Clouds to Coast; Assessing, Monitoring and Managing Forest Carbon in Papua New Guinea

Thesis submitted by

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To Timmy Sowang,

teacher, guide and true friend

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Statement of the Contribution of Others

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Abstract

Considerable investment into tropical forest management for carbon sequestration is now demanding an improved understanding of the state of these forests. This includes management options for forest protection or restoration, as well as addressing the needs of forest dependent communities that forego forest exploitation. These needs are particularly acute in the Papua New Guinea, which houses large tracks of relatively intact tropical forests. This thesis aims to address these need by 1) reviewing global carbon stocks and fluxes in tropical forests and providing an assessment of seven forest carbon management practices, 2) examining the relationship between above ground biomass (AGB) and environmental factors through an extensive field campaign in the Morobe province of Papua New Guinea (PNG) along a 3,100m elevation gradient, 3) assessing the potential for engaging local people to monitor forest carbon stocks by evaluating the robustness of data collected by locally-based monitoring programs and 4) exploring mechanisms to incorporate the needs of forest-dependent people into land-use planning for lowered carbon emissions by testing an approach that integrates socio-economic datasets into a more traditional biophysical land-use planning model.

The seven carbon management or 'recarbonization' practices reviewed in Chapter 2 exhibit a large variation in carbon sequestration potential. These potential to sequester carbon was positively associated with levels of land degradation and resource input. Given the distinct co-benefits, risks and costs associated with each practice, the review outlines the potential for government, community, conservation

and industry initiatives to profit from recarbonization strategies. The review summarizes the benefits of incentivizing a variety of recarbonization actions and moving beyond the current focus on forest protection.

Research conducted along a forested elevation gradient in Papua New Guinea, presented in Chapter 3 of this thesis; found that climatic and edaphic variables were poor predictors of AGB. Instead, natural disturbance was the most significant predictor of AGB. From sampling AGB on very steep forest slopes, up to 80° slope, this research demonstrates for the first time that slope angle can be used to predict the occurrence of natural disturbance and in turn, forest biomass. This finding can be used to further improve models that estimate AGB at the landscape scale, especially in montane areas.

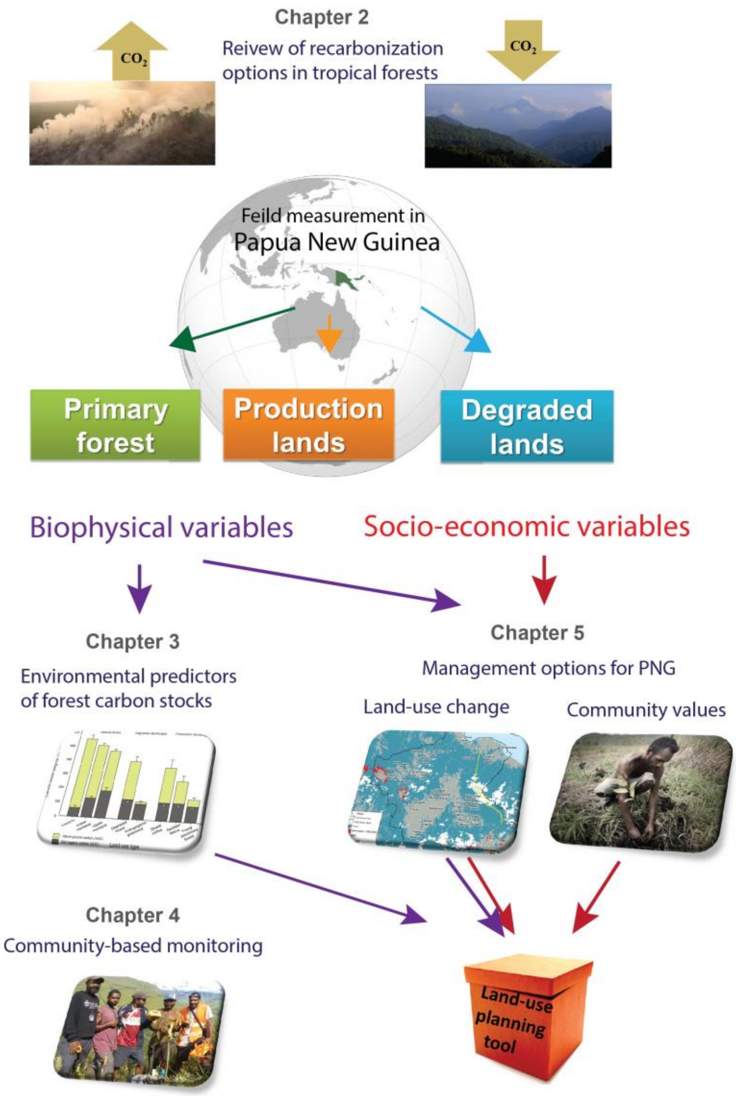
Chapter 3 presents the first field assessment of forest carbon stores in the three main forest types in PNG (Lowland, Montane and Upper-montane) along with secondary grasslands; revealing the highest carbon stocks yet recorded in high altitude forests anywhere in the world. High forest-carbon stocks were best explained by the distribution of a large number of tree species found above 2,200 m asl, which grew to exceptional girth and height. The presence of large trees in high altitude tropical cloud forests is generally uncommon; the large trees in the study coincided with a set of optimal climatic conditions similar to those found in temperate maritime areas which contain the largest trees on Earth. This research challenges the common belief that high altitude tropical forests are stunted and low, with low carbon stocks, and highlights the value of conducting fieldwork in difficult-to-access montane areas.

Involving local people in monitoring forest-carbon stocks could potentially increase monitoring capacity in developing countries, which currently falls short of the requirements by the Intergovernmental Panel on Climate Change (IPCC). Chapter 4 of this thesis assessed the robustness of locally-based monitoring programs by designing a training program that aimed to teach forest-biomass inventory protocols to people with little or no formal education but with remarkable 'traditional' ecological knowledge about their forests. Three communities were involved in the study, and a total of 4,481 'expert' and 'non-expert' measurement pairs of tree diameter, tree height, numbers of trees and plot surface area were compared from 41 sites. The results demonstrate that biomass estimates by experts and non-experts were not statistically different and thus community-based monitoring could be used overcome barriers to reducing forest-carbon emissions in developing countries. The study takes a hierarchical approach to track the types of error in the field that lead to the largest discrepancy in biomass at the landscape scale, and demonstrates that the most common errors are not the most significant errors. In particular the research highlights the importance of accurate recording of measurements on large trees, especially height, and underscores the disproportionate effect on AGB estimates when single large trees are missed from an inventoried plot. This research demonstrates that targeting those errors that cause the large discrepancies could serve to improve forest biomass inventories and training protocols for experts and non-experts alike.

Ensuring the viability of forest carbon projects not only requires a sound knowledge of their carbon stock and an ability to monitor changes in carbon stocks over time, it also requires the implementation of management interventions that are locally relevant and considers the needs of people affected by any interventions. However, integrating societal needs within forest management strategies remains difficult because of the lack of tools for linking socio-economic data to land-use planning models. Chapter 5 explores protection and restoration actions in a landscape where people depend on forests for their livelihood. The study integrates socio-economic data from Poverty Environment Network (PEN) surveys into a more traditional biophysical framework that includes land-cover change analysis along with soil and vegetation carbon stocks associated with different land-use types. Including socio-economic variables significantly altered the scope for emissions reduction, partly because the land-use types not only varied in carbon stocks but also because of the essential environmental products and services they provided to communities. Moreover, the research highlights the importance of local threats to carbon stores in the study area, with per capita fuelwood extraction exceeding emissions from fossil fuel, cement and anthropogenic fires as the main source of emissions in the region, in the absence of industrial logging. These results suggest that the inclusion of fire management in Upper-montane forest should be a priority for emissions reduction in the study region and potentially in PNG as whole. Moreover, the results demonstrate the additional carbon benefits of establishing coffee plantations that use the native *Casuarina,* a common shade tree used in PNG. These shade trees store three times more carbon per volume than the most commonly used shade tree species in coffee plantations worldwide.

By using some of the societal-environmental synergies identified in this research, PNG could become an important contributor to the global fight to curb anthropogenic carbon emissions, while also improving the livelihoods of the PNG population that depends on, owns, or manage these forests, as they have for millennia.

