the different forest types, would allow for some suppositions about their tolerance type, thus whether or not they are upper canopy or pioneer species.

The mean distances of the gaps present in the forest types, in conjunction with the percentage of number recorded, showed that there was a much higher level of light at ground level in the selectively logged forest than in the illegally logged forest. The time that lapsed since the selectively logged forest was logged is another factor that was taken into consideration. The area was selectively logged over 7 years ago, the high ratio of gaps present in the canopy and the mean width of a gap was determined to be much

greater than that of the primary forest and the illegally logged forest gap width. Thus it can be assumed that the majority of the saplings present in the selectively logged forest would be of the pioneer light-loving group.

It was found by Cannon *et al* (1994) that the density of trees and basal area collected by Abdulhadi *et al* (1981) directly after logging were similar to that of a residual forest stand. This was reflected in the mean basal area of the selectively logged MPSF area, though logging occurred over 6 years ago. However, the more recent form of logging - the illegally logged MPSF, which was carried out approximately only a year before sampling, when compared to the primary stand, these findings do not seem to hold true as the basal area is markedly reduced. The tree density of the burnt MPSF is closest to the density of the primary MPSF, whereas the tree density of the selectively logged MPSF is less than half that of the primary MPSF. When this comparison continues to the dbh measurements taken, the selectively logged MPSF was below that of the primary MPSF stand, whilst the illegally logged forest was found to have a greater mean dbh by over a centimetre than the primary MPSF transect mean. Finegan *et al* (1999) found that the annual growth increments for slow growing tree to be 1mm^{-1} year and for fast growing trees up to 16mm^{-1} . Therefore this larger dbh may be attributed to the presence of a high number of fast growing pioneer tree species, and also the size of the remaining remnant climax community of old growth emergent trees.

Both the selectively logged and illegally logged forest showed a reduction in the mean height, when compared to the primary MPSF. In the case of the illegally logged forest this drop was quite dramatic as just below 2 meters. The drop in height of secondary dipterocarp forests upper canopy can be as great as 10m (King and Chapman 1983), though it must be taken into consideration that the height of primary PSF, (with the exception of the Tall Pole PSF), is not as great as other dipterocarp forest. The logged forest types showed a much more dense canopy cover at both the lower and midlevels, whilst the upper level were sparse in comparison. The distinct stratification of forest canopy cover at all four levels is a recognised characteristic of old growth forest (Richards 1996; Oldeman 1993). Whilst Cannon *et al* (1994) stated that lower closure and increased stratification of canopy such as that displayed in these results, was clear evidence of logging.

The height of an individual tree has recently been shown to be relative to both to hydraulic limitations (Ryan and Yoder 1997) and leaf: sapwood ratio (Becker *et al* 2000). Although this ratio was not measured, the percentage of canopy cover can be used to

allow some supposition for this. Thus from the results it can be theorised that the possible reduced height of the trees may also have been due to the reduced canopy cover. As the increased ground level light combined with the erratic El Niño weather front led to the subsequent soil desiccation that occurred in the peat substrate. This would have resulted in a direct increase of water stress on the vegetation for an extended time period and consequently a reduced in the growth increment that had occurred.

The very nature of illegal logging is extremely labour intensive (Riswan and Hartini 1995) and therefore often only occurs in small, sporadic patches. This is reflected in the sample as 3 of the plots (1 to 3) within the belt transect passed through the illegally logged patch. The two remaining plots (4 and 5) passed through semi-primary forest. The transect was taken as a whole sample to reflect the true irregularity of composition resulting from this type of logging.

This type of hand removal or 'Kuda Kuda' is considered to be a much less disruptive form of removal than conventional or rather mechanical logging (Whitmore 1984). A direct result of the high degree of damage to the forest that occurred during tree extraction, can be attributed to the retardation of canopy closure in the selectively logged forest. Burgess (1971) found only 35% of tree that remained in a mechanically logged area where considered to be healthy and viable. Although the mortality rate for this forest was nil, it must be noted that the rate was calculated by measuring dead standing trees only, not dead and fallen trees. Within the transect, particularly in the illegally logged plots of 1 and 2 numerous fallen dead trees were observed, these had created patches with large gaps in the forest canopy.

It has been well documented that logging increases the susceptibility of an area to fire, though this was not so for the burnt forest sample in this investigation as pre-fire it was an established primary MPSF area. The main cause for the fire to take hold in the study area was that the peat swamp had been subject to a high level of desiccation due to the El Niño weather front and the establishment of drainage canals near the site (J. Rieley *pers com*). This would have resulted in a leaf drop (Richards 1996), which would have contributed to the combustible ground cover. Possibly combined with irregular drop of up to 50% in the relative humidity, an ideal environment was created for the underground fires that raged in the sample area for up to 6 months during 1997/98.

Certain tree species that were present in the burnt forest type were found to have a physiological protection against fire damage. Species 32 - Tumih (*combretocarpus*

rutondatus), has an extremely thick bark covering that allowed the fire to continue to burn the lignified outer layer, without reaching the vital core of the tree. The mean scar height was found to be 21 cm where as the tallest scar height was recorded to be was 270 cm, which occurred on a Tumih tree. This species and other tree species such as Tarentang (Species 31) that survived the fires with scarring, would almost certainly be responsible for the large mean dbh and basal area determined for this forest type, both of which were greater than that recorded in the primary mixed swamp.

Forest fires have been found to greatly restrict the regeneration of an area, through the deterioration of seed banks (Uhl *et al* 1981), reduction in the plants which normally resprout post disturbance, and a decline in soil fertility due to the loss of organic material. The regeneration capabilities of burnt primary forest rather than logged forest has previously been found to be a good one (Woods 1989). This investigation can support this as the presence of quite a high number of sapling and a small forest fragment within plot 5 (see Table 3) were found. Some of these seedlings are most likely to be either *M. gigantea* or *G. obscurum* as the germination of these species seeds have been found to be triggered by heat and then augmented by following seasonal rains (Kartawinata *et al* 1989).

If this fragment persists in this area, it will in turn, provide a valuable source of seeds, as Van Shaik *et al* (1993) noted few dipterocarp fruits could be successfully dispersed further than 40m from the parent tree. It will also assist in the protection of seedlings and saplings, which is known to be the most intensive stage of mortality (Whittaker *et al* 1999). Though conversely the Janzen-Connell model (Janzen 1970; Connell 1971) in terms of sapling success, highlighted that the recruitment of an individual species is inhibited in the immediate vicinity of the adult tree, as the mortality rate of saplings increases through species-specific pathogens and herbivores. This in turn allows for rare or competitively inferior species to attain a greater chance of successful, widespread recruitment and persistence. Therefore although the forest fragment had survived the fires at the time the investigation, further study is required to analyse the health of this remnant and subsequent sapling species composition, to establish the long-term viability of the area.

The primary MPSF and primary LPPSF forest stands composition and physiology fell within the previous habitat parameters set by Shepherd *et al* (1997), the only exception being to that of the Tall Pole PSF, with mean height being lower than expected. This can be attributed most likely to the effect of the possible areas of selective logging that

overlapped into the transect plots. The results of the discriminant analysis grouped most of the primary MPSF into either of two other primary habitats, thus marking out the highly stratified nature of MPSF and a greater need for a more of an expansive study using greater area and possibly including more variables. Therefore through a more in depth physiology study to create a more distinct grouping variable, a classification type for the MPSF can be found.

The biomass measurement (Fig. 3) had several limitations, when being used in regenerating areas, these were highlighted by Whittaker *et al* (1999) as a pioneer species would have a reduced biomass than the formula predictor, due to the occurring fast growth spurt. However biomass was found to follow the expected results for the Tall Pole PSF and Low Pole PSF. Though, the MPSF results showed only the selectively logged forest to have a biomass result less than that of the primary forest, with the burnt forest having the highest. This anomaly may again have been due to the fact that height and dbh variables were used and the majority of remaining trees in the burnt MPSF would have been taller and larger than the norm, in order to survive the onslaught of fire.

The vegetation measures of canopy shape and tree architecture are relatively new and dynamic approaches to forest structure studies (Hallé and Oldeman 1975; Hallé *et al* 1978). The main purpose of these approaches was to identify the endogenous processes controlling growth and form of the tree as a whole. For the purpose of this investigation the median canopy shape and tree architecture was used to indicate the past and present growth history of the trees sampled. Therefore giving perhaps some form of grouping for the individual forest types. The tree architecture technique classifies trees by the point of inversion that is the central point where the tree first branches. This is the first permanent reiteration of the individual making it the clear end of the bole (Torquebiau 1986), where the self pruning branches are dropped and replaced by permanent branches (Ng 1983)

In the key created for tree architecture classes (see Appendix i) used, tree class i is an example of trees that have had a continual slow growth until reaching the canopy where it had started to branch. Tree class ii shows permanent branching below half of the height of the tree, this is a tree that is either growing in the open or in the case of a forest tree, in a canopy gaps that has allowed a high enough level of light penetration to cause branching. Tree class iii indicates that in the past the tree has had enough light to stimulate branching, subsequently the light source was removed most likely through canopy closure, resulting in the tree dropping its branches. Lastly, tree class iv shows a branching

below halfway, so again it has been exposed to light below canopy level, therefore an indicator of previous canopy gaps.

The selectively logged MPSF showed a median of architecture class ii and highest percentage in class ii, the median result supports the forest types known history of disturbance. The illegally logged forest type shows a median of class i, which is due to the presence of remnant primary forest and that the logging occurred quite recently. The burnt forest showed a median in class ii again supporting known disturbance. However, the primary MPSF was observed to have a median of class ii as well, this could be the results of Chablis (natural tree fall) or the irregular stratified canopy of the peat swamp forest - particularly for the Mixed Swamp forest, may cause more branching than in other dipterocarp forest types. The primary Tall Pole forest showed no sign of disturbance having a medium of i therefore the selective logging that was assumed to have occurred in this area was of a low level within the transect. The Low Pole forest highlighted as having a medium of class iii, this below half way branching by a symmetrical division may be a specific characteristic of this unusually low canopy forest type.

The median canopy shape for the selectively logged and all the primary transects was found to be shape 6, this is a discontinuous shape that allows for maximum leaf access to the available light. The use of crownlets such as the two shapes found to be most frequent, allows a tree to avoid becoming completely inundated by vines, thus giving the cauliflower canopy shape typical of dipterocarp trees (Putz 1995). The limitations for this type of classification may have been the time of year the areas were sampled. Trees have been found to produce new leaves during the dry season to reduce the rate of herbivory experienced. Leaves have also been noted to drop and regrow during this season, thus reducing the limitation of water stresses on an individual (Frankie 1974). Therefore, consideration must be given in validating the samples taken at the beginning of the dry season, which may have changed through out this season and almost certainly by the start of the wet season in early October.

Insecta

It has been previously shown that there is no significant decline of beetle species in relation to disturbance (Holloway *et al* 1992). If the diversity index (H) is taken into consideration, the results from this investigation supports previous findings, which can be furthered by saying that disturbance particularly in the form of illegal logging increases diversity. This can be attributed to the small-scale habitat mosaic that illegal logging causes. Resulting in patch communities that are surrounded by other communities with markedly dissimilar structures and compositions, this edge effect also results in several overlaps of niches. The results support this to a point, as the most diverse plot of all was located in plot 3 of the illegally logged forest. This is the plot where the transition from logged vegetation to primary vegetation occurs.

