

IMPACTS OF LOGGING ON PRIMARY FORESTS OF  
SIBOMA VILLAGE IN THE MOROBE PROVINCE,  
PAPUA NEW GUINEA

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## ABSTRACT

In a world of changing climate, the relationship between species richness, community composition, and anthropogenic landscape change has become an increasingly critical concern for conservation and management. In tropical regions, species diversity is very high with pressure for development being equally as high due to a growing human population. Papua New Guinea (PNG) is an area of critical concern for conservation because of its high rates of endemism and biodiversity, geologic history, proximity to the equator, and assortment of ecosystem types. However, lowland forests of PNG are becoming more and more threatened by logging operations. Few studies have examined the long-term impacts of logging in the tropics, especially in the primary forests of PNG, which is considered “one of the last great unknowns”. The goal of this project was to assess the long-term impacts of logging on plant diversity. We assessed generic diversity, richness, basal area, and canopy cover of trees in a forest that had been logged about 60 years ago compared to a primary old-growth forest in the coastal lowland rainforests of the Siboma Village, Morobe Province, PNG. In each forest type, we surveyed ten 10m x 50m plots. Student’s t-tests, non-parametric Wilcoxon rank sum tests, and non-metric multidimensional scaling ordinations were used to compare the two sites for the vegetative characters listed. Generic diversity, dbh, basal area, and canopy cover were significantly greater in the primary forest than the previously logged forest. The practical implications of these results may be that 1) a longer interval than 60 years is required for a more complete recovery of the forest and 2) more selective harvesting techniques for future logging operations in these forest types might better preserve long-term diversity.

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## CHAPTER 1. INTRODUCTION

### *Biodiversity*

Biodiversity is defined as the variety and variability among living organisms and the ecological complexes in which they occur (Wilson 1988). A world rich in biodiversity provides numerous food items, clothing, shelter, and medicines, as well as copious intangible benefits. As many scientists have noted, biodiversity provides the important quality of balancing food webs and food chains that keep populations and ecological processes in check (Reaka-Kudla et al. 1997). As primary producers, terrestrial plants form a foundational trophic level, meaning they provide much of the usable energy and biomass for upper trophic levels. When diversity of plants is lost there may be cascading effects on the upper trophic levels (Gardner et al. 2009). With the increased concern about global climate change, conservation of biodiversity has become a priority for many scientists (Reaka-Kudla et al. 1997).

A general characteristic of biodiversity is that the number of species increases as latitudinal distance from the equator decreases (Reaka-Kudla et al. 1997). Areas in close proximity to the equator, with consistent precipitation throughout the year, and with mild fluctuations in temperature, are described as tropical rainforests, which are recognized as areas of high biodiversity (Reaka-Kudla et al. 1997). Increasingly, attention has been paid to tropical rainforests for two main reasons: firstly, tropical rainforests only comprise 7% of Earth's land surface area, but may contain 50% of the world's species; and secondly, these ecosystems are exhibiting rapid loss of biodiversity and increased degradation of habitat due to development and large-scale resource extraction (Gardner et al. 2009; Wilson 1988). Within the whole tropical forest biome, only 9.8% of the land is maintained within protected areas (Gardner et al. 2009). Tropical conservation is further hindered because many equatorial tropical ecosystems lack comprehensive biodiversity surveys and inventories (Wilson 1988). Making precise assessments and conservation recommendations depends upon information regarding species presence and biological/ecological properties of these species (Wilson 1988). Therefore, surveys, inventories, and other assessments of tropical rainforests are critical for conservation management and biodiversity preservation.

*An area important for conservation*

New Guinea is an area of critical tropical conservation concern not only because it contains equatorial rainforests, but also because it has been designated by the International Union for Conservation of Nature (IUCN) as one of the last remaining “High Biodiversity Wilderness” areas in the world ([www.iucn.org](http://www.iucn.org)). Furthermore, New Guinea is the world’s second largest tropical island (Marshall and Beehler 2007; Loffler 1977). The creation of the island was a product of a collision of two tectonic plates—the Indo-Australian and the Sunda shelf-Eurasian, which created the mountain range that runs the length of the island (Laurance et al. 2012; Loffler 1977). As a result of this complex geomorphological history, ecosystems range from mangroves and coastal forests, to savannahs, to montane forests, making New Guinea one of most diverse areas in the world (Laurance et al. 2012; Shearman et al. 2008; Wilson 1988). Due to New Guinea’s location in relation to the equator, its complex geomorphological history, and the vast array of ecosystem types, a high degree of endemism and biodiversity are found here (Gillespie et al. 2008).

The preservation and conservation of ecosystems in Papua New Guinea (PNG), the focus of the present study, are extremely critical from an ecological standpoint. Papua New Guinea’s forests are vital in carbon sequestration and in the maintenance and functioning of ecological processes through watershed protection, soil stabilization, and water filtration (Yosi et al. 2011; Shearman et al. 2008). Importantly, they also provide foundational support for the country’s economy and society through subsistence food and building materials for local communities (Ningal et al. 2008; Shearman et al. 2008). This subsistence culture has sustained the population in PNG for 40,000 years (Shearman et al. 2008). In fact, 80% or more of the population of PNG directly depends upon the natural environment for survival (Ningal et al. 2008; Shearman et al. 2008; NSO 2000). Forests provide 500 different species of food plants growing in primary and secondary forests (Shearman et al. 2008).



### *History of land use*

One of the most interesting aspects of Papua New Guinean culture is the long history of clan and communal ownership of land. This country is one of the few places in the world where clans and families still claim territorial rights to the land (Laurance et al. 2011). Clans claim about 97% of PNG's land area (Laurance et al. 2011; Ningal et al. 2008). Many villagers have, however, sold their land and timber rights in return for the short-term income needed for school fees or health services and other expenses (Laurance et al. 2011). Environmentally sustainable management success depends upon the organization and planning of local communities (Colchester 1994). Protecting the community's management over their land, while maintaining environmentally sustainable land use requires the development of an effective political system that allows local communities the opportunity to participate in the decision-making process regarding land use (Colchester 1994). In other areas of the world, such as in Sarawak (a Malaysian state in Borneo), the timber trade has become corrupted and because this trade largely influences the economy, has aided in corrupting the government to make decisions in the favor of large commercial harvesting enterprises (Colchester 1994). This may not be unlike that which is occurring in PNG (Marshall 1990). Despite customary land rights in PNG, many communities have negotiated rights to timber and land in exchange for short-term money income for schools and health services, but also for royalties (Colchester 1994). Still, many commercial harvesting companies have been able to find loopholes in the law and have exploited the land rights of many communities in order to obtain timber resources (Colchester 1994).

In the 1990s, in response to unfair exchange of services for sale of clan rights, changes were made to PNG's national laws. Clans must now form Incorporated Land Groups (Laurance et al. 2011). Additionally, these laws left final approval for large timber projects to an independent body known as the National Forest Board (Laurance et al. 2011). Papua New Guinea's timber industry is not unlike much of the rest of the world in that it is globalized (Laurance et al. 2011). About 60% of the timber is exported by Rimbunan Hijau, a Malaysian timber company and shipped as logs to Chinese manufacturers and then exported around the world (Laurance et al. 2011). This is of special concern because it limits the employment of PNG communities during the harvest

(Laurance et al. 2011). In fact, this company also owns many PNG businesses, including *The National*, one of only two national newspapers in the country (Laurance et al. 2011). Unfortunately, this makes for an easily corruptible system of timber harvest for local communities. Global aid has attempted to reform this system, but as of yet no positive results have impacted the corruption (Laurance et al. 2011). A study by Bryan et al. (2010) found that in 2001 41% of PNG's carbon emissions could be attributed to deforestation and degradation having risen 20% since 1972.

### *Impacts of logging*

The Food and Agriculture Organization of the United Nations (FAO) estimates that in Southeast Asia a large proportion of deforestation is due to logging (FAO 2000). The FAO (2000) listed the mean regional deforestation rate for Southeast Asia and Oceania between the years of 1981-1990 to be about 1.40-1.43% per year. Land use change and forestry have been estimated to account for about 17-18% of total global emissions (Filer et al. 2009). For PNG between 1972 and 2002 approximately 15% of its rainforests had been cleared and an additional 8.8% had been degraded to secondary forest (Shearman et al. 2008). The primary driver of this degradation and deforestation was logging (Shearman et al. 2008).