Concept Diagram of Thesis



The thesis begins with a global assessment of recarbonization strategies for tropical forests then deals with the biophysical and socio-economic issues of measuring, monitoring and managing forest carbon stocks in Papua New Guinea

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Chapter 1

General Introduction

Author: Michelle Venter

The value of tropical rainforests to society reach far beyond their canopies through their regulation of biogeochemical processes; they create rainfall that supports agriculture and hydropower production, they cool the Earth's surface by evaporation (Sheil and Murdiyarso 2009) and they lock away carbon dioxide that would otherwise contribute to climate change (Malhi and Phillips 2005, Wright 2010, Pan et al. 2011). Tropical forests also provide direct benefits to ~1.6 billion rural-poor in the form of fuelwood, construction material and food (Chomitz 2007). Nonetheless, tropical deforestation and degradation is still rampant, reducing the capacity of these forests to continue to provide these important ecosystem services to a growing global population (Portela and Rademacher 2001).

In the past century economic development has driven the loss of over 200 million hectares of tropical forest (Lamb 2011). As a result of poor management, tropical forests now act as a net source of carbon to the atmosphere, rather acting as a net carbon sink, making them an important contributor to anthropogenic climate change (Malhi 2010). However, this situation may change. The international community has recognized the importance of reducing forest carbon emissions as a central strategy towards combating anthropogenic climate change. It is possible that policy mechanisms that ascribe a value to the carbon stored in forests may lead to more sustainable forest use, where in some cases, forests may be worth more standing than logged or converted to agriculture (Laurance 2007).

Polices under the United Nations Framework Convention on Climate Change (UNFCCC) aim to halt and ultimately reverse the trends in carbon emissions from forests (UNFCCC 2009). More specifically, programs such as Reducing Emission from Deforestation and forest Degradation (REDD+), allow developed countries to support developing countries to reduce forest emissions through changed management practice aimed at retaining or increasing forest carbon stocks.

One of the key features that make REDD+ different to previous tropical forest conservation initiative are the funds available for REDD+ initiatives. Since 2010, over US$200 million has been committed by the UN-REDD program (http://mptf.undp.org/factsheet/fund/ CCF00, accessed April 22, 2015) with further funding coming from the voluntary carbon markets (Peters-Stanley and Yin 2013). Moreover, Norway alone has contributed US$1 billion to REDD+ development in Brazil, and a further US$1 billion to Indonesia (Clements et al. 2010). These large income streams have been a significant motivator for action by eligible countries. However, for a country to benefit from REDD+, rigorous processes must be undertaken, with countries rewarded in proportion to their success in reducing forest emissions. This includes, having an accurate and reproducible assessment of the carbon stored in their forest and of historical rates of forest carbon emissions as well as implementing locally relevant management to reverse trends in forest emissions while having systems in place that can monitor these changes (Maniatis and Mollicone 2010).

In order to demonstrate whether or not changes in tropical forest carbon stocks are occurring, it is necessary to first establish baseline data on the distribution of tropical forest biomass (van der Werf et al. 2009). Our current understanding of tropical forest carbon stocks is largely based on forest carbon density data (carbon per unit area) overlaid onto forest cover maps (Saatchi et al. 2011, Baccini et al. 2012). Though forest cover can be reasonably well measured through remote sensing (Hansen et al. 2013), the amount of carbon stored in forests is best measured by field inventory techniques (Houghton 2012). This is because most of the carbon (60−90%) in tropical forests is stored in above ground biomass (AGB) (Pan et al. 2011) but no satellite can yet measure AGB accurately (Woodhouse et al. 2012, Zhang et al. 2014). Therefore, to participate in initiatives such as REDD+, countries must have a robust field inventory of the carbon stored in their forests and the ability to monitor changes in stocks over time (Asner 2011). The challenges associated with measuring and monitoring forest carbon stocks have been a major roadblock to the implementaion of forest carbon projects in many countries (Romijin et al. 2012).

Most developing countries currently fall short of the requisite technical and scientific capacity to measure and monitor forest carbon. For example, 92 of the 99 non-Annex I countries, which are groups of developing countries recognized by the Convention of Biological Diversity (CBD), are especially vulnerable to the adverse impacts of climate change and do not have the capacity to perform field-based forest carbon inventories (Romijn et al. 2012). Brazil, on the other hand, has the capacity to monitor their forests and scientists have been evaluating their forest carbon stores for decades; for this reason Brazil has been able to demonstrate a reduction in forest emissions and currently has been the only country to receive financial rewards from REDD+ activities (Hargita et al. 2015).

Once national forest inventories and monitoring programs are in place, countries must then decide on locally appropriate forest management (Asner 2011). Two basic management strategies are recognized by REDD+. The first is to reduce forest emissions by protecting forest from the processes that threaten carbon stocks. The second is to increase the forest carbon sink through restorative management that promotes the sequestration of atmospheric carbon into forest biomass (UNFCCC 2010). The suitability of management actions largely depends on the type and extent of land available for management, the local drivers of forest carbon loss and the ability of ecosystems to store or sequester additional carbon (Venter et al. 2012c). In order for a forest carbon project to work on the ground, they must also address the needs of local stakeholders (Blom et al. 2010).

Achieving lowered emissions from tropical forest will invariably change the dynamics of livelihoods in communities that use forest resources, some of which are the poorest and most vulnerable on the planet (Angelsen et al. 2014). The recognition of the community’s role in conservation project success has prompted REDD+ to shift from its initial narrow emissions reduction strategies to also including development objectives for these communities (Sutter and Parreño 2007, Resosudarmo et al. 2012). If improving livelihoods is a target within forest climate mitigation strategies, then approaches that measure and monitor socio-economic factors should be incorporated into management decisions. However, no robust methods have been successfully demonstrated to date (Angelsen et al. 2014).

Papua New Guinea (PNG), covered by dense tropical forests, forms half of the largest tropical island in the world (Hansen et al. 2013). In theory, the country is well placed to participate in forest-carbon initiatives such as REDD+. An estimated 5 million (M) hectare of tropical forest in PNG were lost to industrial logging or converted to agriculture from the years 1970 to 2000; thus there is great scope in PNG for the reductions in forest carbon emissions through restoration activities (Shearman et al. 2009). Moreover, approximately 70-97 % of the forest estate is owned and managed by local people whom often have limited access to roads, electricity, running water sanitation or medical assistance, leaving them vulnerable to poor health and natural disasters. Thus, communities could benefit from REDD+ initiatives that also aim to improve livelihoods (Keenan et al. 2011, Rogers et al. 2012).

In practice however, a number of challenges have constrained PNG's participation in climate mitigation strategies (Babon and Gowae 2013). The geographical isolation imposed by the rugged mountainous terrain of PNG has, on the one hand given rise to outstanding cultural and biological diversity of these forests, but on the other hand, has also presented formidable barriers for access to the region (Flannery 1995). For this reason, PNG has some of the highest uncertainties in their forest carbon stock estimates (Mitchard et al. 2013)**.**

Moreover, PNG's constitution is based on the lawful recognition of indigenous land tenure with livelihoods heavily reliant on forests for food, fiber, fuel, and construction material (Babon 2013). With so much of PNG's forest under local tenureship and management (Keenan et al. 2011), the success of forest carbon projects in PNG is reliant on productive engagement of local landholders and the development of strategies to provide fair and ethical compensation to those who forego forest exploitation. The main objectives of this thesis are therefore to:

1) Improve the understanding of the role of tropical forest in climate mitigation by exploring a suit of management strategies that either reduce forest carbon loss or increase forest carbon gain. To achieve this **Chapter 2** aims to contextualise the costs and benefits of seven recarbonization strategies in tropical forests.
2) Reduce uncertainty in forest carbon estimates in the data deficient PNG. To achieve this **Chapter 3** aims to measure, for the first time, above-ground carbon

stocks for all three broad forest categories in PNG by conducting a large field campaign using a stratified sampling with 193 plots in Lowland, Montane and Upper-montane forests. **Chapter 3** also aims to elucidate the controls on AGB by studying the relationship between forest carbon, forest structure, and climate, edaphic and topographic variables along a 3,100m elevation gradient.

3) To assess the potential of engaging local communities to monitor carbon stocks in their forests and to improve existing protocols for monitoring in PNG. To achieve this, **Chapter 4** aims to evaluate the robustness of data provided by self-directed forest biomass inventory by three communities in the remote tropical forests of Papua New Guinea. **Chapter 4** also aims to evaluate the types of errors in the field that lead to the largest discrepancies in forest biomass estimates by assessing the contribution of errors from tree diameter, height, and numbers of trees and plot surface area with the aim of further improving locally-based monitoring accuracy.
4) Recommend management practices that reduce forest emissions while meeting the needs of forest dependent communities. To achieve this **Chapter 5** aims to test an approach that integrates socio-economic data with more traditional land-use management planning and compares 'restore' and 'protect' interventions for lowering forest carbon emissions. To do this, data from land-cover change analysis (years 1990-2010), field carbon assessments from eight land-cover types (263 forest plots and 115 soil sites) and socio-economic surveys (112 household in 9 communities) are integrated into a land-use planning model that explores future emissions under local threat and industrial logging scenarios.