Conversely, the number of individuals in a species population does become drastically reduced by disturbance. It can be seen from the results that as the beetle number collected almost half with every form of disturbance. Thus on a scale according to that primary MPSF is the least disturbed followed by selectively logged, then illegally logged and finally burnt forest as the most severely damage. A factor that this decline can be attributed to is the amount of food availability in some of the habitats. Although logging would have increased the amount of dead wood available to wood borers such as the Chrysomelidae family and the detritivores such as Curculionidae. On the reverse the increased light exposure through canopy loss would have resulted in a temperature rise in the macroclimate, which would be intolerable to these species, thus causing fluctuations in populations. It must be noted that again seasonality is a limitation of invertebrates and coleopteran sampling. A lower abundance of herbivorous insects has been documented in the dry season as opposed to the wet season (Janzen 1973; Wolda 1983).

Carabidae are predacious ground beetles and are not wood-boring or deep soil inhabiting by nature, as a result of this few were found in the burnt forest. It is possible that the number of beetles and in particular rove beetles (Staphylinidae), found in the burnt forest type may be due to infestation. A consequence of the removal of predator from the immediate upper trophic level and an rise in the mortality rate of other animals in this environment, thus increasing available carrion.

This was highlighted in plot 4 of the burnt area where H was found to be higher than the other plots. This plot was also on the edge of the remaining semi-pristine forest remnant, which would provide a source of both shade and prey. The high level of Scarabidae in the primary MPSF can be directly related to the presence of higher animals. Scarabidae are dung beetles, therefore to thrive in a particular habitat they require the presence of higher herbivorous, dung producing animals. Thus, the reduced numbers recorded in the disturbed habitats of the Mixed Tropical Peat Swamp forest, may be used to highlight the absence or possible reduction in populations of dung producing animals, including gibbons (*Hylobates agilis*) and orangutans (*Pongo pygmaeus*). Therefore, reflecting the further implications of such indicators species for large mammals and habitat conservation.

Conclusion

Though disturbance has been viewed as a natural way of maintaining diversity (Connell 1978), Lovejoy *et al* (1986) reviewed forest fragmentation along with Simberloff (1992) and culminated in Boyles (1995) study, conjectured that forest fragmentation, the size and degree of the resulting isolation it caused was the prime determinants of a species extinction or conservation. The vulnerability of the Tropical Peat Swamp forest vegetation can clearly be seen to be increased by the fragmentation caused by the various forms of deforestation that occurred in the study area. The reduced numbers in tree density in conjunction with the drop in the total number of saplings present, in the disturbed areas highlighted and supported this. The vulnerability of habitat was found to greatly affect the Coleoptera present and which would have a direct bearing on higher more endangered forms of vertebrates.

The frequency of unmanaged illegal deforestation as a direct result of socio-economic pressures could become much higher. Therefore, eventually superseding the negative yet managed effects to the endemic biodiversity, caused by mechanised selective logging. Thus it was found that forest fragmentation on a small scale of disturbance may have increased the diversity of invertebrates but decreased most of the vegetation variables measured and on a larger scale it was found to reduce both floral and invertebrate diversity and abundance of the Peat Swamp habitat as a whole.

The lack of practical investigations into the effects of forest fires on tropical forest ecosystems needs to be addressed and therefore drastically increased. Factors such as smoke haze and smoke particulates affecting the growth of vegetation and the health of fauna, the loss of peat on water sheds and the stability of forest floors are few of the effects of unmanaged forest fires that need immediate attention. Preventative measures need to be implemented to reduce the vulnerability of pristine and secondary forested areas to fires and to regulate and restrict the access to the concessions established by logging concessions, which lead into primary forest.

The viability of all of the areas surveyed in this study should be reassessed at regular intervals to allow for a more long-term understanding of the effects of disturbance. This should be carried out through the establishment of permanent, extended study plots, which are regularly surveyed using the standardised variables used in this investigation.

It is the intention of the author that the worked carried out in this paper can provide some continuity in the understanding of the impacts of anthropogenic and certain natural disturbances on the threatened tropical forest habitat. And assist in establishing an eventual long-term goal, of the compilation of a viable, practicable regeneration and conservation programme, within the Tropical Peat Swamp forests of Kalimantan Tengah.

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Literature Review

Indonesia

Indonesia is an archipelago of approximately 13,677 islands that stretches for over 3400 miles, straddling the equator and spanning three international time zones (Andrews and Freestone 1973). With a population of over 180 million it is one of the worlds most populous nations (Dinerstien and Wikramanayake 1996). It is biogeographically rich enough in species to be considered an ecological hotspot (Myers 1988), or a mega-diversity country (Mittermeir and Werner 1990). It covers 1.3% of the worlds surface and is home to 10% of all known flowering plants, 12% of known mammal species, 16% of all reptiles and amphibians, 17% of known bird species and a quarter all marine and freshwater fish (Kusumaatmadja 1994). The species within this country exhibit a high degree of endemism (Dinerstien and Wikramanayake 1996).

The island of Borneo lies between latitude 70°N and 40°S (MacKinnon *et al* 1996). Kalimantan covers 539,460km³, which is 73% of the total landmass of the island of Borneo (MacKinnon *et al* 1996). The 33 % remaining comprise of the Malaysian states of Sarawak and Sabah and the independent sultanate of Brunei. Kalimantan covers 28.11% of the total land mass of the Republic of Indonesia, but houses only 4.7% of the population of Indonesia (Storey *et al* 1992)

Historically Borneo is on the border of the much disputed Wallacea line (Wallace 1863; Lam 1945; Audley-Charles 1981) which runs through the Makassar strait, separating it biogeographically from Sulawesi and western Indonesia (van Balgooy 1987, George 1987). The Wallacea line divides Borneo into the Asian biogeography (originally Laurasian) group, and with its neighbouring island Sulawesi into the Australasian (originating from Gondawanian) group (Whitmore 1990). Borneo is situated on the Sunda shelf (the largest continental shelf in the world), which it shares with the Malay Peninsula, Sumatra, Java and Palawan (which is a Filipino island). It is separated from the Sahul Shelf by a deep sea, on which lies Sulawesi, Nusa Tenggara and the Philippines (Davis *et al* 1994).

The variety of geological history, geographical, phytogeographical and zoogeographical barriers, has exerted such a huge influence on the dispersal of species within Borneo, that it can be roughly divided into 7 distinct biogeographical units: Meratus Mountains,

Sabah and northern East Kalimantan, North coast (Brunei and Sarawak), North West Borneo (ancient hills north of Kapuas river and western Sarawak), the Southern plains, the East coast of Borneo and lastly Central Borneo (MacKinnon and MacKinnon 1986).

Floristically Borneo has approximately 10000 to 15000 species of flowering plants, the highest of the Sunda Islands group and 20,000-25,000 species of vascular plants (Davis *et al* 1994, MacKinnon *et al* 1996). Of that number 30-50% of species are endemic, which is second only to Papua New Guinea, which has an endemism percentage for vascular plants reaching 70-80%. As a result of its distance from mainland Asia, thus limiting migration, this abundance is not reflected to such a great degree in vertebrate numbers, where it has 222 mammals species, though 44 of these mammals are endemic to Borneo (Payne *et al* 1985). Typically, it has been determined that the tropical forest of Kalimantan, by contrast to other tropical forests such as Amazonia, has a high species diversity (approx. 150-225 species Ha⁻¹), but a comparatively low density of individual taxa (Whitmore 1984; Gentry 1988).

Due to its position on the equator Kalimantan has a moist, tropical climate, West and Central Borneo are the wettest areas, falling into one of the wettest agroclimatic regimes (Oldman *et al* 1990). The climate of Borneo is typically constant at 28±°C (Fatawi and Mori 2000). Combined with this and Borneo's topographical lowland stature (Wilson and Moss 1999), it receives the necessary 60mm of rain per month for it to support evergreen rainforest (Holdridge 1967).

Tropical peat swamp forest

Tropical forests have previously been estimated to be directly accountable for 69% of the earth's biological productivity (Riswan and Hartini 1995). In 1979 Kartawinata, estimated that natural tropical forests covered four fifths of Kalimantan. Borneo has over 3,000 known species of trees, 267 of these are dipterocarps of which 58% are endemic to Borneo (Ashton 1982). Dipterocarps are the most commercially valuable trees in South East Asia (MacKinnon *et al* 1996). This diversity and high percentage of endemism is due to the association between flora and soil distribution over young rock (Aston 1989). The lowland forest of Kalimantan and Sumatra, are considered to be one of the most valuable remaining tropical forest estates of the world (Ashton 1984). Davis *et al* (1994) attributed at least some of the size and diversity of Borneo's rainforest to the theory of the rain forest contraction post-pleistone glaciation was much less severe than that which occurred in Africa and South America.

Of the 7 bio-units mentioned previously (MacKinnon and MacKinnon 1986), the southern lowland plains contains the vast region of tropical peat swamp forest (MacKinnon and MacKinnon 1986). Tropical peat swamp forest within Indonesia can be found mainly in Irian Jaya, Kalimantan and Sumatra, with small areas occurring in Java, Halmahera and Sulawesi (Rieley *et al* 1997). The tropical peat swamp ecosystem can be separated into 2 distinct types: ombrogenous - rainfed swamp, and topogenous - freshwater swamps. Topogenous swamp forests have only a shallow layer of peat deposits, therefore ombrogenous peat swamps are known as true peat swamp forest (Rieley *et al* 1997). Pollen samples from Central Kalimantan have shown that establishment of ombrogenous peat swamp has occurred on top of topogenous freshwater swamp (Morley 1981).

Peat swamp forests cover a large area of Kalimantan, approximately 8 to 11%, (MacKinnon and Artha 1981; Soepradptoahardjo and Driessen 1976), and provides 14.6% of the total forest cover for this area (RePProT 1990). Radiocarbon dating has established the origins of these basin and coastal peatlands from 800 to 5000 BP. (Siefferman *et al* 1988; Diemont and Supardi 1987). Ombrogenous peat swamp forest can retain peat depths of over 20m (Whitten *et al* 1987), with a constant water table near to or above the peat surface (Driessen 1977). The peat composes of former trees and organic material (Andriess 1972; Driessen 1977), resulting in a fibric nature with a low ash and mineral content (Anderson 1983).

Whitmore (1984a) noted that due to the difference in water sources, ombrogenous peat swamps have a more reduced nutrient content and flux, than topogenous peat swamps. This in turn results in a less species diverse vegetation in ombrogenous peat swamp. Despite this, peat swamp forests have been vilified as an important reservoir of diversity in South East Asia (Anderson 1963; Silvius *et al* 1987; Whitmore 1984). The vegetation of ombrogenous peat swamp forest (from this point referred to as peat swamp forest), falls into concentric forest zones, which supersede each other, into the centre of the peat swamp forest, as distance from river sources increases (Whitmore 1984; Dilmy 1965; Anderson 1963). Previously it was thought that the zonation of vegetation in Kalimantan was not so highly developed as in Brunei and Sarawak, (Muller 1965). Recent research into this area has however highlighted the presence of several previously unknown vegetation zones (Rieley *et al* 1997 and Shepherd *et al* 1997).