Commercial-scale logging practices often involve the felling of selected trees based upon species of interest and diameter-breast-height (dbh) size, which legally must be greater than 50cm above the buttress (Shearman et al. 2008). The bole of the tree is cut into logs and these are dragged out of the forest and placed on logging trucks or on barges (Shearman et al. 2008). A critical problem, which may be overlooked by advocates for large-scale logging operations, is the damage and destruction to the environment as a result of the dragging of logs from the site of harvest to the transport mechanism (Jennings et al. 2001; Cannon et al. 1998). The canopy cover of a forest can be reduced by 50% with only a removal of 3.3% of the trees in a stand (Johns 1988; Bawa and Seidler 1998). "Logged forests have seen effects, such as the decline in mean stand tree height, increased abundance of small-diameter species, and decreases in commercial species" (Bawa and Seidler 1998). Additionally, in the case of transport by barge, often a loading zone is needed from the forest edge to the ship, creating unsightly

destroyed areas with large mounds of cleared earth and defoliation. For example, when a tree is felled there is a substantial amount of residual damage because the area around the tree cleared created by the bulldozer or machinery needed to drag the tree from its area of harvest can be up to 4m wide (Shearman et al. 2008). However, this type of removal is still considered ‘selective harvest’ in commercial operations.

Commercial-scale logging increases susceptibility to invasion of alien species, wildfire, high winds, and landslides (Yosi et al. 2011; Shearman et al. 2008; Turner and Corlett 1996). Additionally, fast-growing species are often restricted to forest edges may be able to move inward at the sites of harvest because of residual gaps, landslides, or exposure due to the creation of logging roads, outcompeting the slower-growing interior forest species (Shearman et al. 2008). Studies have also shown a disproportionately high percentage of the tree volume felled not being utilized in the logging mills, which creates a high amount of wasted felled trees (Bertault and Sist 1997). Bertault and Sist (1997) found that only 53.7% of the volume felled during their study reached the logging mill. Even when there is a minimal amount of timber extraction with this method, the risk to the environment of adverse ecological problems, such as fire, is greatly increased for the area of harvest (Shearman et al. 2008).

### *Lowland rainforests*

Lowland rainforests (<1000 m elevation) comprise about 57% of the forest cover in PNG. This forest type contains a vast majority of the country’s commercial timber species (Shearman et al. 2008). The Morobe Province (in which Siboma, the research site for this project is located) contains 2,096,544 hectares of rainforest, 110,129 hectares of which are degraded (Shearman et al. 2008). The lowland tropical forests of New Guinea are among the world’s ten most unique forested regions (Olsen and Dinerstein 1998; Brooks et al. 2006; Bryant et al. 1997; Shearman et al. 2008). Shearman et al. (2008) found that lowland rainforests have seen a disproportionate amount of change due to harvest, because they are the most accessible to logging operations as compared to higher elevation forests, and contain a large number of commercial timber species (Su Mon et al. 2012; Yosi et al. 2011; Shearman et al. 2008; Sist and Gourlet-Fleury 2003; Sist and Nguyen-The 2002). Furthermore, rising sea levels as a result of climate change and the

increasing loss of lowland forests to logging and development may lead to increased pressure on other habitats, such as montane and upper elevation forests, spreading the impacts of forest degradation (Gillespie et al. 2008). The specific community and stand effects of logging in forest communities in PNG are beginning to be studied (Su Mon et al. 2012; Yosi et al. 2011). Studies of the effects of short-term, repeated logging and long-term recovery after logging, however, are lacking in this part of the world. Therefore, studies on the impact of logging to forest communities in PNG are needed and will likely provide critical information on the health of PNG's forest communities and biodiversity, and help with management and conservation decisions.

## CHAPTER 2. STUDY

### INTRODUCTION

New Guinea's location with relation to the equator, its complex geomorphological history, and its vast array of ecosystems contribute to a high degree of endemism and biodiversity on this island (Marshall and Beehler 2007). Marshall and Beehler (2007) conducted an analysis in 2000 showing that the island of New Guinea had about 657,000 square km of forested land, about 82% of the total area. Because of these large relatively untouched areas, PNG's forests are vital in carbon sequestration and in the maintenance and functioning of ecological processes through watershed protection, soil stability, and water filtration (Shearman and Bryan 2011; Shearman et al. 2008; Hunt 2006). These areas also provide foundational support for the country's economy and society through subsistence food and building materials for local communities (Shearman and Bryan 2011; Shearman et al. 2008). This subsistence culture has sustained the population in PNG for 40,000 years (Shearman et al. 2008). In fact, 80% or more of the population of PNG directly depends upon the natural environment for survival (Ningal et al. 2008; Shearman et al. 2008; NSO 2000). Forests provide 500 different species of food plants growing in primary and secondary forests (Shearman et al. 2008). Sustaining these resources will require management of the growing disturbance driven by harvesting and land use change (Banner and LePage 2008). The preservation and conservation of ecosystems in PNG is therefore extremely critical both locally and globally.

The lowland tropical forests of New Guinea are among the world's ten most unique forest regions (Olsen and Dinerstein 1998; Brooks et al. 2006; Bryant et al. 1997; Shearman et al. 2008) and this forest type contains a vast majority of the country's commercial timber source (Shearman et al. 2008). Lowland rainforests (<1000m elevation) comprise about 57% of the forest cover in PNG. Shearman et al. (2008) found that lowland rainforests in Papua New Guinea, however, have seen a disproportionate amount of change, with 6.8 million hectares being degraded or cleared by 2002. Lowland forests are susceptible because they are most accessible to logging operations, and contain a large number of commercial timber species (Su Mon et al. 2012; Shearman et

al. 2008; Sist and Gourlet-Fleury 2003; Sist and Nguyen-The 2002). Between 1972 and 2002 approximately 15% of PNG's rainforests had been cleared and an additional 8.8% had been degraded to secondary forest (Bryan et al. 2010; Shearman et al. 2008). For example, the Morobe Province, where the present study was conducted, contains 2,096,544 hectares of rainforest, 110,129 hectares of which are degraded (Shearman et al. 2008). From 1972 to 2002 the number one driver of forest degradation and deforestation was logging, accounting for 48.2% of net forest change (Shearman et al. 2008). However, the specific effects of logging in forest communities in PNG have not been well studied. The impacts of logging depend upon both the planning that goes into the process prior to extraction and the number of stems removed per stand (Bawa and Seidler 1998).

Papua New Guinea's timber industry is not unlike that of much of the rest of the world in that it is globalized (Laurance et al. 2011). About 60% of harvested timber is exported by Rimbunan Hijau, a Malaysian timber company, and shipped as logs to Chinese manufacturers and then exported around the world (Laurance et al. 2011). In Southeast Asia generally a disproportionate amount of the rate of deforestation is due to logging (FAO 2000). Furthermore, the contribution of greenhouse gas emissions due to logging and deforestation is one of the poorly understood pieces of climate change models to date (Bryan et al. 2010). The FAO lists the mean regional deforestation rate for Southeast Asia between the years of 1981-1990 as between 1.40-1.43% per year (FAO 2000). Logging has continued to rise since the 1990s (Bryan et al. 2010; Shearman et al. 2008). Commercial scale logging practices often involve the felling of selected trees based upon species of interest and dbh (basal) size, which legally must be greater than 50 cm above the buttress (Shearman et al. 2008). The impacts of this scale of logging include increased susceptibility to wildfire, vulnerability to the invasion of weeds, and increased susceptibility to high winds and landslides. Shade-dependent understory species are negatively impacted via the removal of large canopy species, and epiphytes and other organisms that depend on these canopy trees may be adversely impacted (Shearman et al. 2008). The effects of selective logging are of great concern and the impacts are not well known, unlike those of clear cutting and more severe anthropogenic disturbances (Brown and Gurevitch 2004; Verburg and van Eijk-Bos 2003; Cannon et al. 1998). Brown and Gurevitch (2004) found that even after 50 years post-selective harvest,

logging decreases the diversity of trees as described by Fisher's alpha and the Shannon index. Additionally, both clear cutting and selective harvest increased the community's susceptibility to invasion of weeds and non-native plants (Brown and Gurevitch 2004). Hall et al. (2003) suggest from their study that the increased light resulting from logging operations favors the establishment and growth of pioneer and non-pioneer light demanding species. Differences in species composition between size classes post-harvest may occur as a result of disturbance (Verburg and van Eijk-Bos 2003). Logging impacts the community composition, for example, in that fast-growing species that are often along forest edges may be able to move inward from the sites of harvest because of the tree-fall gaps, landslides, or exposure due to the creation of logging roads, outcompeting the slower-growing interior forest species (Shearman et al. 2008; Bawa and Seidler 1998). Cannon et al. (1998) found that logging removed 62% of the commercially viable dipterocarp basal area. Additionally, after one year of post-harvest in a Bornean rainforest, 45% of the lowland forest's canopy was open and dominated by low pioneer vegetation, which included roads and skid trails (Cannon et al. 1998). Hall et al. (2003) identified timber extraction as having an impact on forest structure for both saplings and tree densities 18 years post harvest, and basal area being 19% lower in logged forests as compared to those of unlogged. Studies of both the effects of short-term repeated logging and long-term recovery after logging are lacking in this part of the world. Therefore, studies on the impact of logging to forest communities in PNG are needed and will likely provide critical information on the health of PNG's forest communities and biodiversity.

