Practical Project Report 2014/2015 School of Biological Sciences

Changes in forest structure and tree species composition after logging in tropical peat-swamp forest in Central Kalimantan, Indonesia.

Photo: Katrina Schofield/OuTrop

Katrina Schofield

Supervisor: David Burslem

Work completed in partial fulfilment of the requirements for a degree
in Honours Conservation Biology

Word count (excluding references, tables, figures and appendices): 11,452

Declaration

"I hereby declare that this thesis is my own work. Where I have used the work of other persons or quoted the work of other persons the sources of the other work or information have been detailed explicitly in the presentation."

Signed…Katrina Schofield.

Contents

Abstract

Conservation of tropical peat-swamp forests, which are globally important stores of carbon and biodiversity, is becoming increasingly important due to the large extent of destruction and conversion. This research was carried out in the NLPSF in the Sabangau Forest, Kalimantan in collaboration with OuTrop and CIMTROP to investigate the impacts of selective logging on forest structure and tree species composition. Three 10 x 100 m transects, made up of 10 x 10 m plots were established in relatively undisturbed forest and in logged forest by two timber extraction railways that operated at different times during the official Indonesian Forestry concession system. Within each plot canopy cover and stem density, DBH, basal diameter, height (of trees only) and species identification of trees ≥6 cm DBH and lianas ≥2 cm DBH were recorded followed by calculations of basal area, species richness, diversity and composition. Nested within these, the same characteristics were recorded for small trees <6 cm DBH and small lianas <2 cm DBH within 2 x 2 m plots. The results of the study found evidence of forest recovery with similar overall canopy cover, density and composition of small trees and species richness and diversity of all trees in logged and relatively undisturbed forest. However, marginal differences in canopy layers, height and basal diameter of small trees and higher density of trees with smaller DBH in logged forest implies that the structural community of trees has not fully recovered. Although it is clear that forest recovery and regeneration has occurred, this study demonstrates that impacts of logging disturbance can still be evident 16-20 years after logging. Furthermore, this study provides justification for the urgent conservation of selectively logged forests, which can maintain important levels of diversity and are susceptible to further disturbance.

Keywords:

Borneo; Selective logging; Extraction railways; Disturbance; Regeneration; Forest recovery

1. Introduction

Peatlands occur throughout the world in areas where topography and rainfall contribute to poor drainage, resulting in conditions that are permanently waterlogged and acidic (Page et al., 1999; Yule, 2010). In the tropics, peatlands have been formed under high-rainfall and high-temperature conditions (Page et al., 2009). Approximately 11% of global peatland area occurs in the tropics covering an estimated area of 441,025 km^2 (Page et al., 2011). Indonesia supports the largest area of tropical peatland in the world, making up more than 80% of that found in the Indo-Malayan region and approximately 47% of tropical peatlands across the globe (Page et al., 2011). Peatlands are defined as areas that are covered by at least 80% peat soil; that is, soil which is composed of at least 65% organic matter with a pH of ≤ 4 which is 40 cm deep or more (FAO, 1988; Whitmore, 1982). One major category of peatland found in Indonesia is ombrogenous peatlands, which gain their water and mineral input solely from aerial deposition and have a rain-fed water table that is level with or higher than the peat surface for most of the year (Page, 1999; Whitmore, 1982). The characteristic convex surface of ombrogenous peatland forms between two river catchments with an increasing depth of peat towards the centre of the dome (Whitmore, 1982). The deepest tropical peatlands occur in the Democratic Republic of Congo with maximum-recorded depths of 30-60 m (Page et al., 2011). In their natural state, lowland tropical peatlands are dominated by trees, forming peat-swamp forest (Wösten et al., 2008).

The original extent of peatlands in South East Asia was estimated to be approximately 247,778 km^2, but they have since experienced a dramatic reduction in cover over recent decades, primarily as a result of logging impacts and clearance for conversion to agriculture (Page et al., 2011; Hirano et al, 2014). Since 1990, 51,000 km^2 of peat-swamp forests in Borneo, Sumatra and Peninsular Malaysia was lost to deforestation while the majority of the remaining two-thirds of peat-swamp forest had been selectively logged (Miettinen et al

2010). Until recently, tropical peat-swamp forests have been understudied due to their inaccessibility and the belief that they are generally lower in biodiversity than other lowland rainforests (Prentice and Parish, 1990; Yule, 2010;). Consequently, large areas of peat-swamp forest have been destroyed and often converted to what is considered more productive use of land (Posa et al., 2011). Tropical peat-swamp forests, which are one of the most threatened forest types in Borneo, covering 60,000 km^2 in Indonesian Borneo alone, have long since been under-appreciated and are poorly understood (Rieley et al., 1997; Hamard et al., 2010; Posa et al., 2011). Only recently has their importance been recognised (Posa et al., 2011).

Dominated by peat-swamp forests, tropical peatlands are important reservoirs for carbon and biodiversity (Wösten et al., 2008; Yule, 2010; Posa et al, 2011; Morrogh-Bernard et al., 2003). The forested tropical peatlands of Southeast Asia store around 42,000 Tg of soil carbon alone (Hooijer et al., 2006). Since they act as major carbon sinks and stores, consequently they play an important role in global climate change processes (Page et al., 2011). Degradation of tropical peatlands, for instance by drainage and fire associated with logging and plantation development, can lead to large volumes of carbon being released and reductions in the size of carbon stores (Page et al., 2002; Jauhiainen et al., 2005, 2008; Hooijer et al., 2006, 2010; Rieley et al., 2008). A study found that forest fires in Indonesia in 1997 released of 810 to 2570 Tg of carbon, equivalent to 13-40% of average annual global carbon emissions from fossil fuels (Page et al., 2002). This release of carbon into the atmosphere can occur through removal of above-ground biomass, peat oxidation and combustion (Page et al., 2002; van der Werf et al., 2004, 2008; Hooijer et al., 2006, 2010).

Undisturbed peat-swamp forests are important contributors to both regional and global biodiversity (Andriesse 1988; Page and Rieley, 1998). Across all peat-swamp forests in Southeast Asia, a total of 1524 plant species have been recorded, of which 172 species are restricted to peat-swamp forests (Posa et al., 2011). In comparison with other peatland types across the globe, peat-swamp forests contain the highest floral diversity (Posa et al., 2011). This includes a high proportion of specialised species that exhibit various adaptations for surviving in a challenging environment, including trees with stilt roots and buttresses and many epiphytes and climbers (Yule, 2010; Posa et al., 2011). Many of these are listed as vulnerable or endangered on the IUCN red list including commercial species of the genus *Shorea* (Meranti) and *Gonystylus bancanus* (Ramin) (IUCN, 2014). High tree species diversity of tropical forests is essential for maintaining overall forest biodiversity by providing a variety of habitats and resources to support a high diversity of fauna (Cannon et al., 1998; Yule, 2010). Of all vertebrate species living in tropical forests, 70% rely on undisturbed or marginally disturbed habitat (Grieser Johns, 1997). Tropical peat-swamp forests provide essential habitat for many threatened and rare species of fish, birds, reptiles and mammals (Ismail, 1999; Posa et al., 2011). Of these faunal groups, fish have the highest endemicity to peat-swamp forest (Posa et al., 2011). Peat-swamp forests are well known for supporting important populations of endangered primates including the largest remaining populations of two species endemic to Borneo: Southern Bornean gibbon (*Hylobates agilis*) (Cheyne et al., 2008) and Bornean orang-utan (*Pongo pygmaeus*) (Morrogh-Bernard et al., 2003). Several endangered felid species also inhabit peat-swamp forests (Cheyne et al., 2009; 2010).

Tropical peat-swamp forest ecosystems are entirely dependent on the peat, which in turn relies on canopy cover, leaf litter input and sufficient water supply (Yule, 2010). The interdependence of the entire ecosystem makes tropical peat-swamp forests particularly susceptible to fire, drainage and logging (Yule, 2010). Moreover, many tropical peatlands are located in densely populated areas with fast growing populations, which puts them at great risk of disturbance and conversion through anthropogenic activities (Rieley et al, 1996; Vijarnsorn, 1996). This emphasises the urgent need to conserve them. The level of protection of peat-swamp forest is generally low. Within Kalimantan, the Indonesian part of Borneo, less than 3% of remaining peat-swamp forest is located within the protected area network (MacDicken, 2001; MacKinnon and MacKinnon, 1991). However this figure is likely to have increased since the establishment of the Sebangau National Park in 2004.

Peat-swamp forests are under considerable anthropogenic pressures mainly from timber extraction, agricultural conversion and fire and drainage associated with these activities. Fire itself has severe impacts on the biology of tropical peat-swamp forests including drying of peat, lowering of the water table, loss of vegetation stability, high tree mortality and loss of habitat for forest fauna (Harrison et al., 2009). In addition to natural occurrence, since the 1970s fire has been used in exploitation practices for commercial logging and as a means of clearing land for plantation agriculture from small-scale farming practices to large-scale schemes including the unsuccessful Mega-Rice-Project and the continuously growing palm oil industry (Segah et al., 2010; Harrison et al, 2009; Boehm and Siegert, 2001; Hooijer et al., 2006). Such conversion of land involves the complete removal of natural peat-swamp forest. According to available data, more than a quarter of both logging and oil palm concessions in Indonesia have been established on tropical peatlands and this proportion is expected to increase as deforestation of Bornean peat-swamp forest continues at a rate of >2% each year (Hooijer et al., 2006; Langner et al., 2007).

Like fire, drainage has harmful effects on peat-swamp forests and is also associated with logging and agriculture. Canals dug by illegal loggers in peat-swamp forests to transport timber have rapidly drained the peat (Harrison et al., 2009). Such occurrence of drainage and consequent drying of peat is the main cause of recent increases in intensity and frequency of fires in tropical peat-swamp forest (Harrison et al., 2009). Furthermore, 4000 km of irrigation channels were constructed as part of the Mega-Rice-Project in Central Kalimantan (Notohadiprawo, 1996). This project drained 1 million ha of land, most of which was peatland. Infrastructure constructed as part of this project along with clearance by legal logging allowed areas of forest that were previously inaccessible to become accessible, leading to further degradation by illegal logging, fire and farming (Boehm and Siegert, 2001). It is clear from this that peat-swamp forests are threatened by fire, drainage, agriculture and logging, which are often interlinked. However, it is the threat of logging which is the focus of this paper.

Like other forest types in Indonesia, peat-swamp forests were exposed to intensive logging through the official Indonesian Forestry concession system (Wösten et al., 2008). From 2000-2010, legal logging concessions were responsible for 14.9% of total forest loss in Kalimantan (Abood et al., 2014). Such systems allowed state-owned companies, domestic private companies, cooperatives and foreign private companies to apply to manage an area of forest and utilise its resources (Blaser et al., 2011). Water-logged conditions in peat-swamp forest prevent the use of heavy machinery and construction of roads and so timber was extracted during logging concessions via lightweight railways which were elevated above the peat on a stable platform of felled trees laid horizontally across the surface (Allinson, 2002). Logging concessions only allowed the felling of mature commercial species including particularly *Gonystylus bancanus*, a species that is endemic to the peat-swamp forests of Southeast Asia, and *Shorea* spp if they met prescribed conditions (Blaser et al., 2011; Husson, 2014). However, logging concessions were frequently irresponsibly managed and only complied with the minimum felling diameter limit, failing to obey the other regulations including residual stand inventory, post-harvest tending and enrichment planting (Siry et al., 2005; ITTO, 2001). Furthermore, logging concession agreements were usually too short to cover a complete harvesting cycle; therefore companies often lacked incentive to invest in the long-term productivity of the forest (Barbier, 1993). Currently in Kalimantan, most remaining forest is located within logging concessions with approximately 57% found within industrial concessions (Abood et al., 2014). However, Indonesian President Joko Widodo has imposed a temporary moratorium as of October 2014 that has forbidden the issue of new logging permits and renewal of existing ones in an attempt to control and minimise the environmental impacts of legal logging concessions (Eshelman, 2014).

Nevertheless many peat-swamp forest ecosystems are still threatened by illegal logging (Boehm and Siegert, 2002). In 1998 Indonesia was hit by a wave of illegal logging following a political and economic crisis as democracy made its first appearance whilst simultaneously the country experienced an enormous decline in the value of its currency (McCarthy, 2002; Husson, 2014). These changes resulted in lack of governance and enforcement ultimately creating ideal conditions for illegal logging (Husson, 2014). Illegal logging occurs in most accessible areas and will target smaller trees (10-20 cm DBH) if larger, commercial trees have already been cut (Boehm and Siegert, 2001). The trees are then transported out of the forest using illegally dug canals, which cause continued drainage of the peat and increase its susceptibility to fire long after loggers have moved out of an area (Harrison et al., 2009).

Logging, whether by legal or illegal methods, has impacts on the physical environment of tropical peat-swamp forests. In some cases although timber harvesting may be selective much of the forest is destroyed during extraction to the extent that levels of incidental damage may be higher than damage caused by actual felling of timber (Grieser Johns, 1997). However modern concessions are unlikely to cause as much incidental damage (Mark Harrison, pers. comm., December 2014). Following timber extraction tropical forests are sometimes left as a mosaic of undisturbed forest, cleared patches, fallen trees that damage lower forest layers and extraction tracks (Whitmore, 1982). Opening of the canopy after tree removal alters the microclimate causing patches to become drier, hotter and more exposed to windy conditions (Grieser Johns, 1997). A drier microclimate along with loss of living biomass disrupts many ecosystem functions, particularly those that relate to hydrology (Siegert et al., 2001). Changes in microclimate and physical damage associated with logging can lead to seedling damage (Grieser Johns, 1997). In tropical forests of Southeast Asia and Western Africa 30-40% of seedlings are killed or damaged from being covered by logging debris (Grieser Johns, 1997). Moreover, leaf litter is likely to dry out with changes in microclimatic conditions, which may bring about changes in decomposition rates (Grieser Johns, 1997). Such changes together with residual organic matter associated with logging increase the susceptibility of logged areas to fire (Brown, 1998). During the El Niño Southern Oscillation event of 1982-83 droughts caused extensive fires in East Kalimantan (Gill and Rasmusson, 1983; Beaman et al., 1985). The area of logged forest destroyed by fire was twice that of primary forest (Grieser Johns, 1997). This has also been observed in more recent events in Kalimantan (Page et al., 2002; Harrison et al., 2009).

Logging also impacts tropical peat-swamp forests by altering forest structure plus tree species diversity and composition. A study in West Kalimantan found that selective, commercial logging reduced density of small and large trees (Cannon et al., 1998). The same study found a 43% reduction in basal area of trees. In general, logging operations in tropical forests of Malaysia and Indonesia tend to experience a 50-60% decrease in basal area in both monocyclic and polycyclic systems (Grieser Johns, 1997). However, studies have reported regenerating selectively logged forest to contain similar tree density, DBH and basal area to primary forest in Ghana (Asase et al., 2014) and higher density and basal area of medium sized trees than primary forest in Malaysia (Supardi, 1999; Okuda et al., 2003). Due to the opening up of niche space re-growth in selectively logged areas can be rapid (Grieser Johns, 1997). In dipterocarp forest in Sumatra, mean DBH and height of trees after one year were 2 cm and 2 m, which after ten years had increased to 35 cm and 22 m, respectively (Geollegue and Hue, 1981). However, in peat-swamp forest natural regeneration following selective or clear-cut logging can be extremely difficult since the opening up of the forest leads to drainage of the peat and increased susceptibility to fire (Page et al., 2009). This leads to further problems as fire not only destroys the above-ground biomass but also damages the underlying peat leading to further disruption of hydrology and loss of propagules for re-establisment of vegetation (Page et al., 2009). Removal of emergent trees through selective logging can reduce canopy height and cover (Yule, 2010). Canopy in selectively logged forest in Malaysia had significantly lower cover than primary forest 39 years after logging (Okuda et al., 2003). In addition, selective logging gives rise to a decrease in stratification of forest layers, allowing more light to reach the forest floor (Yule, 2010). Following a single tree fall, light levels can increase by up to five times (Brown, 1993). One year after selective logging in tropical rainforest in West Kalimantan, 45% of the canopy was open or covered by low pioneer species (Cannon et al., 1998). Such changes in canopy along with gap formation in the soil after logging disturbance encourage the germination and establishment of new recruits by increasing insolation levels and altering competition (Duah-Gyamfi et al., 2014). Increases in seedling recruitment, density and diversity have been found to be higher in logging skid trails than in unlogged areas up to 7 years after logging (Duah-Gyamfi et al., 2014). Conversely, no differences in seedling density were reported in 25 year logged forest and unlogged forest in Uganda (Chapman and Chapman, 1997).

Diversity and species richness of trees has been found to be similar between logged and unlogged sites (Slik et al., 2002; Bischoff et al., 2005; Berry et al., 2008) and higher in logged areas (Asase et al., 2014). Some studies have reported the effect of logging on diversity and tree species richness to be related to scale (He et al., 2002; Cleary et al., 2005). At a large scale, diversity is affected by the volume of timber removed and variation in severity of disturbance following individual tree removal while smaller scale logged areas may contain less heterogeneous forest structure (Cannon et al., 1994; Okuda et al., 2003). In

18-year logged forest in Borneo, logging had no effect on tree species diversity on a small scale but overall logged forest contained a higher number of species than unlogged forest (Berry et al., 2008). Similarly, Cannon et al (1998) found a lower number of species per plot in 8-year logged tropical rainforest in Indonesia compared to unlogged forest but in samples of the same number of trees, logged forest supported a significantly higher number of species per individual than unlogged forest. Sheil et al (1999) suggest that these changes are related to influx of pioneers or invasive species. Conversely, Cannon (et al 1998; 1999) argue that most of the species characteristic of 8-year logged forest were the same as those found in mature, unlogged forest. Nevertheless, species composition is likely to differ in logged areas from primary forest due to a combination of random chance arrival of pioneers and alteration of microclimatic conditions which influences recruitment, germination and establishment (Whitmore, 1984). Logging disturbance also encourages growth of lianas, which respond strongly to increased light levels. For example, liana densities have been found to increase with frequency of tree fall gaps (Malizia et al., 2010). Thus, lianas will often thrive after logging operations and may hinder the recovery of the forest (Whitmore, 1984). A study in tropical forests in the Solomon Islands found more species and individual lianas in disturbed forest compared to primary forest (Whitmore 1974).

2. Hypotheses

This study investigated changes in forest structure and species composition in areas of the Sabangau Forest in Central Kalimantan, Indonesia, that had been under the control of logging concessions on behalf of the Orang-utan Tropical Peatland Conservation Project (OuTrop) in order to contribute to achieving their ecological monitoring objectives. Three main hypotheses were tested regarding the effects of logging, distance from the forest edge and distance from the timber extraction railways on forest structure, tree species composition and regeneration. All hypotheses were based on the assumption that more timber was likely to have been removed through logging activities in forest closest to the edge and to the extraction railways since these areas were more accessible.

Three main hypotheses were tested:

1. Forest structure will differ between logged and relatively undisturbed forest, with increasing distance from the forest edge and with increasing distance from the railways due to the selective removal of tall trees with large DBH through logging activities, which is expected to be higher in more accessible areas.

    a) Tree size (height, DBH, basal diameter), density and basal area of trees ≥6 cm DBH will be lower in more heavily logged forest and in areas closer to the forest edge and extraction railways where large trees have been removed.

    b) Small trees (i.e. seedlings and saplings) <6 cm DBH will have a greater density and basal area and be greater in size (height, DBH, basal diameter) in more heavily logged forest and in areas closer to the forest edge and extraction railways due to opportunities created by logging such as increased space and light (forest regeneration).

c) Lianas will have a greater density and basal area and be greater in size (DBH, basal diameter) in more heavily logged forest and in areas closer to the forest edge and extraction railways due to changes in forest structure (particularly canopy) and increased light levels associated with logging.

d) Assuming that H1a and H1b are supported, low canopy cover will be proportionally more prevalent in more heavily logged forest and in areas closer to the forest edge and extraction railways due to an increased abundance of small trees and removal of large trees. Mid- and upper canopy cover will be proportionally more prevalent in relatively undisturbed forest and in areas further from the forest edge and extraction railways due to a greater density of adult trees and presence of taller trees. Overall canopy cover will not change in response to logging due to the aforementioned changes in the canopy layers.

2. Tree species composition, diversity and richness differ between more heavily logged and relatively undisturbed forest, with increasing distance from the forest edge and with increasing distance from the railways due to the opportunities provided by logging.

a) Species richness and diversity of trees ≥6 cm DBH will be lower in more heavily logged forest and in areas closer to the forest edge and extraction railways due to removal of species through the direct impact of logging and additional incidental damage and composition will differ between logged areas and undisturbed forest.

b) Species richness and diversity of small trees (i.e seedlings and saplings) <6 cm DBH will be higher in more heavily logged forest and in areas closer to the forest edge and extraction railways due to changes in forest structure and canopy cover associated with logging which encourage germination and establishment of light demanding pioneer species and composition will differ between logged areas and undisturbed forest.

c) Species richness and diversity of lianas will be higher in more heavily logged forest and in areas closer to the forest edge and extraction railways due to changes in forest structure (particularly canopy) and increased light levels associated with logging which encourage germination and establishment of species and composition will differ between logged areas and undisturbed forest.

3. Although I expect changes in forest structure and species composition (as stated above), I expect that differences between more heavily logged and relatively undisturbed forest and with increasing distance from the extraction railways will be less pronounced in this study compared with a previous study carried out by Allinson (2002) due to natural forest recovery and regeneration during the intervening 12 years.

3. Materials and methods

3.1 Study site

Research was carried out in the Natural Laboratory for the study of Peat Swamp Forest (hereafter NLPSF) in the Sabangau Forest in Central Kalimantan, Indonesia during July and August, 2014 (Figure 1). This research was conducted as part of the Orang-utan Tropical Peatland Conservation Project (OuTrop) in collaboration with the Centre for International Cooperation in Sustainable Management of Tropical Peatland (CIMTROP), who manage the NLPSF research site. The Sabangau Forest, which is centred on the Sabangau river catchment, is the largest lowland rainforest in Borneo covering 8,750 km^2 of tropical peatland (Ehlers-Smith and Ehlers-Smith, 2013). The NLPSF covers an area of 500 km^2 in the north east of the Sabangau Forest, 20 km south of Palangkaraya, the Provincial capital of Central Kalimantan.