Typically species diversity decreases across this catena of vegetation (Anderson 1963). Tall canopy forest situated at the edge of the swamp is replaced with denser low growing

forest (Rieley *et al* 1997). Individual trees characteristically have dbh of < 30 cm and a height that rarely exceeds 20m. With this ‘low pole forest’; trees display a greater frequency of xeromorphism and chlorosis of foliage. (Rieley *et al* 1997).

It has been thought that although many species of mammals and birds do occur in peat swamp forest, few are specific to this habitat and therefore diversity is low (Merton 1962; Janzen 1974; Medway 1977; Davis and Payne 1982; Wells 1985; Whitten *et al* 1987). Only half the amount of known species of birds has been recorded in a peat swamp forest, when taken in comparison to a lowland evergreen forest sample (Wells *et al* 1979). Of the mammals that do appear in the peat swamp forest habitat, a high number of them occur in and around river edges (Wilson and Wilson 1975; Marsh and Wilson 1981; Davis and Payne 1982; Bennett 1988). Out of the endemic mammal species of Borneo, 15 of them are known to have a distribution limited to only lowland habitats (MacKinnon *et al* 1996).

Fauna

Borneo has over 200 land mammal species (Payne *et al* 1985), more than 500 resident and transient bird species (Smythies 1960), 166 species of snake (Stuebing 1991), 183 species of amphibians (Inger 1966) and an unknown countless number of invertebrates (MacKinnon *et al* 1996). Borneo has 44 endemic species of mammal, almost twice as many as mainland Sumatra (Payne *et al* 1985; MacKinnon 1990). This number includes 13 species of primate, 10 species of tree shrews and larger mammals such as the banteng (*Bos javanicus*), Sumatran Rhino (*Dicerorhinus sumatrensis*) and the Asian Elephant (*Elephas Maximus*) (MacKinnon *et al* 1996). The majority of the endemic species to be found however, are small rodents and bats, with a few small carnivorous species including the clouded leopard, (*Neofelis nebulosa*) and the Sunbear (*Helarctos malyanus*) (Payne *et al* 1985). An endemic species of the pygmy squirrel (*Exilisciurus exilis*) is the smallest known species of squirrel in the world (Payne *et al* 1985).

The avifaunal species of Borneo has strong affinity to those of Malaysia and Sumatra. It has a strong representation of forest bird families, including the hornbills (8 species), woodpeckers (18 species) and Pittas (13 species) (MacKinnon *et al* 1996). Borneo has approximately 420 resident bird species; with over 37 of them being endemic, this includes white fronted falconet (*Microhierax latifrons*), mountain serpent eagle (*Spilorins kinabaluensis*), Bulwer’s pheasant (*Lophura bulweri*) and blue-banded pitta (*pitta baudi*). Several of Borneo endemic species feature on the ICUN red data books, such as the

Orangutan (*Pongo pygmaeus*) (Groombridge 1993) and the blue banded-pitta area cited as endangered (ICUN 2000)

Davis (1962) estimated that due to the rich variety of canopy food sources, 45% of non-flying mammals in Borneo are arboreal. The faunal population of most rainforest, including Borneo can be roughly divided into 2 distinct behavioural groups, that of the diurnally active and the nocturnal (Harrison 1962; MacKinnon 1974). As with vegetational distribution, animal distribution can be delineated within the tropical forest according to elevation (MacKinnon 1997). This was classified by Harrison (1962), for tropical forest in the Malay Peninsula, into 6 distinct communities of birds and mammals. This division into communities was found to follow trends of food availability and feeding habit patterns.

This niche separation has allowed the evolution of animals to be successful whilst operating within numerous guilds (MacKinnon and MacKinnon 1978, Payne 1979). However a particular niche cannot be utilised by two closely related species, studies of groups of frugivorous bird species (Diamond 1972; Diamond 1973) and primate communities (Van Schaik *et al* 1993; Spalinger *et al* 1992; Chivers 1980; Rijksen 1978) highlight this. Fruiting trees are a main food source for many of Borneo's frugivorous mammals and birds. One of the main fruiting tree species favoured by primates is the fig tree (Snow 1981). It provides over 22% of the diets of gibbons (Gittins and Raemakers 1980) and a larger proportion of fruit for orangutans (MacKinnon 1974; Rijksen 1978). Figs also feature strongly within the diets of all eight species of Bornean hornbills (Cranbrook 1982).

Canopy birds have been found to be particularly vulnerable in tropical forests (Terborgh 1974; Willis 1974; Hilty 1985; Levey and Stiles 1994). This may partly be due to the lack of knowledge about these birds (Munn and Lioselle 1995). The only known in-depth study of this group was carried out by Nadkarni and Matelson (1989), on passerines and hummingbirds foraging on canopy epiphytes in the Monte Verde cloud forest in Costa Rica. It is proposed that they play a vital role as pollinators and in seed dispersal within the forest (Stiles 1985). And although canopy predators such as raptors occur at lesser densities (Thiollay 1989), their impact on canopy dwelling mammals such as shrews and mice may be large (Karr 1982; Lioselle and Hopps 1983). However Ramsen (1985) did note the huge restriction and difficulties of carrying out such research on this unknown group or guild.

Similarly to the distribution of the vegetation of Borneo, the diversity of faunal species is high but the actual abundance of individual species is low (MacKinnon *et al* 1996). The distribution of primate species supports this, with a mean of 13.5 groups/km² (Marsh and Wilson 1981), which is relative low when drawn in comparison with other tropical forest in Asia and Africa. One of Borneo's primates subject to the most intensive studies is the Orangutan (*Pongo Pymaeus*), can be used to illustrate this further. Due to its low natality rates, approx. 8 year (Galdikas and Wood 1990) and late reproductive age of approx. 12 years (Harvey *et al* 1987), fecundity rates for the orangutan are low. High maintenance food requirements for an individual requires a large and often solitary home ranges (MacKinnon 1974). These factors result the low densities of individuals in any given area (Rijksen *et al* 1995).

As mentioned previously, figs are an important source of food for frugivorous animals (Snow 1981). However they tend only to occur only with larger dipterocarp trees, and the abundance of this important food source has found to drop drastically in area that have been subjected to any form of disturbance, thus posing a serious threat to mammals and birds (Leighton and Leighton 1983). Primates such as gibbons and orangutans have been found to feed on young leaves, shoots and even bark of young dipterocarp species during times of food shortages (Chivers 1980; MacKinnon 1974). Though despite the large distribution of dipterocarp tropical forest, most mammals and birds do not feed on dipterocarp species even in times of food shortage (Medway 1972b). The only animals species that has been found to use dipterocarp as their main food source are bearded pigs (*Sus barbatus*) (Medway 1972; Pfeffer and Caldecott 1986), fruit eating bats and parakeets (Wyatt-smith 1963).

Deforestation

The Indo-Pacific region, of which Indonesia is a constituent part of, has some of the most species rich forest habitat in the world, however it is under increasing threat from deforestation (Myers 1988). In FAO/UNEP (1981) the blanket term of deforestation was clarified into two distinct terminologies: Deforestation: meaning alienation or conversion of forest areas to permanent or shifting agriculture or to other uses i.e. settlement, roads and mining. Degradation: meaning a rather more gradual reduction of biomass resulting from fire, overgrazing, over exploitation for fuel wood and charcoal etc. Though deforestation is still the term most frequently used to cover the majority of forms of tropical forest removal and will suffice for the purposes of this review.

Large-scale removal of timber for economic gain began in 1967, when the forested areas of Indonesia were declared property of the state under the basic forestry law and foreign capital 1967, and domestic capital law of 1968 (Riswan and Hartini 1995). Up until that time, the majority of forest removal had been on a subsistence level by the indigenous people e.g. Bornean Dyaks, who practised 'swidden' (Slash and burn) a low density method with a long fallow period (Avé and King 1986). Only a few concessions were being granted in the upper Barito in 1904 (Potter 1998). The impact of logging on the forests of Kalimantan steadily increased in the 1970's with a greater number of timber concession allotted to foreign timber companies (Avé and King 1986). This was in turn compounded by a drastic change in logging techniques, with the suspension of small scale logging in favour of mechanised logging (MacKinnon *et al* 1996).

The majority of logging has taken place in the lowland dipterocarp forest, including tropical freshwater and peat swamp forest (MacKinnon *et al* 1996). It has been carried out to such a degree that the forest has been reduced by at least a third (MacKinnon and MacKinnon 1986). By 1990 Kalimantan had lost over 7 million hectares in 20 years and this lost was calculated to occur at the rate of 1,000 hectares per day (MacKinnon *et al* 1996). A shockingly high number when out of the 3000 known tree species of Borneo, less than a 100 species are targeted by loggers for removal (Burgess 1966), and only 12 of those species are exported commercially (Myers 1984).

The method/techniques of logging used by timber companies has also been subject to some investigation (Burgess 1978; Meijer 1970). In government guidelines selective/polycyclic cutting systems, had to leave individual trees of a dbh <50 cm in a 'normal' production forest and <60cm in a limited production forest. These areas were then supposed to left for 25-40 years before extraction could resume. Cable logging in Sabah was found to cause extensive damage to adult tree, though little to saplings and seedlings (MacKinnon *et al* 1996). Mechanical ground logging was found by Johns (1988) to damage up to 40% of seedling through log haulage. Regeneration of a site of clear felled trees is an extremely slow process, which resulted in clear felled areas being replanted with non-native fast growing species *Acaia*, *Eucalyptus* and important cash species such as teak *tectona grandis*, (Tatawi and Mori 2000)

This focus on only a small number of species has led to a highly fragmented forest structure. Burgess (1971), study on the effects of such a style of logging, determined that if 10% of trees were targeted in an area for removal, post removal 55% of the remaining trees were damaged and only 35% left undisturbed. This theory was extrapolated by

Johns (1983a), by looking at a logging operation in its entirety. Of the 50.9% of trees removed from the forest; 3.3% were marketable, 4.8 % were lost to road building, 3.6% to logging skids and spar-tree sites and 39.2 were lost to incidental damage. This left only 49.1% trees remaining and out of that number a further 6% of those were damaged.

The communications established to allow access to the logging sites has been found to be extremely detrimental to the surrounding forest (Burgess 1973), 20 to 30% of the logged area was found to be bare soil created by roads and storage areas (Kartawinata 1979; Meijer 1970). The compaction of soil can lead to a decrease in water filtration and an increase in soil erosion and surface runoff (Liew 1974). In East Kalimantan clear felling of trees has been determined to increase soil erosion by almost 4 times that of a primary forest. (Stadtmuller and Hardiwanarto 1990). The increase of runoff, in turn causes the silt carrying capacity of nearby rivers to increase by up to 30 times more than in undisturbed areas (Hamzah 1978). Seed mortality is also greatly increased by soil compaction, as germination is highly unlikely to occur (Johns 1988).

If polycyclic extraction was carried out in a successful in this way, islands of pristine forest could act as a seed bank and a refuge for frugivorous animals vital to seed dispersal (Johns 1988). It has been estimated that it may take more than 100 years for those canopy species with a dbh <50 cm to mature to a climax state (Riswan *et al* 1985; Toma *et al* 1997). The forests regenerating to replace ancient tropical forests, have been found to have a vastly different species composition, with pioneer species dominating the habitat and the regrowth of commercial species being extremely restricted (Riswan 1982). The remaining forest landscape is a complex mosaic of different levels of canopy disturbance within patches of forest types (Cannon *et al* 1994).