-E N D OF C H A P T E R 1-

Chapter 2

Recarbonization of the Humid Tropics

Authors: Michelle Venter, Oscar Venter, Susan Laurance and Michael Bird

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2.1 Overview

This chapter presents the current state of knowledge on carbon stocks and fluxes in humid tropical forests, the relevant policies that could be harnessed to encourage recarbonization and investigate management interventions for the recarbonization the humid tropics. The carbon sequestration potential of seven recarbonization options is explored as well as co- benefits, risks and costs.

2.2 Introduction

To curb the effects of anthropogenic carbon (C) emissions, it is essential to explore the capacity of natural ecosystems to store and sequester atmospheric carbon (Phillips et al. 1998, Malhi et al. 1999). Humid tropical forests store most of their carbon in their above ground biomass a carbon pool that can be rapidly depleted by extractive industries or replenished more slowly through natural regeneration or reforestation (Laurance 2007, van der Werf et al. 2009). This rapid exchange of carbon dioxide between the forest and the atmosphere makes humid tropical forests one of the most dynamic carbon pools on Earth (Malhi 2010). They act as either a store or as sink for C, but currently are the source of almost all anthropogenic carbon emissions from the land-use and land-use change (uncertainty in estimates of tropical forest carbon would greatly improve existing models of global carbon stocks and fluxes, as well as help prioritize forest protection for climate mitigation (Mitchard et al. 2013).

Logging and agriculture, the main drivers of deforestation, play an important role in the economic development of most tropical countries. In some cases, forestry practices have been sustainable and productive agricultural lands have been established (Lamb 2011). However in many areas, tropical forests landscapes were altered through poor management practices and exist in a degraded sate, providing reduced economic value, productivity or ecosystem services (Sasaki et al. 2011).

Degraded lands are widespread in the humid tropics covering about 40% of the humid tropical area (ITTO 2002). Degraded lands range from fragmented primary forests to secondary forest to completely altered ecosystems such as non-native grasslands (Lamb 2011). These lands represent enormous potential for carbon sequestration and the rate of biomass recovery is largely dependent on the level of their degradation. However, in many tropical areas, biomass cannot recover effectively without human intervention (Chazdon 2008).

Recarbonization of the humid tropics is defined in this chapter as any land management strategy that aims to maintain or increase the carbon density of humid tropic regions. Land-use change in the tropics typically reduces the carbon density of a landscape (Ebeling and Yasué 2008, Sandbrook et al. 2010). In the past there has been little economic incentive to avoid reducing the carbon densities of forest areas (Fearnside 2000, Pongratz et al. 2009). However, since the adoption of the Kyoto protocol in 1997, new policies that incentivise recarbonization activities mean that active restoration and forest conservation could become an economic alternative to logging and agriculture (UNFCCC 2006).

The Clean Development Mechanism (CDM) within the Kyoto protocol was the first agreement that bound industrialized countries to offset their emissions through projects in developing countries. This marked-based mechanism was meant to give some flexibility to industrialized countries in how they met their emissions reduction targets. It also included wider objectives such as sustainable development and poverty alleviation in developing countries. The CDM was predicted to stimulate reforestation and afforestation in the tropics; however, of the 1600 projects registered under the mechanism in 2010, only four were forest-related projects (Thomas et al. 2010). Instead developing countries invested in renewable energy and energy efficiency projects because of the high transaction costs associated with relatively small land-use change projects and problems associated with leakage (UNEP 2009).

More recently, Reducing emissions from deforestation and forest degradation (REDD+) has emerged under the United Nation Framework Convention on Climate Change as a potential means to promote recarbonization in the tropics (UNFCCC 2009). Because REDD+ will set baselines and award credits at the national scale, it has the potential to allow tropical forests to play a much larger role in mitigating climate change, essentially sidestepping the effects of within-country leakage and reducing the transaction costs of carbon credits by creating a single Measurement, Reporting and Verification (MRV) scheme to cover an entire country (Sloan and Pelletier 2012). Individual site-based projects can still participate in national-scale REDD+ through sub-national baselines and MRV schemes, but presumably these would be less stringent and costly than the national scale scheme and those required under Kyoto's CDM (Cacho et al. 2005).

The scope of REDD+ has been a major issue throughout its policy development. Originally it was intended only to reduce emissions from deforestation (RED), but it quickly expanded to consider emissions from forest degradation (REDD). In 2009, REDD further expanded to include the role of forest conservation, the sustainable management of forests and the enhancement of forest carbon pools in developing countries, and in so doing became REDD+. While its exact scope remains to be determined, REDD+ has the potential to go beyond simply maintaining forest carbon pools to also include incentives to improve industrial logging operations, and promote forest carbon sequestration through re/afforestation. The expanded scope of REDD+ and the scale of the incentives breathes new life into ongoing efforts to re-carbonize the tropics.

2.3 Humid tropical forest

Humid tropical forests are largely concentrated within 10° latitude of the equator but may reach as far as 25° on the east coast of continents and Pacific Islands (Richards

1952). They are the characteristic vegetation type of the humid tropics and cover most of the land surface where climate is hot and rainfall is heavy and non-seasonal, except for swampy or volcanic areas (Richards 1952).

High levels of plant productivity, and hence carbon sequestration of humid tropical forests is attributable to the year-round growing season. Tropical humid climate has been defined as monthly precipitation exceeding 60 mm, annual rainfall above 2,000 mm and average monthly temperatures above 18°C at sea level (Huston and Wolverton 2009). The transition between humid forest and dryer landscapes are seldom static and are driven by environmental gradients or cyclicity in rainfall, temperature and anthropogenic disturbance (Murphy and Lugo 1986, Hirota et al. 2010).

2.4 Carbon flux in tropical forests

Sound knowledge of carbon stocks and fluxes in tropical humid forest are the cornerstone of effective recarbonization strategies. The best way to underscore the importance of tropical forest in recarbonization of the biosphere is through their role in the global carbon cycle. Basically, the dynamics of global carbon cycle is driven by five major carbon pools (Fig. 2.1).

In the past century, the atmospheric carbon pool has been increasing at an unprecedented rate. The major sources of carbon are the fossil fuel and the biosphere pools, while the largest sinks are the ocean and the biosphere pools. For example, of the carbon emitted by humans between years 2000 and 2008, 45 % remained in the atmosphere, 26 % was absorbed by the oceans, and 29 % was absorbed by the biosphere (Le Quere et al. 2009). Only recently have we understood the role of forests in the terrestrial carbon sink (includes soil and terrestrial biosphere), before that, it was referred to as the ‘unknown terrestrial sink’. Now estimates show that the carbon stored in soil and biomass of the world’s tropical forests probably make up half of the terrestrial sink (Fig. 2.1, (Malhi 2010, Pan et al. 2011).

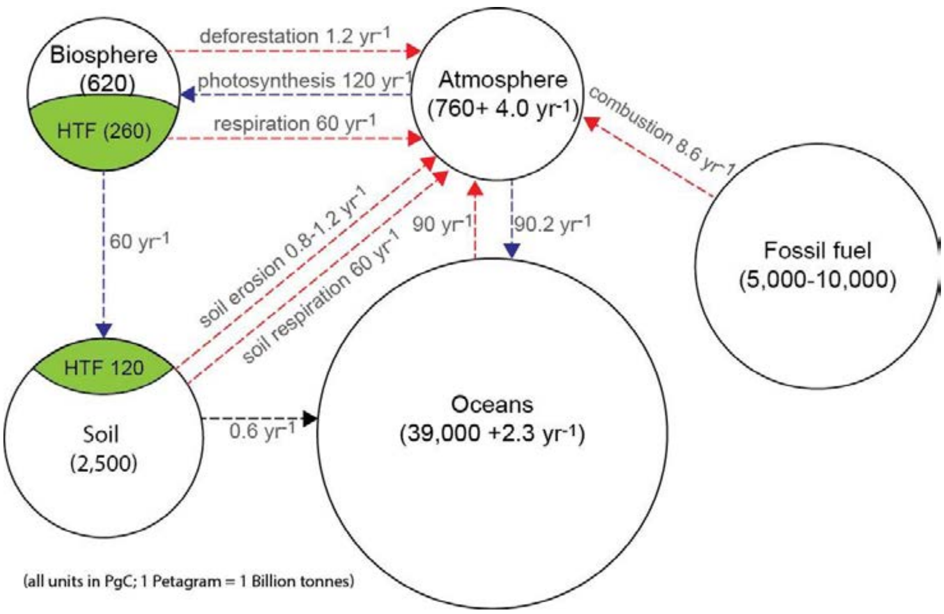

Figure 2.1. Five major carbon pools forming the global carbon cycle adapted from Lal (2010). *Deforestation rates are from Pan et al. (2011), fossil fuel emissions from JRCNEA (2011), humid tropical carbon from Pan et al. (2011) and soil carbon to 1m depth from Scharlemann et al. (2010).*

Carbon enters the terrestrial biosphere via photosynthesis, and then makes its way to the more stable soil carbon pool by biomass decomposition and mineralization, after accounting for losses in soil erosion and soil respiration. About 56 % of the carbon in humid tropical forests is stored in biomass, while 32 % is stored in soil up to 1m of depth and 12 % in necromass (Pan et al. 2011). Storing carbon in biomass rather than soil has the effect that humid tropical forests accumulate or release significant quantities of carbon over shorter time periods compared to higher latitude ecosystems (Phillips et al. 2004, Houghton 2005). When forests are degraded through logging, or converted to non-forest vegetation types, the carbon stored in the biomass is rapidly released into the atmosphere. In comparison, soil carbon loss happens more slowly through erosion/remineralisation after land-use change (Achard et al. 2004).