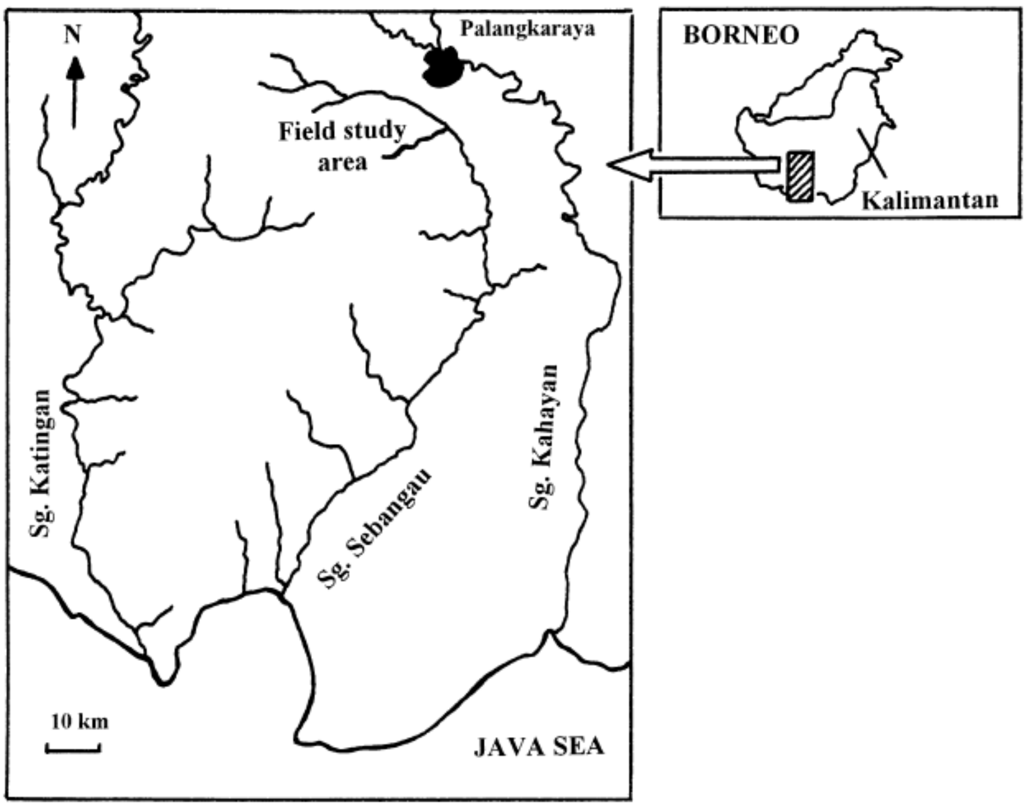


Figure 1. The location of the Sabangau Forest on mainland Borneo, indicating the field study area, the city of Palangkaraya and the larger extent of the forest which lies to the south of the field study area (taken from Morrogh-Bernard et al., 2003)

The Sabangau Forest can be described as a dual ecosystem that is composed of diverse tropical forest and a thick layer of peat up to 15 m deep that has formed over the past 18,300 years under high rainfall and high temperature conditions (Page et al., 1999; 2009). The growth and maintenance of the ecosystem relies on processes which occur within both the forest and the peat and the point of contact at which these interact (Page et al., 1999). The ombrogenous peatland displays the characteristic dome profile, reaching a maximum of 20 m above the water level of the Sabangau River (Page et al., 1999). Tropical peatland forest, which is the dominant habitat type in the Sabangau, can be categorised into several forest classes including riverine forest, mixed peat-swamp forest, low pole forest, tall interior forest and very low open canopy forest (Page et al., 1999). Four of these forest classes can be found within the NLPSF (Page et al., 1999). The plots for this survey were located within mixed peat-swamp forest (Central plot location: 2^0 19.665 S, $113^0$53.989 E; Elevation: 34 m). This habitat stretches 5-6 km into the forest from the research base and is characterised by three canopy layers of forest on top of a 6 m layer of peat (OuTrop, n.d; Morrogh-Bernard et al., 2003; Page et al., 1999). Species that dominate the upper canopy and reach a maximum height of 35 m are *Gonystylus bancanus*, *Shorea spp*., *Cratoxylon glaucum* (Gerrongang) and *Dactylocladus stenostachys* (Mertibu) (Page et al., 1999; Morrogh-Bernard et al., 2003). This transitions into low pole forest that contains only two canopy layers, few commercial species and is dominated by species including *Combretocarpus rotundatus* (Tumih) and *Calophyllum spp* (Page et al., 1999; Morrogh-Bernard et al., 2003). Towards the centre of the dome on peat 10-13 m deep is tall interior forest which contains four canopy layers with emergent trees up to 45 m tall and many commercial genera including *Agathis*, *Dactylocladus* (Mertibu), *Gonystylus*, *Koompassia* (Kempas), *Palaquium* (Nyatoh) and *Shorea* (Page et al., 1999; Morrogh-Bernard et al., 2003; Shepherd et al., 1997). At the far side of the NLPSF, is very low open canopy forest that has a permanently high water table and supports trees around 1.5 m tall including mainly *Calophyllum* spp, *Cratoxylum* spp, *Litsea* spp and *Dactylocladus stenostachys* (Page et al., 1999).

The site is home to many threatened or endangered species including Red langur monkey (*Presbytis rubicunda rubida*) (Ehlers-Smith and Ehlers-Smith, 2013), Sunda clouded leopard (*Neofelis diardi*), Marbled cat (*Pardofelis marmorata*), Leopard cat (*Prionailurus bengalensis*), Flat-headed cat (*Prionailurus planiceps*), Otter civet (*Cynogale bennettii*) (Cheyne et al., 2009; 2010), Sun bear (*Helarctos malayanus*) and the endemic Southern Bornean gibbon (*Hylobates agilis*) (Morrogh-Bernard et al., 2003). The Sabangau Forest is also home to the world's largest population of Bornean orang-utans (*Pongo pygmaeus*) at around 6,900 individuals (Wich et al*.,* 2008).

Prior to the establishment of the NLPSF research base in 1998, the study site had been under the control of legal logging concessions since 1966 (Manduell et al., 2012). This logging continued until around 1998 when logging ceased (Husson, 2014). A lightweight extraction railway was constructed, most likely between 1987 and 1991 (hereafter "old railway" see appendix 1), that ran through mixed peat-swamp forest, low pole and transition forest before terminating 10 km from the research centre (Rieley, 2003; 2014). This railway was dismantled and replaced by the "new railway" (see appendix 2) in 1994 which ran parallel and to the west of the old railway and remained in use until around 1998 when the logging concession ended (Husson, 2014). Following the end of the legal logging activity the area was subject to illegal logging up until 2004 (Husson, 2014). Illegal loggers used canals as the main means of extraction to float timber down the river (Harrison et al, 2009). The new railway remains intact as far as the research base and remains in use as an access route to the research centre for scientists and local staff with only the remaining path extending 12 km into the forest. Due to the determined efforts of the CIMTROP Community Patrol Team, the area of the NLPSF has now been free of logging for 10 years (Mark Harrison pers. comm., December 2014).

3.2 Plot establishment

Plots were located along transects extending perpendicular to the old railway, new railway and along a pre-existing hand-cut study transect in relatively undisturbed, control forest (hereafter referred to as "undisturbed"). A GPS waypoint was marked in each plot in order to construct a map indicating the plot locations (Figure 2). The control transect was located so that the distance from the new railway to the control transect was the same as that from the new railway to the old railway. Five contiguous 10 x 10 m plots were established on each side of and perpendicular to the new railway, old railway and control transect using raffia tape along compass bearings (Figure 3). The first plots were situated 3 m from the edge of the railway or transect in order to mitigate direct edge effects associated with the railways. Subplots measuring 2 x 2 m were established in the NE or NW corner of each 10 x 10 m plot. This arrangement of plots along transects was repeated along each of the railways and the control transect at three locations, approximately 0.7 km apart with increasing distance from the forest edge (Figure 2). This differed from the experimental design of Allinson's (2002) study, in order to include the effect of distance from the forest edge on forest structure and tree species composition. At each of the locations, the plots were located 20 m away from pre-existing study transects established by other researchers (Shown in yellow on Figure 2) to avoid including the transects in the sampling plots.

Figure 2. The study site in the NLPSF indicating the research base and location of plots along the two railways and the control transect, where the tree symbols represent 10 plots (see Figure 3). Plots were located at 0.8 km, 1.5 km and 2.25 km from the north forest edge where the research base is located. Yellow lines represent pre-existing study transects. This map was constructed using Garmin MapSource.

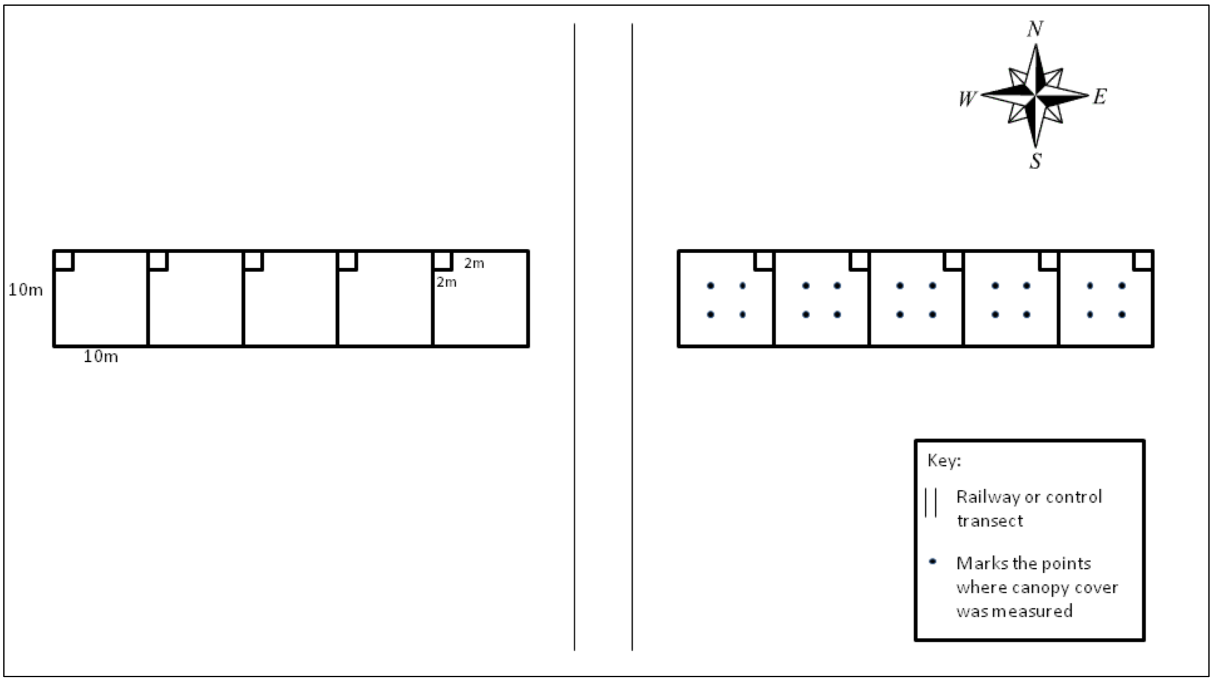


Figure 3. Layout of plots and subplots along transects indicating the size of the large plots where trees (≥6 cm DBH) and lianas (≥2 cm DBH) were sampled and the nested subplots plots where small trees (<6 cm DBH) and small lianas (<2 cm DBH) were sampled. Points where canopy cover was measured are indicated.

3.3 Vegetation sampling

Within each 10 x 10 m plot all trees ≥6 cm diameter at breast height (DBH) and all lianas (woody climbers) ≥2 cm DBH were recorded, measured and identified (See appendix 3 for a complete list of recorded species). These will be hereafter referred to as "trees" and "lianas". Trees and lianas were identified in the field by OuTrop botanical experts, using local names and these names were later translated to the equivalent botanical binomials. Basal circumference (at the base of the liana or tree trunk or directly above the tallest buttress) and circumference at breast height (1.3 m above the basal circumference) of each tree and liana were measured using a tape measure and later converted to basal diameter and DBH, respectively (Figure 4). A consistent height of 1.3 m was chosen as this is the standard measurement for breast height. In addition, tree height was estimated by eye in 5 m classes (0-5 m, 6-10 m, 11-15 m, 16-20 m, 21-25 m and 26-30 m) and converted to the mid-point of the 5 m class for data analysis. Liana height was not recorded due the nature of their growth which made estimation too subjective. Canopy cover was estimated by eye using a densiometer to determine percentage cover at three height ranges (0-10 m, 11-20 m and 21-30 m) and also for overall canopy cover, rather than one measurement over 10 m as done in Allinson's (2002) study in order to capture differences in canopy layers. This was recorded at four points, each 3.3 m apart within the centre of each plot, rather than in the corners of each plot, where canopy cover was recorded in Allinson's (2002) study. This was because samples taken from corners of adjacent plots would overlap.

Within each 2 x 2 m plot all tree seedlings and saplings <6 cm DBH and all liana seedlings and lianas <2 cm DBH were recorded, measured and identified. These will be referred to throughout this study as "small trees" and "small lianas". Basal diameter and DBH, if ≥1.3m in height, were measured for each seedling and sapling using manual callipers. Height of tree seedlings and saplings up to 2 m was measured using a tape measure and those >2 m (i.e. too tall to measure) were estimated by eye to the nearest half metre (Figure 4).

Figure 4. Top left: Measuring the DBH of a tree with tall buttresses. Top right: Measuring height of seedlings. Bottom: The author estimating canopy cover using a homemade densiometer. (Photos: Katrina Schofield/OuTrop)

3.4 Data analysis

Data were analysed using General Linear Mixed Models (GLMMs), implemented with Minitab v17. GLMM was chosen because the residuals of the data were not all normally distributed. Normality was tested using the Anderson-Darling test. Several models were tested which used a nested design and fixed and random effects to find the best model to account for the sampling design. However, the differences in the tested models were marginal. The final model to test the effects of "treatment" (i.e. the difference between relatively undisturbed forest, new railway or old railway) and "edge" (distance from the forest edge) used treatment and edge as fixed effects, "distance" (from the railways or control transect) nested within treatment as a random effect and included an interaction term. The second model, which tested the effect of treatment and distance, used treatment and distance as fixed effects, edge nested within treatment as a random effect and included an interaction term. An alternative model was tried that included treatment, distance and edge as fixed effects and labelled each 10 x 100 m transect (as illustrated in Figure 3) as a random effect. However, this model was rejected by Minitab v17. Where the results of the GLMMs were found to vary significantly, post-hoc comparison tests were carried out to establish where the differences were. Tukey tests were used but if these did not produce a significant result Fisher's tests, which were more sensitive to differences in the data, were applied. Bar charts of the means (± SE) were constructed on Microsoft Excel to display these results.

Diversity values, expressed as Shannon's diversity index, were calculated using a function add-in to Microsoft Excel. The Shannon index was chosen due to its sensitivity to the presence of rare species (Nagendra, 2002), which is important in forest that has been selectively logged where rare species are particularly prone to reduction or elimination even if these are not commercial species (Grieser Johns, 1997; Slik et al., 2002). Unknown species were included for richness and diversity calculations since the number of unknown species per plot was known (i.e. species were distinguished in the field as “unknown sp. 1”, “unknown sp. 2” etc.). Nonmetric Multidimensional Scaling (NMS) ordinations were carried out on PCORD v5.10 (McCune and Mefford, 2006) to determine similarity of species composition among plots. Plots that contained no individuals (as was sometimes the case for lianas) were omitted from the NMS analysis. Species area values and standard deviations were calculated using PCORD 5.10 in order to construct species area curves on Microsoft Excel. Unknown species were omitted from this analysis since the total number of unknown species across all plots was unknown.

4. Results

4.1 Forest structure

Increasing distance from the forest edge had significant effects on canopy cover at low, mid- and upper canopy heights (Table 1. See appendices 4-6 for mean and SD values for all variables throughout results). Mean canopy cover at 0-10 m height was significantly lower in plots furthest from the forest edge (2.25 km) across all treatments (Figure 5a) but showed no significant effect of distance from the railways or between relatively undisturbed forest and the railways (Tables 1-2). Mean canopy cover at 11-20 m height was significantly higher in plots at 1.5 and 2.25 km from the forest edge across all treatments but showed no significant effect of distance from the railways (Figure 5b; Table 2). Old railway plots contained significantly higher cover of canopy at 11-20 m than new railway plots but both were similar to undisturbed forest (Figure 5c). However this result should be treated with caution since it was only significant in one model (Tables 1-2). Mean canopy cover at 21-30 m height was significantly greater in new railway plots located at 2.25 km than old railway and undisturbed plots at the same location, while plots located at 0.8 and 1.5 km were similar across all treatments (Figure 6). Patterns of mean canopy cover at 21-30 m height with increasing distance from the railways were consistent with distance from the control transect.

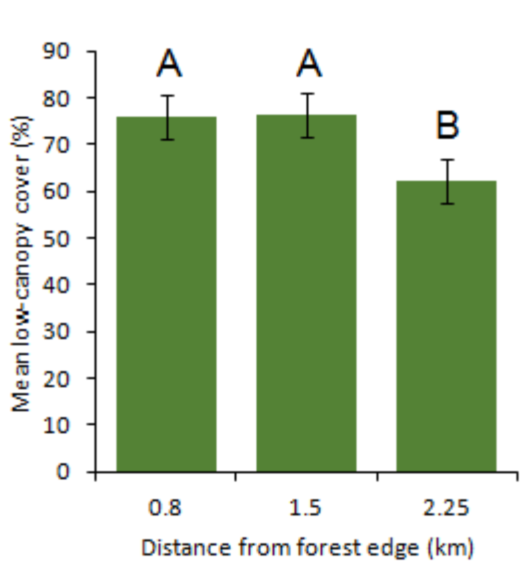
However, cover was only significantly different between 30-40 m and 0-10 m. Overall, there were no significant differences in canopy cover at 21-30 m height between undisturbed forest and the railways (Tables 1-2). Overall canopy cover exhibited no significant differences in response to logging treatments (Tables 1-2).

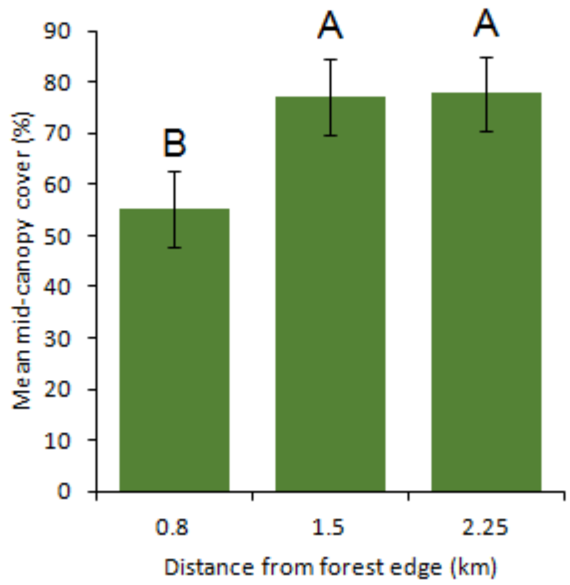
Table 1. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on canopy cover. Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Low canopy (0-10 m)	2	1.10	0.365	2	8.97	**<0.001**	4	1.47	0.219
Mid-canopy (11-20 m)	2	12.58	**0.001**	2	10.03	**<0.001**	4	0.88	0.480
Upper canopy (21-30 m)	2	0.20	0.823	2	5.58	**0.006**	4	7.64	**<0.001**
Overall canopy	2	3.15	0.079	2	0.44	0.648	4	0.69	0.600

Table 2. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on canopy cover. Significant values are in bold.

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Low canopy (0-10 m)	2	0.30	0.754	4	2.34	0.063	8	0.44	0.895
Mid-canopy (11-20 m)	2	1.52	0.293	4	0.72	0.584	8	0.35	0.941
Upper canopy (21-30 m)	2	0.03	0.970	4	2.73	**0.036**	8	0.23	0.984
Overall canopy	2	4.76	0.058	4	0.91	0.463	8	0.92	0.505





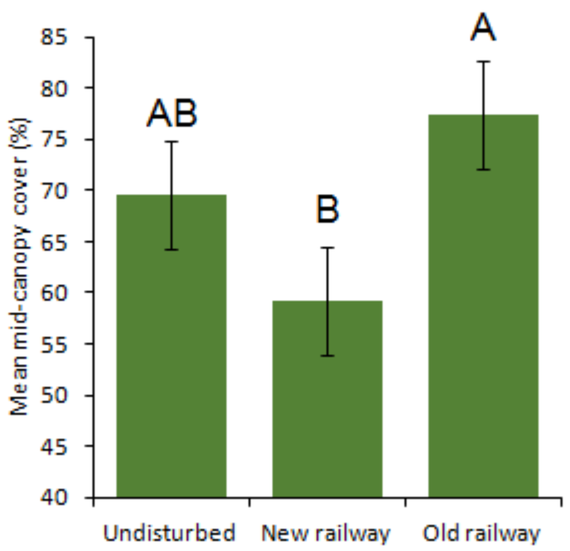


Figure 5. Mean (± SE) canopy cover (%) for: a) low canopy (0-10 m) in relation to increasing distance from forest edge within each treatment; b) mid-canopy (11-20 m) in relation to increasing distance from forest edge within each treatment and; c) mid-canopy in relation to treatment. Tukey test results are indicated where means that do not share a letter are significantly different.