The overall goal of this project was to assess the effects of logging on plant diversity in the Morobe Province, PNG. The supporting objectives and hypotheses tested were:

1. Quantify the generic richness and diversity of trees for logged and non-logged (primary) forest communities,
  - a. Hypothesis 1: Richness of trees will be lower on the previously logged site than the non-logged forest:
    - i.  $H_a: m_{\text{non-logged}} > m_{\text{logged}}$  (Null Hypothesis:  $H_0: m_{\text{non-logged}} = m_{\text{logged}}$ )

- b. Hypothesis 2: Diversity of trees will be lower on the logged site than the non-logged site:
  - i.  $H_a: m_{\text{non-logged}} > m_{\text{logged}}$  (Null Hypothesis  $H_0: m_{\text{non-logged}} = m_{\text{logged}}$ )
- c. Hypothesis 3: Desired commercial timber species will be less prevalent on the logged site than the non-logged site:
  - i.  $H_a: m_{\text{non-logged}} > m_{\text{logged}}$  (Null Hypothesis  $H_0: m_{\text{non-logged}} = m_{\text{logged}}$ )
- 2. Quantify forest stature and basal area in terms of diameter at breast height (dbh),
  - a. Hypothesis 4: The forest stature will be different in the non-logged site from the site that has been logged:
    - i.  $H_a: m_{\text{non-logged}} \neq m_{\text{logged}}$  (Null Hypothesis:  $H_0: m_{\text{non-logged}} = m_{\text{logged}}$ )
- 3. Quantify canopy cover for the two sites
  - a. Hypothesis 5: The canopy cover density of the non-logged site will be greater than the previously logged site
    - i.  $H_a: m_{\text{non-logged}} > m_{\text{logged}}$  (Null Hypothesis:  $H_0: m_{\text{non-logged}} = m_{\text{logged}}$ )

## METHODS

### *Study site*

The village of Siboma, known locally as Numbami, is located on the Huon Coast, south of Lasanga Island in the Morobe Province, PNG. Coastal lowland rainforests form the dominant vegetation type within the village territory. The elevation of this village territory reaches to about 800 m. Approximately 50 years ago, forests within the village's territory were contracted to South Pacific Timber for harvesting of logs. However, there are also untouched stands of forest within Siboma. As a result, this is an interesting area for comparative ecological assessment, e.g., understanding the impacts of logging on the ability of forests to recover from logging activities. The first study site (7.52361°S, 147.301°E) was the non-logged (primary) forest site. This site was 0.74 miles to the south of the village proper. Ten plots (numbered 1-10) were delineated alongside the ridge of the mountain. Plots were placed perpendicularly and parallel to the ridgeline. The second site (7.51611°S, 147.3289°E) was logged in the early 1960s and 2.08 miles west of the village. Plots 11-20 were placed on either side of a logging road that ran along the ridge,



both perpendicular and parallel to the road at similar elevations as the first (primary forest) site.

#### *Data collection and computation*

To quantify the richness and diversity of trees, I used a rectangular plot design (Gibson 2002; personal communication Tamara Ticktin). I placed 10 (10m x 50m) rectangular plots approximately 100m apart along the ridgeline at each site (previously logged forest and non-logged primary forest) for a total of 20 plots for the study. Each corner of the plots was marked and GPS coordinates recorded. All of the trees greater than 10cm dbh within plots were identified to genus and measured for dbh using a dbh tape (Verburg and van Eijk-Bos 2003). Identification of trees was possible with the help of a field technician from the Papua New Guinea Forestry Research Institute. Identification to species was not possible during the course of the study due to lack of flowering/fruitletting and size of the individuals. The trees were sequentially marked with aluminum tree tags. If buttresses were present, diameter measurement was taken above the buttress where the bole straightened, as seen in Webb (1997), Bertault and Sist (1997), and Sist and Nguyen-The (2002). For individuals with multiple trunks, the equation:  $\sqrt{(DBH_1^2 + DBH_2^2 + DBH_3^2)}$  was utilized for the diameter for the individual tree (as described by usda.gov) Canopy cover was estimated using a densiometer at four random points within each plot, where one canopy density measurement was taken for each of the four compass directions. The four canopy data points were averaged for each plot producing an average percentage of open canopy. The average percent open canopy value was subtracted from 100 for each of the plots to give an average percent canopy cover. This was done for each of the 20 plots, creating a total of 20 percent canopy cover values for the study.

The number of genera and number of stems were counted for each plot. The Shannon-Wiener Index ( $H'$ ) (Brown and Gurevitch 2004; Hall et al. 2003; Sagar et al. 2003) and Simpson's Index ( $D$ ) (Sagar et al. 2003) were calculated as estimates of the diversity of genera for the two sites. The equations were as follows:

$$H' = -\sum(p_i)(\ln p_i)$$

where  $p_i$  = proportion of individuals belonging to the  $i$ th genus

$$D = \sum(n/N)^2$$

where  $n$  = number of entities belonging to the  $i$ th genus and  $N$  = the total number of entities

The basal area (BA) for each tree was calculated using the dbh (cm) for each individual tree. The basal area per individual was computed using the formula described by Hedl et al. (2009):

$$BA_i = (\pi(d/2)^2) \times 10^{-4},$$

where the diameter ( $d$ ) of an individual tree measured in cm is converted to area in  $m^2$ .

#### *Data analysis*

For the first and third objectives, two-sample t-tests were used to identify differences between the two sites. The set of dbh values for each site was not normally distributed and there were outliers present; therefore, nonparametric Wilcoxon rank-sum tests were used for the analysis of differences between sites for dbh and basal area.

I also used post hoc chi-square contingency table analyses and non-metric multidimensional scaling (NMS) ordinations using the Sørensen (Bray-Curtis) distance measure and random starting configurations to identify the relationship between the genera present as well as the basal area within plots of the two sites (Banner and LePage 2008). Non-metric multidimensional scaling ordinations are useful tools for ecological studies because of their practicality with data that are not normally distributed, such as basal area within a stand (McCune and Grace 2002). Non-metric multidimensional scaling ordinations create a visual representation of the similarities and dissimilarities in the data by ranking each data point and assigning it a location within an N-dimensional space (McCune and Grace 2002). This type of ordination avoids the assumptions of a linear relationship among variables, but rather utilizes ranked distances to understand the dimensionality of the data set (McCune and Grace 2002). The program PC-ORD (Banner

and LePage 2008) was used to perform this ordination with a primary matrix consisting of presence/absence scores for each genus within each plot (presence=1; absence=0), and a secondary matrix consisting of the categorical identity of site (non-logged or logged) for each of the 20 plots. The ordination results were then plotted on two-dimensional and three-dimensional graphs (Banner and LePage 2008). A chi-square contingency table analysis was used to assess if generic composition varied between the two forest sites.

## RESULTS

### *Generic richness and diversity*

The inventory and analysis of all trees >10cm dbh within a primary forest and a 50 year post-harvest forest identified no significant difference in the generic richness between forests, supporting the null hypothesis that the generic richness in the logged forest was no different than that of the primary forest (Fig. 1; Table 1). The frequencies of genera by site are shown in Figures 2a-d with a number of genera only present within one site. A post hoc chi-square contingency table analysis of the dependency of generic composition on forest type showed a significant dependence, i.e., there were differences in the generic composition of the two sites (Table 2). Thirty-six genera occurred at both sites, whereas 18 genera were found only at the non-logged sites and 15 genera were unique to the previously logged, secondary forest site.