Humid tropical forest continuously uptake CO_2 from the atmosphere. A number of recent studies have argued that the carbon sink capacity within intact humid tropical forest might be increasing (Lewis et al. 2009b). The evidence supporting this argument comes from long-term observation of increased biomass within forest plots. Globally, tropical humid forests seem to be reaching a higher biomass state. Lewis et al. (2009b) compiled biomass estimates collected in forest inventory plots for intact humid tropical forest and showed a mean increase of 0.49 Mg C ha^{-1} yr^{-1} between 1987 and 1997. There are two main schools of thought to explain this increase in biomass. The first is a response to CO_2 fertilization from increased anthropogenic carbon emissions (Phillips et al. 1998, Lewis 2006, Malhi 2010) and the second is that is represents a recovery response from a widespread disturbance, such as drought, that happened in the past (Wright 2005). The extent to which this increase in biomass sink will persist is still unknown (Lal 2010).

When considering the combined effects of deforestation and degradation and sinks from intact forest and forest regrowth, humid and dry tropical forest are found to be a net source of carbon to the atmosphere, emitting -1.3 Pg C and sequestering 1.1 Pg C (Malhi 2010). However, it is important to note that the net balance reported in this study is not statistically different from zero (Malhi 2010). Therefore recarbonization of tropical forests, through forest protection or restoration, holds enormous potential of shifting humid tropical forests from a CO_2 source to a CO_2 sink.

2.5 Options for recarbonizing the humid tropics

The recarbonization strategies in the humid tropics include those that 1) protect forests carbon pools from processes that threatens them (Laurance 2008), 2) reduce the impacts of degrading activities (Silver et al. 2000), 3) enhance the rehabilitation process of degraded areas (Brown et al. 1986, Lamb 1998) and 4) convert non-forest lands into forests (Brown and Lugo 1994, Niles et al. 2002, Lamb et al. 2005, Zomer et al. 2008, Torres et al. 2010). In Table 2.1, seven recarbonization options discussed in this chapter are presented under the four aforementioned strategies

Table 2.1 Recarbonization options for different land types in humid tropical regions.

Land type	Results from no action in this land type	Recarbonization option	Risks	Costs	Sequest-ration	Co-benefits
Primary or degraded forest	Timber and fuelwood harvest, fires, drought, storms, climate change.	Forest protection (see 2.5.1)	Leakage Weak governance Reliance on carbon economy	High start-up Low running	High	Ecosystem services Cultural conservation
Degraded forests only	Weeds and vines impede recovery. Soil loss, compaction reduces fertility.	Reduced impact logging (RIL) (See 2.5.2)	Transaction costs of collaborating with industry. Ongoing logging, fire threats	Low	Low	Employment Biodiversity benefits
		Accelerated natural regeneration (ANR) (See 2.5.3)	Tree mortality Other activities more profitable	Medium	Medium	Employment
Degraded non-forest lands	Altered disturbance regime lead to lack of seed stock, soil loss, and lack of micro-fauna, altered hydrology, dryer habitat, new stable state.	Agroforestry (See 2.5.4.1)	Very few risks because practice is well established	Low	Low	Food production Cash economy Biodiversity
		Monoculture plantation (See 2.5.4.2)	Species extinctions Market dependent	Medium	Low	Income generation
		Polyculture plantation (See 2.5.4.3)	Access to markets Long return on investments	Medium	Medium	Income generation. Native species improve habitat for wildlife
		Restoration Plantings (See 2.5.4.4)	Fire, poor seedling establishment	High	High	Full range of ecosystem services

2.5.1 Reducing deforestation

Forest protection

The most obvious way to reduce emissions from deforestation is to protect the forest from extractive activities. If the clearing of tropical forests were to continue unabated, 87 Pg C to 130 Pg C will be released into the atmosphere by year 2100 (Houghton 2005). Conversely, if business as usual deforestation rates were reduced by half by 2050 and then held constant until year 2100, 50 Pg C of emissions would be avoided (Gullison et al. 2007). The Eliasch Review (Eliasch 2008) estimated that including reductions to tropical deforestation in global climate initiatives would halve the cost of reducing global anthropogenic carbon emissions (Eliasch 2008), making this a politically and economically attractive strategy.

Keeping forests standing can have many additional, non-carbon benefits as intact natural forests provide ecosystem services that contribute to human well-being. These services include the provision of non-timber forest products, the filtration of water and air (Sheil and Murdiyarso 2009), mitigation of floods (Bradshaw et al. 2007) and the provision of pollinators for adjacent farm crops (Ricketts et al. 2004). Natural forests also provide habitat for many species. carbon payments to reduce tropical deforestation could yield substantial benefits for biodiversity conservation, especially if the payments favour areas of high biodiversity (Venter et al. 2009).

Though reducing emissions from deforestation is now backed by substantial political support (UNFCCC 2009), there are challenges that may yet prevent broad implementation. A number of these issues are technical, such as establishing accurate reference emissions levels against which emissions reductions can be measured (Olander et al. 2008, Griscom et al. 2009). Accurately measuring forest cover change across large areas and in a timely manner remains complex and expensive as it requires large field campaigns (Gibbs et al. 2007, Grainger 2009). One of the major roadblocks in using forest protection as recarbonization intervention is difficulty in minimizing or measuring the 'leakage' or displacement of deforestation from one place to another (Ewers and Rodrigues 2008). Other impediments include politicking in corrupt countries (HRW 2009) and ethically engaging local stakeholders, who may both gain or lose from forgoing forest extraction for climate mitigation (Pfaff et al. 2007).

2.5.2 Reducing forest degradation

Reduced impact logging

Forest degradation is defined here as the loss or removal of forest canopy cover not resulting in deforestation, usually describing forest that retain at least 10 − 30 % cover (Sasaki and Putz 2009). Forest degradation can occur directly and indirectly due to human activities. Planned degradation occurs primarily through selective timber and fuel-wood harvesting, and is a major cause of forest cover change in the tropics (Wright 2010). An estimated 20 % of the humid tropics underwent some form of timber harvesting in the period 2000 to 2005 (Asner 2009) and that despite the intensity of the impacts from logging, forest degradation and associated emissions are difficult to measure over broad areas.

While its impacts vary, in Southeast Asia, where selective timber harvest rates are highest, selective logging removes 33 - 56 % of the biomass stored in forests (Pinard and Putz 1996). Early estimates concluded that forest degradation across the tropics was responsible for 4.4 % of emissions from land-use change (Achard et al. 2004). However a more recent estimate attributed roughly 20 % of emissions from the Amazon to selective logging (Asner et al. 2005). At the global scale, selective logging is responsible for the release of 0.5 Pg C yr^{-1} into the atmosphere (Putz et al. 2008). No data are available to quantify the emissions from other degrading activities, such as fuel wood harvesting (Griscom 2009).

The footprint of selective logging in the humid tropics is 20 times larger than that of deforestation (Asner et al. 2009b). To reduce emissions degrading activities would require working with industry to reduce its impact. In the case of industrial logging, much of the emissions come from the building of roads. Building roads has additional collateral impacts associated with the removal of non-commercial trees (Bertault and Sist 1997). Reduced impact logging techniques (RIL) involves careful planning and implementation of logging operations in order to minimize impact on the residual stand. RIL has shown to reduce carbon emissions up to 30% (Pinard and Cropper 2000, Keller et al. 2004). Implementing RIL techniques across the production in the humid tropics could have the potential to reduce carbon emissions by 0.16Pg·C (Putz et al. 2008).