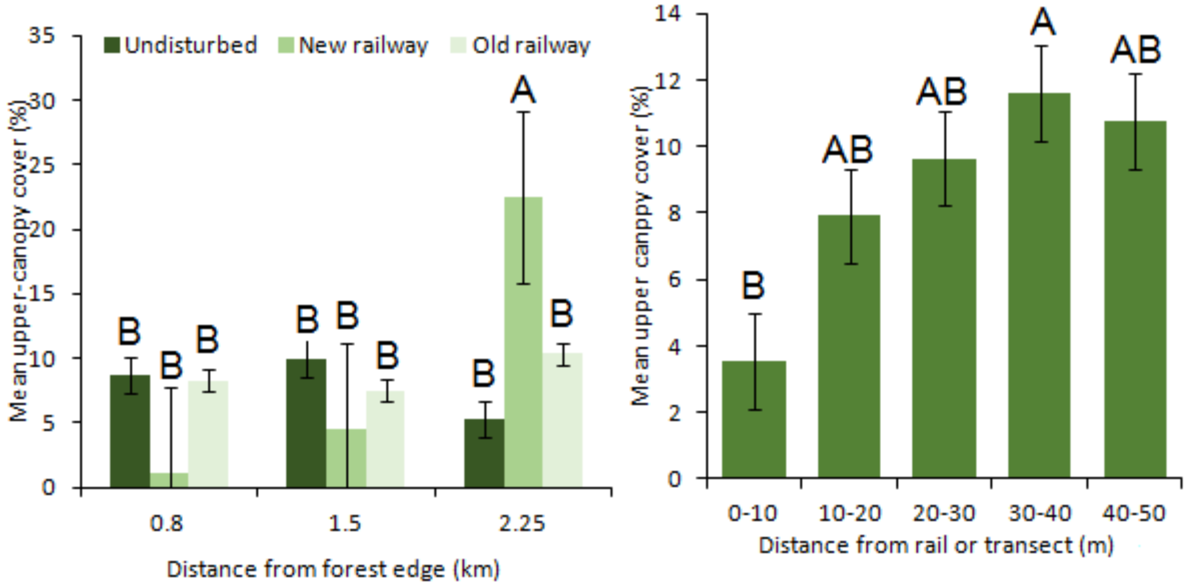


Figure 6. Mean (± SE) upper canopy cover (21-30 m) in relation to increasing distance from the forest edge within each treatment and with increasing distance from the railway or control transect. Tukey test results are indicated where means that do not share a letter are significantly different.

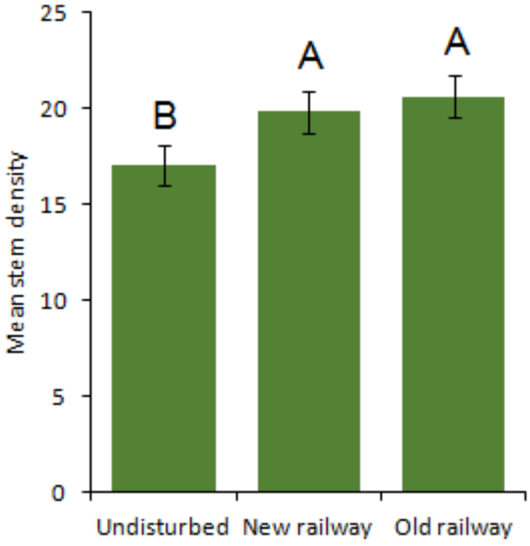
Mean tree stem density was significantly higher along the two railways than in undisturbed forest (Tables 3-4; Figure 7). There was no significant effect of increasing distance from the forest edge or railways on stem density (Tables 3-4). Mean DBH and basal diameter of trees were significantly higher in undisturbed forest (Tables 3-4; Figures 8-9). There was no significant effect of increasing distance from the forest edge or railways on DBH or basal diameter (Tables 3-4). Mean height increased significantly with increasing distance from the forest edge across all treatments (Table 3; Figure 9). Height also differed with distance from the railways, however only tree height at 0-10 and 10-20 m were significantly different (Table 4; Figure 9). There were no significant differences in mean tree height between undisturbed forest and the railways (Tables 3-4). Basal area of trees did not differ significantly in response to logging treatment (Tables 3-4).

Table 3. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on stem density, diameter at breast height (DBH), basal diameter, basal area and height of trees (≥6 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	3.94	**0.048**	2	0.22	0.803	4	1.06	0.381
DBH (cm)	2	12.34	**0.001**	2	1.11	0.329	4	1.56	0.183
Basal diameter (cm)	2	12.37	**0.001**	2	1.48	0.229	4	0.69	0.600
Basal area (cm²)	2	1.51	0.260	2	0.89	0.416	4	2.42	0.057
Height (m)	2	0.86	0.449	2	42.71	**<0.001**	4	2.10	0.079

Table 4. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on stem density, diameter at breast height (DBH), basal diameter, basal area and height of trees (≥6 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	7.76	**0.022**	4	1.53	0.230	8	1.55	0.157
DBH (cm)	2	9.39	**0.014**	4	0.63	0.640	8	1.33	0.224
Basal diameter (cm)	2	19.14	**0.002**	4	1.31	0.263	8	1.54	0.140
Basal area (cm²)	2	0.67	0.547	4	0.56	0.695	8	0.99	0.453
Height (m)	2	0.11	0.900	4	2.73	**0.028**	8	1.66	0.102



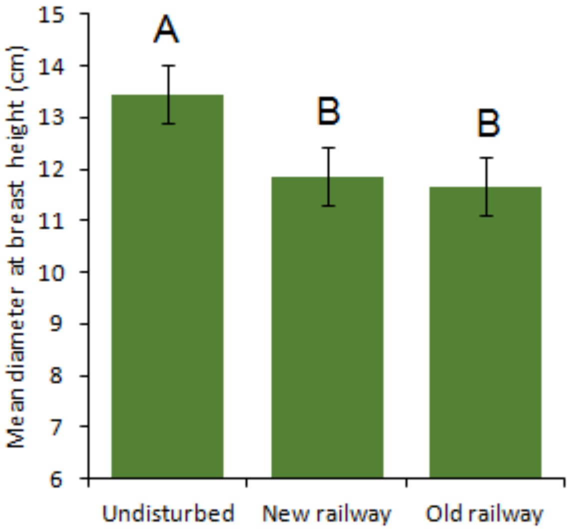


Figure 7. Mean (± SE) stem density of trees (≥6 cm DBH) per 100 m^2 across different treatments. Tukey test results are indicated where means that do not share a letter are significantly different.

Figure 8. Mean (± SE) DBH (cm) of trees (≥6 cm DBH) across different treatments. Tukey test results are indicated where means that do not share a letter are significantly different.

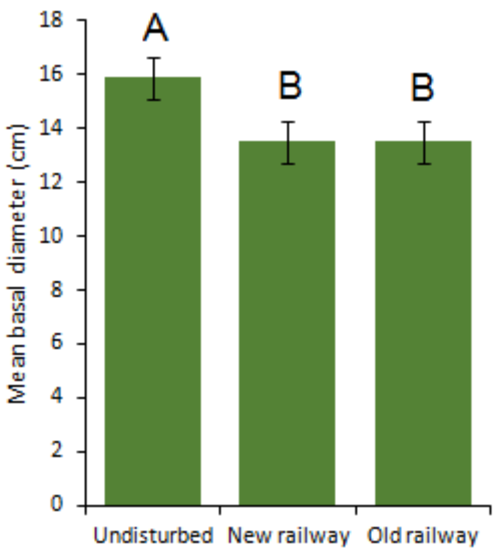


Figure 9. Mean (± SE) basal diameter (cm) of trees (≥6 cm DBH) across different treatments. Tukey test results are indicated where means that do not share a letter are significantly different.

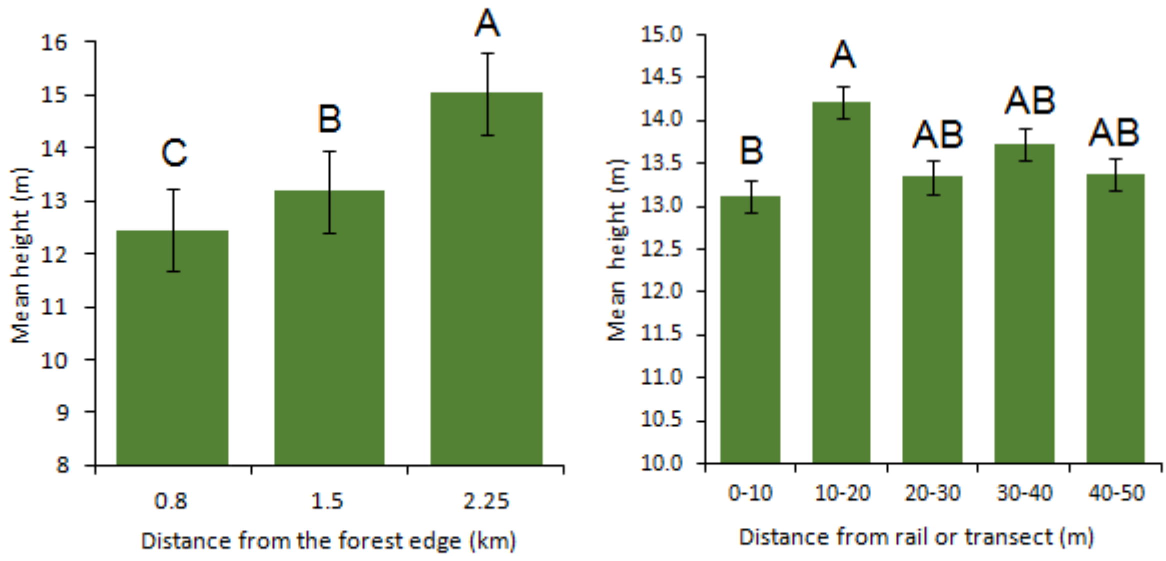


Figure 10. Mean (± SE) height (m) of trees (≥6 cm DBH) in relation to increasing distance from the railways or control transect and increasing distance from the forest edge within each treatment. Tukey test results are indicated where means that do not share a letter are significantly different.

Mean stem density of small trees i.e. seedlings and saplings, differed significantly with increasing distance from the railways or control transect, however differences were marginal and there were no apparent trends among plots that were significantly different (Table 6, Figure 11). Furthermore, there were no significant differences in stem density of small trees between treatments or with increasing distance from the forest edge (Tables 5-6). Mean basal diameter and height of small trees differed significantly with increasing distance from the forest edge (Table 5) and with increasing distance from the railways or control transect (Table 6). Mean basal diameter in new railway plots was significantly greater than old railway and undisturbed plots at 0.8 km and significantly greater than undisturbed plots at 1.5 km, while plots at 2.25 km were similar across treatments (Figure 12). Mean height of small trees was significantly highest at 0.8 km and 1.5 km from the forest edge within new railway plots and at 2.25 km within undisturbed forest and along the old railway (Figure 13). Both basal diameter and height of small trees fluctuated with increasing distance from the railways or control transect but, although significant, showed no obvious trends within or across treatments (Figures 12-13). Basal diameter and height of small trees did not differ significantly between treatments overall (Tables 5-6). DBH of small trees was discarded from the analysis due to insufficient sample size, i.e. not enough small trees ≥1.3 m tall.

Table 5. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on stem density, basal diameter and height of small trees (<6 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	1.45	0.272	2	1.80	0.172	4	1.34	0.264
Basal diameter (cm)	2	0.52	0.607	2	1.39	0.250	4	2.83	**0.023**
Height (m)	2	0.19	0.829	2	1.06	0.346	4	3.89	**0.004**

Table 6. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on stem density, basal diameter and height of small trees (<6 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	2.20	0.192	4	1.12	0.354	8	2.83	**0.009**
Basal diameter (cm)	2	0.83	0.479	4	1.60	0.172	8	4.82	**<0.001**
Height (m)	2	0.19	0.835	4	1.74	0.139	8	3.41	**0.001**

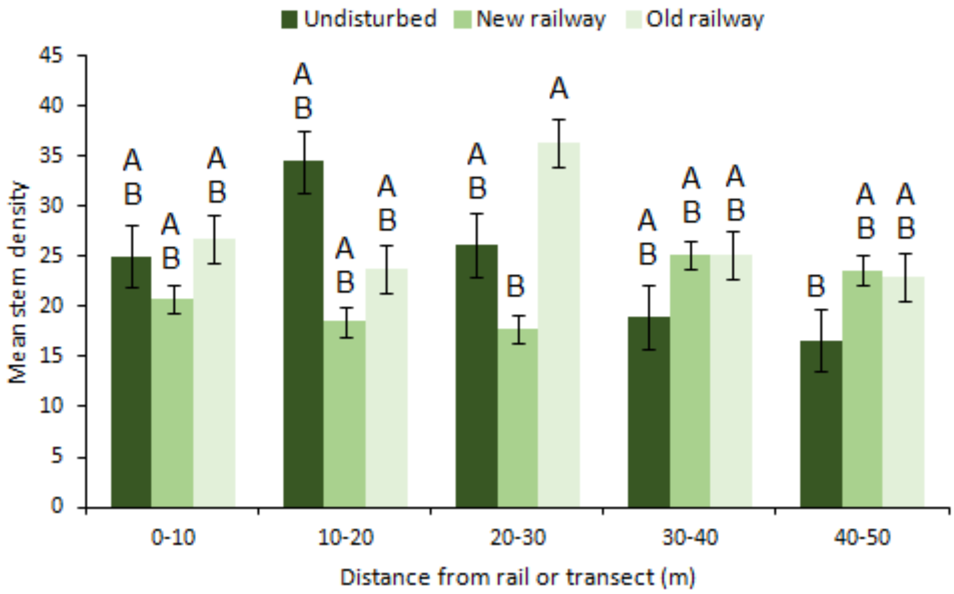


Figure 11. Mean (± SE) stem density of small trees (<6 cm DBH) per 4 m^2 in relation to increasing distance from the rail or control transect within each treatment. Tukey test results are indicated where means that do not share a letter are significantly different.

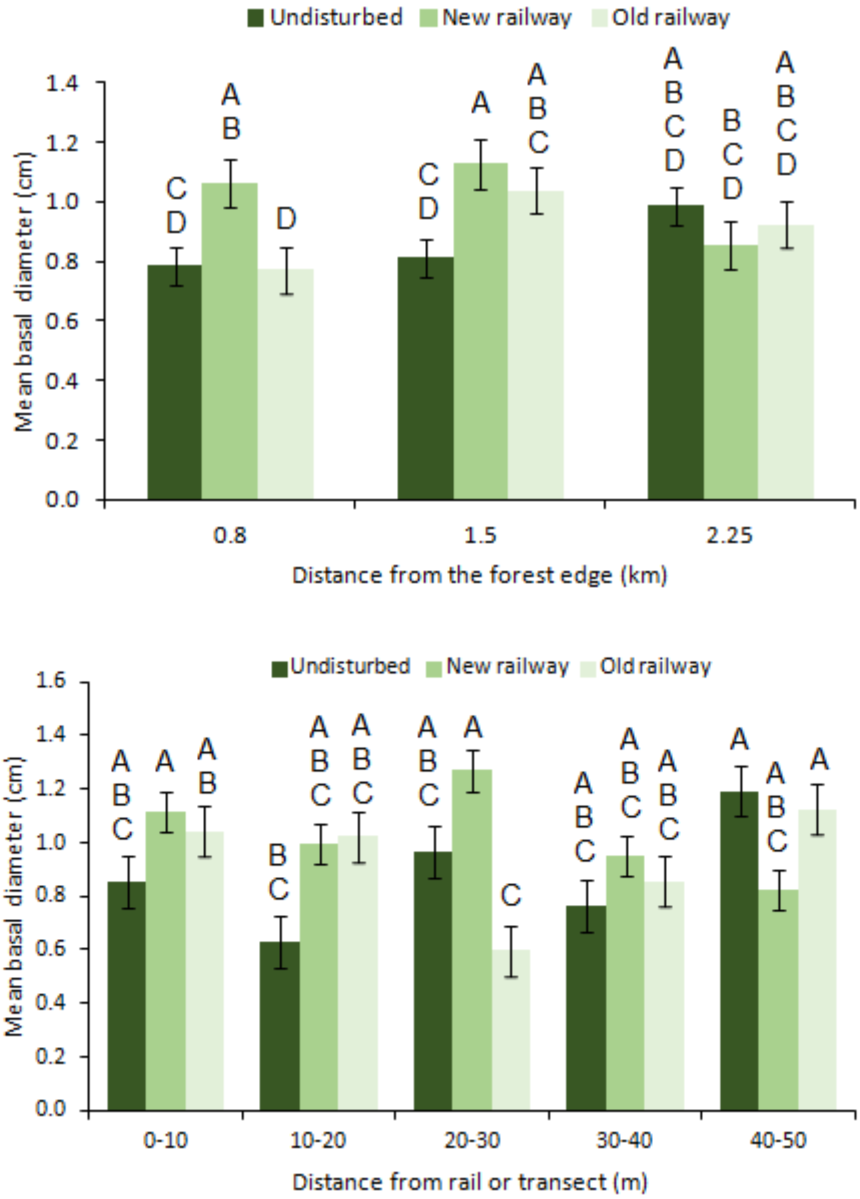


Figure 12. Mean (± SE) basal diameter (cm) of small trees (<6 cm DBH) in relation to increasing distance from the rail or control transect and increasing distance from the forest edge within each treatment. Fisher (top) and Tukey (bottom) test results are indicated where means that do not share a letter are significantly different.

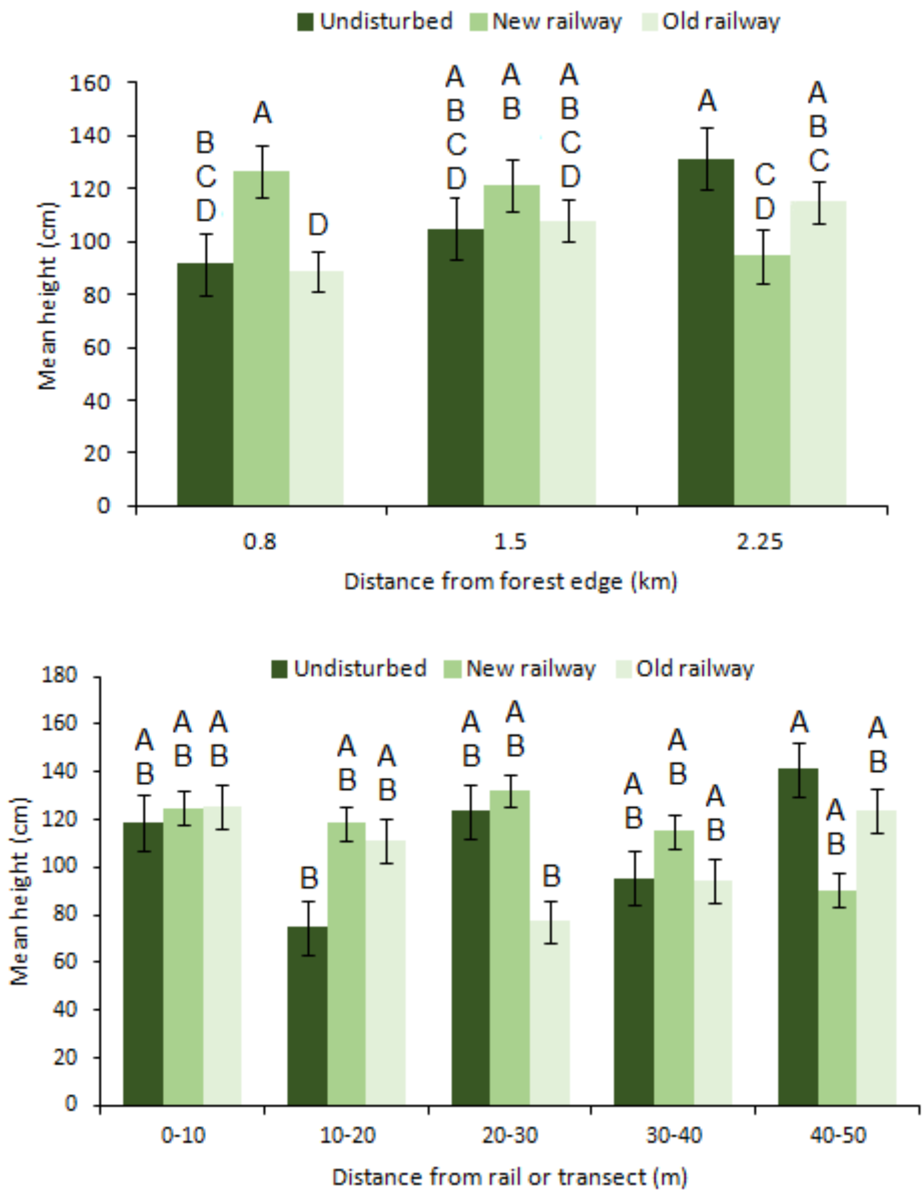


Figure 13. Mean (± SE) height (cm) of small trees (<6 cm DBH) in relation to increasing distance from the rail or control transect and increasing distance from the forest edge within each treatment. Fisher (top) and Tukey (bottom) test results are indicated where means that do not share a letter are significantly different.

Mean stem density of lianas differed significantly with increasing distance from the forest edge within treatments (Table 7). New railway plots contained a significantly higher density of lianas at 1.5 km whereas old railway and undisturbed forest plots were similar across all distances (Figure 14). However, there was no significant effect of treatment or distance from the railways on stem density of lianas (Tables 7-8). Mean basal area of lianas was significantly higher in plots located at 2.25 km along the old railway and at 1.5 and 2.25 km in undisturbed forest, while new railway plots contained similar basal area of lianas across all distances (Figure 15). Mean basal area of lianas did not differ significantly in relation to distance from the railways or between treatments (Figure 14). DBH of lianas was significantly greater at 2.25 km than 0.8 km from the forest edge across all treatments and also differed significantly with distance from the railways (Tables 7-8; Figure 16). However the differences were marginal and there were no clear trends among plots that were significantly different (Figure 16). Basal diameter of lianas exhibited no significant changes in response to logging (Tables 7- 8).

Table 7. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on stem density, diameter at breast height (DBH), basal diameter and basal area of lianas (≥2 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	2.84	0.098	2	1.95	0.149	4	5.55	**0.001**
DBH (cm)	2	0.29	0.754	2	3.28	**0.039**	4	1.09	0.363
Basal diameter (cm)	2	0.43	0.658	2	2.05	0.130	4	0.92	0.454
Basal area cm²)	2	2.65	0.111	2	2.26	0.112	4	3.09	**0.021**

Table 8. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30m, 30-40 m, 40-50 m) and the interaction between treatment and distance on stem density, diameter at breast height (DBH), basal diameter and basal area of lianas (≥2 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	0.55	0.602	4	0.76	0.555	8	0.89	0.531
DBH (cm)	2	0.36	0.712	4	2.44	**0.046**	8	2.24	**0.024**
Basal diameter (cm)	2	0.65	0.556	4	1.96	0.100	8	1.76	0.083
Basal area cm²)	2	0.81	0.487	4	2.06	0.095	8	0.26	0.975

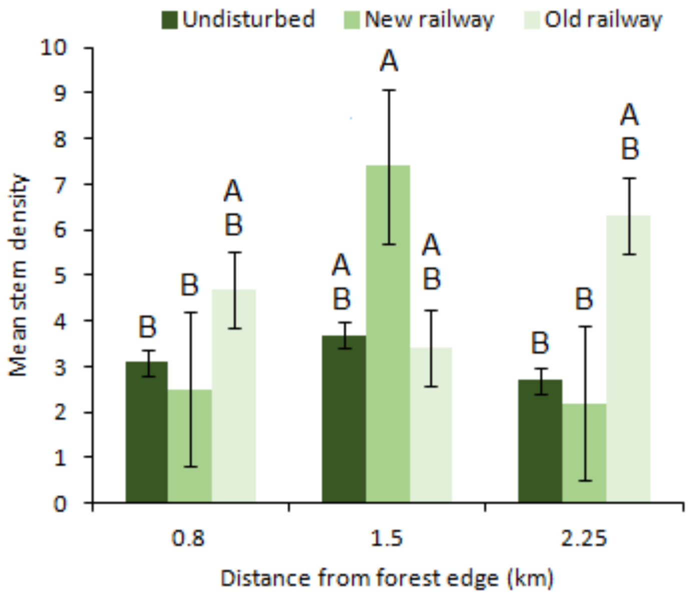


Figure 14. Mean (± SE) stem density of lianas (≥2 cm DBH) per 100 m^2 in relation to increasing distance from forest edge. Tukey test results are indicated where means that do not share a letter are significantly different.