Under the Indonesian government's 7th Development plan for 2000-2004, logging production for the whole of Indonesia is set to rise to 57.2 million m³/year. Industrial forests are set to produce 22.1 million m³ /year of this total and the remaining 18.3 million m³/year are to be extracted from natural forest. This number is twice as much as the target set for 1996 (Tatawi and Mori 2000)

Anthropogenic deforestation of Indonesia's rain forests has been compounded by natural events. The phenomenon of the cyclic southern oscillation weather front of El Nino ENSO (Cane 1983; Gurhardja *et al* 2000) has in the last 20 years caused severe drought within Borneo; this in turn lead to uncontrollable fires raging through the forests of Borneo. The two most severe occurring in 1983 (Woods 1989, Beaman *et al* 1985;

Mackie 1984) and in 1997 (Meijaard and Dennis 1997). In 1983 over 4.6 million hectares were burnt in Sabah and Kalimantan (Beaman *et al* 1985; Kinnaird and O'Brien 1998).

The scale of these fires, were mirrored again in 1997, were the number of hectares in Indonesia, damaged by fires was estimated to reach up to 5 million Ha (Schweithelm 1998). Of the millions of hectares burnt 1983, only an estimated 15 % of the burnt forest in Sabah (Beaman *et al* 1985) and 23% in Kalimantan (Woods 1989) were primary forest. A test site analysed using high resolution, multi-temporal ERS-2 SAR images and NOVA-AVHRR hotspot data, found that 71 % or 1.3 million ha of a 1.85 million ha sized test site was damaged in the 1997 forest fires (Siegert and Hoffman 2000). A huge percentage of a large site and a strong indicator of the immense damage that had been done to the forests of Kalimantan, and Borneo as a whole.

The majority of the remaining percentage of forest damaged by the fires were areas that had been subject to deforestation, this supported the theory that fire is not a common feature of everwet tropical primary rainforest (Whitmore 1983), due to their high moisture content (Mutch 1970). Though Bartlett (1955) highlighted that fire was the most frequently method employed in the destruction and disturbance of tropical forest for forest clearance. Leighton (1984) estimated that droughts of similar stature occurred in 1887-88, 1903-04, 1914-15, 1941-42 and 1972, but were not accompanied by fires. Woods (1989) observed that logged areas in the state of Sabah suffered much more damage in the 1983 fires than primary forest. One main factor for this was the abundance of combustible shrubs and vines present in disturbed forest (Wirawan 1984; Lennertz and Pazer 1984; Tagawa *et al* 1988)

Poore and Sayer (1991) noted that tropical forest species are particularly prone to extinction, due to a very narrow niche specification, but in turn this specificity has allowed for an even higher level of diversity and speciation. Many insects have restricted distribution to such as narrow are that it may be restricted to only a few species of tree: Erwin and Scott (1980) recorded 945 species of beetle from only a single tree species. However, this natural rate of extinction has been drastically increased as deforestation has been found to affect species richness, abundance and frequency of flora and fauna (Thiollay 1992). In general the rapid exploitation of these forest resources has resulted in the extinction of species that may have has some future economic benefit, loss of genetic resources, erosion, soil compaction, loss of water reserves and impact on local and global climate (Riswan and Hartini 1985)

Insects

Invertebrate studies have become a proven viable method in highlighting the level of the disturbance in a sampled area. To such a degree that a criteria has been established to assist in the selection of insects to be used as a biological indicator (Speight 1986; Luff and Worwood 1995; Holloway 1987) and in an aid in forest management (Niemela 1997). In tropical forest habitats the increased abundance, diversity and availability of invertebrates when compared to other habitats (Erwin 1983; Stork 1988; Knight and Holloway 1990). Has allowed them to become a group often used to indicate the effects of disturbances such as deforestation, in areas of tropical forest (Wolda 1987; Sutton and Collins 1991; Didham *et al* 1998). Two key groups of invertebrate sampled in tropical forest are macro-lepidoptera, specifically moths and butterflies (Hill *et al* 1995; Lovejoy *et al* 1986) and Coleoptera (Davis 1998; Stork 1987; Watt *et al* 1997; Klein 1989).

The preference for these two groups arises from their short life span, abundance and a clear niche affiliation (Niemela 1997). A large and diverse number of Lepidoptera and Coleoptera species have been found to be herbivorous (Gullan and Cranston 1994; Watt *et al* 1997), thus giving them an increased sensitivity to a change in a vegetation stand (Janzen 1973). Beyond this niche Coleoptera families fall within the primary consumer role for many of the tropical forest ecosystem products; Soil saprophages e.g. Curculionidae (weevil), decaying wood and litter xylophages, e.g. Platypodidae (Ambrosia beetle), dung coprophages e.g. Scarabaeidae (dung beetle) and carrion necrophages e.g. Cleridae (Stork 1997). These groups represent niches in the five types of principal biotypes of the lowland rainforest; the canopy, tree trunks, ground and low vegetation, leaf litter and soil (Stork 1988)

At present invertebrate abundance, diversity and spacio-temporal distribution in the tropics is poorly understudied (Sutton and Collins 1991; Stork 1988; Southwood 1978), particularly within Indonesia. The work that has been carried out on butterflies (Collins and Morris 1985; Hill *et al* 1995) and beetles (Hanski 1983; Stork 1987; Stork 1988; Paarmann and Stork 1987; Hammond 1990; Davis 1998). There is a strong focus in beetle sampling for either a specific habitat type such as tree canopies (Stork 1987; Paarmann and Stork 1987) or for a specific feeding habitat, in particularly coprophages and necrophages (Hanski 1983; Davis 1998).

In comparison between undisturbed and disturbed sites, species composition for butterflies was found overall, to increase in richness with and increase in disturbance

(Brown 1991; Lovejoy *et al* 1986). However, endemism or biogeographical distinctiveness of butterfly species was found to decrease with increased levels of disturbance (Hill *et al* 1995). Species richness and abundance for dung and carrion beetles has been found to decrease with disturbance except for the genus *Glaphyrocathion* where 97% of the species sampled were located only in highly disturbed (clear-cut) forested areas (Klein 1989), no fieldwork has been published concerning beetle endemism.

Table 1 - Summary of site history and sampling methods used for sample plots

Research transect	Forest type/ status	Location (GPS) ³	Survey date ³	Vegetation Plot area	Coleoptera Plot area/type	Number of trapping days
T1	Selectively logged M.S.F (19)	2°19'63"0S 113°54'53"5E	12.8.99	5 x 30 m ²	Carrion baited 75 x pitfall	3
T2	Illegally logged M.S.F (1997)	2°19'75"2S 113°54'30"7E	1.8.99	5 x 30 m ²	Carrion baited 75 x pitfall	3
T3	Completely Burnt M.S.F (1997/8)	2°20'52"9S 114°02'36"9E	21.8.99	5 x 30 m ²	Carrion baited 75 x pitfall	3
T4	Primary M.S.F ¹	2°20'67"3S 114°02'21"1E	20.9.99	5 x 30 m ²	Carrion baited 75 x pitfall Unbaited 75 x pitfall	3 2
T5	Primary/logged Tall Pole S.F ²	2°23'95"6S 113°50'47"8E	28.9.99	3 x 30 m ²	Unbaited 75 x pitfall	2 2
T6	Primary Low Pole S.F	2°22'16"8S 113°52'30"9E	29.8.99	3 x 30 m ²	Unbaited 75 x pitfall	2

¹Mixed Swamp Forest ²Swamp Forest ³ Start points only

Table 2 Showing the level at which the variables were subject to statistical analysis

<i>Data set to be analysed</i>	<i>Statistical analysis carried out (using SPSS 6.0)</i>	
	<i>One way ANOVA</i>	<i>Discriminant Analysis</i>
Level 1	All Mixed Peat Swamp Forest types samples (n=4)*	All of the forest types sampled (n= 6)
Level 2	All Primary forest type samples (n=3)	All of the plots sampled in all the forest types (n=30)
Level 3	All of the forest types sampled (n=6)	-

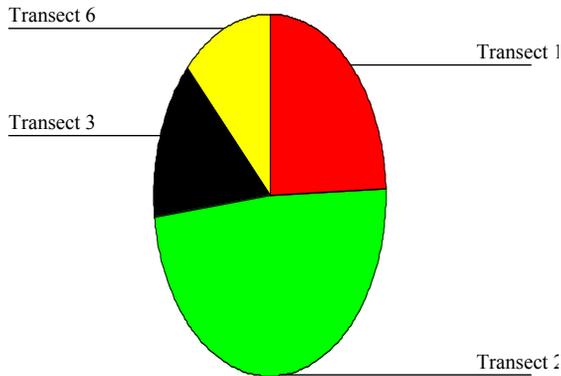
* n= number of individual sample sets.

Table 7 Descriptive of total number of coleoptera found in each transect

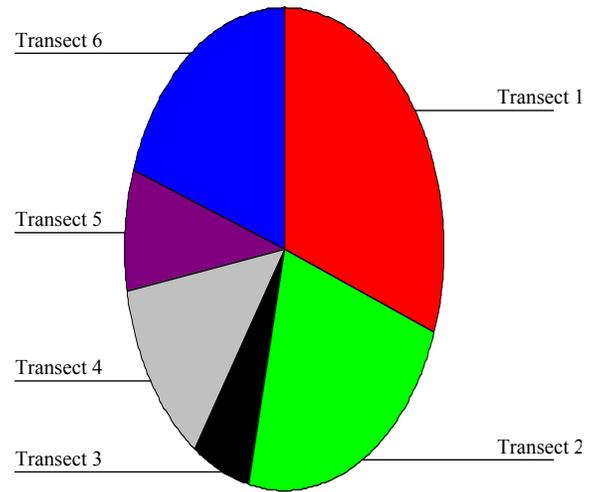
	Forest type	Total number of individuals trapped	Mean number (plot/transect)	Total number of species found
Carrion baited pitfall	Selectively logged M.S.F (Tran 1)	309	61.80 ± 13.55	42
	Illegally logged M.S F (Tran 2)	198	39.60 ± 12.76	39
	Burnt M.S.F (Tran 3)	84	18.60 ± 13.09	29
	Primary M.S.F (Tran 4)	727	145.40 ± 13.94	36
Unbaited pitfall	Primary/selective logged Tall Pole Forest (Tran 5)	38	7.60 ± 7.30	19
	Primary Low Pole Forest (Tran 6)	3	0.60 ± 0.89	1

Figure 2 Charts to show the percentage of canopy cover for all 6 transects at each of the canopy stratification level measured

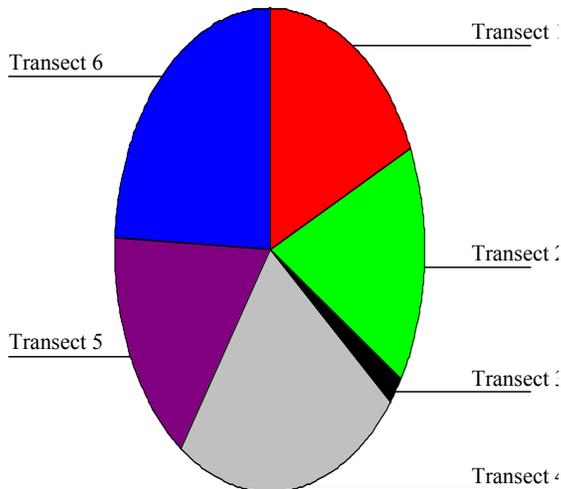
Ground canopy level (0 to 2 m)
5m)



Lower canopy level (2 to 5m)



Middle canopy layer (5 to 15m)
(15m+)



Upper canopy layer

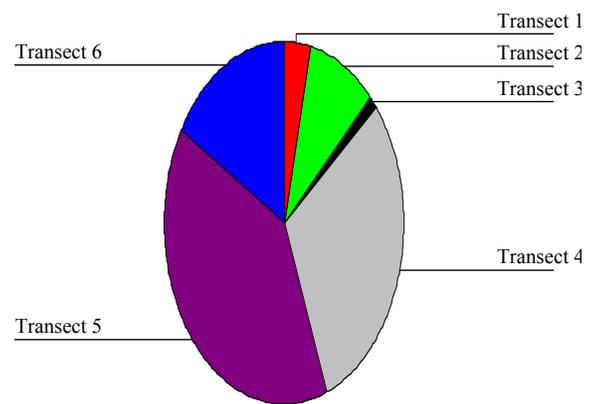


Figure3 Bar char to illustrate the biomass (Mg Ha⁻¹) distribution^{-plot}, within each forest type.