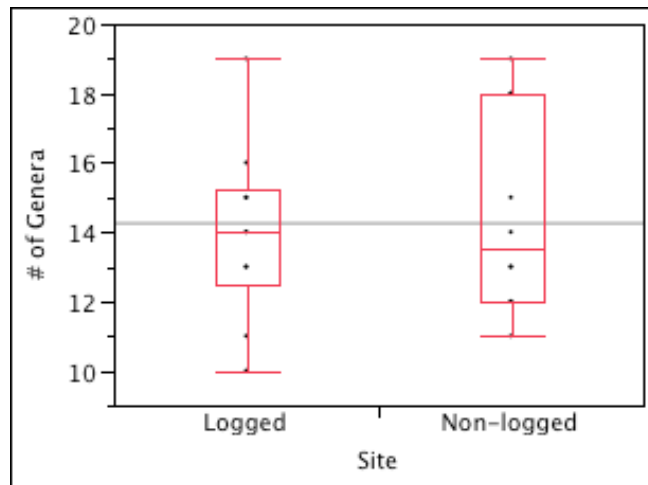


Figure 1. Number of genera per plot by site ( $t_{(17.73)} = 0.412082$ ,  $p=0.3426$ ).

Table 1. Descriptive information for student t-test for number of genera

Level	Number	Mean	Std Err Mean
Logged	10	14.0000	0.80277
Non-logged	10	14.5000	0.90982

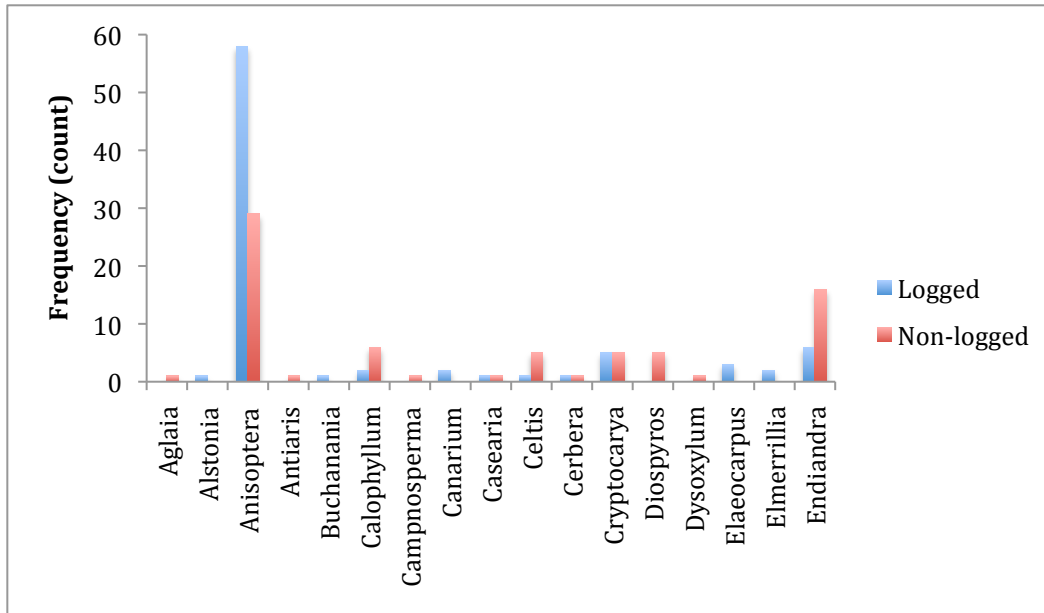


Figure 2a. Frequency of genera by site (part 1)

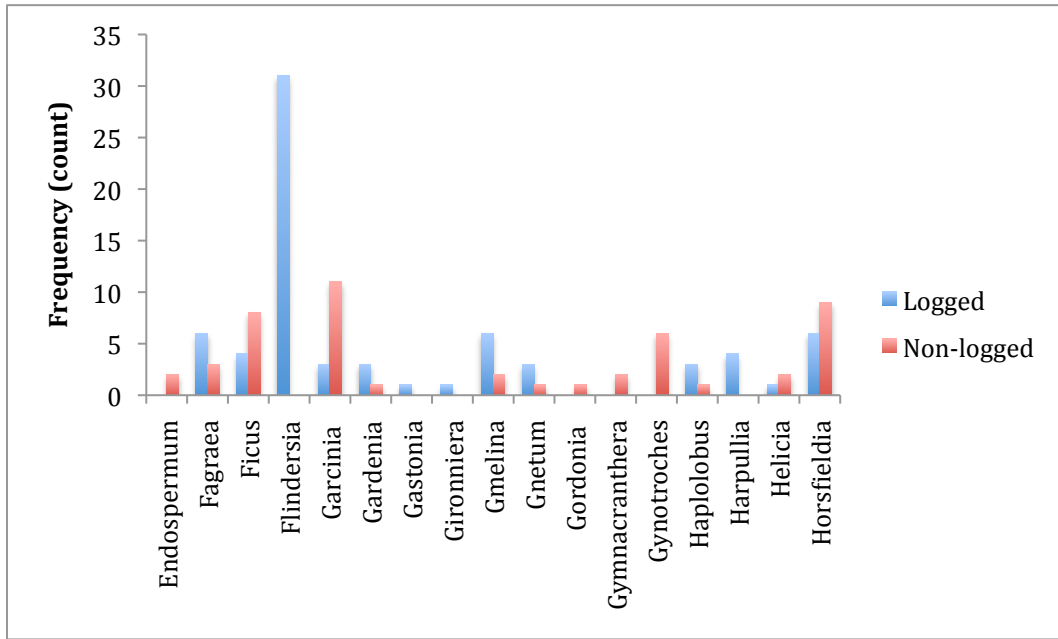


Figure 2b. Frequency of genera by site (part 2)

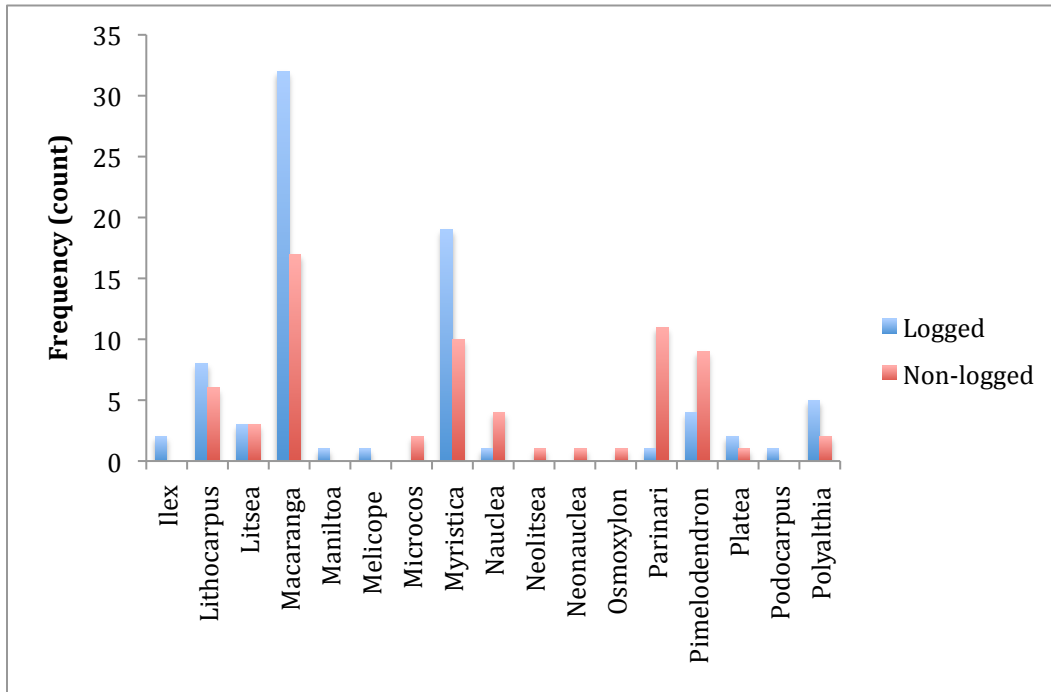


Figure 2c. Frequency of genera by site (part 3)