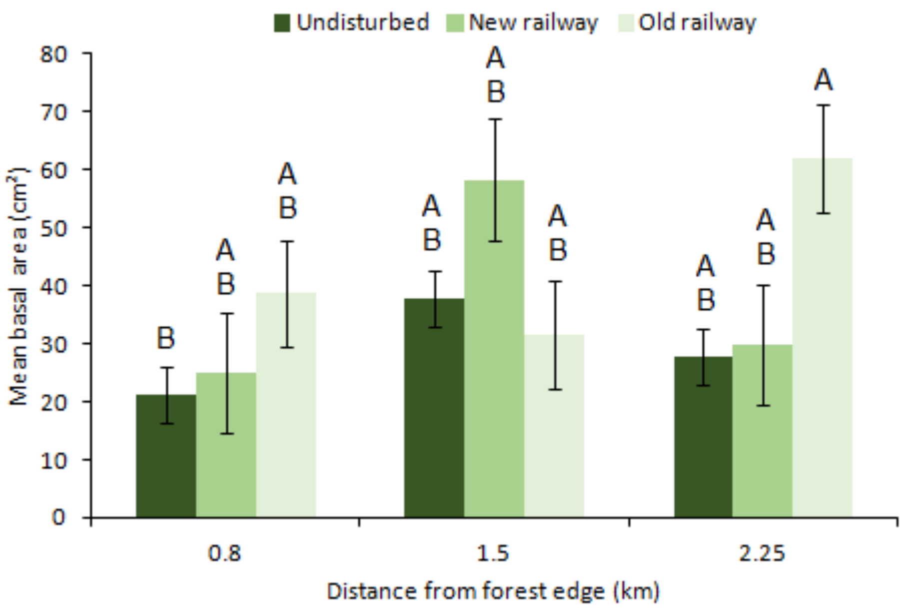


Figure 15. Mean (± SE) basal area (cm²) of lianas (≥2 cm DBH) per 100 m^2 in relation to increasing distance from the forest edge within treatments. Tukey test results are indicated where means that do not share a letter are significantly different.

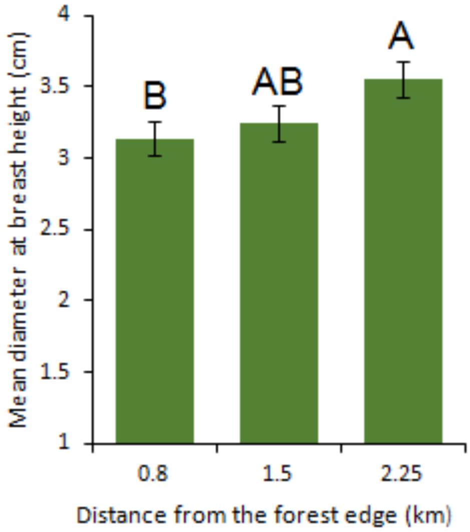
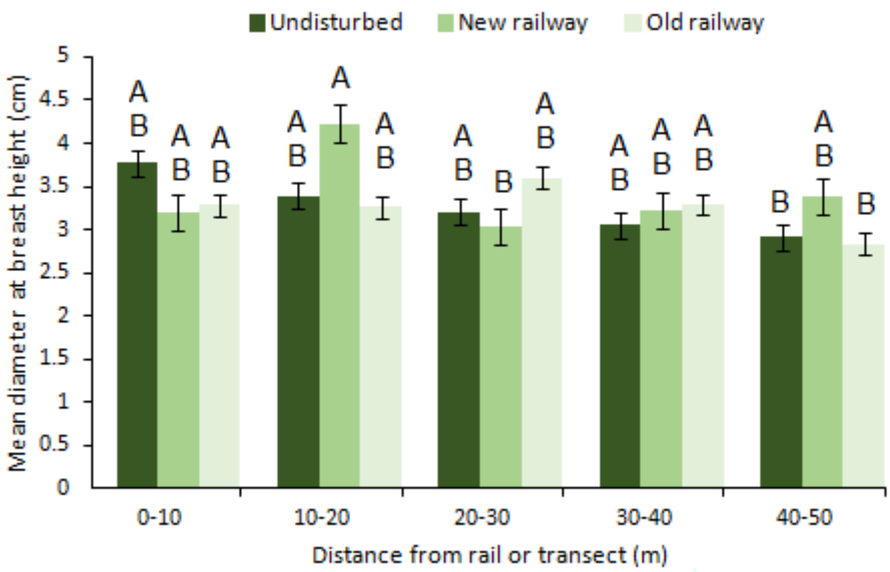


Figure 16. Mean (± SE) DBH of lianas (≥2 cm DBH) in relation to: a) increasing distance from the forest edge and; b) with distance from the railways or control transect within treatments. Tukey test results are indicated where means that do not share a letter are significantly different.



Mean stem density of small lianas did not differ significantly in response to logging (Tables 9-10). Mean basal diameter of small lianas was significantly higher in plots located along the new railway than those along the old railway or in undisturbed forest (Tables 9-10; Figure 17). However, basal diameter of small lianas did not differ significantly with increasing distance from the forest edge or railways (Tables 9-10). DBH of small lianas was discarded from the analysis due to insufficient sample size, i.e. not enough small trees ≥1.3 m tall.

Table 9. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on stem density, basal diameter and height of small lianas (<2 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	2.43	0.130	2	0.16	0.856	4	0.85	0.497
Basal diameter (cm)	2	4.58	**0.031**	2	0.34	0.715	4	0.69	0.602

Table 10. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on stem density, basal diameter and height of small lianas (<2 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Stem density	2	4.08	0.076	4	1.28	0.285	8	0.92	0.507
Basal diameter (cm)	2	6.91	**0.026**	4	1.20	0.312	8	0.76	0.603

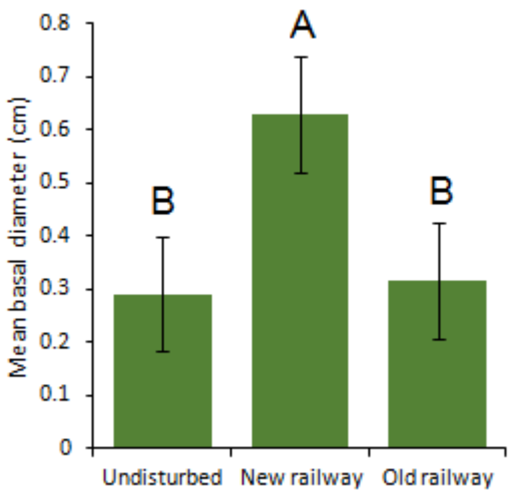


Figure 17. Mean (± SE) basal diameter (cm) of small lianas (<2 cm DBH) across different treatments. Tukey test results are indicated where means that do not share a letter are significantly different.

4.2 Species composition, richness and diversity

Tree species richness did not change significantly in response to logging (Tables 11-12). The species-area curve starts to reach a plateau towards 30 plots indicating that this study is likely to have captured most of the tree species and also confirms no significant differences in species richness between treatments (Figure 18). Tree diversity differed significantly with distance from the forest edge (Table 11) with highest diversity occurring closest to the forest edge at 0.8 km and significantly lower diversity at 2.25 km across all treatments (Figure 19). There were no significant differences in tree species diversity between undisturbed forest, new railway and old railway plots or with increasing distance from the railways (Tables 11-12). NMS ordination results indicate that most undisturbed plots were similar in species composition (Figure 20). Species composition of new railway and old railway plots showed greater variation than undisturbed plots and were not distinctly different from each other. Species composition in most plots located at 0.8 km from the forest edge was distinctly different from those at 2.25 km but both were similar to those located at 1.5 km (Figure 21). There were no significant differences in species composition at different distances from the railways or control transect (Figure 22).

Table 11. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on species richness and Shannon's diversity index of trees (≥6 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	1.85	0.199	2	1.93	0.152	4	1.43	0.233
Shannon's diversity index	2	0.72	0.509	2	3.57	**0.034**	4	1.84	0.130

Table 12. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on species richness and Shannon's diversity index of trees (≥6 cm DBH).

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	1.12	0.386	4	1.16	0.337	8	0.87	0.544
Shannon's diversity index	2	0.19	0.830	4	0.53	0.715	8	0.71	0.683

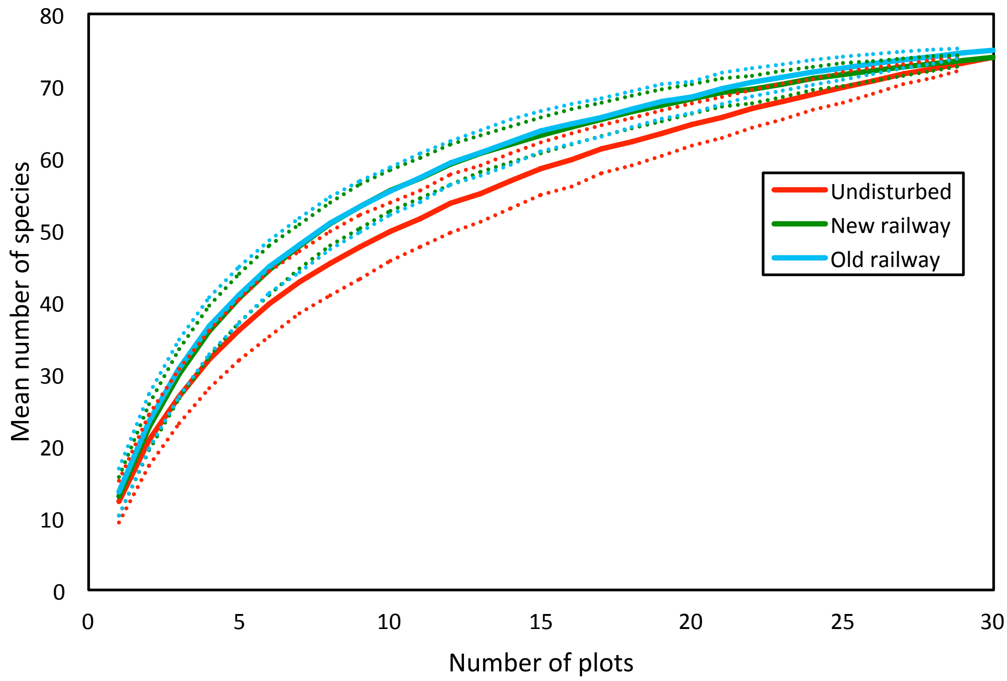


Figure 18. Species-area relationship for mean number of tree species (± 95% confidence intervals) present within each treatment.

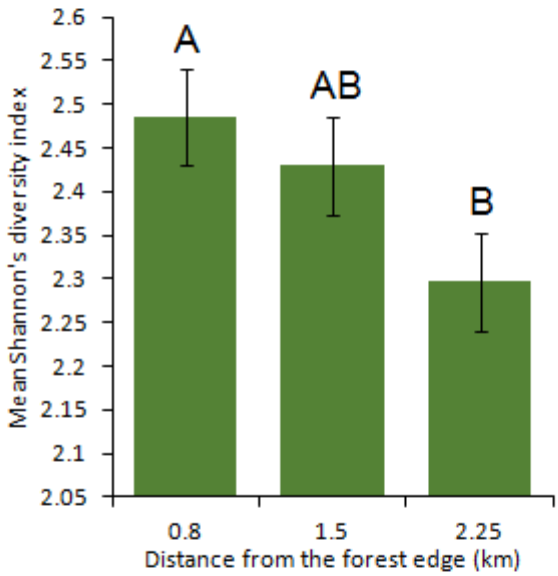


Figure 19. Mean (± SE) Shannon's diversity of trees (≥6 cm DBH) per 100 m^2 in relation to increasing distance from the forest edge. Tukey test results are indicated where means that do not share a letter are significantly different.

Figure 20. NMS ordination result indicating species composition of trees (≥6 cm DBH) within plots across different treatments, where each symbol represents a plot. Plots closer together are more similar.

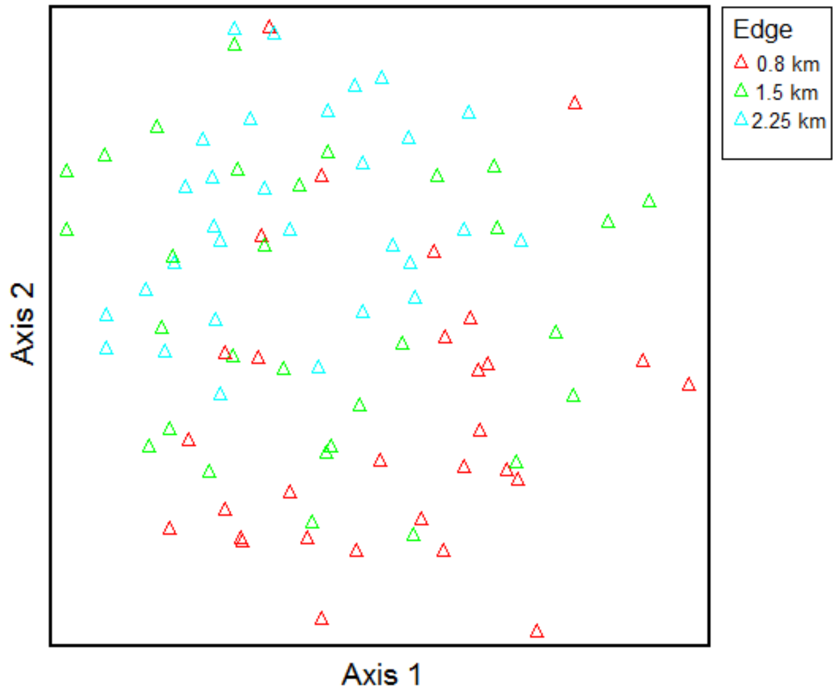


Figure 21. NMS ordination result indicating species composition of trees (≥6 cm DBH) within plots in relation to distance from the forest edge, where each symbol represents a plot. Plots closer together are more similar.

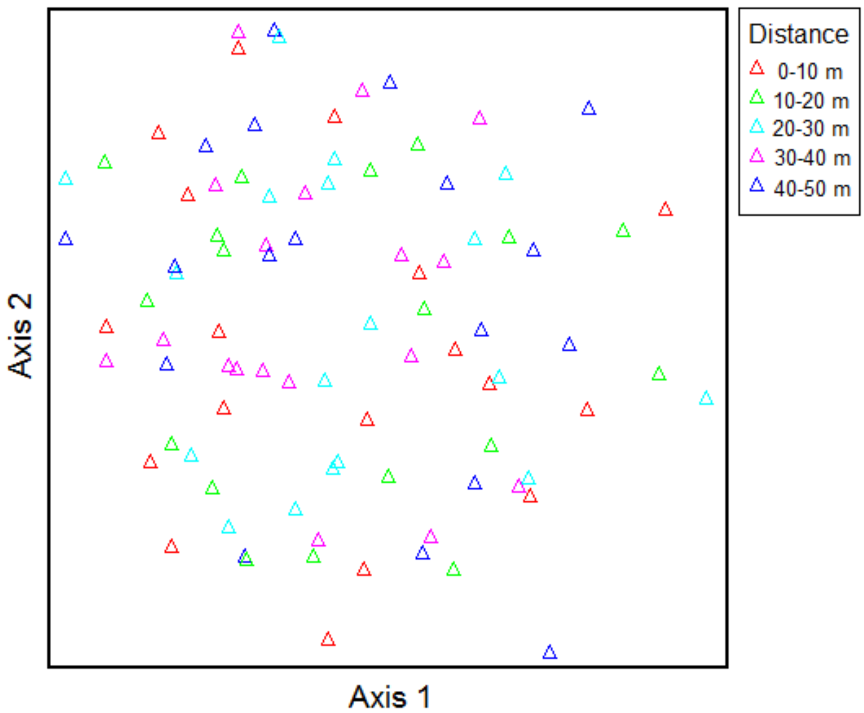


Figure 22. NMS ordination result indicating species composition of trees (≥6 cm DBH) within plots in relation to distance from the railway or control transect, where each symbol represents a plot. Plots closer together are more similar.

There were no significant changes in species richness or diversity of small trees in response to logging (Tables 13-14). The species-area curve almost reaches a plateau indicating that although this study has captured a lot of the small tree species, it would have been likely to capture the majority of species with an additional c.5 plots (Figure 23). It also confirms no significant differences in species richness between treatments (Figure 23). NMS ordination results showed no differences in species composition of small trees between treatments, with increasing distance from the forest edge or increasing distance from the railways or control transect (Figures 24-26).

Table 13. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway,), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on species richness and Shannon's diversity index of small trees (<6 cm DBH).

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	2.90	0.094	2	1.95	0.151	4	0.56	0.694
Shannon's diversity index	2	2.10	0.165	2	0.96	0.387	4	0.34	0.853

Table 14. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on species richness and Shannon's diversity index of small trees (<6 cm DBH).

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	3.52	0.097	4	0.82	0.516	8	1.45	0.192
Shannon's diversity index	2	3.82	0.085	4	0.80	0.529	8	1.08	0.385

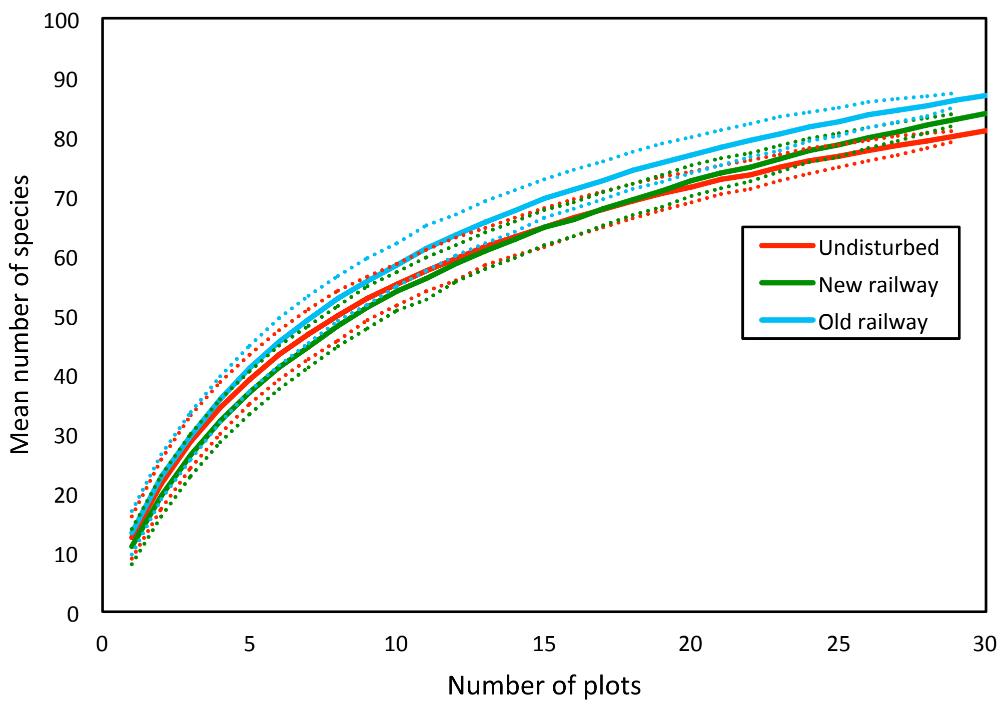


Figure 23. Species-area relationship for mean number of small tree species (± 95% confidence intervals) present across all plots.

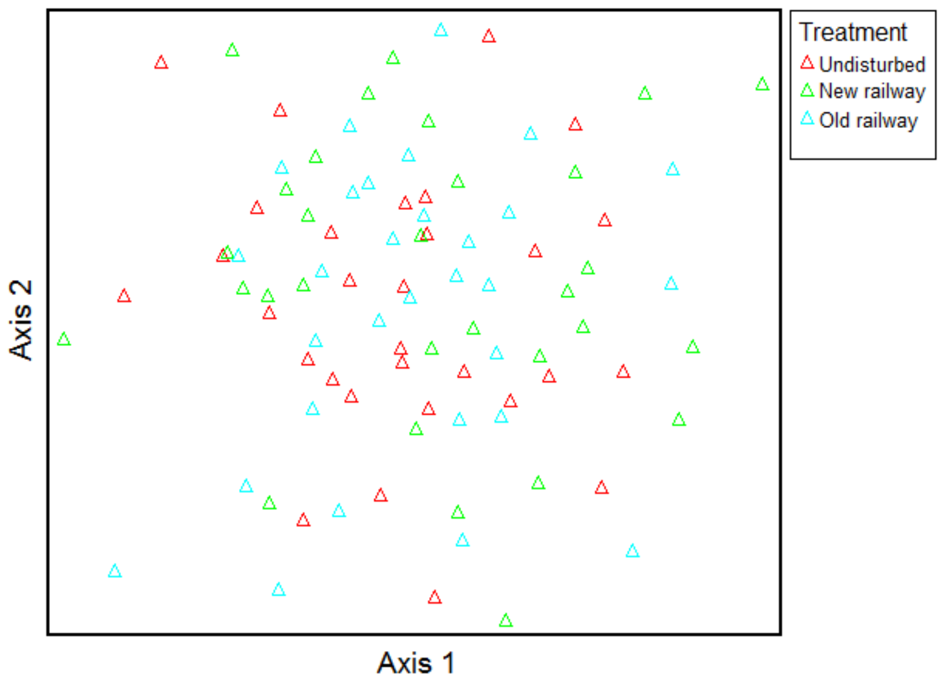


Figure 24. NMS ordination result indicating species composition of small trees (<6 cm DBH) within plots across different treatments, where each symbol represents a plot. Plots closer together are more similar.