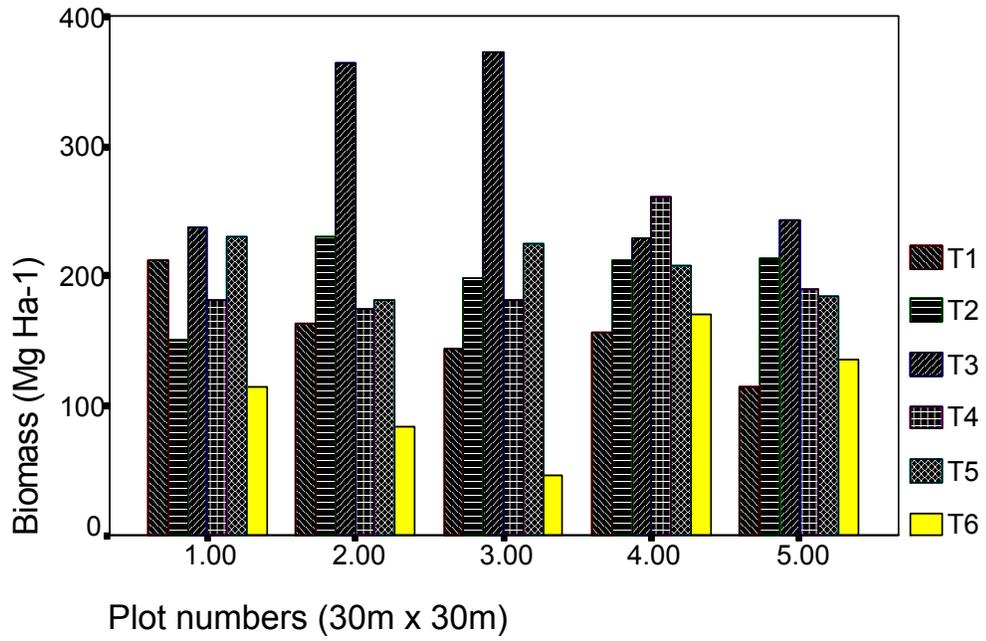
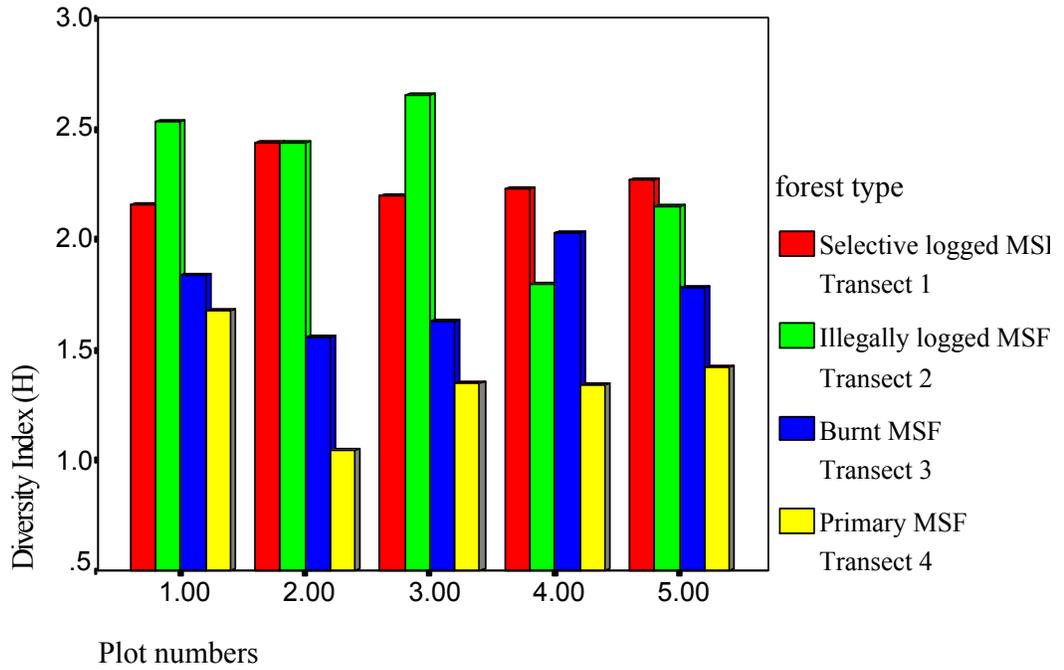


Fig 4 A bar chart to illustrate the Shannon weaver (H) insect diversity of each plot within each transect



Appendix i

Key for Tree Species for the Tropical Peat Swamp Forest. Setia Alam, Sungai Sebangau catchment area, Palangka Raya Administration, Central Kalimantan, Indonesia 1999.

Tree species local name (Dyak)	Tree species taxonomic name	Reference number
Banitan (Kambilitan)	<i>Polytia sp</i>	1
Balawan	<i>Tristania sp</i>	2
Benggaris		3
Bintangur	<i>Calopyllum sp</i>	4
Cemara	<i>Casuarina sp</i>	5
Hantangen	<i>Camnusperma auriculatum</i>	6
Hangkang	<i>Xylophia fusca</i>	7
Hangkapas		8
Galam Tikus	<i>Eugenia sp</i>	9
Geronggang	<i>Cratoxylon arborescen</i>	10
Gemur	<i>Alseodapne sp</i>	11
Rambutan-rambutan	<i>Nephelium sp</i>	12
Jambu – Jambuan	<i>Eugenia sp</i>	13
Jelutung(Pantung)	<i>Dyra cotulata</i>	14
Kapur naga	<i>Dryobalanop sp</i>	15
Katiau	<i>Palaquium sp</i>	16
Kayu sepat	<i>Erythroxylum cuneatum</i>	17
Lalas	<i>Homalium caryophyllaceum</i>	18
Mahalilis	<i>Palaquium sp</i>	19
Malilis	<i>Palaquium sp</i>	20
Manggis Manggisian	<i>Gacinia sp</i>	21
Marutan		22
Meranti	<i>Shorea sp</i>	23
Nyatuh Bawui	<i>Palaquim sp</i>	24
Perupuk	<i>Cantopilum calopylum</i>	25
Rambutan	<i>Nephelium sp</i>	26
Ramin	<i>Gunystilus bancanus</i>	27
Rasak	<i>Vatica sp</i>	28
Rasak Napu	<i>Vatica sp</i>	29
Tapagagas		30
Tarentang	<i>Camnusperma sp</i>	31
Tumih	<i>Combretocarpus rutondatus</i>	32
Uring Pahe	<i>Diospyros sp</i>	33
Balaugiran		34
Gamer		35

Appendix iii Insect abundance and total species numbers for transect 1-6 representing each forest type.

<i>Coleoptera</i> <i>Morph-Species -</i> <i>Family</i>	<i>Carrion Baited pitfall traps</i>				<i>Unbaited pitfall traps</i>	
	<i>Transect 1</i>	<i>Transect 2</i>	<i>Transect 3</i>	<i>Transect 4</i>	<i>Transect 5</i>	<i>Transect 6</i>
1-Scarabaeidae	1(5)	1(6)	-	1(40)	-	-
2-Scarabaeidae	1(9)	-	-	1(19)	-	-
3-Hybosoridae	1(14)	1(18)	-	1(41)	-	-
4-Scarabaeidae	-	-	1(4)	1(5)	1(1)	-
5-Scarabaeidae	1(26)	1(2)	-	1(10)	-	-
6-Scarabaeidae	1(51)	1(20)	2(19)	4(457)	-	-
7-Scolytidae	-	1(1)	1(1)	1(1)	-	-
8-Curculionidae	-	-	-	1(2)	-	-
9-Chrysomelidae	1(2)	-	1(1)	1(3)	-	-
10-Curculionidae	-	-	-	1(1)	-	-
11-Elateridae	-	-	-	1(1)	-	-
12-Staphylinidae	1(1)	-	1(2)	1(1)	-	-
13-Staphylinidae	1(1)	-	2(20)	1(10)	-	-
14-Staphylinidae	2(87)	2(1)	1(14)	1(50)	-	-
15-Staphylinidae	-	-	-	1(1)	-	-
16-Staphylinidae	-	-	-	1(5)	-	-
17-Staphylinidae	-	1(7)	-	-	-	-
18-Erotylidae	-	-	-	1(3)	-	-
19-Curculionidae	-	-	-	1(1)	-	-
20-Staphylinidae	-	-	1(1)	1(1)	1(1)	-
21-Hydrophilidae	-	1(9)	-	-	-	-
22-Hydrophilidae	1(5)	1(9)	-	-	-	-
23-Cerambycidae	1(1)	-	-	-	-	-
24-Carabidae	1(4)	1(2)	-	1(1)	-	-
25-Carabidae	1(3)	1(1)	1(1)	-	-	-
26-Curculionidae	1(1)	-	-	-	-	-
27-Histeridae	1(1)	-	-	1(1)	-	-
28-Curculionidae	1(1)	-	-	-	-	5 (5)
29-Chrysomelidae	1(1)	-	-	-	-	-
30-Chrysomelidae	-	-	-	1(1)	-	-
31-Curculionidae	1(1)	-	-	1(1)	-	-
32-Carabidae	1(1)	1(1)	-	-	-	-
33-Staphylinidae	1(9)	1(5)	1(5)	-	-	-
34-Carabidae	1(1)	-	-	-	-	-
35-Chrysomelidae: Alticinae	1(1)	-	-	-	-	-
36-Staphylinidae:	1(1)	-	-	-	1(2)	-
Scaphidiinae						
37-Tenebrionidae	1(1)	-	-	-	-	-
38-Buprestidae	1(4)	-	-	-	-	-
39-Lycidae	1(1)	-	-	-	-	-
40-Tenebrionidae	1(1)	-	-	1(1)	-	-
41-Unknown *	1(1)	-	-	-	-	-
42-Scaptidae	-	-	-	1(1)	-	-
43-Staphylinidae	-	-	-	1(1)	-	-
44-Anaspidae	-	-	-	1(1)	-	-
45-Carabidae	1(3)	-	1(3)	-	1(1)	-
46-Buprestidae	-	1(1)	-	-	-	-
47-Curculionidae	-	1(1)	-	-	-	-
48-Noteridae	1(1)	-	-	-	-	-
49-Staphylinidae:	-	1(1)	-	-	-	-
Scaphadinae						
50-Coccinellidae	-	1(1)	-	-	-	-
51-Elateridae	1(1)	-	-	-	-	-
52-Staphylinidae	1(28)	1(21)	1(3)	1(36)	1(7)	-
53-Scarabaeidae	-	-	-	1(1)	-	-
54-Ceratocanthidae	-	-	-	1(23)	-	-
55-Coccinellidae	-	1(1)	-	-	-	-
56-Curculionidae	-	1(2)	-	-	-	-
57-Curculionidae	-	-	-	1(1)	-	-
58-Erotylidae	1(1)	-	-	-	-	-
59-Curculionidae	-	-	-	1(1)	-	-
60-Curculionidae	-	-	-	1(2)	-	-
61-Curculionidae	1(1)	-	-	-	-	-
62-Atelabidae	1(1)	-	-	-	-	-
63-Unknown *	-	-	-	1(1)	-	-
64-Anthiciidae	-	1(1)	-	-	-	-
65-Carabidae	-	1(1)	-	-	-	-
66-Histeridae	-	1(1)	-	-	-	-
67-Elateridae	1(1)	-	-	-	-	-
68-Histeridae	-	-	-	1(1)	-	-
69-Chrysomelidae	1(7)	1(8)	1(2)	-	-	-
70-Staphylinidae	-	1(1)	-	-	-	-
71-Anthribidae	-	1(1)	-	-	-	-
72-Carabidae	1(11)	1(6)	-	-	-	-
73-Cleridae	-	1(1)	-	-	-	-
74-Curculionidae	1	1(5)	-	-	-	-
75-Carabidae	-	1(1)	-	-	-	-
76- Colydiidae	-	1(1)	-	-	-	-