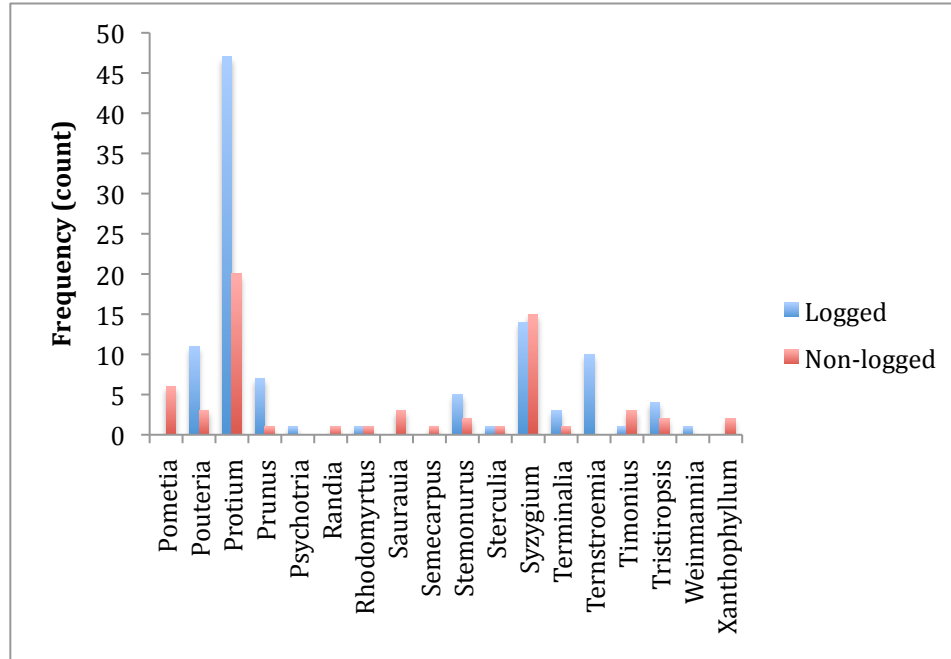


Figure 2d. Frequency of genera by site (part 4)

Table 2. Contingency Analysis of Genus By Site

N	DF	-LogLikelihood	RSquare (U)
592	68	103.81819	0.0512

Test	ChiSquare	Prob>ChiSq
Likelihood Ratio	207.636	<0.0001*
Pearson	168.207	<0.0001*

For the diversity of the two sites, defined in this study as the generic richness in relation to the generic evenness, both the Shannon-Weiner and Simpson's diversity indices were significantly different between the two sites, therefore, rejecting the null hypothesis that there was no significant difference in the diversity of trees between sites (Figs. 3, 4; Tables 3, 4). Values of both indices were greater for the non-logged site than the logged. The NMS ordination provided a three-dimensional graphic depiction of the relationship between presence of each genus and site location, distinguishing the plots by site (Fig. 5). Figure 5 shows a separation of plots by site with a majority of the non-

logged plots falling to the negative end of Axis 1, and a majority of the logged plots at the positive end. The amount of variation accounted for by each of the axes is shown in Table 5.

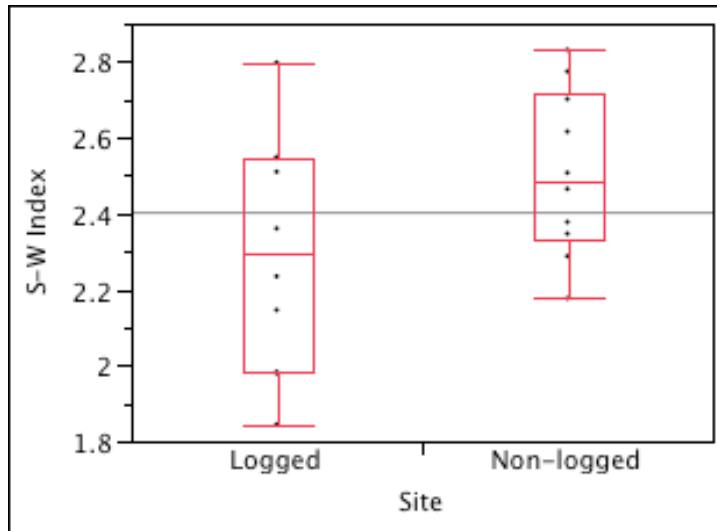


Figure 3. S-W Index values per plot by site ( $t_{(16,196)}= 1.787066$ ,  $p=0.046$ )

Table 3. Means and Std Deviations of S-W Index

Level	Number	Mean	Std Err Mean
Logged	10	2.29465	0.09754
Non-logged	10	2.50811	0.06894

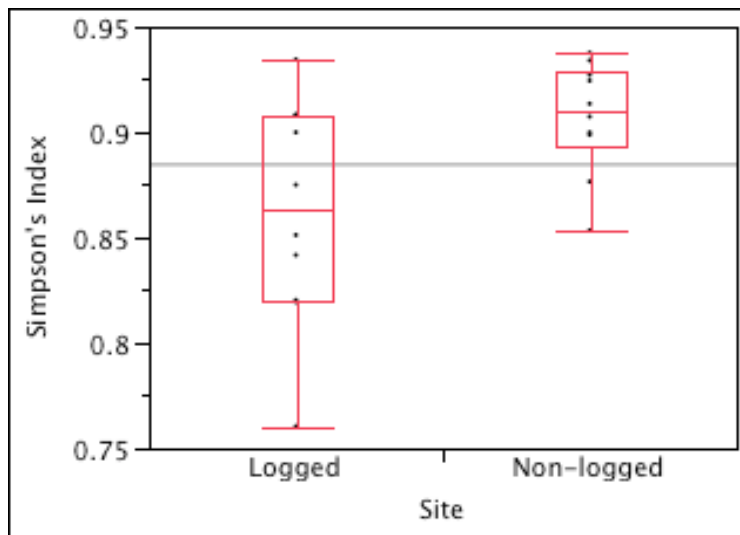


Figure 4. Simpson's Index values per plot by site ( $t_{(13,215)}= 2.1412$ ,  $p=0.0155$ )

Table 4. Means and Std Deviations of Simpson's Index

Level	Number	Mean	Std Err Mean
Logged	10	0.861876	0.01684
Non-logged	10	0.907261	0.00840

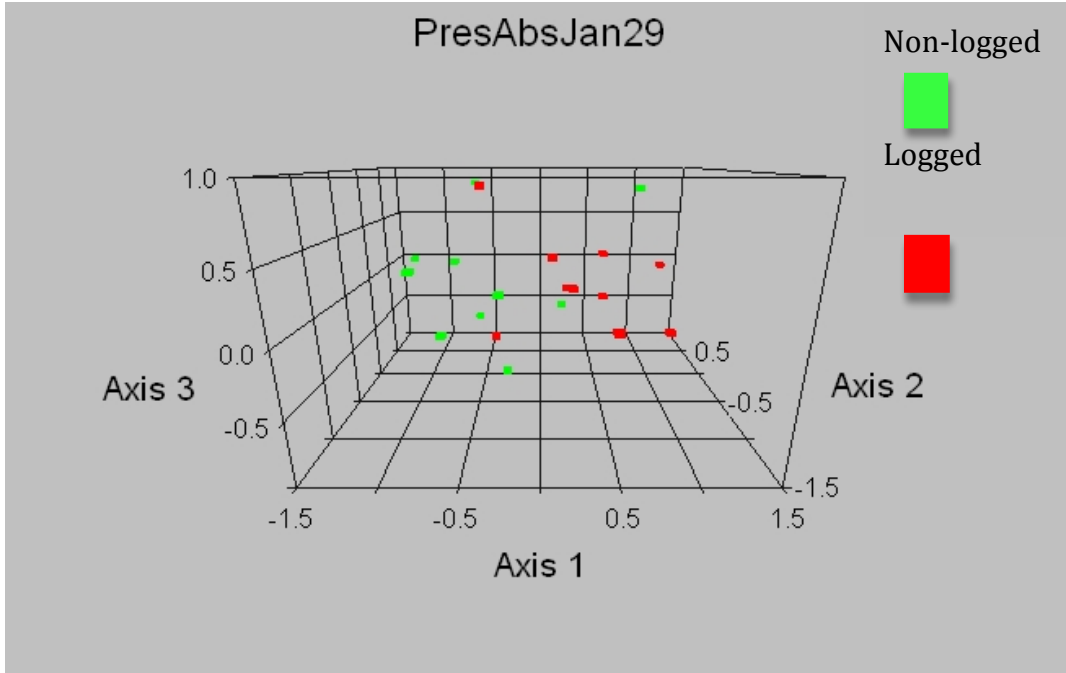


Figure 5. Three-dimensional depiction of NMS ordination of presence/absence data among plots.