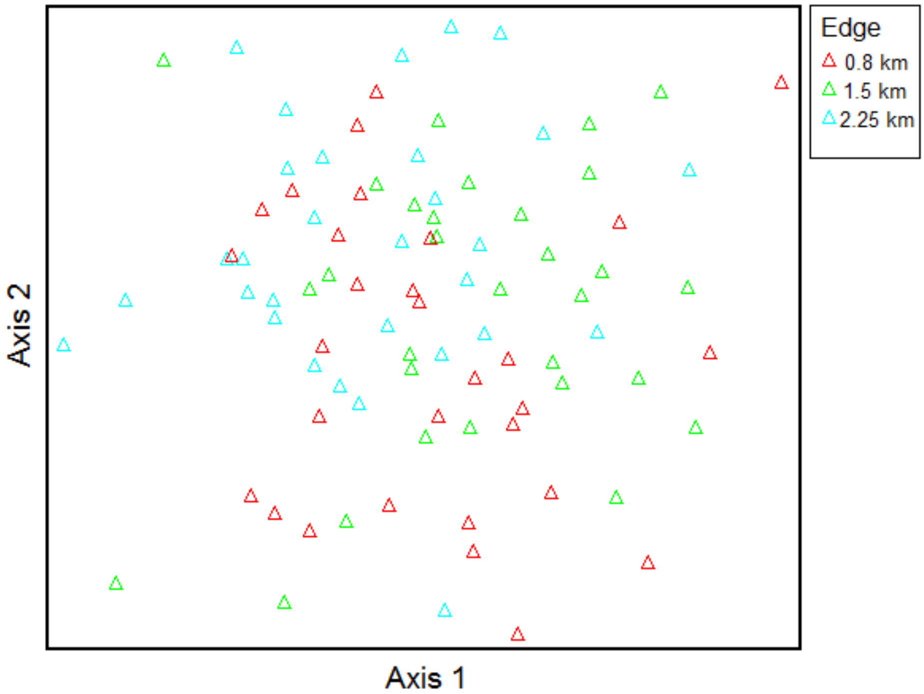


Figure 25. NMS ordination result indicating species composition of small trees (<6 cm DBH) within plots in relation to distance from the forest edge, where each symbol represents a plot. Plots closer together are more similar.

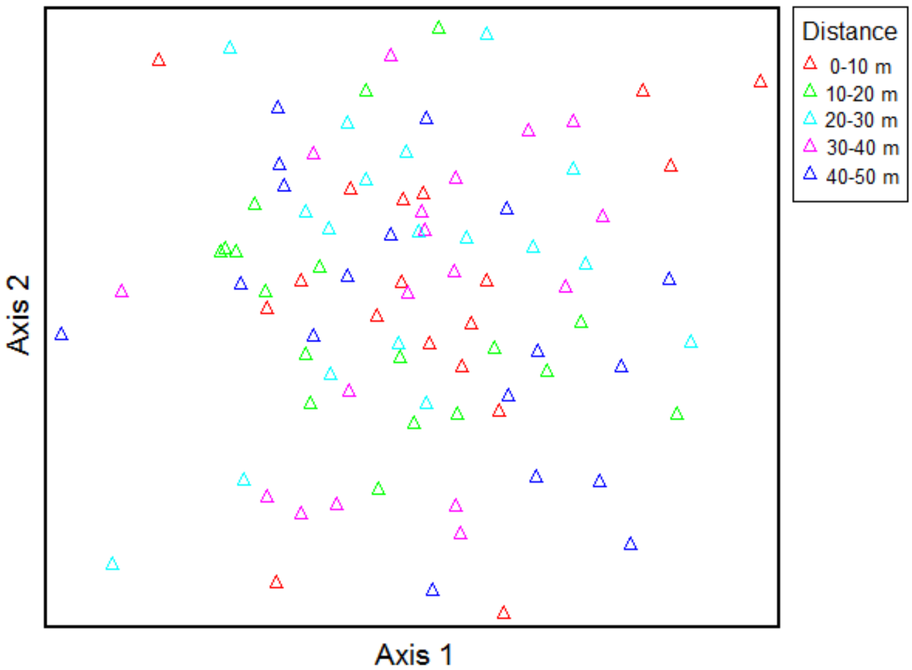


Figure 26. NMS ordination result indicating species composition of small trees (<6 cm DBH) within plots in relation to distance from the railway or forest edge, where each symbol represents a plot. Plots closer together are more similar.

The species-area relationship almost reaches a plateau indicating that most of the liana species are likely to have been captured in this study and suggests no significant differences in species richness when unknown species are excluded (Figure 27). Conversely, liana species richness was found to be significantly higher in old railway plots than in undisturbed forest, while diversity was higher along both of the railways when unknown species were included (Figures 28-29). However, this result was only found to be significant in one model (Table 15) therefore should be interpreted with caution. Increasing distance from the forest edge had significant effects on species richness and diversity of lianas, within treatments (Table 15). Liana species richness differed significantly between plots located at 0.8 and 1.5 km within new railway plots and between 1.5 and 2.25 km within old railway plots, while richness of lianas did not differ significantly with distance from the forest edge within undisturbed forest (Figure 28). Diversity of lianas showed similar patterns with distance from the forest edge as species richness but exhibited fewer significant differences (Figure 29). There were no significant effects of increasing distance from the railways on richness of diversity of lianas (Table 16). NMS ordination results showed no significant differences between undisturbed forest and the two railways or with increasing distance from the forest edge, railways or control transect (Figures 30-32).

Table 15. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on species richness and Shannon's diversity index of lianas (≥2 cm DBH). Significant values are in bold.

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	8.53	**0.005**	2	0.06	0.946	4	6.76	**<0.001**
Shannon's diversity index	2	7.46	**0.008**	2	0.03	0.975	4	5.07	**0.001**

Table 16. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on species richness and Shannon's diversity index of lianas (≥2 cm DBH).

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	0.79	0.497	4	0.81	0.525	8	0.22	0.986
Shannon's diversity index	2	0.96	0.434	4	0.98	0.424	8	0.17	0.995

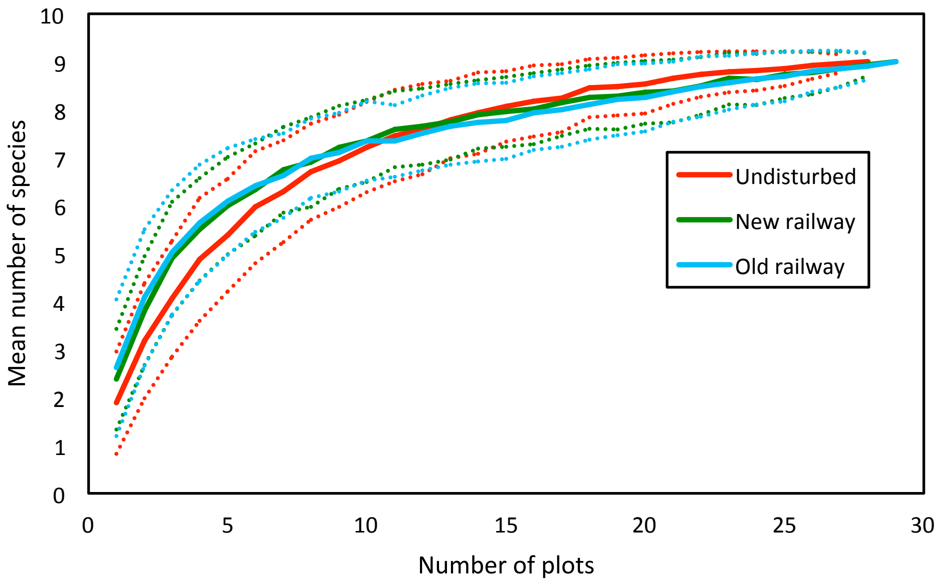


Figure 27. Species-area relationship for mean number of liana species (± 95% confidence intervals) present across all plots excluding unknown species.

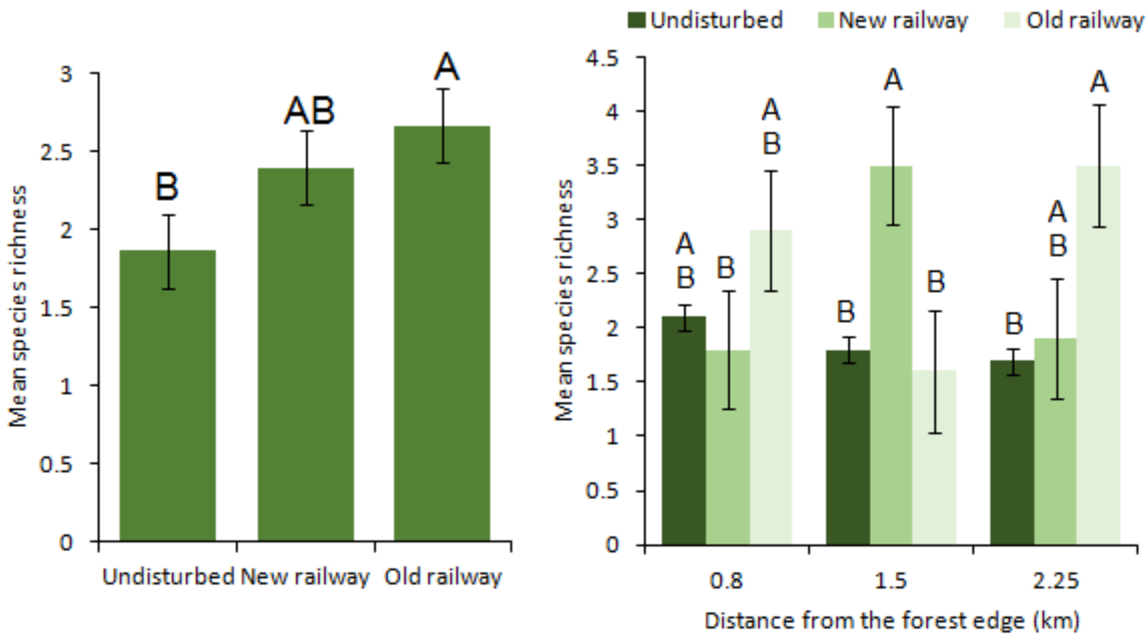


Figure 28. Mean (± SE) species richness of lianas (≥2 cm DBH) per 100 m^2 across treatments and with increasing distance from the forest edge within treatments. Tukey test results are indicated where means that do not share a letter are significantly different.

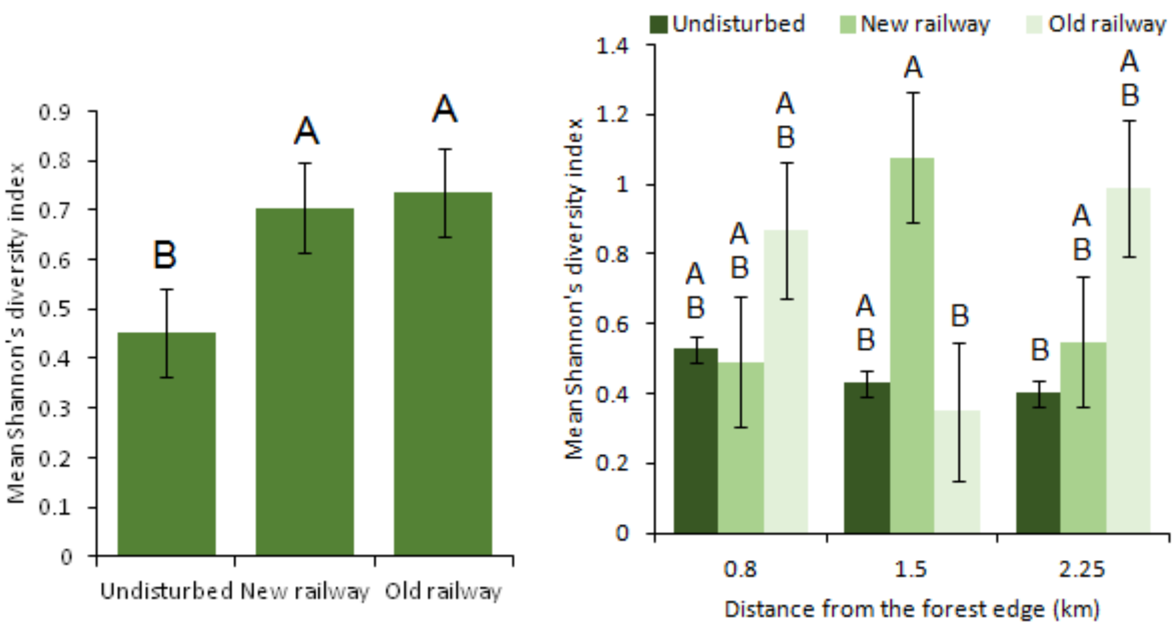


Figure 29. Mean (± SE) Shannon's diversity of lianas (≥6 cm DBH) per 100 m^2 across treatments and with increasing distance from the forest edge within treatments. Fisher (left) and Tukey (right) test results are indicated where means that do not share a letter are significantly different.

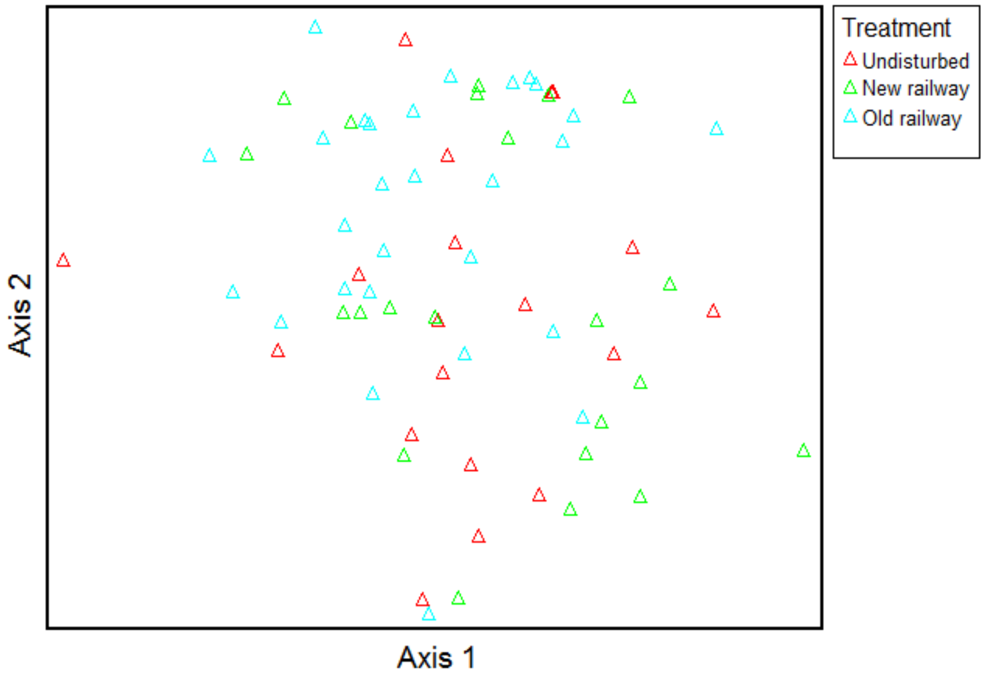


Figure 30. NMS ordination result indicating species composition of lianas (≥2 cm DBH) within plots across different treatments, where each symbol represents a plot. Plots closer together are more similar.

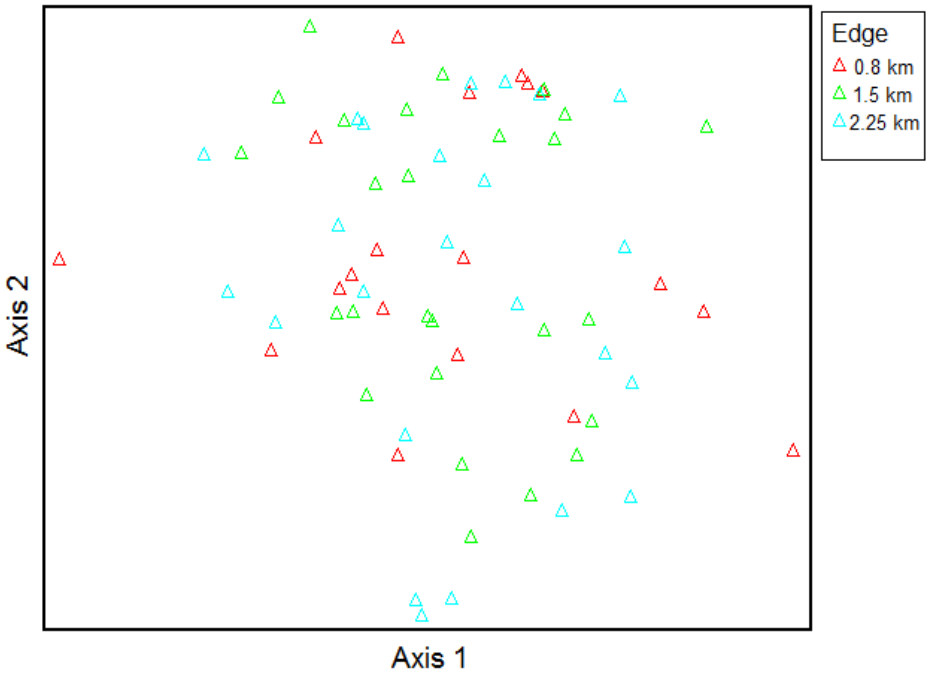


Figure 31. NMS ordination result indicating species composition of lianas (≥2 cm DBH) within plots in relation to distance from the forest edge, where each symbol represents a plot. Plots closer together are more similar.

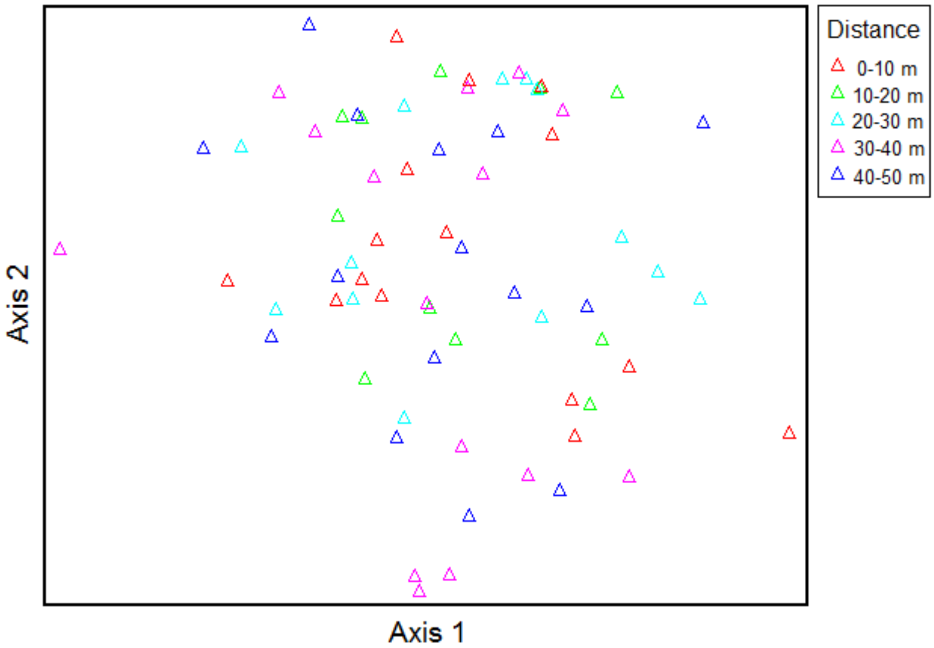


Figure 32. NMS ordination result indicating species composition of lianas (≥2 cm DBH) within plots in relation to distance from the railway or control transect, where each symbol represents a plot. Plots closer together are more similar.

No significant differences in species richness or diversity of small lianas occurred in response to logging when unknown species were included (Tables 17-18). The species-area relationships plateau indicating that the study was likely to have captured most of the small liana species (Figure 33). However, the species-area relationships also suggested significant differences between all treatments when unknown species were excluded from analysis, with old railway and new railway plots containing the highest and lowest number of species, respectively. NMS ordination results showed that plots in undisturbed forest were not distinctly different in species composition from plots along the two railways, however there was more variation in species composition among plots by the railways (Figure 34). Similarly, there were no distinct differences between plots at different distances from the forest edge but those at 0.8 km showed less variation in species composition (Figure 35).

There were no significant differences in species composition at different distances from the railways or control transect (Figure 36).

Table 17. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the edge of the forest (0.8 km, 1.5 km and 2.25 km) and the interaction between treatment and edge on species richness and Shannon's diversity index of small lianas (<2 cm DBH).

	Effect of treatment			**Effect of edge**			**Effect of treatment*edge**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	0.01	0.992	2	0.79	0.459	4	1.46	0.224
Shannon's diversity index	2	0.07	0.932	2	0.74	0.480	4	1.09	0.369

Table 18. GLMM results showing the effect of treatment (relatively undisturbed forest, new railway, old railway), distance from the railway or control transect (0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m) and the interaction between treatment and distance on species richness, and Shannon's diversity index of small lianas (<2 cm DBH).

	Effect of treatment			**Effect of distance**			**Effect of treatment*distance**		
	d.f.	F-value	P-value	d.f.	F-value	P-value	d.f.	F-value	P-value
Species richness	2	0.00	0.996	4	1.48	0.217	8	0.30	0.962
Shannon's diversity index	2	0.04	0.960	4	1.21	0.316	8	0.25	0.980

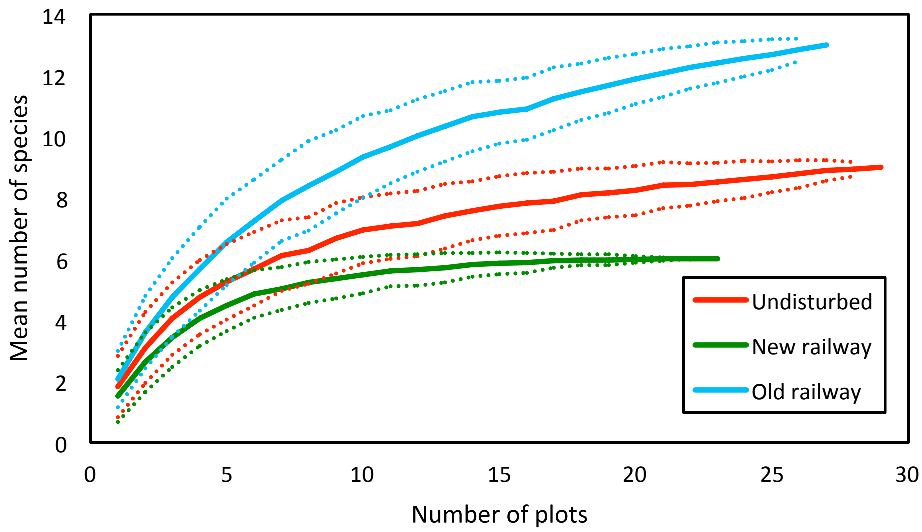


Figure 33. Species-area relationship for mean number of small liana species (± 95% confidence intervals) present across all plots excluding unknown species.