77-Chrysomelidae	-	1(1)	-	-	-	-
78-Staphylinidae	-	1(1)	-	-	-	-
79-Scirtidae	-	1(1)	-	-	-	-
80-Scolytidae	-	1(1)	-	-	-	-
81-Hydrophilidae	1(7)	1(1)	1(1)	-	1(6)	-
82-Chrysomelidae	1(7)	1(2)	1(1)	1(1)	1(4)	-
83-Staphylinidae: Pselaphinae	1(1)	-	-	-	-	-
84-Chrysomelidae	1(1)	-	-	-	-	-
85-Leiodidae	-	1(1)	-	-	-	-
86-Unknown*	-	1(1)	-	-	-	-
87-Scirtidae	1(2)	1(1)	-	-	-	-
88-Scolytidae	-	-	1(2)	-	-	-
89-Cerambycidae	-	-	1(1)	-	-	-
90-Unknown*	-	-	1(1)	-	-	-
91-Tenebrionidae	-	-	1(1)	-	-	-
92-Unknown*	-	-	1(1)	-	-	-
93-Chrysomelidae	-	-	1(1)	-	-	-
94-Staphylinidae	-	-	1(1)	-	-	-
95-Carabidae	-	-	1(1)	-	-	-
96-Chrysomelidae	-	-	1(1)	-	-	-
97-Carabidae	-	-	1(1)	-	-	-
98-Chrysomelidae	-	-	1(1)	-	-	-
99-Chrysomelidae	-	-	1(1)	-	-	-
100-Curculionidae	-	-	1(1)	-	-	-
101-Chrysomelidae	-	-	1(1)	-	-	-
102-Scarabaeidae+	-	-	-	-	-	-
103-Staphylinidae	-	-	-	-	1(1)	-
104-Curculionidae	-	-	-	-	1(1)	-
105-Clambidae+	-	-	-	-	-	-
106-Staphylinidae	-	-	-	-	1(1)	-
107-Staphylinidae	-	-	-	-	1(2)	-
108-Buprestidae	-	-	-	-	1(1)	-
109- Carabidae	-	-	-	-	1(1)	-
110-Histeridae+	-	-	-	-	-	-
111-Staphylinidae	-	-	-	-	1(4)	-
112-Staphylinidae+	-	-	-	-	-	-
113-Staphylinidae	-	-	-	-	1(1)	-
114-Staphylinidae	-	-	-	-	1(1)	-
115-Staphylinidae+	-	-	-	-	-	-
116-Leodidae+	-	-	-	-	-	-
117-Histeridae+	-	-	-	-	-	-
118-Corylophidae+	-	-	-	-	-	-
119-Scolytidae+	-	-	-	-	-	-
120-Scolytidae+	-	-	-	-	-	-
121-Unknown *	-	-	-	-	1(1)	-
122-Unknown *	-	-	-	-	1(1)	-
123-Unknown * +	-	-	-	-	-	-
Total No. Species	42	39	29	36	19	1
Total No. Individuals	309	198	72	727	38	3

* Coleoptera that are currently under analysis by South East Asian specialist in Japan due to identification is unable to be carried out with expertise currently available in U.K and/or possibly being a species new to science. + Species that were hand collected and were not analysed in this project, however for complete listing of morpho-species they were included.

Bibliography

- Abudulhadi, R Karatawunata, K and Sukardjo, S., (1981) Effects of mechanised logging in the lowland dipterocarp forest at Lemake, East Kalimantan. *Malay. For.* 44:407-418
- Aide, T.M (1992) Dry season leaf production: an escape from herbivory. *Biotropica* 24: 532-537
- Alvarez-Buylla, E.R., Garcia-Barrios, R., Lara-Moreno, C. and Martinez-Ramos, M., (1996) Demographic and genetic models in conservation biology: Applications and perspectives for tropical rainforest.
- Brown, N.D., (1993) The implications of climate and gap microclimate for seedlings growth conditions in a Bornean Lowland rainforest. *Jnl of Tropical Ecology* 9: 153-168
- Brown, S. and Iverson, L.R (1992) Biomass estimates for Topical forest. *World Resources. Rev.* 4: 366-384
- Beaman, F.A., Beaman, J.H., Marsh C. and Woods, P., (1985) Drought and forest fires in Sabah in 1983 *Sabah Society Journal* 8: 10-30
- Becker, P., Meinzer, F.C and Wullschlegers, P., (2000) Hydraulic limitations of tree height: a critique *Functional Ecology* 14:4 –11
- Beckman, T., (1981) Structural dynamic aspects of the occurrence and development of lianas in the tropical rainforest. Dept. Forestry Agric.Univ. Wageningen
- Burgess, P.F (1971) The effects of logging on hill dipterocarp forest. *Malay. Nat. Jnl.* 24 :231-237
- Boyle T.J.B and Sayer J.A (1995) Measuring, monitoring and conserving biodiversity in managed tropical rainforest. *Commonwealth Forestry Review* 74: 21-25
- Box, E (1995) Factors determining distributions of tree species and plant functional types *Vegetatio* 121: 101-116
- Brown, S and Iverson, L.R (1992) Biomass estimates for tropical forest. *World Resources Rev.* 4: 366-384
- Cane, M.A (1983) Oceanographic events during El Niño *Science* 222:1189-1195
- Cannon, C.H, Peart, D.R Leighton, M and Kartawinta (1994) The structure of lowland rainforest after selective logging in West Kalimantan. *Forest Ecology and Management* 67: 49- 68
- Chai, D.N.P and Udarbe, M.P, (1977) The effectiveness of current silviculture practice in Sabah, Malay. *Forest.* 38:271-277
- Cedergren, J. (1996) A silvicultural evaluation of stand characteristics, pre-felling climber cutting and directional felling in a primary dipterocarp forest in Sabah, Malaysia. PhD Thesis, Swedish University of Agricultural Sciences, Umeå. Sweden

- Cochrane, M.A and Schulze, M.D (1998) Forest fires in the Brazilian Amazon, Conservation biology 12: 948–950
- Coley, P.D (1983) Herbivory and defensive characteristics of tree species in a lowland tropical forest. Ecol. Monogr. 53:209-233
- Connell, J.H (1978) Diversity in a tropical rainforest and coral reefs. Science 199:1302-1310
- Connell, J.H (1971) On the role of natural enemies in preventing competitive exclusion in some marine animals and in rain forest trees. In. dynamics of populations (Eds) P. J den Boer and G. R Gradwell . Pp 298-312. Proceedings of the Advanced Study Institute on Dynamics of Numbers in Populations, Oosterbeek, The Netherlands, Sept. 1970. Centre for Agricultural publishing and Documentation Wageningen
- Corner, E.U (1988) Wayside trees of Malaysia. 3rd Edn. United Selangor Press Kuala Lumpur, Malaysia
- Davis, A (1998) Dung beetle Abundance and Diversity in the Maliau Basin, Sabah, Malaysian Borneo Malaysian Nature Journal 52 (3 & 4): 181-191
- Debinski, D and Brusard, P.F (1992) Biological diversity assessment in Glacier National Park Montana. Sampling design. In D.H Mackensie, D.E Hyatt and V.J McDonald, (Eds.) Ecological Indicators (2 vols.) Elsevier London pp 393-407
- Didham, R.K., Hammond, P.M., Lawton, J.H., Eggleton, P., Stork, N.E (1998) Beetle responses to tropical forest fragmentation. Ecological Monographs 68: 295-323
- Eyre, M.D and Luff, M.L (1990) An initial classification of the habitats of grassland ground beetles in the United kingdom. Entomologist Gazette, 41:197-208
- Finegan B, Camacho, M and Zamora, N (1999) Diameter increment patterns among 106 tree species in a logged and silviculturally treated Costa Rican rain forest. For.Ecol. and Manag. 121:159-176
- Fox, J.E.D (1978) Some data on the growth of *Anhocephalus cadamba* (Roxb) Miq. Insba. Malaya. Forest. 31: 89-100
- Fox, J.E.D (1976) Constraints on the natural regeneration of tropical moist forest. Forest Ecology and Manag. 1:37-65
- Fox, J.E.D (1972) The natural vegetation of Sabah and natural regeneration after logging in tropical rainforest. Jnl of forest Ecol. And Manag. 1:512-536
- Frankie, G.H, Baker H.G., Opler, P.A , (1974) Comparative phenological studies of trees in tropical wet and dry forests in the lowlands of Costa Rica. Jnl. Ecol. 62:881-919
- Gentry, A.H. (1988) Changes in plant community diversity and floristic composition on evolutionary and geographic gradients. Ann. of the Missouri Botanical Gardens 75: 1-34
- Gillis, M (1988) Indonesia: public policies, resource management and the tropical forest. In Public policies and the misuse of forest resource. (Eds), R.Repetto and M Gillis, pp 43-114. Cambridge Univeristy Press. Cambridge U.K