It is possible that RIL has both the lower costs and risks involved than other recarbonization actions. Mostly because RIL minimizes or even avoids opportunity costs and increases the permanence of emissions reductions (Berault and Sist 1997). Compared to forest protection, where the threat of forest clearance will persist, RIL reduces threat to forest and overcomes issues of leakage (Ewers and Rodrigues 2008). The main drawback, however, is that compared to other options, RIL has a far lower mitigation potential per hectare (Putz et al. 2008).

2.5.3 Forest rehabilitation

Accelerated natural regeneration

Degraded forests make up almost half the 11 million km^2 of humid tropical forest (ITTO 2002). Because of the current extent of degraded forest, management that accelerates forest regeneration (ANR) poses a great opportunity to increase carbon density across the tropics (Parrotta et al. 1997). Though forest recover naturally to some extent, the rate at which they do can affected by the competition with weeds or pioneer species barriers, sometime causing long-lasting ecosystem changes (Pariona et al. 2003). Recovery is particularly slow following a disturbance that has severely altered soils and vegetation communities (Chazdon 2003). There are a variety of silvicultural techniques that can overcome these barriers by assisting natural regeneration and therefore increasing carbon density.

Enrichment planting (EP), a technique that involves planting existing forest with trees is commonly used in forestry to increase the density of target species under canopy gaps (Lamb 2005). This practice it is often supplemented by other treatments that accelerate carbon sequestration by reducing competition from vines (liberation cuttings) and certain pioneer trees (thinning) (Kosonen et al. 1997, Peña-Claros et al. 2008). Forestry practices that have included EP and liberation cutting during second and third timber harvest rotations have reported increased productivity (Keefe et al. 2009).

ANR practices could also target tree species of conservation value, and assist in the propagation exploited species such as mahogany (*Swietenia macrophylla*) and those of large fruiting species that cannot self-propagate (Lamb 2005; Keefe et al. 2009). Other benefits could include the improvement wildlife and riparian corridors and decreasing the risk of fires (Parrotta et al. 1997, Brown et al. 2004, Chazdon et al. 2009). Aside from mortality of planted trees or failure to compete with more profitable activities, there is very little risk associated with ANR relative to the small costs of labour, equipment, and training and site preparation, moreover it could be an effective way to engage local communities by providing work opportunities (Keefe et al. 2009). Possibly ANR is the most economically feasible recarbonization option in areas where local people depend on forests and threats of logging are not immanent.

2.5.4 Converting degraded non-forest lands to forests

The most ambitious of the recarbonization strategies available in the humid tropics is to convert degraded non-forest lands to forested lands. The term 'reforestation' is used when trees are planted on sites where forest has been present during the last 50 years, and 'afforestation' refers to planting trees on sites where forest was absent for longer than 50 years (FAO 2000).

Anthropogenic grasslands, common throughout the tropical humid regions, can be slow to revert to forest if frequent fires and grazing pressures are present (Garrity et al. 1996). Shade-intolerant grasses and ferns become well established in areas where forest once existed if disturbance regimes are not altered (Cramer et al. 2008). Therefore, carbon poor anthropogenic grassland can persists despite being surrounded by forest (Fig. 2.2 andMacDonald (2004).

Figure 2.2. Photo of lowland forest in Morobe province of Papua New Guinea. *A) Mosaic of primary forest and anthropogenic grasslands. B) Lowland primary forest fragment (Photos by Michelle Venter).*

Re/afforestation techniques are resource intensive as they all require seeds or seedlings to be planted (Lamb 2011). In this section, I discuss re/afforestation in the form of agroforestry for crop production, monoculture plantations on short rotations for fibre production, longer rotation of mixed hardwoods for timber production and finally ecological planting (EP) for enhanced ecosystem services (section 2.3.4.4).

Agroforestry

Agroforestry involves planting trees as part of a system that will provide food or timber for subsistence or cash income (Lamb 2011). Agroforestry areas can be established by planting seedlings in the early fallow stage of a shifting cultivation cycle or on degraded land (de Jong

2002). Agroforests are multi-strata systems with a complex mixture of native and exotic trees and crop species, which distinguishes them from monocultures crops for food production (such as oil palm plantations). They are a commonly cultivated in many parts of rural tropics, usually as home or community gardens (de Jong 2002, Albrecht and Kandji 2003).

Biomass and soil carbon stocks in agroforests across the tropics are up to 228 Mg C ha^{-1}, with a median value of 95 Mg C ha^{-1} (Albrecht and Kandji 2003). The carbon sequestration rate varies with types of species planted, the density to which they are plated and rotation length (Albrecht and Kandji 2003). While carbon stocks are relatively low in agroforests compared to other tree plantations, they usually have higher carbon sequestration rates than pastures, field crops and degraded lands (Ramachandran Nair et al. 2009).

Permaculture techniques used in agroforestry originate from a mix of traditional knowledge and agricultural sciences. For example, in Papua New Guinea, planting nitrogen fixing tree (e.g. *Casuarina oligodon*) to increase soil fertility is a common gardening practice (Bourke 1997). Permaculture techniques are also used to reduce weeds and pest and to compost, therefore few non-organic inputs are required (Michon et al. 2007, Vieira et al. 2009).

Though carbon sequestration benefits are low, agroforestry could be a feasible option as a climate mitigation strategy where food security is a priority (Vieira et al. 2009). Very little risk is associated with this option because agroforestry systems are proven to be successful. The major drawbacks are caused by low nutrients in top soils in the humid tropics (Schroth et al. 2002), and the unpredictable impact of a changing climate on food production performance (Cacho et al. 2005). Agroforests also have a rich understory of plants which can provide habitat for birds and invertebrates as well as contributing to landscape biodiversity (Bhagwat et al. 2008).

Monocultures in short rotations

Tropical forest plantations increased from 18 M ha to more than 70 M ha from 1980 to 2000 (FAO 2001), forming 25 % of global forest plantations (FAO 2006). Most of the plantations established after 1980 have been for wood fibre production on short rotation cycles (5-10 years) dominated by *Acacia* and *Eucalyptus* (Brown et al. 1986) They usually are privately owned in partnership with large corporations (Lamb 2011).

Pulpwood species have rapid growth rates and can be a good temporal option for mitigating CO_2 emissions if established on degraded lands (Silver et al. 2000). On average, biomass carbon stocks in 13 year old plantations are 62 Mg C ha^{-1} accumulating at a rate of 1.9 Mg C ha^{-1} yr^{-1} (Liao et al. 2010). However the effect of plantation establishment on soil carbon stocks is debatable (Laganière et al. 2010). Some studies have shown that monoculture forestry causes a decrease in soil carbon pools (Guo and Gifford 2002). However, the overall soil carbon change is generally positive (Paul et al. 2010).

Monocultures are easy to establish and to maintain to over large areas, and provide income for smallholders (Putz and Redford 2009, Bremer and Farley 2010). If natural regeneration is permitted in the understorey, plantations may contribute additional habitat for wildlife and act as corridors for wildlife migration. These benefits however are temporary, lasting only until the next harvest.

Monocultures are the most controversial among re/afforestation strategies. This is mostly, because they are characterized by limited ecosystem services, low biodiversity values and lower carbon benefits (Bremer and Farley 2010). These issues are exacerbated if they replace native forests. Replacing degraded grasslands with monocultures is a viable recarbonization strategy, as plantations can provide better watershed protection (McElwee 2009) and hold great value to the local people who harvest products from them. In Vietnam, reforestation has been strongly promoted by the government in a program known as the '5 M hectare project'. This re/afforestation program has helped to reverse the trend of forest loss but it has had a multitude of unforeseen negative consequences. The worst of these was the increase in poverty amongst the poorest people in the region (McElwee 2009). Therefore, careful consideration of social and cultural welfare should be assessed before ambitious re/afforestation projects are undertaken.

Polyculture in long rotations

Polyculture plantations typically have rotations of 20 years or more and can include a mix of native and exotic tree species. Because polyculture plantations are more difficult to manage than monoculture plantations and have higher start-up costs, they are generally less favoured by industry and thus cover very limited areas in the humid tropics (Lamb 2011).

Carbon stocks in polyculture plantations range from 110 - 173 Mg C ha^{-1} and sequestration rates vary with plantation types, but have higher average carbon sequestration than monocultures (Paul et al. 2002) and are most likely to improve to biodiversity (Bremer and Farley 2010). The most profitable plantations are those that nurture highly valuable timbers such as teak (*Tectona grandis*) (Griess and Knoke 2011). Using a mixture of species also makes the practice more resilient to markets responses, as mixed species plantation have less stringent harvest times and provide a wider range of goods (Lamb 2011).