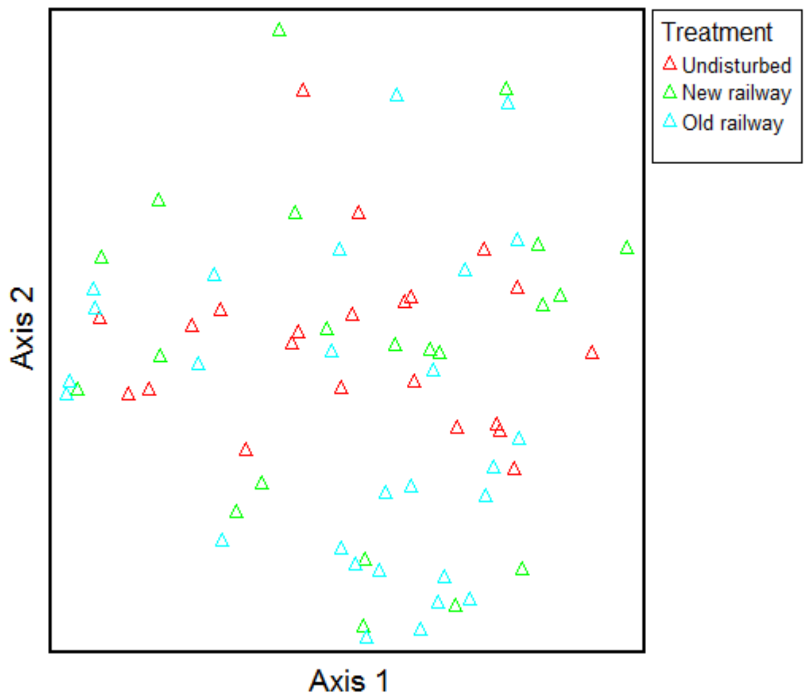


Figure 34. NMS ordination result indicating species composition of small lianas (<2 cm DBH) within plots across different treatments, where each symbol represents a plot. Plots closer together are more similar.

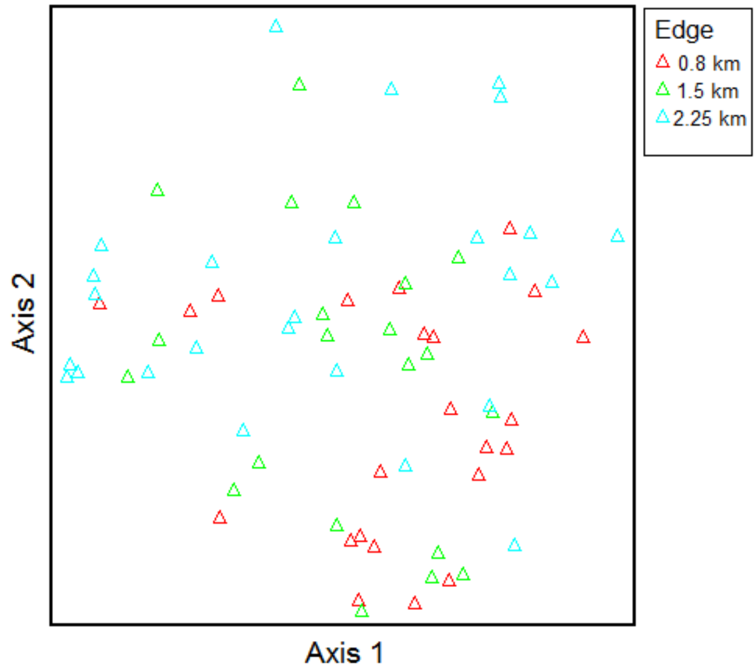


Figure 35. NMS ordination result indicating species composition of small lianas (<2 cm DBH) within plots in relation to distance from the forest edge, where each symbol represents a plot. Plots closer together are more similar.

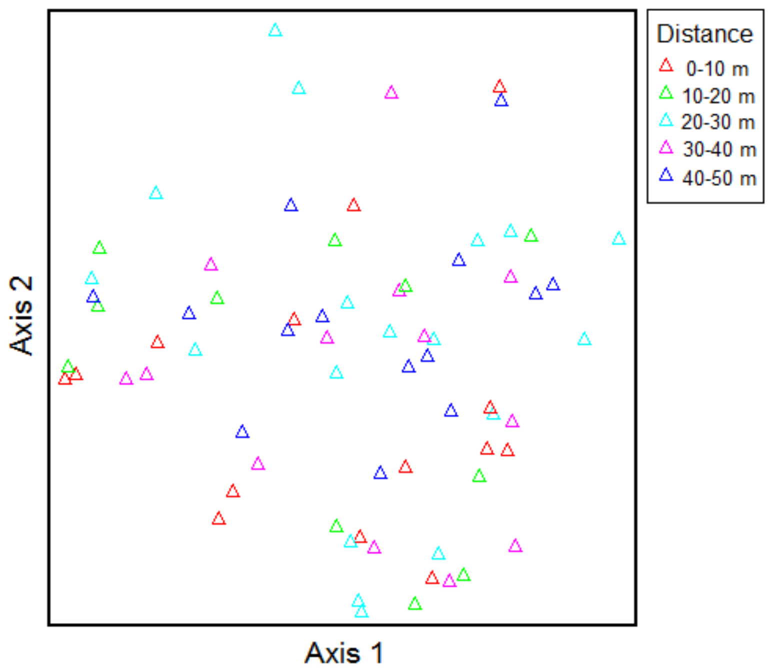


Figure 36. NMS ordination result indicating species composition of small lianas (<2 cm DBH) within plots in relation to distance from the railway or control transect, where each symbol represents a plot. Plots closer together are more similar.

4.3 12 years of forest regeneration

In his 2002 study, Allinson found that canopy cover >10 m was significantly higher in old railway plots than undisturbed and new railway plots (Table 19). In comparison, the present study found higher cover of mid-canopy (11-20 m) in old railway plots than new railway plots but both were similar to undisturbed forest. However, there were no differences in upper (21-30 m) or overall canopy between treatments in the present study. In 2002, canopy cover increased with distance from the railways but not the control transect. Such trends in this study were not apparent as only upper canopy differed with distance but this was consistent across all treatments. Tree height, DBH and basal area in 2002 were all found to be significantly higher in undisturbed forest than in railway plots. The present study also found DBH to be significantly higher in undisturbed forest but found no significant effect of logging treatment on tree height and basal area. Tree height, DBH and basal area in 2002 increased with distance from the railways but not the control transect. In comparison, the present study only found tree height to differ significantly with distance within the first 20 m from the railway or control transect and this was consistent across logged and undisturbed forest. There was no significant effect of distance on DBH or basal area in the present study. Stem density of trees was significantly higher in old railway plots than undisturbed forest in both studies, however new railway plots only differed significantly from undisturbed forest in 2014. Neither studies found significant differences in stem density with distance from the railways.

Tree species richness was found to be higher in new railway plots and lowest in undisturbed plots in 2002 but these results were not tested for significance. Comparatively, the present study found no significant differences in richness between logged and undisturbed forest or diversity, which was not measured in Allison’s study. Both richness and diversity peaked at 20-30 m from the edge of the railways in 2002, while no such trend occurred with distance from the control transect, whereas the present study found no significant effect of distance on either variable. Species composition in the 2002 study showed no greater variation between railway and undisturbed plots than between the two railways. Conversely, this study found greater differences between the railways and undisturbed plots and fewer distinct differences between the two railways.

Allinson found that height and richness of small trees was greater in the railways than undisturbed forest, however again richness was not tested for significance. The present study found no significant effects of logging treatment on height, richness or diversity (which was not calculated in Allinson's study) of small trees. In 2002, small tree height increased with distance from the railways between. The present study was significant but only found marginal differences with distance within treatments and no apparent trends. Neither study found significant differences in species richness or diversity of small trees with distance from the railways.

Allinson (2002) did not include lianas or examine the effect of distance from the forest edge on forest structure, composition or regeneration. Therefore no comparisons could be made regarding these variables.

Table 19. Comparison of results of structural and compositional variables that were measured in both Allinson's (2002) study and the present study with respect to treatment and distance from the railways or control transect. Treatments are labelled where means that do not share letters indicate those that were significantly different. Distances that were significantly different are indicated in bold while those that were not significantly different have been excluded from the table. Abbreviations: UF = Undisturbed Forest, NR = New Railway, OR = Old railway, sig. = significant/ significance, N/A signifies variables that were not measured or calculated.

		Effect of treatment		Effect of distance	
		2002	**2014**	**2002**	**2014**
Trees	Canopy cover >10 m	OR(a) > UF(b) and NR(b)	Upper canopy: No sig. effect. Mid-canopy: OR(a) >UF(ab) > NR(b). Overall: No sig. effect	Increased with distance from railways	Mid canopy: No sig. effect. Upper canopy: **30-40 m** > **0-10 m**. Overall: No sig. effect.
	Height	UF(a) > NR(b) and OR(b)	No sig. effect	Increased with distance from railways	**10-20 m** > **0-10 m.**
	DBH	UF(a) > NR(b) and OR(b)	UF(a) > NR(b) and OR(b)	Increased with distance from railways	No sig. effect
	Basal area	UF(a) > NR(b) and OR(b)	No sig. effect	Increased with distance from railways	No sig. effect
	Stem density	OR(a) >NR(ab) >UF(b)	NR(a) and OR(a) > UF(b).	No sig. effect	No sig. effect
	Species richness	NR>OR>UF (but not tested for sig.)	No sig. effect	Peaked at 20-30 m for both railways	No sig. effect
	Shannon's diversity	N/A	No sig. effect	Peaked at 20-30 m for both railways	No sig. effect.
Small trees	Height	NR(a) and OR(a)>UF(b)	No sig. effect	Increased with distance from railways	Marginal sig. differences but no apparent trends.
	Species richness	NR>OR>UF (but not tested for sig.)	No sig. effect	No sig. effect	No sig. effect
	Shannon's diversity	N/A	No sig. effect	No sig. effect	No sig. effect

5. Discussion

5.1 Forest structure

Changes in canopy cover at 0-10 m and 11-20 m height with distance from the forest edge could be explained by the finding from the present study that height of adult trees increased further from the forest edge. Since this increase in tree height occurred across all treatments, these findings are unlikely to be explained by logging activities associated with the railways. Taller trees further into the forest may be related to edge effects such as less wind exposure deeper into the forest. However, no such effect of distance from the forest edge was found regarding other measures of size (DBH or basal), which may support this. It is also possible that taller trees were removed by illegal loggers in more accessible forest, including relatively undisturbed forest, closer to the north edge up until illegal logging ceased in 2004. However, this study did not find an increase in stem density or in small tree height closer to the forest edge that would support the changes in low canopy cover. This may be because this study measured seedlings and saplings in 2 x 2 m plots, which may not have been representative of all small trees within the 10 x 10 m plots in which canopy cover was measured.

Differences in mid-canopy cover (11-20 m) between treatments may have occurred due to different stages in recovery; for instance, new railway plots may have less mid-canopy cover because pioneers that colonised after logging are not yet as tall as those in old railway plots. However it is important to note that this result was only significant in one model and that estimating canopy cover may be subjective to observer bias, with mid-canopy the most difficult to estimate. Greatest cover of canopy at 21-30 m height in new railway plots at 2.25 km may be associated with former logging activities since this forest is less accessible so fewer trees are likely to have been removed therefore still representing a larger proportion of the upper canopy in comparison with forest closer to the edge. Comparatively, upper canopy may have had less cover in more mature forest i.e. along the old railway and in undisturbed forest due to higher frequency of gap formations associated with natural tree falls. Studies have confirmed more frequent occurrence of gap formation as forest matures (Knight, 1975; Brokaw, 1982; Lang and Knight, 1983) while less frequent gap formation has been recorded in logged and regenerating tropical forests (Chapman and Chapman, 1997). Changes in upper canopy cover and tree height with distance from the railways and control transect suggest that differences were not due to logging activity through use of the extraction railways. Although plots were located back from the railways and hand-cut control transect, these results suggest that the direct impact of these was still in effect within 0-10 m. Alternatively, timber may have been extracted by illegal loggers, predominantly within the first 0-10 m from the control transect within undisturbed forest.

Lack of significant differences in overall canopy cover in response to logging could be explained by the observed changes in different canopy layers since changes in low canopy were generally accompanied by reverse changes in mid- and upper canopy. In comparison, Okuda et al (2003) found that mean canopy had greater cover in primary forest than 39-year selectively logged forest in Malaysia, suggesting that canopy in the NLPSF is recovering at a faster rate. This partially supports H1d, however this study found less differences in canopy cover than were expected and most differences were consistent across logged and unlogged forest, which provides evidence of canopy recovery since the use of logging railways.

Higher density of trees in railway plots with smaller mean DBH and basal diameter was consistent with studies by Supardi, 1999 and Okuda et al., 2003 that found higher densities of medium sized trees in selectively logged areas than in primary forest. Combined changes in density and DBH of trees explain why there were no significant differences in basal area between the railways and undisturbed forest. These results are likely to be due to removal of trees with large DBH by logging concessions. Conversely, Asase et al (2014) found no differences in DBH, density or basal area in 29-35 year logged moist deciduous and moist evergreen forest in Ghana. This suggests that as forests regenerate, density is naturally reduced as smaller trees are out-competed for resources such as space and light, allowing remaining trees to increase in size. Since this study looked at 16-20 year logged forest, it is likely that if it were to be repeated in another 10-15 years there may be no significant differences in DBH or density between logged and unlogged forest. H1a was only partially accepted since differences in diameter were as expected between logged and unlogged forest, however variation in tree height did not appear to be related to impacts of the logging concessions and differences in basal area and density contradicted expectations. Nevertheless, although contrary to predictions, density results suggest that impacts of logging activities on forest structure are still prevalent.

Greater basal diameter and height of seedlings closer to the forest edge in new railway plots may be due to opportunities created by disturbance such as changes in soil, which would support the assumption that more trees were removed through logging activities closer to the forest edge but were unlikely to be related to canopy since changes in canopy with distance from the edge were mostly consistent across treatments. Changes in old railway and undisturbed plots with distance suggest that other factors contributed to seedling growth such as opportunities created by natural tree falls. However, similar overall density of small trees across logged and unlogged forest supports the findings of Chapman and Chapman (1997) who found no differences in seedling density between 25-year logged and unlogged forest in Uganda. Although logging disturbance can encourage recruitment and growth of seedlings through gap formation, it is likely that such disturbance only had temporary impacts on seedling density as has been suggested in studies by Chapman and Chapman (1997) and Duah-Gyamfi et al (2014) which found that numbers of seedlings were initially increased by logging disturbance. Convergence of small tree density with undisturbed forest could be due to canopy closure, which supports the finding of the present study that overall canopy cover is similar across logged and undisturbed forest. Small tree density along 50 m transects from the railways and control transect was generally lower where seedlings and saplings were greater in height and basal diameter and higher where trees were smaller in height and basal diameter (Figures 11-13) which may be due to self-thinning (as recorded by Swaine and Hall., 1983 and suggested by Duah-Gyamfi et al., 2014). However, lack of clear trends in density or size of seedlings with distance from the railways suggests that the immediate impacts of logging disturbance on microclimate are again no longer apparent. H1b was only partially supported in relation to size of small trees and distance from the forest edge, however lack of differences in mean density and size of small trees overall suggest regeneration is similar between logged and undisturbed forest.

Inconsistent trends among treatments for density and basal area of lianas with distance from the forest edge, supported by lack of significant differences in liana characteristics overall between logged and undisturbed forest, suggests that these results were not strongly associated with impacts of the extraction railways. A study carried out in the Solomon Islands found 470 and 194 large lianas per 10,000 m^2 in logged and unlogged forest, respectively (Whitmore, 1974). In comparison, this study found 403, 480 and 320 large stems per 10,000 m^2 in new railway, old railway and undisturbed plots, respectively which when compared to Whitmore's results suggest much less variation between logged and unlogged sites but higher density within undisturbed forest, suggesting higher levels of disturbance in undisturbed plots within this study. However, these results may not be directly comparable depending on Whitmore's definition of "large lianas". Given that very few lianas occur in 10 x 10 m plots and that these figures had to be extrapolated since only 3000 m^2 were sampled per site, a larger plot or sample size is required to capture more accurate density and measurement data for lianas to be confident with these results. Higher DBH of lianas further from the forest edge may be related to tree fall gaps in mature forest. A study in subtropical forest in Argentina found that total density of lianas (>2cm DBH) increased with density of recent tree fall gaps (≤6 years old), which was also found to be the most influencing factor on liana communities (Malizia et al., 2010). This result may also be associated with other forms of disturbance across all plots such as illegal logging and fire. Although DBH was found to be significant with distance from the railways and control transect, marginal differences suggest that a larger sample size may have reduced these differences. Other factors that may have contributed to liana density and growth throughout the forest which have been found to be important in determining liana colonisation include tree crown size, shape and height (Alvira et al., 2004).

Conversely liana seedlings were larger in new railway plots, which implies higher levels of disturbance along the new railway than along the old railway, or in undisturbed forest. This is supported by the findings from this study that there was lower mid-canopy cover by the new railway and upper canopy had very low cover in all plots (<30%) therefore higher light levels would be expected to reach the forest floor in new railway plots. However, results for density of small lianas found no significant differences among treatments suggesting that levels of disturbance did not affect density. Given that few liana seedlings occurred in each plot, again it may be that a larger sample size would alter these results. Therefore H1c was neither rejected nor accepted due to uncertainty of results.

5.2 Species composition, richness and diversity

Similarity in diversity and richness of large trees between logged and unlogged forest, as found in the present study, was consistent with studies carried out in Kalimantan c.15 years after selective logging (Slik et al., 2002) in 13 year-logged forest in Sabah, Borneo (Bischoff et al., 2005) and 18 years after logging (Berry et al., 2008). In comparison, logging disturbance has been found to increase tree species richness in larger samples (Cannon et al., 1998). However, the species area relationship indicates that this study is likely to have captured most of the tree species in the forest therefore it is unlikely the same result would emerge from a larger sample in the present study. Higher diversity and richness of large trees in 8 year-logged forest as found by Cannon et al. (1998) suggests that initially species richness is enhanced by logging disturbance but with increased time since disturbance, richness and diversity of logged forest is likely to be similar to that of unlogged forest since such trends were no longer apparent in the present study. On the other hand, logged and unlogged forest in the present study showed marked differences in species composition of adult trees, probably as a result of logging impacts. It may be that the number of commercial species and pioneer species differs between the railways and undisturbed forest, although overall diversity and richness is not affected. Asase et al (2014) found that there was significantly higher proportion of pioneer species among trees >10 cm DBH in post-logged forest even 29-35 years after logging due to destruction of canopy through logging activities. Initial destruction of canopy through logging activities in this study is likely to have encouraged establishment of pioneer species that still persist as adult trees. However Asase et al (2014) found significantly higher Shannon's diversity of large trees associated with higher number of pioneer species in post-logged than unlogged forest which this study did not find. Therefore differences in species composition in this study might be better explained by differential mortality of trees associated with timber harvesting (Cannon et al., 1994; Slik et al., 2002). In western Kalimantan, differential mortality of remaining tree species after logging gave rise to differences in overall species composition in logged and unlogged forest, even when pioneers were discounted Cannon et al (1994). Further analysis could be done on data collected in this study to determine dominant species or plant types that give rise to differential species composition of logged and undisturbed sites in the Sabangau Forest.

Variation in diversity and species composition of trees with distance from the forest edge might be due to edge effects or the additional impact of illegal logging across the forest across all treatments. A study in tropical forest in Brazil, which supports either of these theories, found that pioneer species were more common closer to the forest edge or in disturbed areas (Oliveira-Filho et al., 1997). Such variation in pioneers species could explain overall differences in diversity and composition within the present study. Alternatively variation in tree species composition may be due to the decline in similarity of plots with increasing distance as has been recorded in tropical forests (He et al., 1997; Condit et al., 2002) and can be due to many factors including limited seed dispersal (He et al., 1997) and disturbance (Moloney et al., 1992). H2a was partially supported as this study found differences in species composition between logged and unlogged sites. However contradictory to predictions, species richness and diversity of logged forest appears to have recovered to be similar to that of undisturbed forest.

Similarity in diversity and richness of small trees between logged and relatively undisturbed forest supports studies by Duah-Gyamfi et al. (2014), Bischoff et al (2005) and Berry et al (2008) which found no differences in diversity and richness of small trees between 7-year, 13-year and 18-year logged forest and unlogged forest, respectively. However, Berry et al (2008) found higher diversity of small trees (2.5-10 cm DBH) in logged forest at a large scale but this is unlikely to be the case in the present study since the species area relationship indicates that this study is likely to have captured most of the tree species in the forest.

Similarities in species composition between logged and undisturbed forest in the present study supports the work of Duah-Gyamfi et al (2014) who found that after 3 years, species composition of logged tropical forest converged with unlogged forest. Since this study was carried out 16-20 years after logging, the immediate impacts of logging disturbance on diversity and composition of small trees were no longer apparent therefore H2b was rejected.

Higher diversity of lianas along the railways and higher richness along the old railway when unknown species were excluded may be due to higher levels of disturbance associated with the logging railways. However, the difference in number of species per plots was only between approximately 2 and 3 species therefore might have little biological significance. Since there were small numbers of individuals and species per plot in this study, identification of unknown species together with a larger sample size would give a more confident result. Insufficient sample size might explain inconsistent trends in diversity and richness of large lianas with distance from forest edge. As with the small trees, similarity in richness, diversity and composition of small lianas between the railways and undisturbed forest may be because the immediate impacts of logging on liana seedling density are no longer apparent due to forest recovery with increased time since logging. However since species richness results for small lianas differed when unknown species were excluded from analysis, this suggests that unknown species affect richness and diversity results, especially given the small sample size of lianas. Liana seedlings are very difficult to identify therefore more accurate identification of small lianas may reveal different results. Therefore although results suggest there may be significant differences in richness and diversity of lianas between logged and unlogged areas, H2c was not accepted due to uncertainty of results.