- Gillison A.N and Brewer, K.R.W (1985) The use of gradient directed transects or gradsects in natural resource surveys. *Jnl. of Environmental Management* 20: 103-127
- Greenslade, P.J.M (1964) The distribution, dispersal and size of a population of *Nebria brevicollis* (F.) with comparative studies on three other Carabidae. *Jnl. Animal Ecology*. 33: 311-333
- Guhardja, E., Fatani, M., Suitisna, M., Nori, T., Ohta, S. (2000) Rainforest ecosystems of East Kalimantan. Springer Verlag, Tokyo pp331
- Haila, Y and Margules, C.R (1996) Survey research in conservation biology *Ecography*, 19: 232-331
- Haila, Y and Levins, R (1992) *Humanity and Nature, Ecology, Science and Society* Pluto Press London.
- Haila, Y and Hanski, I.K (1984) Methodology for studying the effects of habitat fragmentation on land birds. *Ann. Zool. Fennici* 21: 393 – 397
- Haila, Y. (1992) Measuring nature quantitative data in field biology. In Clarke, A.E and Fujimura J.H (Eds.) *The right tools for the jobs. At work in twentieth-century life sciences*. Princeton University Press. N.J pp 233-253
- Hallé, F. and Oldman R.A.A (1975) *an Essay on the architecture and dynamics of growth of tropical trees*. Penerbit Universiti Malaya, Kuala Lumpur Malaysia
- Hallé, F. (1995) Canopy architecture in tropical trees: a pictorial approach. In: *Forest Canopies* (Eds) M.D Lowman and N.M Nadkarni Pp27-44. Academic Press.
- Hallé, F. (1978) Architectural variation at the specific level in tropical trees. In *Tropical trees as a living systems* (eds) .P.B Tomlinson and M.H Zimmerman pp 209-221. Cambridge University Press, Cambridge. U.K
- Heywood, V.H (1995) *Global biodiversity assessment*. Cambridge university Press, Cambridge U.K
- Holdsworth, A.E and Uhl, C (1997) Fire in Amazonian selectively logged rainforest and the potential for fire reduction *Ecological applications* 7: 713-725
- Holloway, J.D., Kirk-Spriggs, A.H and Chey von Khen (1992) The response of some rainforest insect groups to logging and conversion to plantations *Phil. Trans. Royal Soc. London B* 335:425-436
- Holloway, J.D (1987) Macrolepidoptera diversity in the Indo-Australian tropics: geographic, biotopic and taxonomic variations. *Biol. Jnl. Linnean Soc.* 30:35-341
- Huth, A., Ditzer, T., Bossel, H., (1997) Rainforest growth model FORMIX3: A tool for forest management planning towards sustainability. Model development and case study for Dermakot Reserve in Sabah, Malaysia. *Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ) GmbH, Eschborn, Pp 78*.
- Janzen, D.H (1973) Sweep samples of Tropical foliage insects: effects of season, vegetation types, elevation, time of day and insularity. *Ecology* 54: 687-708

- Janzen, D.H (1970) Herbivores and the number of tree species in tropical forest. *American Naturalist* 104:501-528
- Kahn, F (1983) Architecture comparée de forêt tropicales humides et dynamique de la rhizosphere. Thesis, University of Montpellier, France
- Kartawinata, K. Jessup, T.C, Vayda, A.P (1989) Exploitation in Southeast Asia. In *Tropical rainforest ecosystems* (Eds) H. Leith and MJA Weger Pp 591-610 Elsevier, Amsterdam. The Netherlands
- Kartawinata, K., Adisoemarto, S., Riswan, S., Vayda, A.P (1981) The impact of man on tropical forest of Indonesia. *Ambio* 10: 115-119
- Kennedy, D.N and Swaine, M.D (1992) Germination and growth of colonizing species in artificial gaps of different sizes in dipterocarp rainforest. *Phil. Trans. Royal Soc. of London Series B* 335:357-368
- King, G.C and Chapman, W.S (1983) Floristic composition and structure of a rainforest 25 years after logging. *Aus. Jnl. of Ecol.* 8: 415-423
- Kuuluvainen, T., (1988) Troopisen sademetsän arkkitehuuri. Summary: the architecture of tropical rainforests- relationships between structure and function. *Silva Carelica* 12:41-56
- Lovejoy, T.E Bierregard R.O, Rylands Jnr A.B, Malcom, J.R, Quintela, C.E, Harper, L.H, Brown, K.S, Powell, Jnr A.H., Shubart, H.O.R and Hays, M.B (1986) Edge and other effects of Isolation on Amazon forest fragments. Pp 257-285 In M.E Soule (Ed) *Conservation Biology: the science of scarcity and diversity*. Sinauer Sunderland. Massachusetts, USA
- MacKinnon, J. (1974) The behaviour and ecology of wild orangutans (*Pongo Pymaeus*) *Animal behaviour* 22:3-74
- Makihara, H., Kinuura, H., Yahiro, K. and Soeyamto, C (2000) The effects of droughts and fires on Coleopteran insects on lowland Dipterocarp forest in Bukit Soeharto, East Kalimantan. In: *Rainforest ecosystems of East Kalimantan*. (Eds) E.Guhardja, M. Fatwai, M.Sutisna, T.Mori, S.Ohta. pp 153-163 Springer Verlag, New York and Tokyo
- Marsden, S.J (1998) Changes in Bird abundance following selective logging on Seram, Indonesia *Conservation Biology* 12:605-611
- Melborne, B.A. (1999) Bias in the effect of habitat structure on pitfall traps: An Experimental evaluation. *Australian Jnl. of Ecology* 24:228-239
- Mori, T. (2000) The effects of forest fires and drought on dipterocarp forest in East Kalimantan. In *Rainforest ecosystems of East Kalimantan* (Eds) E .Guhardja, M. Fatani, M., Suitsisna, T. Nori, S. Ohta, (2000) Springer Verlag, Tokyo
- Mutch, R.W (1970) Wildland fires and ecosystems a null hypothesis *Ecology* 51:1046-1051
- Myers, N (1990) The Biological challenge: extended hotspot analysis. *Environmentalist* 10:243-56

- Niemela, J., (1997) Invertebrates and Boreal Forest Management. *Conservation Biology* 11:601-610
- Ng, F.S.P (1983) Ecological principles of tropical lowland rain forest conservation. In (Eds) S.L Sutton, T.C Whitmore and A.C Chadwick. *Tropical rainforest: ecology and management*, pp0 359-375. Blackwells Scientific Publ .Oxford U.K
- Okimori, Y and Matius, P. (2000) Tropical secondary forest and its succession following traditional agriculture in Menicai, East Kalimantan. In Pp 185-197
- Oldeman, R.A.A., (1983) Tropical rainforest, architecture, silvigenesis and diversity. In: *Tropical rainforests: ecology and management*. (Eds.) S.L Sutton, T.C Whitmore and A.C. Chadwick, pp 139-150. British Ecological Soc. Special publication 2.
- Oldeman, R.A.A., (1974) L'architecture de la forêt guyanaise. Memoire ORSTOM No. 73 Paris France
- Page, S.E., Rieley, J.O., Shotyk, Ø. W and Weiss, D. (1999) Interdependence of peat and vegetation in a tropical peat swamp forest. *Phil. Trans. R. Soc London B* 354:1885-1897
- Page, S.E., Rieley, J.O. and Neuzil, S.G., (1997) The role of tropical peatlands and peat swamp forests in the global carbon balance: Preliminary findings from the high peats of Central Kalimantan, Indonesia. 5th International symposium on the biogeochemistry of wetlands, Royal Holloway College, University of London, Sept. 1997
- Pinnard, M.A and Cropper, W.P. (2000) Simulated effects of logging on carbon storage in dipterocarp forest. *Jnl. Applied Ecol.* 37:267-283
- Pinnard, M.A., Barker, M.G and Tay J (2000) Soil disturbance and post logging forest recovery on bulldozer paths in Sabah, Malaysia. *Forest ecology and Manag.* 130: 213-225
- Putz, F.E and Pinnard, M.S (1993) Reduced –impact logging as a carbon offset methods. *Conservation Biology* 7:755-757.
- Putz, F.E (1995) Vines in treetops: Consequences of mechanical dependence. In *Forest canopies* (Eds) M.D Lowman and N.M Nadkarni pp311-321 Academic Press
- Raison, R.J, Khanna, P.K Woods, P.V (1985) Transfer of elements to the atmosphere during low-intensity prescribed fires in three Australian subalpine euclypt. forest . *Can Jnl. for Res.* 15: 657-664
- Radjagukguk, B. (1997) Peat soils of Indonesia: Location, classification and problems for sustainability In *Tropical Peatlands* (Eds) J.O Reiley and S.E Page. pp45-53 Samara Publishing Limited, Cardigan press U.K
- Rieley, J. O., Page, S.E., Limin, S.H and Winarti, S. (1997) .The peatland resource of Indonesia and the Kalimantan peat swamp forest research project. In *Tropical Peatlands* (Eds) J.O Reiley and S.E Page. Samara Publishing Limited, Cardigan U.K
- Richards, P.W (1996) *The tropical rainforest* 2nd Edn. Cambridge University Press. Cambridge, U.K
- Riswan, S. and Hartini, L. (1995) Human Impacts on tropical forest dynamics *Vegetatio* 121:41-52

- Ryan, M.G and Yoder, B.J (1997) Hydraulic limit to tree height and growth *Bioscience* 47: 235-242
- Schweithelm, J. (1998) The fire this time. An over view of Indonesia's forest fires in 1997/98. Indonesian discussion paper WWF Jakarta
- Shepherd, P.A, Rieley, J.O and Page, S.E (1997) The relationship between forest vegetation and peat characteristics in upper catchment of Sungai Sebangau, Central Kalimantan. In *Tropical Peatlands* (Ed) J.O Rieley and S.E Page, Samara Publishing Ltd, Cardigan U.K
- Simberloff, D (1992) Species-area relationships, fragmentation and extinction in tropical forests. In *In harmony with nature* (Eds) S.K Yap and S.W Lee, pp 398-413. Proc. Int. Conf. On Conservation of Tropical Biodiversity.
- Soepadmo, E., (1995) Plant diversity of the Malesian tropical rainforest and its phytological and economic significance. In: *Ecology, Conservation and Management of the South East Asian Rainforests*. (Eds.) R.B Primark and T.E Lovejoy, pp 19-40. Yale University Press, New Haven and London.
- Soulé, M.E (199) Conservation: tactics for a constant crisis. *Science* 253: 744-750
- Speight, M.C.D., (1986) Criteria for the selection of insects to be used as bio-indicators in nature conservation pp 485-488. Proceedings of 3rd European Congress of Entomology 3: 65-75, Amsterdam
- Stocker G.C (1981) Regeneration of a North Queensland rainforest following cutting and burning. *Biotropica* 13:86-92
- Stork N.E (1987) Guild Structure of Arthropods from Bornean rainforest trees. *Ecological entomology* 12: 69-80
- Tagawa. H., Suzuki, E., Wirawan, N., Miyagi, Y., Oka, N., (1988) Changes in vegetation in Kutai National Park, East Kalimantan. In *Research progress of early recovery , tropical rainforest after large scale fires, Kalimantan Timor*. Indonesia (Eds) H Tagawa and N Wirawan. Pp12-50. Indonesia Research centre, South Pacific, Kagoshima University , Japan
- Tang, H.T., (1987) Problems and strategies for regenerating dipterocarp forest in Malaysia. In: *Natural Management of topical moist forests: silvicultural and management prospects of sustained utilisation*, (Eds.) F. Mergen and J.R Vincent, pp 23-46. Yale University, New Haven C.T,
- Tatawi, M and Mori, T (2000) Description of forest and forestry in East Kalimantan. In *Rainforest ecosystems of East Kalimantan*. (Ed) E.Guhardja, M. Fatwai, M.Sutisna, T.Mori, S.Ohta. p 3-11 Springer Verlag, New York and Tokyo.
- Torquebiau, E. F., (1986) Mosaic Pattern in dipterocarp rainforest in Indonesia, and their implications for practical forestry. *Journal of Tropical Ecology*. 2: 301-325
- Turner, I.M (1990) The seedling and growth of 3 *Shorea* spp. in a Malaysian tropical rainforest. *Jnl. of Tropical ecology* 6:469-478
- Uhl, C. and Kauffman, J.B (1990) Deforestation, fire susceptibility and potential rainforest tree responses to fire in the Eastern Amazon.. *Ecology* 71: 437-449