Restoration plantings

Restoration plantings (RP) are a reforestation strategy that attempts to overcome dispersal and recruitment limitation of natural regeneration by planting an assortment of native species. Dispersal limitation accounts for all the factors that limit the arrival of a seed at a site, including seed source and production and seed dispersal. Recruitment limitation encompasses the next life history stage where the seed has arrived at the site but cannot germinate or successfully establish (Young et al. 1987, Shono et al. 2007). In time, the aim is to produce a forest of similar structure and community to native forests.

Restoration plantings, compared with other recarbonization options areas, have the highest overall potential to sequester carbon in both above- and belowground pools. Because the duration of project involving restoration plantings are usually unlimited, carbon sequestration can continue to be positive over 80 years during forest establishment, and possibly longer (Silver et al. 2000).

The practical barriers to RP are the time it takes to collect and germinate seeds and then to plant them, the long-time commitment and the risks from wild fires (Brown et al. 2004). Other barriers to implementation come from a lack of understanding about assembling ecosystems. Creating any viable ecosystem requires deliberation. But trying to replicate a specific ecosystem, such as the one present before clearing, can be almost impossible. Moreover, with changing climates, the persistence of certain tree species will be compromised, and planning for future climates and unknown outcomes is even more challenging.

This method is extremely intensive and currently is mostly used for rehabilitating strategic locations in the landscape such as wildlife corridors and creek banks (Chazdon 2008). The potential benefits that arise from RP, especially in the conservation of biodiversity and natural heritage, outweigh many of the other recarbonization options. The success of RP will probably depend in the motivation and engagement of multiple parties seeking multiple benefits in the long term.

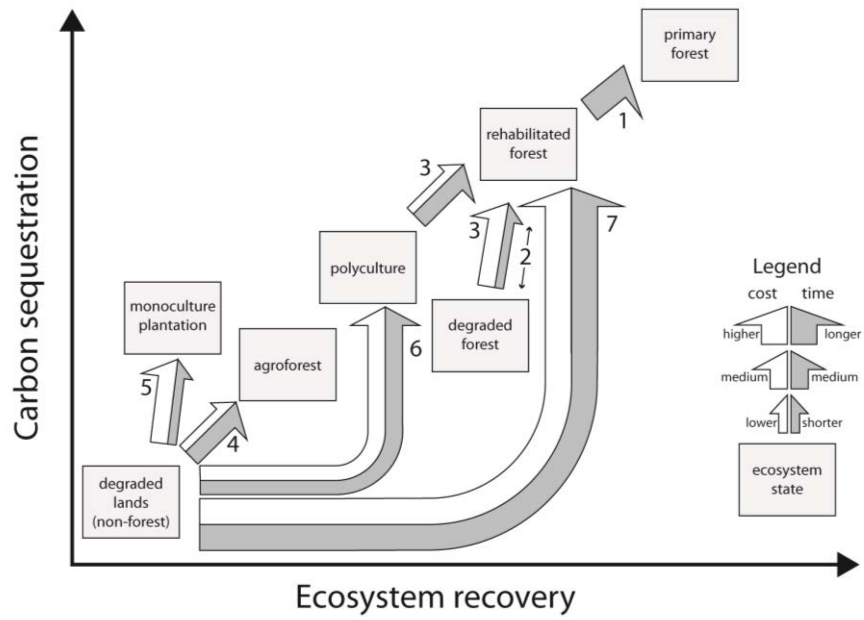

Figure 2.3. Relative level of carbon sequestration to ecosystem recovery of different recarbonization strategies*. The arrows indicate the relative time and costs for the implementation of recarbonization options. The numbers refer to following recarbonization option; 1) Forest protection 2) Reduced impact logging 3) Accelerated natural regeneration 4) Agroforestry 5) Monoculture plantation 6) Multi-species plantation 7) Restoration planting.*

2.6 Concluding remarks

Humid tropical forests contain some the most dynamic carbon pools on Earth, although there are still many uncertainties regarding the exact figure of carbon fluxes in the humid tropics. Given the extent of degraded humid tropical forests and non-forest lands, it is necessary to look beyond halting forest loss to consider options to accelerate the recovery of carbon stocks through increasing biomass.

The sink capacity of humid tropical forests can be enhanced beyond the natural phenomenon of forest recovery. In theory, the recarbonization capacity of humid tropical forest equates to the historic depletion of carbon from the biome (Rhemtulla et al. 2009). Positive or negative deviations from this potential could arise due to increased CO_2 fertilization from anthropogenic emissions or changes in global or regional climates (Lewis et al. 2009b). The potential role of humid tropical forests in recarbonizing the biosphere is determined by their rates of carbon emissions and capacity to sequester and store carbon.

While humid tropical forests remain a net source of carbon emissions, a more in-depth assessment of these figures gives reason to be hopeful; the carbon sequestration in intact and recovering forests is almost equal to the net emissions from forest loss. Given the distinct co-benefits, risks and costs associated with each practice, the review has outlined the potential for government, community, conservation and industrial initiatives to profit from recarbonization strategies. Incentivizing a variety of recarbonization actions and moving beyond the current focus on forest protection could tip the balance, making humid topical forests part of the solution for climate change instead of part of the climate change problem.

2.7 Summary of Chapter 2

- If resources and time were unlimited, most natural systems could be restored to their full carbon potential with human intervention contributing to important reduction in atmospheric carbon
- There is no lack of options for recarbonizing tropical humid forest, each having distinct benefits, costs and risks. Thus, it is likely that management action can be tailored to meet targets of emissions reduction.
- The most appropriate recarbonization options depend on the state of land degradation, the resources available to restore it which are in turn based on biophysical and socio-economic constraints.
- Finally, careful planning must be an integral part of any recarbonization project to optimize benefits as well as to avoid unforeseen environmental or social consequences.

-END OF CHAPTER 2-

Chapter 3

Large Trees and Natural Disturbances Drive Forest Biomass on a 3000 m Elevation Gradient in Papua New Guinea

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3.1 Overview

This chapter presents, for the first time, estimates of above-ground carbon stocks for all three broad forest categories in PNG measured in the field. Using a stratified sampling design using 193 plots in primary forest distributed along a 3,100 m elevation gradient, we also studied the relationship between above ground carbon store, forest structure, and climate, edaphic and topographic variables.

3.2 Introduction

Tropical forests play an important role in the global carbon cycle, containing about half the carbon stored in terrestrial plant biomass (Pan et al. 2011). As a result, changes in carbon stocks of tropical forests can exert significant influence on the global carbon cycle (Chapter 2). If intact tropical forests increase in biomass as the global climate changes, this could lessen the rate of growth in atmospheric carbon dioxide, providing a negative feedback on global warming(Malhi 2010). If the converse occurs, as predicted by some models (Cox et al.

2000, Huntingford et al. 2013) the loss of carbon from tropical forests will induce positive feedback, accelerating the pace of global climate change.

The largest uncertainties in tropical forest carbon stocks come from areas that have had few direct field inventories of their AGB (Mitchard et al. 2013). For example, Papua New Guinea (PNG), the Democratic Republic of Congo (DRC) and the Central Amazon region of Brazil are the areas with the highest uncertainties in their forest carbon stocks because of the difficulty associated with performing field inventories in these remote forests (Mitchard et al. 2013). Because AGB is highly spatially variable, measurements requires significant field efforts to reduce these uncertainties (Chave et al. 2003). Reducing uncertainties can most efficiently be achieved by uncovering the environmental controls on AGB. By using information of the relationship between AGB and environmental variables, field observations can be translated into modelled estimates of AGB at the landscape scale (Clark and Kellner 2012, Chave et al. 2014).

In the tropics, elevation gradients are well suited to this approach, as they present gradual and directional changes in environmental variables that are known or thought to affect AGB. These variables can include temperature, rainfall, insolation and soil condition across a defined bio-geographical region (Malhi et al. 2010). Tropical elevation gradients have also become a powerful tool to help predict the effect of a changing climate on forest carbon stocks and fluxes. Recently, important insights have been gained through studies of elevation gradients conducted in the Neotropics (Vázquez and Givnish 1998, Girardin et al. 2010, Homeier et al. 2010, Girardin et al. 2013), in South East Asia (Kitayama and Aiba 2002), and in Africa (Marshall et al. 2012).