5.3 12 years of forest regeneration

Overall, this study found few differences in structural characteristics of large and small trees between the two railways, which supports the findings of Allinson (2002). However, Allinson (2002) suggested that a greater time period might be required to see noticeable differences in forest recovery and regeneration between the two railways. Considering there were few differences, it is likely that the short time (approximately 4 years) separating the use of the two railways had little impact on the recovery in forest structure between these two sites. Fewer differences in structural characteristics of all trees between the railways and undisturbed forest were found in 2014 than in 2002 (Table 19) which suggests that forest along the railways has recovered in the intervening 12 years. Similarly the general lack of trends with distance from railways in this study, which were apparent in Allinson's study, suggests that forest recovery and regeneration since 2002 now obscures the former differences. Based on his results, Allinson suggested that logging disturbance increased species richness of both large and small trees along the two railways. This suggests that logging disturbance initially increased species richness along the railways but has since converged to be the similar to that of undisturbed forest. Although composition was measured differently in both studies so is not directly comparable, results suggest that railways may have been more similar to each other in 2014. This is likely to be due to changes that have occurred by the new railway as part of the forest recovery process, for instance reduction in pioneer species. Since there were fewer differences between logged and undisturbed forest and with distance from the railways in the present study than in 2002, H3 was accepted.

6. Conclusions and recommendations

The results of the present study provided much evidence of forest recovery in 16-20 year logged forest. When compared with Allinson's (2002) study, the results of the present study indicate evidence of forest recovery and regeneration along the extraction railways with regard to forest structure and tree species composition. Overall this study found few marked differences in forest structure or species composition with distance from the railways suggesting that forest recovery has obscured the former differences associated with the impact of the extraction railways. Density and species richness, diversity and composition of small trees were found to be similar in logged and relatively undisturbed forest, although logging may still have impacted size of small trees. Large trees were similar in species richness and diversity within logged and relatively undisturbed forest but differences in composition implied that impacts of the logging railways were still apparent. Further research into plant functional types in this forest would explain these differences. While canopy closure seems to have occurred in logged forest and trends in tree height were consistent in logged and relatively undisturbed forest, marginal differences in canopy layers and higher density of trees with smaller DBH in logged forest implies that the structural community of trees has not fully recovered. Liana results provided some evidence for lasting disturbance impacts of the logging railways, however a larger, more extensive study would provide more reliable results on the impacts of logging on liana communities, particularly in peat-swamp forests. Further research into the effects of edge within primary forest and disturbed forest could reveal whether such differences as found in this study occur due to edge effects or the impact of illegal logging. Although it is clear that forest recovery and regeneration has occurred, this study demonstrates that impacts of logging disturbance can still be evident 16-20 years after logging. The implications of this study are important for future management decisions as it is clear that areas of recovering forest are as important for tree diversity as relatively undisturbed forest, which is fundamental to overall forest biodiversity (Cannon et al., 1998). Although evidence of forest recovery is clear from this study, selectively logged areas are more susceptible to fire, which emphasises the critical need for human intervention of restoration projects to ensure their successful conservation (Page et al., 2009). Furthermore, there remains scope for further research into the interactive effects of disturbance types on peat-swamp forest.

Acknowledgements

Permission to study in the NLPSF and in Indonesia was kindly granted by the Centre for the International Cooperation in Sustainable Management of Tropical Peatlands at the University of Palangka Raya. I'd like to thank Mark Harrison, Megan Cattau and David Burslem for overseeing this project and providing invaluable guidance and advice and Megan for her help in the field. Thanks to all the staff and volunteers at OuTrop who helped me collect data in the field and most importantly made the experience enjoyable and unforgettable: Hendri, Salahudin, Santiano, Ari Purwanto, Suwarjono, Azis, Azis K, Tommy, Unyil, Uzhi, Kineng, Roby, Beth, Emma, Eilidh, Marianne, Jen, Lizzy, Harry, Ben, Jack and Christian and to the other staff and volunteers at camp. Without all these people this research would not have been possible.

References

Abood, S.A, Lee, J.S.H., Burivalova, Z., Garcia-Ulloa, J. and Koh, L.P., 2014. Relative contributions of the logging, fiber, oil palm and mining industries to forest loss in Indonesia. *Conservation Letters,* 0(0), 1-10.

Allinson, T., 2002. *Analysis of the level of damage and regeneration caused by logging railways in a tropical peat swamp forest.* Bachelors (Hons). University of East Anglia.

Alvira, D., Putz, F.E. and Fredericksen, T.S., 2004. Liana loads and post-logging liana densities after liana cutting in a lowland forest in Bolivia. *Forest Ecology and Management*, 190, 73-86.

Andriesse, J.P., 1988. Nature and Management of Tropical Peat Soils. FAO Soils Bulletin, 59. Rome: Food and Agriculture Organization of the United Nations.

Asase, A., Asiatokor, B.K. and Ofori-Frimpong, K., 2014. Effects of selective logging on tree diversity and some soil characteristics in a tropical forest in south west Ghana. *Journal of Forestry Research*, 25 (1), 171-176.

Barbier, 1993. Economic aspects of tropical deforestation in South-East Asia. *Global Ecology and Biogeography Letters*, 3, 1-20.

Beaman, R.S., Beaman, J.H., Marsh, C.W. and Woods, P.V., 1985. Drought and forest fires in Sabah in 1983. *Sabah Society Journal*, 8, 10-30.

Blaser, J., Sarre, A., Poore, D. and Johnson, S., 2011. Status of Tropical Forest Management 2011. ITTO Technical Series, No 38, International Tropical Timber Organization, Yokohama, Japan.

Boehm, H.D.V. and Siegert, F., 2001. Ecological Impact of the One Million Hectare Rice Project in Central Kalimantan, Indonesia, using Remote Sensing and GIS. Paper presented at the *22nd Asian Conference on Remote Sensing (ACRS)*. Singapore, 5-9 November 2001. Singapore: Centre of Remote Imaging, Sensing and Processing (CRISP); National University of Singapore; Singapore Institute of Surveyors and Valuers (SISV); Asian Association on Remote Sensing (AARS).

Boehm, H.D.V. and Siegert, F., 2002. Land use change and (il)-legal logging in Central Kalimantan, Indonesia. In: Rieley, J.O., Page, S.E. (Eds.), Peatlands for people: natural resource functions and sustainable management, Proceedings of the International Symposium on tropical peatland, Jakarta, 22-23 August 2001. Agency for the Assessment and Application of Technology and Indonesian Peat Association, Jakarta, Indonesia, pp. 132–140.

Berry, N. J., Philips, O. L., Ong, R. C. and Hamer, K. C., 2008. Impacts of selective logging on tree diversity across a rainforest landscape: the importance of spatial scale. *Landscape Ecology*, 23, 915-929.

Bischoff, W., Newbery, D.M., Lingenfelder, M., Schnaeckel, R., Hubert Petol. G., Madani, L. and Ridsdale, C.E., 2005. Secondary succession and dipterocarp recruitment in Bornean rain forest after logging. *Forest Ecology Management*, 218, 174–192.

Brokaw, N.V.L., 1982. Treefalls: frequency, timing and consequences. In: Leigh Jr., E.G., Rand., A.S., Windsor, D.W. (Eds), The Ecology of a Tropical Forest: Seasonal Rhythms and Long Term Changes. Smithsonian Institution Press, Washington, DC, 101-108.

Brown, N., 1993. The implications of climate and gap microclimate for seedling growth conditions in a Bornean lowland rainforest. *Journal of Tropical Ecology*, 9, 153-168.

Brown, N., 1998. Out of control: Fires and forestry in Indonesia. Trends in Ecology and Evolution, 13, 41.

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

Cannon, C.H., Peart, D.R and Leighton, M., 1998. Tree species diversity in commercially logged Bornean rainforest. *Science*, 281, 1366-1368.

Cannon, C.H., Peart, D.R and Leighton, M., 1999. Tree species diversity in logged rainforests. *Science*, 284, 1587a.

Chapman, C.A. and Chapman, L.J., 1997. Forest regeneration in logged and unlogged forests of Kibale Nation Park, Uganda. *BIOTROPICA*, 29(4), 396-412.

Cheyne, S.M., 2008. Gibbon feeding ecology and diet characteristics. *Folia Primatologica,* 79, 320.

Cheyne, S.M., Husson, S.J. and MacDonald, D.W., 2010. First Otter Civet *Cynogale bennettii* photographed in Sabangau Peat swamp Forest, Indonesian Borneo. *Small Carnivore Conservation*, 42, 25-26.

Cheyne, S.M. and Macdonald, D.W., 2011. Wild felid diversity and activity patterns in Sabangau peat-swamp forest, Indonesian Borneo. *Oryx*, 45, 119-124.

Cheyne, S.M., Morrogh-Bernard, H. and MacDonald, D,W., 2009. First flat-headed cat photo from Sabangau peat-swamp forest, Indonesian Borneo. *Cat News*, 51, 16.

Cleary, D.F.R., Genner, M.J., Boyle, T.J.B., Setyawati, T., Angraeti, C.D. and Menken, S.B.J., 2005. Associations of bird species richness and community composition with local and landscape-scale environmental factors in Borneo. *Landscape Ecology*, 20, 989–1001.

Condit, R., Pitman, N., Leigh, E.G.J., Chave, J., Terborgh, J., Foster, R.B., Núñez, P., Agular, S., Valencia, R., Villa, G., Muller-Landau, H.C., Losos, E. and Hubbell, S.P., 2002. Beta-diversity in tropical forest trees. *Science,* 295, 666-669.

Duah-Gyamfi, A., Swaine, E. W., Adam, K.A., Pinard, M.A. and Swaine, M.D., 2014. Can harvesting for timber in tropical forest enhance timber tree regeneration? *Forest Ecology and Management*, 314, 26-37.

Ehlers-Smith, D. A. and Ehlers-Smith, Y. C., 2013. Population density of red langurs in Sabangau tropical peat-swamp forest, Central Kalimantan, Indonesia. *American Journal of Primatology,* 75**,** 837-847.

Eshelman, R., 2014. Indonesia imposes moratorium on new logging permits. *Mongabay,* [online] 20 Novemeber 2014. Available at: http://news.mongabay.com/2014/1120-eshelman-indonesia-logging-moratorium.html

FAO, 1988. *Nature and management of tropical peat soils.* Rome: Food and Agriculture Organization of the United Nations.

Geollogue, R.T. and Hue, R., 1981. Early stages of forest regeneration in south Sumatra. *BIOTROP Special Publications*, 13, 153-161.

Gill, A.E. and Rasmussen, E.M., 1983. The 1982-83 climate anomaly in the Equatorial Pacific. *Nature*, 306, 229-234.

Grieser Johns, A., 1997. *Timber Production and Biodiversity Conservation in Tropical Rain Forests*. Cambridge: Cambridge University Press.

Hamard, M., Cheyne, S.M. and Nijman, V., 2010. Vegetation Correlates of Gibbon Density in the Peat-Swamp Forest of the Sabangau Catchment, Central Kalimantan, Indonesia. *American Journal of Primatology*, 72, 607-616.

Harrison, M., Page, S.E. and Limin, S.H., 2009. The Global Impact of Indonesian Forest Fires. Biologist, 56 (3), 156-163.

He, F., Legendre, P. and LaFrankie, J.V., 1997. Distribution patterns of tree species in a Malaysian tropical rain forest. *Journal of Vegetation Science*, 8, 105-114.

He, F., LaFrankie, J.V. and Song, B., 2002. Scale dependence of tree abundance and richness in a tropical rain forest, Malaysia. *Landscape Ecology*, 17, 559–568.

Hirano, T., Kusin, K., Limin, S. and Osaki, M., 2014. Carbon dioxide emissions through oxidative peat decomposition on a burnt tropical peatland. *Global Change Biology*, 20, 555–565.

Hooijer, A., Page, S.E., Canadell, J.G., Silvius, M., Kwadijk, .J, Wösten, H. and Jauhiainen, J., 2010. Current and future CO2 emissions from drained peatlands in Southeast Asia. *Biogeosciences*, 7, 1505–1514.

Hooijer, A., Silvius, M., Wösten, H. and Page, S. 2006. PEAT-CO2, Assessment of CO2 emissions from drained peatlands in SE Asia. Delft Hydraulics report Q3943.

Husson, S., 2014. Ten years no timber from OuTrop's director of conservation. *OuTrop blogspot*, [online] 14th June. Available at: http://outrop.blogspot.co.uk/2014/06/ten-years-no-timber-from-outrops.html [Accessed 2 Oct 2014].

Ismail, Z., 1999. Survey of fish diversity in peat swamp forest In: Yuan, C.T., Havmoller, P. (Eds.), Sustainable Management of Peat swamp Forest in Peninsular Malaysia. Forestry Department Peninsular Malaysia, Kuala Lumpur, 173–198.

ITTO, 2001. Achieving sustainable forest management in Indonesia. Report of the diagnostic mission. Presented at the thirty-first session of the International Tropical Timber Council, November 2001. ITTO, Yokohama, Japan.

IUCN, 2014. *The IUCN Red List of Threatened Species. Version 2014.3*. [online]. Available at: http://www.iucnredlist.org [Accessed 12 Jan 2015].

Jauhiainen, J., Limin, S., Silvennoinen, H., Vasander, H., 2008. Carbon dioxide and methane fluxes in drained tropical peat before and after hydrological restoration. *Ecology*, 89, 3503–3514.

Jauhiainen, J., Takahashi, H., Heikkinen, J.E.P., Martikainen, P.J. and Vasander, H., 2005. Carbon fluxes from a tropical peat swamp forest floor. *Global Change Biology*, 11, 1788–1797.

Knight, D.H., 1975. A phytosociological analysis of species-rich tropical forest on Barro Colorado Island, Panama. *Ecological Monographs*, 45, 259-284.

Lang, G.E. and Knight, D.H., 1983. Tree Growth, mortality, recruitment, and canopy gap formation during a 10-year period in a tropical moist forest. *Ecology*, 64, 1075-1080.

Langner, A., Miettinen, J. and Siegert, F., 2007. Land cover change 2002–2005 in Borneo and the role of fire derived from MODIS imagery. *Global Change Biology,* 13, 2329-2340.

MacDicken, K.G., 2001. Cash for tropical peat: land use change and forestry projects for climate change mitigation. In: Rieley, J., Page, S. (eds) Peatlands for people: natural resources function and sustainable management. Proceedings of the international symposium on tropical peatland. Jakarta: BPPT and Indonesian Peat Association, 1–6, 22–23 Aug 2001

MacKinnon, K. and MacKinnon, J., 1991. Habitat protection and reintroduction programs. In: Gipps, J.H.W. (Ed.), Beyond Captive Breeding. Re-introducing Endangered Mammals to the Wild, Vol. 62. Zoological Society of London Symposia. Oxford: Oxford University Press, 173–198.

Malizia, A., Grau, H. R. and Lichstein, J.W., 2010. Soil phosphorus and disturbance influence liana communities in a subtropical montane forest. *Journal of vegetation Science*, 21, 551-560.

Manduell, K.L., Harrison, M. E. and Thorpe, S.K.S., 2012. Forest structure and the support for Orang-utan locomotion in Sumatra and Borneo. *American Journal of Primatology*, 74, 1128-1142.

McCarthy, J.F., 2002. Turning in Circles: District Governance, Illegal logging, and Environmental Decline in Sumatra, Indonesia. *Society and Natural Resources*, 15, 867-886.

McCune, B. and M. J. Mefford., 2006. PC-ORD. Multivariate analysis of Ecological Data, Version 5.0 for Windows.

Miettinen, J. and Liew, S.C., 2010. Degradation and development of peatlands in Peninsular Malaysia and in the islands of Sumatra and Borneo since 1990. *Land Degradation and Development*, 21, 285-296.

Moloney, K.A., Levin, S.A., Chiarello, N.E. and Buttel, L., 1992. Pattern and scale in a serpentine grassland. *Theoretical Population Biology*, 41, 257-276.

Morrogh-Bernard, H., Husson, S., Page, S.E. and Rieley, J.O., 2003. Population status of the Bornean orang-utan (Pongo pygmaeus) in the Sabangau Peat Swamp Forest, Central Kalimantan, Indonesia. *Biology and Conservation*, 110, 141–152.

Nagendra, H., Opposite trends in response for the Shannon and Simpson indices of landscape diversity. *Applied geography*, 22, 175-186.

Notohadiprawiro, T., 1998. Conflict between problem solving and optimising approach to land resources development policies- the case of Central Kalimantan wetlands. In: Sopo, R. (ed) *Proceedings of the International Peat Symposium - The Spirit of Peatlands.* Jyväskylä Finland, 7-9 September 1998. 14-24. Jyväskylä Finland: International Peat Society.

Oliveria-Filho, A.T., Marcio de Mello, J. and Scolforo, R.S., 1997. Effects of past disturbance on tree community structure and dynamics within a fragment of tropical semideciduous forest in South-Eastern Brazil over a five-year period (1987-1992). *Plant Ecology*, 131, 45-66.

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

OuTrop, n.d. *The Sabangau Forest*. [online] Available at: http://www.outrop.com/sabangau-forest.html [Accessed 2 Oct 2014].

Page, S., Hosciło, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., Ritzema, H., Tansey, K., Graham, L., Vasander, H., and Limin, S., 2009. Restoration Ecology of Lowland Tropical Peatlands in Southeast Asia: Current Knowledge and Future Research Directions. *Ecosystems*, 12, 888-905.

Page, S.E. and Rieley, J.O., 1998. Tropical peatlands: a review of their natural resource functions, with particular reference to Southeast Asia. *International Peat Journal*, 8, 95–106.

Page, S.E., Rieley, J.O., Shotyk, W. and Weiss, D., 1999. Interdependence of peat and vegetation in a tropical peat swamp forest. *Philosophical Transactions of the Royal Society of London B*, 354, 1885–1897.

Page, S.E., Rieley, J.O. and Banks, C.J., 2011. Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*, 17, 798–818.

Page, S.E., Siegert, F., Rieley, J.O., Boehm, H-D.V, Jaya, A. and Limin, S.H., 2002. The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, 420, 61–65.

Posa, M.R.C., Wijedasa, L.S. and Corlett, R.T., 2011. Biodiversity and Conservation of Tropical Peat Swamp Forests. *BioScience*, 61, 49-57.

Prentice, C. and Parish D., 1990. Conservation of peat swamp forest: A forgotten ecosystem. Pages 128–144 in Proceedings of the International Conference on Tropical Biodiversity, 12–16 June 1990. International Conference on Tropical Biodiversity.

Rieley, J.O., 2003. Personal communication, In: Allinson, T., 2002. Analysis of levels of damage and regeneration caused by logging railways in a Tropical Peat Swamp forest. *Unpublished works*.

Rieley, J. 2014. Ten years no timber. *OuTrop blogspot*, [online] 1st June. Available at: < http://outrop.blogspot.co.uk/2014/06/ten-years-no-timber.html> [Accessed 2 Oct 2014].

Rieley, J.O., Ahmad-Shah, A.A. and Brady, M.A., 1996. The extent and nature of tropical peat swamps. In: Maltby, E., Immirzi, C.P., Savord, R.J. (eds) Tropical lowland peatlands of southeast Asia. Proceedings of a workshop on integrated planning and management of tropical lowland peatlands held at Cisarua, IUCN, Gland, 3–8 July 1992.

Rieley J.O. and Ahmad-Shah A.A., 1996. The vegetation of tropical peat swamp forests. In: Maltby E., Immirzi C.P. and Safford R.J. (eds), Tropical Lowland Peatlands of Southeast Asia. Proceedings of a Workshop on Integrated Planning and Management of Tropical Lowland Peatlands. IUCN, Gland, Switzerland.

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: Rieley, J.O., Page, S.E., editors. Tropical peatlands. Cardigan: Samara Publishing Limited. 37–44.

Rieley, J.O.,Wüst, R.A.J., Jauhiainen, J., Page, S.E., Wösten, H., Hooijer, A., Siegert, F., Limin, S.H., Vasander, H. and Stahlhut, M., 2008. Tropical peatlands: carbon stores, carbon gas emissions and contribution to climate change processes. In: Peatlands and Climate Change (ed. Strack M), 148–181. International Peat Society, Jyväskylä, Finland.

Segah, H., Tani, H. and Hirano, T., 2010. Detection of fire impact and vegetation recovery over tropical peat swamp forest by satellite data and ground-based NDVI instrument. *International Journal of Remote Sensing*, 31 (20), 5297-5314.

Shepherd, P.A., Rieley, J.O. and Page, S.E., 1997. The relationship between forest structure and peat characteristics in the upper catchment of the Sungai Sebangau, Central Kalimantan. In: Rieley J.O., Page, S.E. (Eds.), Biodiversity and Sustainability of Tropical Peatlands. Samara Publishing, Cardigan, UK, 191–210.

Shiel, D., Sayer, J.A. and O'Brien, T., 1999. Tree species diversity in logged rainforests. *Science*, 284, 1587a.

Siegert, F., Rücker, G., Hindrichs, A. and Hoffman, A.A., 2001. Increased damage from fires in logged forests during droughts caused by El Niño. *Nature*, 414, 437–440.

Siry, J.P., Cubbage, F.W. and Ahmed, M.R., 2005. Sustainable forest management: global trends and opportunities. *Forest Policy and Economics*, 7, 551–561.

Slik, J.W.F., Verburg, R.W. and Keßler, P.J.A., 2001. Effects of fire and selective logging on tree species composition of lowland dipterocarp forest in East Kalimantan, Indonesia. *Biodiversity and Conservation*, 11, 85-98.

Supardi, N.Md.N., 1999. The impact of logging on the community of palms in the lowland dipterocarp forest of Pasoh, Peninsular Malaysia.Ph.D. Dissertation. University of Reading, Reading.

Swaine, M.D., Hall, J.B., 1983. Early succession on cleared forest land in Ghana. Journal of Ecology, 71, 601–627.

van der Werf, G.R., Dempewolf, J., Trigg, S.N., Randerson, J.T., Kasibhatla, P.S., Giglio, L., Murdiyarso, D., Peters, W., Morton, D.C., Collatz, G.J., Dolman A.J. and DeFries, R.S., 2008. Climate regulation of fire emissions and deforestation in equatorial Asia. *Proceedings of the National Academy of Sciences of the United States of America*, 105, 20350–20355.

van der Werf, G.R., Randerson, J.T., Collatz, G.J., Giglio, L., Kasibhatia, P.S., Arellano, A.F., Olsen, S.C. and Kasischke, E.S., 2004. Continental-scale partitioning of fire emissions during the 1997 to 2001 El Niño/La Niña period. *Science*, 303, 73–76.