Table 5. Amount of variation accounted for by each of the axes of the NMS ordination

Stress in relation to dimensionality (number of axes)							
Stress in real data 50 run(s)				Stress in randomized data Monte Carlo test, 50 runs			
Axes	Minimum	Mean	Maximum	Minimum	Mean	Maximum	p
1	32.484	47.316	54.708	36.279	48.934	54.772	0.0196
2	19.15	20.877	37.316	18.338	24.571	29.835	0.0392
3	12.545	13.414	23.527	11.687	15.381	26.58	0.0392
4	9.05	9.387	10.693	8.12	10.56	12.756	0.098



*Stand structure (dbh and basal area)*

Both the post-harvest forest and the primary forest exhibited distributions of dbh values that were skewed with a long right tail with many smaller diameter trees and few larger diameter trees. However, a nonparametric rank-sum test of median dbh (cm) was significantly different between the two sites, therefore, rejecting the null hypothesis that the dbh is the same between sites (Fig. 6; Table 6). The rank-sum median dbh was greater for the non-logged site showing that on average the dbh values of trees in the primary forest are greater than those of the previously logged forest. Furthermore, the distributions of values for basal area of trees within each site were also skewed with a long right tail. The nonparametric rank-sum test of the median basal area (m<sup>2</sup>) was significantly different between the two sites, therefore, rejecting the null hypothesis that the basal area is the same between sites (Fig. 7; Table 7). The median basal area was greater for the non-logged site providing a similar result to the dbh that the average basal area of trees within the primary forest is greater than that of the previously logged forest.

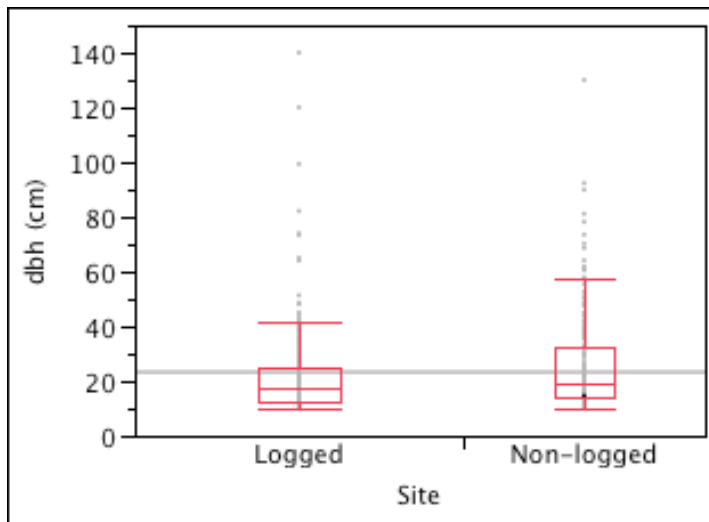


Figure 6. Summary of dbh (cm) values by site ( $S=80164.5$ ,  $p=0.0052$ ).

Table 6. Wilcoxon and Kruskal-Wallis Tests (Rank Sums) for dbh

Level	Count	Score Sum	Expected Score
Logged	341	95363.5	101107
Non-logged	251	80164.5	74421.5

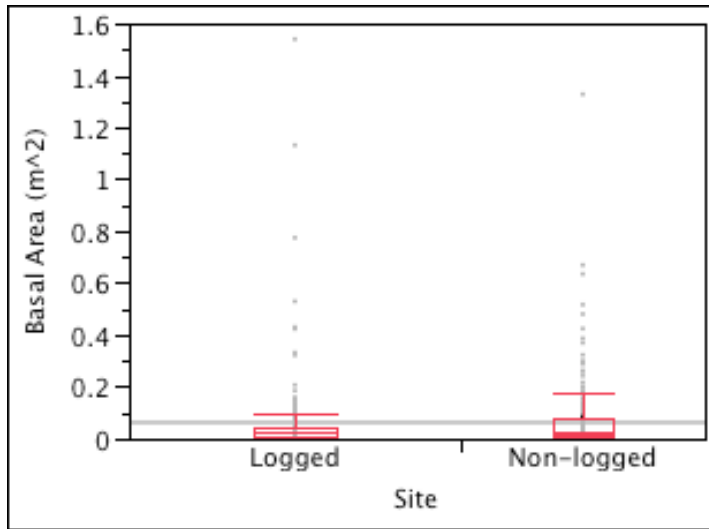


Figure 7. Summary of basal area values per plot by site (S=80163, p=0.0052)

Table 7. Wilcoxon and Kruskal-Wallis Tests (Rank Sums) for basal area

Level	Count	Score Sum	Expected Score
Logged	341	95365.0	101107
Non-logged	251	80163.0	74421.5

#### *Stand structure- timber value*

The data collected for this study also included a descriptive categorization of each tree genus based upon the timber value. In Figure 8, which shows the frequency of each category separated by timber value for each site, “Yes” represents trees that are desirable commercial timber genera, “Yes, sometimes” are tree genera that are sometimes taken, but may depend on the use as well as the particular species, and “No” are genera that are not desirable for timber. Notably, there was a greater number of trees in all three categories found in the logged site as compared to the primary forest (Figure 8).

However, when converted to average basal area per timber category, there are no overall significant site differences (Figure 9, Table 8). Though there does appear to be a trend within the “Yes” category that suggests the non-logged site as having a greater basal area than that of the previously logged site as shown by the non-overlapping error bars (Figure

9).