Though some generic patterns in AGB have emerged from studies on elevation transects, little consensus exists on the effect of climate, edaphic conditions or topography on AGB in tropical forests (Baraloto et al. 2011, Selmants et al. 2014). Moreover, local variation caused by natural disturbances can be significant and can obscure the effect of climatic or edaphic variables (Fox et al. 2011, Stegen et al. 2011). In Montane forest especially, AGB is likely to be controlled by disturbance processes, due to the prevalence of steep slopes (Baraloto et al. 2011). However, incorporating natural disturbance into models that predict AGB is challenging, as proxies for natural disturbances are not readily available and detecting forest disturbance through remote sensing is challenging (Kasischke et al. 2013).

In tropical forests, the bulk of the variation in AGB has been attributed to the distribution of large trees (Slik et al. 2013). As a result of the disproportionate effect of large trees on AGB variation, understanding the environmental factors that promote the growth and distribution of large trees is also essential to reduce uncertainties in AGB (Larjavaara and Muller-Landau 2012).

The island of New Guinea is rugged, with a mountainous spine extending 2,500km along its length, making it one of the world's great mountain systems (Hall 1984). The region is ideal for the study of controls on AGB, with forests up to 4,000 m above sea level (asl) with high local relief causing abrupt changes in environmental conditions (Paijmans 1976). Little is known about the forest carbon stock in Papua New Guinea (Bryan et al. 2011). For example, the best available global dataset of forest AGB used 4,000 field sites from across the tropics with not a single site from PNG (Saatchi et al. 2011). Most of PNG's forests lie beyond the reach of roads, coastal or river access, thereby imposing major logistic constraints on field biomass surveys (Bryan et al. 2011). Consequently, the forests of PNG remain some of the most under-researched areas of tropical forest worldwide (Marshall and Beehler 2007). The most significant report of PNG's forest stocks is a cross-country network of permanent forest plots (Fox et al. 2010). However, these permanent forest plots are mostly concentrated in Lowland forests near roads with no plots in Upper-montane forests. Therefore, a third of PNG's forest found above 1,000 m asl, have been underrepresented by research (Shearman and Bryan 2011).

We present the first field inventory of primary forest biomass across all three broad forest categories in PNG (Lowland, Montane and Upper-montane forests) (Johns 1982). Using a stratified sampling design, we studied the relationship between forest structure attributes, AGB, climate, edaphic and topographic variables along a 3,100 m elevation transect. We also gained new insights into the relationships between AGB, natural disturbance and topography by sampling on slopes ranging from gentle to very steep (up to 80º).

3.3 Methods

3.3.1 Study Area

The study was conducted in the Saruwaged Range in Morobe Province, Papua New Guinea (PNG; 6°04'S, 146°48'E). This was the YUS Conservation Area, a region of 182,000 ha covered by 70 % primary forest ranging from 50 m asl to 3,100 m asl. The climate is perhumid, with a mean annual precipitation ranging between 2,600 mm in the Lowlands and 4,200 mm in Upper-montane forests (Hijmans et al. 2005). Mean annual temperature in the Lowlands is 26˚C, decreasing by about 5.4˚C per 1,000 m of elevation gain, reaching 10˚C at 3,100 m asl.

More specifically, the work for this chapter consisted of field surveys in a contiguous area of primary forest along an elevation transect extending 35 km from near the Bismark sea coast at 50m asl, (5°53.9' S, 146°52.0' E) to 3,115m asl (6° 5.7' S and 146°55.3' E) (Fig. 3.1). The elevation gradient follows a SW-NE trending ridgeline that is covered by primary wet tropical forest, including Lowland (below 1000 m), Montane (1,000 to 2,800 m asl) and Upper-montane (2,800 to 3,100 m asl) forest (Johns 1982).

In our study area the cloud immersion zone, as defined by remote sensing (Gillieson et al. 2011), occurs between 2,200 m and 3,100 m. All soils at our sites have developed on limestone bedrock, with the exception of the lowermost sites, where a limestone-colluvium derived soil with a 50 cm A−horizon is overlain onto alluvial deposits (for more details about soils see Table 3.1 and Dieleman et al 2013).

Table 3.1 Edaphic, climatic and topographic variables along a 3,100m elevation gradient in Papua New Guinea.

Forest Type[þ]	Elevation zone	Altitude range (m asl)	Mean slope (°) (95% C.I.)	MAT* (°C)	MAP (mm)	Soil type[§]	Mean soil depth[β] (mm)	N^{o} of plots
Lowland	50	50 – 150	2 (1 - 5)	26.3	2598	Hapludolls	97	21
	500	470 – 610	15 (8 - 22)	23.4	2806	Rendolls	92	16
	800	610 – 1030	26 (17 - 45)	22.1	2911	Troporthents	128	19
Montane	1400	1300 – 1500	19 (12 - 25)	18.8	3207	Troporthents	118	18
	1800	1750 – 1930	13 (8 - 17)	16.2	3484	Troporthents	134	21
	2200	2090 – 2230	13 (9 - 17)	14.4	3643	Troporthents	>200	29
	2400	2240 – 2500	15 (11 - 19)	13.0	3788	Troporthents	175	22
Upper-montane	2800	2720 – 2886	12 (9 - 13)	10.5	4090	Troporthents	180	25
	3000	2900 – 3115	17 (14 - 21)	10.0	4218	Cryorthents	194	22

Þ Forest types are based on Johns (1982). *Climatic variables are from BIOCLIM (Hijmans et al. 2005), MAT = Mean Annual Temperature and MAP = Mean Annual Precipitation. [§]Soil types are from PNGRIS database, [β] Soil depths are from Dieleman et al. (2013)

The entire study area lies within the Yopno-Uruwa-Som (YUS) conservation area. The YUS conservation area is named after the three major water catchments that drain a rugged, road-less mountain region of the Huon Peninsula. About 35 communities surround the YUS area, with a population of c.a. 12,000 who have a subsistence lifestyle and depend on the forest for building material, food and fuel. The authority that issued the permit to work in the YUS (Yopno-Uruwa-Som) Conservation Area was the Tree Kangaroo Conservation Program (TKCP).

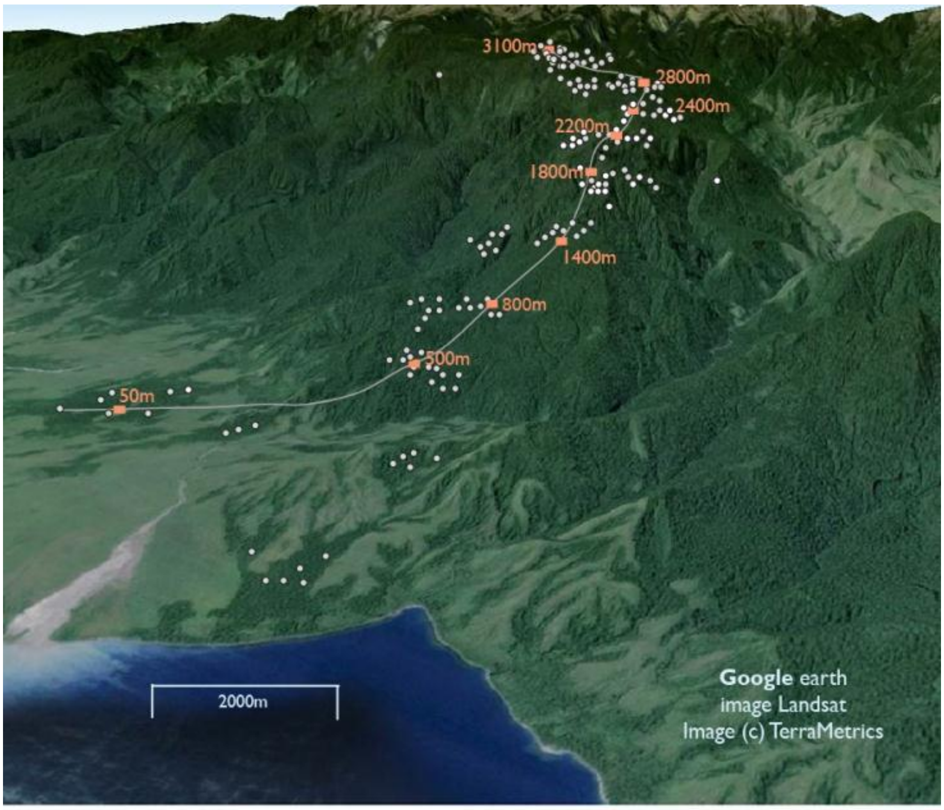

Figure 3.1. Digital representation of the study sites along the elevation gradient. *White dots represent study sites. Orange squares represent elevation zones, and the grey line represents the transect. Source: Google earth, image Landsat, TerraMetrics © 2014.*

3.3.2 Study Design

Nine elevation zones were selected prior to the commencement of fieldwork via remotely sensed images with the aims of 1) capturing the range of forest types and the changes in forest structure attributes along the elevation gradient and 2) sampling in areas far from human disturbance (e.g. requiring more than one day's return walk from a village). The lowermost sites did not satisfy the second criterion as villagers could access the forest within a day's walk; therefore they were excluded from analyses investigating natural disturbance.