Vijarnsorn P., 1996. Peatlands in Southeast Asia: a regional perspective. In: Maltby E., Immirzi C.P. and Safford R.J. (eds), Tropical Lowland Peatlands of Southeast Asia. Proceedings of a Workshop on Integrated Planning and Management of Tropical Lowland Peatlands. IUCN, Gland, Switzerland.

Whitmore, T.C., 1974. Change with time and the role of cyclones in tropical rainforest on Kalombangara, Solomon Islands. *Commonwealth Forestry Institute*, Paper 46.

Whitmore, T. C., 1984. *Tropical Rainforests of the Far East.* 2nd ed. Oxford: Oxford University Press.

Wich, S.A., Meijaard, E., Marshall, A.J., Husson, S., Ancrenaz, M., Lacy, R.C., van Schaik, C.P., Sugardjito, J., Simorangkir, T., Taylor-Holzer, K., Doughty, M., Supriatna, J., Dennis, R., Gumal, M., Knott, C.D. and Singleton, I., 2008. Distribution and conservation status of the orang-utan (Pongo spp.) on Borneo and Sumatra: How many remain? *Oryx,* 42, 329-339.

Wösten, J.H.M., Clymans, E., Page, S.E., Rieley, J.O. and Limin, S.H., 2008. Peat–water interrelationships in a tropical peatland ecosystem in Southeast Asia. *Catena*, 73, 212-224.

Yule, C.M., 2010. Loss of biodiversity and ecosystem functioning in Indo-Malayan peat swamp forests. *Biodiversity Conservation*, 19, 393-409.

Appendix 1. Path of the "old" timber extraction railway where it used to operate from c.1987-1994. Photo: Katrina Schofield/OuTrop.

Appendix 2. Path of the "new" timber extraction railway where it used to operate from 1994-1998. Photo: Ben Thomas/ OuTrop.

Appendix 3. Complete list of recorded plant species (excluding unknown species).

Botanical binomial	Local/ Indonesian name	Tree/ Liana
Adenanthera pavonina	Tapanggang	Tree
Aglaia rubiginosa	Kajalaki	Tree
Aglaia sp 1	Bangkuang Napu	Tree
Alseodaphne coriacea	Gemur	Tree
Alyxia sp 1	Kelanis	Liana
Ampelocissus sp 1	Liana anggur	Liana
Antidesma coraeceum	Dawat	Tree
Antidesma phanerophleum	Matan undang	Tree
Ardisia cf. sanguinolenta	Kalanduyung himba	Tree
Ardisia sp 2	Kamba sulan	Tree
Artobotrys cf. roseus	Kalalawit hitam	Liana
Artobotrys suaveolins	Bajakah Balayan	Liana
Baccaurea bracteata	Rambai hutan	Tree
Baccaurea stipulata	Kayu tulang	Tree
Blumeodendron elateriospermum / tokbrai	Kenari	Tree
Calophyllum hosei	Jijnjt	Tree
Calophyllum sclerophyllum	Kapurnaga jangkar	Tree
Calophyllum soulattri	Takal	Tree
Calophyllum sp	Kapurnaga kalakai	Tree
Calophyllum sp 2	Mahadingan	Tree
Campnosperma coriaceum	Terontang	Tree
Campnosperma squamatum	Teras nyating	Tree
Canthium dydimum	Kopi kopi	Tree
Cariliia brachiata	Gandis	Tree
Castanopsis foxworthyii / jaherii	Takurak	Tree
Chisocheton sp 1	Latak manuk	Tree
Combretocarpus rotundatus	Tumih	Tree
Combretum sp 1	Bajakah Tampelas	Liana
Cratoxylon arborescens	Geronggang putih	Tree
Cratoxylon glaucum	Geronggang merah	Tree
Ctenolophon parvifolius	Bintan rambut merah	Tree
Cyathocalyx biovulatus	Kerandau	Tree
Dactylocladus stenostachys	Mertibu	Tree
Dialium patens	Prupuk Keras	Tree
Diospyros bantamensis	Malam malam	Tree
Diospyros cf. evena	Gulung haduk	Tree
Diospyros confertiflora	Arang	Tree
Diospyros siamang	Ehang	Tree

Diospyros sp	Kayu arang	Tree
Dipterocarpus borneensis	Keruing	Tree
Dipterocarpus sp	Rasak	Tree
Dyera lowii	Jelutong	Tree
Elaeocarpus acmocarpus	Patanak galaget	Tree
Elaeocarpus marginatus	Mangkinang 2	Tree
Elaeocarpus mastersii	Mangkinang	Tree
Eugenia spicata	Galam tikus or Kayu lalas	Tree
Fibraurea tinctoria	Liana Kuning	Liana
Ficus cf. spathulifolia	Lunuk punai	Tree (fig)
Ficus sp 2	Lunuk bunyer	Liana (fig)
Fragraea sp 1	Kalamuhe	Liana
Garcinia bancana	Manggis	Tree
Garcinia sp 1	Aci	Tree
Garcinia sp 11	Mahalilis	Tree
Garcinia sp 3	Gantalan	Tree
Glochidion cf. glomerulatum	Buah bintang	Tree
Gnetum sp 1	Bajakah luaa	Liana
Gnetum sp 2	Bajakah Oto oto	Liana
Gonystylus bancanus	Ramin	Tree
Gymnacranthera farquhariania	Mendarahan daun kecil	Tree
Horsfieldia crassifolia	Mendarahan daun besar	Tree
Ilex cymosa	(No local name)	Tree
Ilex hypoglauca	Sumpung	Tree
Isonandra sp	Nyatoh palanduk	Tree
Ixora havilandii	Keranji	Tree
Knema intermedia	Kerandau merah	Tree
Koompassia malaccensis	Kempas	Tree
Licania splendens	Bintan peter-peter	Tree
Lithocarpus conocarpus	Pampaning bayang	Tree
Lithocarpus sp	Pampaning	Tree
Lithocarpus sp. 1 cf. dasystachys	Pampaning bitik	Tree
Litsea cf. rufo-fusca	Tampang	Tree
Litsea ochirea	Madang 2	Tree
Litsea sp	Madang	Tree
Litsea sp 1 cf. resinosa	Medang marakuwung	Tree
Lucinea sp 1	Bajakah tabari	Liana
Macaranga caladiifolia	Mahang bitik	Tree
Madhuca mottleyana	Katiau	Tree
Magnolia bintulensis	Asam asam or Madang limo	Tree
Memecylon sp 2	Milas	Tree
Memecylon sp 3	Tabati himba	Tree
Mesua sp 1	Tabaras akar tinggi	Tree
Mezzetia leptopoda / parviflora	Pisang pisang besar	Tree
Mezzetia spp	Pisang pisang	Tree
Mezzetia umbellata	Pisang pisang kecil	Tree

Microcos (Grewia) sp 1	Brania himba	Tree
Neoscortechinia kingii	Pupu palanduk	Tree
Nephellium lappaceum	Rambutan hutan	Tree
Nephellium maingayi	Kelumun buhis	Tree
Palaquium cochlearifolium	Nyatoh gagas	Tree
Palaquium leiocarpum	Hangkang	Tree
Palaquium pseudorostratum	Nyatoh babi	Tree
Palaquium sp	Nyatoh burung	Tree
Parartocarpus venenosus	Lilin lilin	Tree
Phoebe cf. grandis	Tabitik	Tree
Polyalthia glauca	Kayu bulan	Tree
Polyalthia hypoleuca	Alulup or Rewoi	Tree
Sandoricum beccanarium	Papong	Tree
Santiria cf. griffithi	Teras bamban	Tree
Santiria cf. laevigata	Irat	Tree
Schleffera sp 3	Bajakah tabulus	Liana
Shorea teysmanniana	Meranti semut	Tree
Shorea uliginosa	Meranti batu	Tree
Stemonorus cf. scorpiodes	Tabaras tidak ada akar	Tree
Sterculia rhoiidifolia	Loting	Tree
Sterculia sp	Pendu	Tree
Sterculia sp 2	Muara bungkan	Tree
Syzygium havilandii	Tatumbu	Tree
Syzygium sp 13	Tampohot himba	Tree
Syzygium sp 15	Hampuak Galaget	Tree
Syzygium sp 2	Kemuning putih	Tree
Syzygium sp. 5 cf. E.spicata	Kayu lalas daun kecil	Tree
Syzygium spp	Jambu spp	Tree
Ternstroemia magnifica	Tabunter	Tree
Ternstroemia sp	Tabunter daun kecil	Tree
Tetractomia tetrandra	Rambangun	Tree
Tetramerista glabra	Ponak	Tree
Tristaniopsis obovata	Blawan	Tree
Tristaniopsis sp 2	Blawan merah	Tree
Tristaniopsis sp 4	Blawan punai	Tree
Tristaniopsis sp. 3 cf. merguensis	Blawan putih	Tree
Uncaria sp 1	Kalalawit merah	Liana
Uncaria spp	(No local name)	Liana
Unknown latin name	Tulang ular	Tree
Unknown latin name	Tawas ut	Liana
Willughbeia sp 1	(No local name)	Liana
Xanthophyllum cf. ellipticum	Kemuning	Tree
Xerospermum laevigatum / noronhianum	Kelumun biasa	Tree
Xerospermum sp	Kelumun	Tree
Xylopia cf. malayana	Tagula	Tree

Xylopia coriifolia	Jangkang merah or Nonang	Tree
Xylopia fusca	Jangkang Kuning	Tree
Zyzyphus angustifolius	Karinat	Liana

Appendix 4. Mean (+SD) of all measured variables for canopy cover, trees (≥6 cm DBH), lianas (≥2 cm DBH), small trees (<6cm DBH) and small lianas (<2cm DBH) across all plots in relatively undisturbed forest and by the new and old railways.

		Undisturbed		New railway		Old railway	
		Mean	**StDev**	**Mean**	**StDev**	**Mean**	**StDev**
Canopy	Low canopy (0-10m)	68.54	17.299	74.42	15.65	71.79	16.13
	Mid canopy (11-20m)	69.50	23.155	59.25	20.52	77.46	21.50
	Upper canopy (21-30m)	8.00	7.374	9.38	12.70	8.71	8.42
	Overall canopy	90.96	14.876	96.38	4.49	96.00	5.71
Trees	Stem density	17.07	3.88	19.83	4.39	20.63	4.54
	DBH (cm)	13.45	7.25	11.86	5.95	11.67	5.96
	Basal diameter (cm)	15.86	8.22	13.65	6.80	13.54	6.82
	Basal area (cm^2)	3127.75	1037.06	2741.15	889.76	2781.55	1203.69
	Height (m)	13.74	5.26	13.64	4.97	13.23	4.77
	Species richness	12.23	2.93	13	2.73	13.67	3.27
	Shannon's diversity	2.37	0.28	2.41	0.26	2.43	0.32
Small trees	Stem density	24.27	9.38	21.20	7.56	27.03	11.68
	Basal diameter (cm)	0.85	1.07	1.02	1.34	0.90	1.34
	Height (cm)	106.67	144.67	114.27	165.34	103.24	157.39
	Species richness	12.47	3.56	11.00	3.01	13.30	3.66
	Shannon's diversity	2.27	0.36	2.13	0.36	2.31	0.35
Lianas	Stem density	3.17	2.41	4.03	3.79	4.80	3.18
	DBH (cm)	3.26	1.24	3.22	1.25	3.26	1.02
	Basal diameter (cm)	3.82	1.39	3.87	1.43	3.87	1.30
	Basal area (cm^2)	28.83	27.40	37.73	29.64	44.07	30.34
	Species richness	1.87	1.17	2.40	1.13	2.67	1.49
	Shannon's diversity	0.45	0.47	0.71	0.45	0.73	0.54
Small lianas	Stem density	3.60	2.62	3.23	3.11	5.40	5.50
	Basal diameter (cm)	0.29	0.27	0.63	1.11	0.32	0.42
	Species richness	2.03	1.19	2.00	1.80	2.00	1.20
	Shannon's diversity	0.52	0.52	0.55	0.57	0.55	0.42

Appendix 5. Mean (+SD) of all measured variables for canopy cover, trees (≥6 cm DBH), lianas (≥2 cm DBH), small trees (<6cm DBH) and small lianas (<2cm DBH) across all plots in relation to increasing distance from the forest edge.

		0.8 km		1.5 km		2.25 km	
		Mean	**StDev**	**Mean**	**StDev**	**Mean**	**StDev**
Canopy	Low canopy (0-10m)	76.08	14.39	76.46	14.76	62.21	16.15
	Mid canopy (11-20m)	55.33	24.86	73.13	18.70	77.75	18.29
	Upper canopy (21-30m)	6.04	8.77	7.33	7.54	12.71	11.30
	Overall canopy	93.92	7.94	95.79	5.09	93.63	14.18
Trees	Stem density	19.4	4.92	19.37	4.51	18.77	4.17
	DBH (cm)	12.01	6.01	12.52	6.79	12.25	6.42
	Basal diameter (cm)	14.08	6.72	14.63	7.83	14.08	7.39
	Basal area (cm^2)	2747.77	966.91	3084.19	1248.71	2818.50	922.05
	Height (m)	12.43	4.48	13.15	4.97	15.04	5.14
	Species richness	13.57	2.73	13.20	2.87	12.13	3.32
	Shannon's diversity	2.49	0.23	2.43	0.27	2.30	0.33
Small trees	Stem density	26.63	11.45	23.27	6.85	22.60	10.56
	Basal diameter (cm)	0.86	1.15	0.98	1.29	0.92	1.33
	Height (cm)	100.19	145.21	110.61	155.41	113.48	166.94
	Species richness	13.17	3.69	12.13	3.07	11.47	3.66
	Shannon's diversity	2.30	0.33	2.23	0.35	2.17	0.40
Lianas	Stem density	3.43	2.28	4.83	3.79	3.73	3.31
	DBH (cm)	3.14	1.12	3.14	1.17	3.49	1.15
	Basal diameter (cm)	3.76	1.42	3.75	1.31	4.08	1.37
	Basal area (cm^2)	28.22	17.88	42.55	31.24	39.86	35.43
	Species richness	2.27	1.28	2.30	1.24	2.37	1.43
	Shannon's diversity	0.63	0.49	0.62	0.50	0.65	0.53
Small lianas	Stem density	4.13	5.47	3.77	3.02	4.33	3.22
	Basal diameter (cm)	0.34	0.52	0.44	0.79	0.40	0.67
	Species richness	1.77	1.17	2.23	1.72	2.03	1.30
	Shannon's diversity	0.48	0.45	0.63	0.58	0.51	0.47

Appendix 6. Mean (+SD) of all measured variables for canopy cover, trees (≥6 cm DBH), lianas (≥2 cm DBH), small trees (<6cm DBH) and small lianas (<2cm DBH) across all plots in relation to increasing distance from the railways or control transect.

		0-10 m		10-20 m		20-30 km	
		Mean	**StDev**	**Mean**	**StDev**	**Mean**	**StDev**
Canopy	Low canopy (0-10m)	65.83	20.00	68.40	13.07	73.13	14.22
	Mid canopy (11-20m)	65.42	24.77	75.21	21.10	65.21	22.93
	Upper canopy (21-30m)	3.54	4.62	7.92	10.13	9.65	10.23
	Overall canopy	90.90	17.78	95.42	7.43	93.89	7.70
Trees	Stem density	20.61	5.38	19.67	4.39	17.44	3.82
	DBH (cm)	11.75	6.10	12.50	6.59	12.43	6.51
	Basal diameter (cm)	13.42	6.83	14.54	7.56	14.66	7.59
	Basal area (cm²)	2834.30	1165.92	3082.08	1124.18	2694.69	1330.55
	Height (m)	13.18	4.64	14.14	5.02	13.33	5.12
	Species richness	13.94	3.26	13.22	2.82	12.50	3.22
	Shannon's diversity	2.46	0.31	2.44	0.25	2.38	0.32
Small trees	Stem density	24.22	8.08	25.61	9.48	26.78	14.20
	Basal diameter (cm)	1.00	1.32	0.84	1.23	0.87	1.19
	Height (cm)	122.76	171.19	96.53	152.35	103.29	152.80
	Species richness	11.56	4.08	12.83	3.13	12.78	3.44
	Shannon's diversity	2.14	0.44	2.30	0.30	2.27	0.31
Lianas	Stem density	4.39	3.36	4.39	3.03	4.50	4.66
	DBH (cm)	3.34	1.06	3.54	1.38	3.13	1.04
	Basal diameter (cm)	3.93	1.21	4.17	1.72	3.73	1.18
	Basal area (cm²)	42.16	34.61	49.67	33.61	35.75	33.31
	Species richness	2.44	1.25	2.61	1.42	2.39	1.33
	Shannon's diversity	0.73	0.51	0.74	0.44	0.63	0.50
Small lianas	Stem density	5.50	6.63	3.00	2.79	4.83	3.40
	Basal diameter (cm)	0.39	0.71	0.24	0.25	0.45	0.90
	Species richness	2.28	1.45	1.39	1.04	2.44	1.58
	Shannon's diversity	0.61	0.55	0.32	0.40	0.64	0.51

Appendix 6 continued.

		30-40 m		40-50 m	
		Mean	**StDev**	**Mean**	**StDev**
Canopy	Low canopy (0-10m)	79.86	11.60	70.69	19.33
	Mid canopy (11-20m)	69.72	18.74	68.13	26.65
	Upper canopy (21-30m)	11.60	12.23	10.76	8.30
	Overall canopy	96.53	5.19	95.49	4.68
Trees	Stem density	19.61	3.16	18.56	5.19
	DBH (cm)	12.52	6.51	12.15	6.37
	Basal diameter (cm)	14.65	7.30	14.15	7.34
	Basal area (cm^2)	3066.15	907.86	2740.21	677.72
	Height (m)	13.69	5.20	13.25	4.93
	Species richness	13.17	3.11	12.00	2.54
	Shannon's diversity	2.40	0.33	2.34	0.22
Small trees	Stem density	23.11	9.24	21.11	6.92
	Basal diameter (cm)	0.87	1.21	1.03	1.34
	Height (cm)	102.31	151.29	115.26	146.80
	Species richness	12.72	3.75	11.39	3.18
	Shannon's diversity	2.30	0.39	2.17	0.37
Lianas	Stem density	3.50	2.26	3.22	2.29
	DBH (cm)	3.17	1.14	2.97	1.08
	Basal diameter (cm)	3.80	1.38	3.56	1.20
	Basal area (cm^2)	31.16	21.68	25.64	16.66
	Species richness	2.00	1.08	2.11	1.45
	Shannon's diversity	0.48	0.47	0.58	0.57
Small lianas	Stem density	3.83	3.09	3.22	2.65
	Basal diameter (cm)	0.40	0.57	0.46	0.51
	Species richness	1.83	1.29	2.11	1.53
	Shannon's diversity	0.50	0.48	0.63	0.53

School of Biological Sciences

Academic Supervisor:
David Burslem

Honours Project Risk Assessment Form

This form must be used to record the significant risks discussed with the project supervisor.

A copy must be given to the School Safety Adviser BEFORE commencement of the project work.

Name of student: Katrina Schofield

Names of others who will be involved e.g. Post doc, technician:

Dr Mark E. Harrison (Primary OuTrop Research Supervisor), Wiwit Sastramidjaja (Secondary OuTrop Research Supervisor), Simon Husson, (Secondary OuTrop Research Supervisor)

Description of Work

Investigating the impact of logging railways on forest structure and composition in a tropical peatswamp forest. Characteristics such as DBH, basal diameter, height, stem density and species identifications will be measured in tree plots.

Intended location(s) of fieldwork or laboratory number

Sabangau forest, Central Kalimantan, Indonesia.

Intended start date: 3rd July 2014.

Hazard identification **Describe those aspects of the work that could create significant risks**	**Control measures** **List those to be used to reduce the risks to an acceptable level**
Chemical	n/a
Biological	

Diseases – Hepatitis A, Hepatitis B, Typhoid, Tetanus, Polio, Diptheria, Rabies, Japanese Ecephalitus, Malaria. Mosquitoes Poisonous snakes, spiders and scorpions	I have had vaccinations prior to my trip against all the named diseases (except malaria). I will take Doxycycline tablets for malaria 3 days before, during and for a month after my trip. Avoid direct contact with animals and uncooked meat. Only drink bottled water. To avoid mosquito borne diseases, use an insect repellent with a high DEET concentration, wear long clothing (insect repelling if possible) and a mosquito net where possible. Have creams and antibiotics to treat bites. Do not touch. Learn which harmful species are common in area. Always carry a first aid kit. Check clothing and footwear each day for spiders and scorpions.
Sharps	Be careful using any sharp equipment. Carry a first aid kit including anti-bac treatments, plasters and dressings. Vaccinations have been taken against diseases likely to be caught through infection of cuts e.g. Tetanus and Hepatitis B.
Electrical	Use silica gel to protect electrical equipment against humidity.
Radiation	Wear a high factor sun cream, long sleeves (UV protective clothing if possible) and a hat. Stay in the shade where possible.
Field Smoke from forest fires Forest fires Tree falls Unpredictable downpours/ tropical storms Tree roots, hummocks, holes, slippery/ broken board walks Waist high water	Carry an inhaler at all times. Be aware. Assess work site each day for risks. Be aware. Assess work site each day for risks. Always carry waterproofs and know the way to base camp. Choose a careful route when walking and wear appropriate footwear. Wear appropriate footwear and waterproofs.
Manual handling	When handling trees be careful of thorns (See “sharps”) and biting insects (See “biological”).
Workstation	See field.

Other Dehydration Poor road safety Unpredictable political events and natural disasters	 Ensure fluid levels are kept high (drink at least 3L a day of clean water). Carry sachets of rehydration solutions. Use only metered taxis and/or reputable transport companies; avoid unregulated or informal transport operators; take local advice Check area-specific advice on fco.gov website for regular updates.
Animal House/Aquarium Access required Y/N	Not required.

Prepared by _Katrina Schofield___Signature _*K.Schofield*__ Date____08.06.2014_

Approved by _David Burslem_____Signature _*D.Burslem*___ Date____08.06.2014_

Others involved with the work with whom the assessment has been discussed:

Name __Dr Mark E. Harrison____ Signature _ Date _16/06/2014_

Name ________________________ Signature ____________________ Date _______

Name ________________________ Signature ____________________ Date _______

Record of Review

Date	Approved	Comments