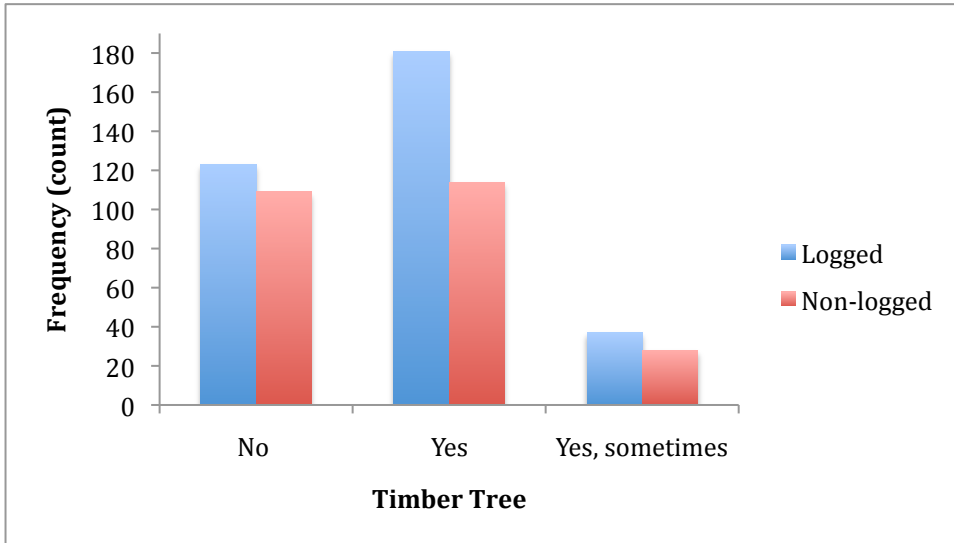


Figure 8. Summary of frequency of timber trees by site mean for site

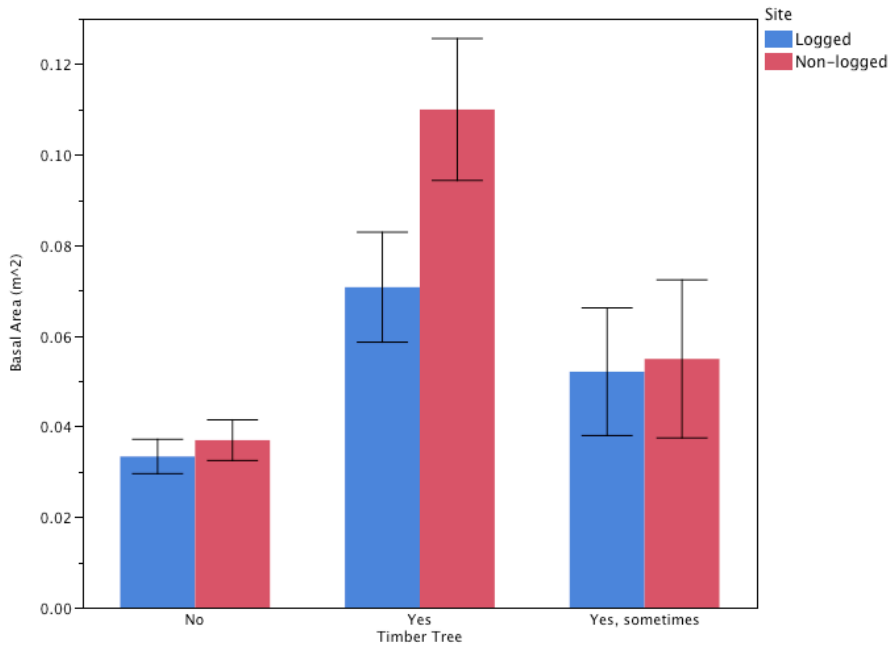


Figure 9. Mean and standard error of trees by timber use for each site.

Table 8. Contingency Analysis of Timber Tree By Site

N	DF	-LogLikelihood	RSquare (U)
592	2	1.8552122	0.0033

Test	ChiSquare	Prob>ChiSq
Likelihood Ratio	3.710	0.1564
Pearson	3.711	0.1564

*Canopy cover*

The amount of open canopy as a result of anthropogenic disturbance is also a critical concern and we know little about the long-term impacts of increases in open canopy. The canopy density differed significantly between the two sites (Fig. 10; Table 9) with the non-logged forest showing significantly greater canopy cover than the logged forest. However, the difference between an average of 93% closure and 95% closure in the field may be negligible for forest communities 50 years post harvest. The difference in canopy disturbance may be more important in the initial years after logging rather than in the long term.

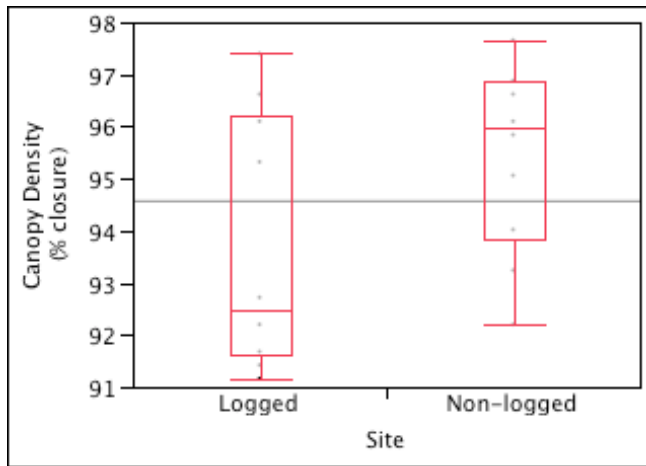


Figure 10. Two sample t test summary of canopy density values per plot by site ( $t_{(16,612)} = 1.871166, p=0.0395$ ).

Table 9. Means and Std Deviations of canopy closure

Level	Number	Mean	Std Err Mean
Logged	10	93.6820	0.75857
Non-logged	10	95.4500	0.56333

## DISCUSSION

A primary concern of the effects of logging is the impact of harvest on the community composition and diversity of a forest. The S-W Index and Simpson Index in this study describe a marked significant difference between the diversity (whereas richness is simply a count of genera, diversity takes into account the evenness or abundances of each genus) of the logged and non-logged site. The non-logged, primary lowland forest had greater diversity index values for both diversity indices. Therefore, logging negatively impacted the diversity of tree genera, resulting in lower diversity of logged sites even after 50 years. Previous studies in both the tropics and temperate regions have, similarly, found that logging can negatively impact the diversity of a forest. Clark and Covey (2012) found that because many logging operations are “unplanned and fail to incorporate the regeneration ecology” they can severely alter the ecosystem, including among other impacts, decrease in species diversity of late succession tree species, which are a defining feature of old growth and primary forests.

As can be seen from the frequency of genera by site, there is a difference in the actual genera found at each site. Some genera are only found in the non-logged forest, whereas others are found only in the logged site (Figure 2a and 2b). The two sites may provide different ecological conditions, each of which are differentially conducive to the survival of different suites of genera or the different generic compositions represent stochastic processes of community assembly post-logging, or both may be occurring.

Another primary concern for researchers studying previously logged forests is the change in basal area of a stand. With the removal of large canopy trees for lumber there is a concern about the length of time required for canopy trees to regenerate post-harvest. Though the last harvest for this logged stand was in the early 1960s, the mean dbh and basal area of trees were significantly less at the logged site than in the non-logged forest,

suggesting that insufficient time has passed since logging to allow trees to reach the size of those observed in the primary forests. This pattern has previously been found, in which stem density and basal area decrease with increasing disturbance intensity (Sager et al. 2003; Ramirez-Marcial et al. 2001; Smiet 1992).

Furthermore, the results for canopy cover were consistent with the suggestion that logging negatively impacts the average amount of canopy cover in a stand. The amount of canopy cover (and resultant gap disturbance) has been found in other studies to be negatively impacted (via reducing canopy closure) by logging at varying intensities (Bawa and Seidler 1998; Webb 1997; Felton et al. 2006; Johns et al. 1996). Furthermore, the creation of gaps in canopies alters the competition for light, as well as carbon and other nutrient cycling in the soil below the canopy (Arihafa and Mack 2012). Logging gaps are often larger and have a greater impact than natural tree fall gaps; logging gaps therefore allow for the better establishment of invasive species (Arihafa and Mack 2012; Webb 1997). Although there is a statistically significantly less amount of canopy cover in the previously logged forest for this study, in the field there is, realistically, a negligible amount of difference between 92 and 95% canopy cover. Therefore, gap disturbance resulting from logging may be more important for initial colonization and regeneration in the short-term post-harvest.

Though the previously logged forest has grown considerably since harvesting in the 1960s, the results of this study suggest that the time needed for recovery will impact the amount of time needed between harvesting cycles (Sist et al. 2003). Johns et al. (1996) found that uncontrolled logging with a lack of preplanning may require 75-100 years for recovery prior to secondary harvest. Sist et al. (2003) found that the time of recovery for a “pristine pioneer population...even under low harvesting intensities [is] at least 90 years”. Sist et al. (2003) also suggested that when there are frequent harvesting operations in one area (i.e., 40-year intervals), these stands risk a change towards a species composition different from that of the original primary forest. The results of the present study are consistent with that information, in that after approximately 60 years post-harvest the diversity of the previously logged site at Siboma is still considerably different from the primary, non-logged forest, and basal area of the previously logged forest is still significantly less than the primary forest. However, because there are no true

baseline diversity data for the logged area (prior to harvest), the differences in sites may be a result of inherent differences in the sites, as a result of logging, or both. Furthermore, the influence of logging on community composition is dependent upon the nature of the logging practices and the intensity of the harvest (Seng et al. 2004; Webb 1997). There are current efforts, such as the creation of “natural-disturbance-based management”, to create sustainable harvesting cycles while also decreasing the risk of community compositional and diversity changes that may result from logging (Tittler et al. 2012). Additionally, concentrating logging areas may aid in the reduction of fragmentation at the landscape level (Tittler et al. 2012). The concentration of logging areas may aid in the reduction of fragmentation on the landscape level and preservation of stands that include important timber species for future sustainable harvest (Tittler et al. 2012). Because the current study identified differences in the diversity and stand structure (via basal area) between the two sites, my recommendation to the village of Siboma would be for any future commercial harvest contracted to be concentrated in the area already previously disturbed by logging.