- Utterera, J., Tokola, T. and Maltamo, M. (2000) Differences in the structure of primary and managed forests in East Kalimantan, Indonesia. *Forest Ecology and Management* 129: 63-74
- Van Schaik, C.P., Terborgh, J.W and Wright, S. J (1993) The Phenology of Tropical forest: Adaptive significance and consequences for Primary consumers. *Annu. Rev. Ecol. Syst.* 24: 353-377
- Webb, L.J, Tracey, J.G, Williams W.T (1972) Regeneration and pattern in the subtropical rainforest. *Jnl. Ecolo.* 601: 675-695
- Whittaker, R.H (1972) Evolution and Measurement of Species diversity *Taxon* 21:213-251
- Whittaker, R.J., Partomihardjo, T., and Jones, S. H. (1999) Interesting times on Krakatau: stand dynamics in the 1990's : *Phil. Trans. R. Soc. Lond. B* 354: 1857-1867
- Whitmore, T.C (1990) An introduction to tropical rainforest Clarendon Press, Oxford
- Whitmore, T.C (1984) Tropical rainforest of the Far East 2nd Edn Clarendon Press, Oxford U.K
- Whitmore, T.C (1983) Secondary succession from seed in tropical rainforest. *Forestry abstracts* 44:767-779
- Whitmore, T.C (1978) Gaps in the forest canopy. In *Tropical Trees as Living Systems* (Eds) P.B Tomlinson and M.h Zimmerman, pp 639-655. University Press, Cambridge U.K
- Whitmore, T. C. (1975). *Tropical rain forests of the Far East*, Clarendon Press, Oxford, England
- Wirawan, N, (1984) Kutai National Park and the great Kalimantan fire. *WWF monthly Report* June 1984: 125-131
- Wolda, H. (1983) Diversity and diversity indices and tropical cockroaches. *Oecologia (Berlin)* 58:290-298
- Woods, P. (1989) Effects of logging, drought and Fire on Structure and composition of Tropical Forest in Sabah, Malaysia. *Biotropica* 21:4, 290-298
- Wright, S.J and Cornejo, F.H (1990) Seasonal drought and the timing of flowering and leaf fall in a neotropical forest. In *Reproductive ecology of tropical forest plants* (Eds) K.S Bawa and M Hadley, pp 46-61. Paris/Carnforth: UNESCO/Parthenon
- Wolda, H (1983) Fluctuations in abundance of tropical insects. *American Nat.* 112:1017-45
- Yahata, H., (2000) Photographic estimation of light environments on forest floors and effects of light on growth of Dipterocarp seedlings. In: *Rainforest ecosystems of East Kalimantan*. (Eds) E.Guhardja, M. Fatwai, M.Sutisna, T.Mori, S.Ohta. pp259-267 Springer Verlag, New York and Tokyo.
- Yeager, C. (1998) Interim report on the fire impacts on Tanjung Puting National Park, *WWF Indonesia August Report*, Jakarata Indonesia

Appendix i

Key for Tree Species for the Tropical Peat Swamp Forest. Setia Alam, Sungai Sebangau catchment area, Palangka Raya Administration, Central Kalimantan, Indonesia 1999.

Tree species local name (Dyak)	Tree species taxonomic name	Reference number
Banitan (Kambilitan)	<i>Polytia sp</i>	1
Balawan	<i>Tristania sp</i>	2
Benggaris		3
Bintangur	<i>Calopyllum sp</i>	4
Cemara	<i>Casuarina sp</i>	5
Hantangen	<i>Camnusperma auriculatum</i>	6
Hangkang	<i>Xylophia fusca</i>	7
Hangkapas		8
Galam Tikus	<i>Eugenia sp</i>	9
Geronggang	<i>Cratoxylon arborescen</i>	10
Gemur	<i>Alseodapne sp</i>	11
Rambutan-rambutan	<i>Nephelium sp</i>	12
Jambu – Jambuan	<i>Eugenia sp</i>	13
Jelutung(Pantung)	<i>Dyra cotulata</i>	14
Kapur naga	<i>Dryobalanop sp</i>	15
Katiau	<i>Palaquium sp</i>	16
Kayu sepat	<i>Erythroxylum cuneatum</i>	17
Lalas	<i>Homalium caryophyllaceum</i>	18
Mahalilis	<i>Palaquium sp</i>	19
Malilis	<i>Palaquium sp</i>	20
Manggis Manggisian	<i>Gacinia sp</i>	21
Marutan		22
Meranti	<i>Shorea sp</i>	23
Nyatuh Bawui	<i>Palaquim sp</i>	24
Perupuk	<i>Cantopilum calopylum</i>	25
Rambutan	<i>Nephelium sp</i>	26
Ramin	<i>Gunystilus bancanus</i>	27
Rasak	<i>Vatica sp</i>	28
Rasak Napu	<i>Vatica sp</i>	29
Tapagagas		30
Tarentang	<i>Camnusperma sp</i>	31
Tumih	<i>Combretocarpus rutondatus</i>	32
Uring Pahe	<i>Diospyros sp</i>	33
Balaugiran		34
Gamer		35

Figure 2

Tree Canopy Shape Index (after Hallé)

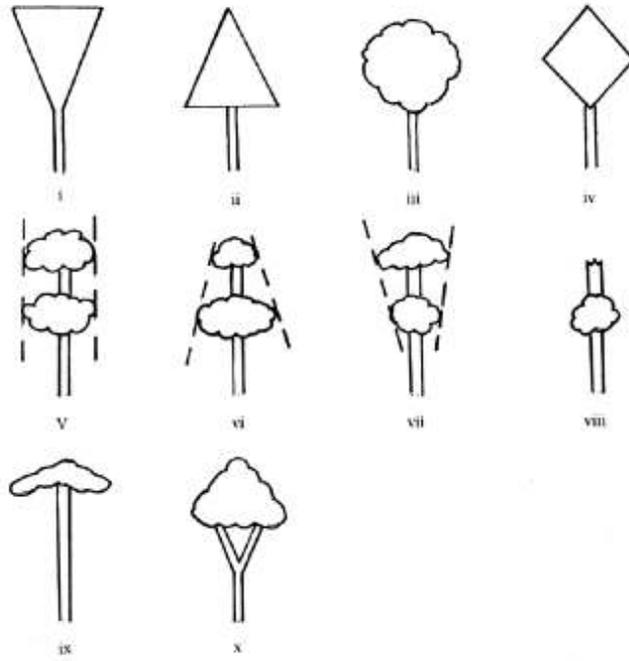
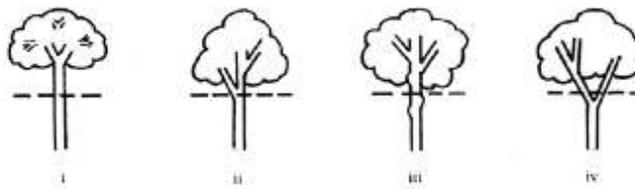


Figure 3

Tree Architecture Index (after Torquebiau)



Appendix ii

Key for Coleoptera morpho-species for the Tropical Peat Swamp Forest. Setia Alam, Sungai Sebangau catchment area, Palangka Raya Administration, Central Kalimantan, Indonesia 1999.

No.	Coleoptera Family	No.	Coleoptera Family	No.	Coleoptera Family
1	Scarabaeidae	44	Anaspidae	87	Scirtidae
2	Scarabaeidae	45	Carabidae	88	Scolytidae
3	Hybosoridae	46	Buprestidae	89	Cerambycidae
4	Scarabaeidae	47	Curculionidae	90	unknown
5	Scarabaeidae	48	Noteridae	91	Tenebrionidae
6	Scarabaeidae	49	Staphylinidae: Scaphadinae	92	Unknown *
7	Scolytidae	50	Coccinellidae	93	Chrysomelidae
8	Curculionidae	51	Elateridae	94	Staphylinidae
9	Chrysomelidae	52	Staphylinidae	95	Carabidae
10	Curculionidae	53	Scarabaeidae	96	Chrysomelidae
11	Elateridae	54	Ceratocanthidae	97	Carabidae
12	Staphylinidae	55	Coccinellidae	98	Chrysomelidae
13	Staphylinidae	56	Curculionidae	99	Chrysomelidae
14	Staphylinidae	57	Curculionidae	100	Curculionidae
15	Staphylinidae	58	Erotylidae	101	Chrysomelidae
16	Staphylinidae	59	Curculionidae	102	Scarabaeidae
17	Staphylinidae	60	Curculionidae	103	Staphylinidae
18	Erotylidae	61	Curculionidae	104	Curculionidae
19	Curculionidae	62	Attelabidae	105	Clambidae
20	Staphylinidae	63	Unknown *	106	Staphylinidae
21	Hydrophilidae	64	Anthicidae	107	Staphylinidae
22	Hydrophilidae	65	Carabidae	108	Buprestidae
23	Cerambycidae	66	Histeridae	109	Carabidae
24	Carabidae	67	Elateridae	110	Histeridae
25	Carabidae	68	Histeridae	111	Staphylinidae
26	Curculionidae	69	Chrysomelidae	112	Staphylinidae
27	Histeridae	70	Staphylinidae	113	Staphylinidae
28	Curculionidae	71	Anthribidae	114	Staphylinidae
29	Chrysomelidae	72	Carabidae	115	Staphylinidae
30	Chrysomelidae	73	Cleridae	116	Leodidae
31	Curculionidae	74	Curculionidae	117	Histeridae
32	Carabidae	75	Carabidae	118	Corylophidae
33	Staphylinidae	76	? Colydidae	119	Scolytidae
34	Carabidae	77	Chrysomelidae	120	Scolytidae
35	Chrysomelidae: Alticinae	78	Staphylinidae	121	Unknown *
36	Staphylinidae: Scaphidiinae	79	Scirtidae	122	Unknown *
37	Tenebrionidae	80	Scolytidae	123	Unknown *
38	Buprestidae	81	Hydrophilidae		
39	Lycidae	82	Chrysomelidae		
40	Tenebrionidae	83	Staphylinidae: Pselaphinae		
41	Unknown *	84	Chrysomelidae		
42	Scraptiidae	85	Leiodidae		
43	Staphylinidae	86	unknown		

* Coleoptera that are currently under analysis by South East Asian specialist in Japan as either identification is unable to be carried out with current keys available in the U.K. or possibly being new to science