## CONCLUSION

Few studies have examined the long-term impacts of logging, especially in the primary forests of PNG, which is considered “one of the last great unknowns” (Marshall and Beehler 2007). The goal of the present study was to assess the impacts of previous harvesting on the tree community composition decades after harvest. The practical and esoteric implications of these results may be that more selective harvesting techniques for future logging operations in these forest types of the Morobe Province of PNG might better preserve long-term diversity. Of course, this would render the need for further study on the appropriate method for the least negative impact on the environment. The primary constraint and consideration in reviewing the results of this study is the lack of replication. Only one replicate of 10 plots per site were utilized for analysis. Therefore, these results are specific to the particular village within this study. Furthermore, due to the length of time since logging, it is important to consider the inherent differences in the two sites apart from harvest influence. Future studies should consider setting up long

term research plots in order to sample and collect data on sites prior to harvest in addition to post-harvest. This research adds to the literature on the long-term impacts of logging within a village community in the Morobe Province and may aid in the direction of future research in the forest communities in Papua New Guinea.



## APPENDICES

*Appendix A- List of genera and their presence (x) or absence from each of the two study sites*

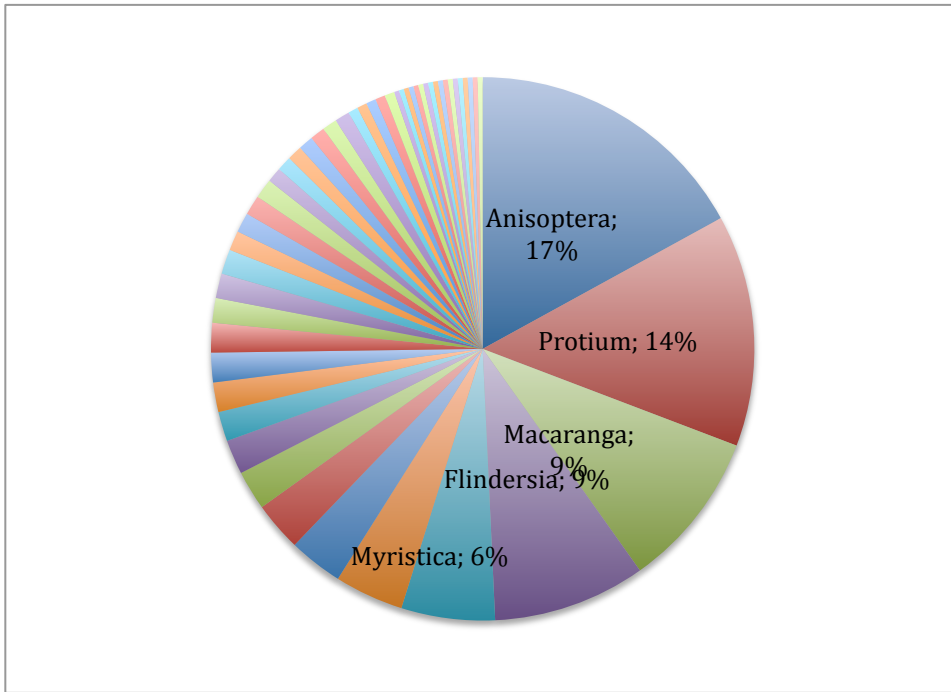
<b>Family</b>	<b>Genus</b>	<b>non-logged</b>	<b>logged</b>
Annonaceae	Polyalthia	x	x
Apocynaceae	Cerbera	x	x
Burseraceae	Haplolobus	x	x
Burseraceae	Protium	x	x
Calophyllaceae	Calophyllum	x	x
Cannabaceae	Celtis	x	x
Chrysobalanaceae	Parinari	x	x
Clusiaceae	Garcinia	x	x
Combretaceae	Terminalia	x	x
Dipterocarpaceae	Anisoptera	x	x
Euphorbiaceae	Macaranga	x	x
Euphorbiaceae	Pimelodendron	x	x
Fagaceae	Lithocarpus	x	x
Salicaceae	Casearia	x	x
Gentianaceae	Fagraea	x	x
Gnetaceae	Gnetum	x	x
Icacinaceae	Platea	x	x
Lamiaceae	Gmelina	x	x
Lauraceae	Cryptocarya	x	x
Lauraceae	Endiandra	x	x
Lauraceae	Litsea	x	x
Moraceae	Ficus	x	x
Myristicaceae	Horsfieldia	x	x
Myristicaceae	Myristica	x	x
Myrtaceae	Rhodomyrtus	x	x
Myrtaceae	Syzygium	x	x
Protaceae	Helicia	x	x
Rosaceae	Prunus	x	x
Rubiaceae	Gardenia	x	x

Rubiaceae	Nauclea	x	x
Rubiaceae	Psychotria	x	x
Rubiaceae	Timonius	x	x
Sapindaceae	Tristiropsis	x	x
Sapotaceae	Pouteria	x	x
Stemonuraceae	Stemonurus	x	x
Malvaceae	Sterculia	x	x
Actinidiaceae	Saurauia	x	
Anacardiaceae	Camptosperma	x	
Araliaceae	Osmoxylon	x	
Ebenaceae	Diospyros	x	
Euphorbiaceae	Endospermum	x	
Anacardiaceae	Semecarpus	x	
Lauraceae	Neolitsea	x	
Meliaceae	Aglaia	x	
Meliaceae	Dysoxylum	x	
Moraceae	Antiaris	x	
Myristicaceae	Gymnacranthera	x	
Polygalaceae	Xanthophyllum	x	
Rhizophoraceae	Gynotroches	x	
Rubiaceae	Neonauclea	x	
Rubiaceae	Randia	x	
Sapindaceae	Pometia	x	
Theaceae	Gordonia	x	
Malvaceae	Microcos	x	
Anacardiaceae	Buchanania		x
Apocynaceae	Alstonia		x
Aquifoliaceae	Ilex		x
Araliaceae	Gastonia		x
Burseraceae	Canarium		x
Cannabaceae	Gironniera		x
Cunoniaceae	Weinmannia		x
Elaeocarpaceae	Elaeocarpus		x
Fabaceae	Maniltoa		x

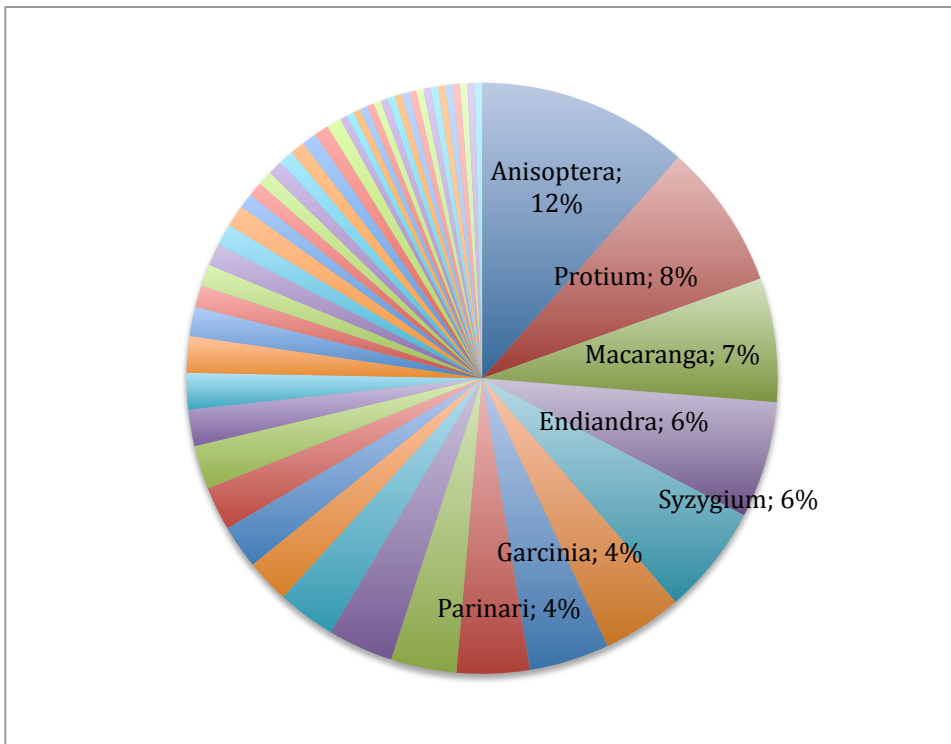
Magnoliaceae	Elmerrillia		x
Podocarpaceae	Podocarpus		x
Rutaceae	Flindersia		x
Rutaceae	Melicope (Euodia)		x
Sapindaceae	Harpullia		x
Pentaphylacaceae	Ternstroemia		x

*Appendix B- Dominant genera found at each site*

*Dominant genera found in logged forest*



*Dominant genera found in non-logged (primary) forest*



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