Elevation zones (N = 9) were further stratified to include three slope categories (gentle 0˚ to 15˚, moderate 16˚ to 25˚, and steep 26˚ to 80˚) and three aspect categories (east, west and ridge top), where possible. In plots established on very steep slopes, ranging from 45˚ to 80˚, climbing equipment was used to establish the plot and to measure the trees (Fig. 3.2).

Figure 3.2. Photo of a steep sample plot with an average slope of 68.0˚.

A total of 193 rectangular plots (20 x 50 m) were established (Fig. 3.3), with each of the nine elevation zones containing 16 to 29 sites with a minimum of 120 m distance between plots. Methods for biomass inventories and plot establishment were largely based on the widely used Land Use, Land-Use Change and Forestry (LULUCF) protocols (Pearson et al. 2005). We chose to sample many small plots across the landscape, instead of fewer larger plots, to capture more landscape variability (Laumonier et al. 2010). The southwest corner of each plot was selected randomly by throwing a spear into the air. From the point where the spear landed, we followed a random compass bearing for 15m. This technique was employed to ensure that sites were sampled regardless of their disturbance history. Natural disturbance was recorded for a site if a landslide or wind throw resulted in visual damage to forest structure causing at least six tree-falls within the site. The plots were delineated using compasses and survey tapes and pegs were used to mark the corners.

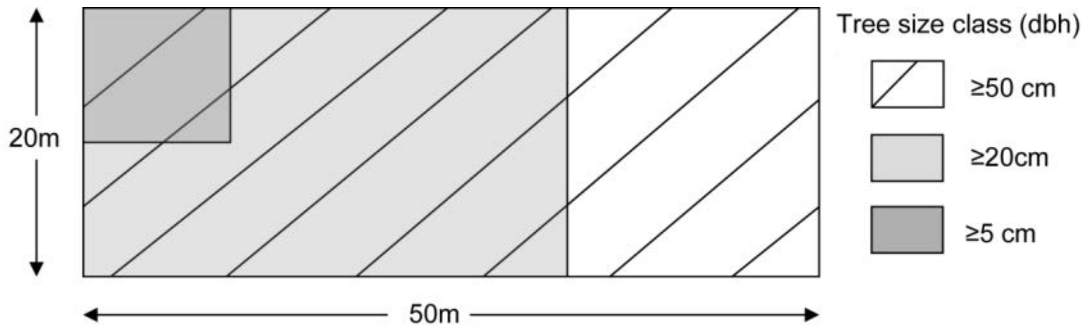

Figure 3.3. Diagram of a forest-biomass inventory plot.

Parameters measured in the field to estimate AGB included diameter at breast height (DBH), tree height (H) and Wood Specific Gravity (WSG) and tree taxa. DBH values were measured to the nearest millimeter with a diameter tape at 1.3 m above the ground. For trees with buttresses, bush ladders were built to measure the diameter from a point immediately above the buttress with distance to the ground recorded. Height was measured by standing directly below the crown and measuring the highest point visible in the canopy with a rangefinder (LaserAce® hypsometer) multiple times until the highest point was reliably identified (Feldpausch et al. 2011). Heights were measured for 75.4 % of the stems; for the remainder, heights were estimated using a taxa and altitude-specific height-diameter model (Appendix 3.0). Canopy height was determined by using the mean of all stems >50cm DBH. The height of emergent trees is reported as the mean of the tallest tree of each plot, reported for each elevation zone (Girardin et al. 2013). WSG values were derived from several rainforest datasets (Eddowes 1977, Chave et al. 2006, IPCC 2006b, Zanne et al. 2009) and from wood cores for 25 tree species with high contributions to AGB (Chave et al. 2006). For trees without WSG values, average WSG for the elevation zone was used (Fox et al. 2010). We measured lianas, palms, pandanus and lying dead trees according to protocols in Pearson et al. (2005).

Tree identification was carried out from the collection of fertile vouchers, pictures of fertile botanical specimens, DNA-barcoding analysis from leaf tissue and from local knowledge of tree names. DNA barcoding analysis was performed for 50 of the most common trees in the study area (Appendix 3.1). For the 6,791 stems recorded in this study, 71 % of stems were identified to family level or below, with more than half of the sites having 85 % or more of their stems identified. In total, we identified 75 families and 140 genera. However, botanical surveys are ongoing since 2010 and taxonomy is still being defined.

3.2.3 Biomass Estimates

Trees, as a broad category, included woody broadleaf and conifers, palms, woody lianas, tree ferns and pandanus. For woody trees (broadleaf and conifers), which comprised 80.2 % of the stems, we used Eq. 3.1 by Chave et al.(2005) to estimate AGB as this equation performs well across a broad range of tropical forests (Rutishauser et al. 2013). For other stem types, including standing dead trees and lying dead trees, we used allometric equations outlined in Appendix 3.2, Table S3.1). We used Eq. 3.2 to transform AGB estimates on slopes to a horizontal projection (Pearson et al. 2005).

Eq.3.1 *AGB (dry biomass)* = $[0.0776 \times (\rho \times D^2 \times H)^{0.940}]$

Equation 3.1. Allometric equation that estimates live tree dry biomass in wet tropical forests (Chave et al. 2005). Above ground biomass 'AGB' in kilograms is estimated using three parameters; tree diameter measured at 1.3 m above the ground 'D' in centimetres, tree height 'H' in meters and wood specific gravity 'ρ' in grams per cubic centimetre.

$$\boldsymbol{Eq.3.2} \quad A_P = W(L_F \times cos\,\mathrm{S})$$

Equation 3.2. AGB values were transformed from a slope projection area to a horizontal projection area (Pearson et al. 2005), where 'A_P' is horizontal projected area, 'W' is width of plot measured in the field and 'L_F' is length measured in the field, and 'S' is slope of predominant angle in degrees measured with a clinometer (Pearson et al. 2005).

3.2.4 Climate and Edaphic Variables

Site latitude and longitude obtained by hand-held GPS. Mean annual precipitation (MAP), mean annual temperature (MAT), and intra-annual temperature range (maximum temperature of the warmest month minus the minimum temperature of the coldest month) were derived from BIOCLIM (Hijmans et al. 2005) with a spatial resolution of 10' for temperature and 5' for rainfall variables. Evapotranspiration (E) was obtained from CGIAR-CSI Global-PET and Global Aridity Index (Global-Aridity) database (http://www.cgiar-csi.org) and the ratio of precipitation to E (P/E) was used as a measure of water availability for plants (Bowman et al. 2014). Radiation was derived from topographic position calculated without taking into account cloud cover (Ramachandran Nair et al. 2009, Kriticos et al. 2012). Soil organic carbon (SOC), soil pH, root mass, carbon and nitrogen ratios (C/N) and soil depth was provided from Dieleman et al. (2013) who analyzed 497 samples from 87 plots covering all nine elevation zones in the study area.

3.2.5 Data Analysis

Exploratory analyses

We undertook a range of exploratory analyses that directed subsequent analysis. We inspected pairwise plots to confirm that the various slope and aspect categories were well represented in each of the nine elevation zones along the altitudinal gradient; omitting elevation zone 50 from this test as sites in this zone had zero aspect and slope (Appendix 3.3, Fig. S3.1). We also examined pairwise relationships between AGB, altitude, climatic and edaphic variables (Appendix 3.4 Fig. S3.2). This revealed two results relevant to our subsequent analysis; 1) the relationship between AGB and altitude was bimodal rather than linear and 2) most of the climate and edaphic variables were strongly correlated with altitude and with each other. We therefore tailored our various subsequent analyses to accommodate these preliminary findings.

Models of disturbance

We were interested in two aspects of disturbance; 1) how natural disturbances affects AGB along the altitudinal gradient and 2) how the probability of disturbance changes with slope, given that a major cause of natural disturbance is land slip in this system. For the first analysis, we used mixed-effects generalized additive models (GAMs) to fit non-linear spline functions to the relationship between ln-transformed AGB and altitude (Wood 2006). GAMs are semi-parametric tests that are based on generalized linear models (GLM) but provide smooth response to the explanatory variable without setting a priori relationships (e.g. linear, logarithmic, power). Two models were fitted; one with separate splines for disturbed and intact sites and the other with the same spline fitted to disturbed and intact sites, with an additive disturbance effect. In both models, elevation zone was included as a random effect to account for clustered sampling within each zone. The two models had similar explanatory power, but the simpler model that included the same spline function for disturbed and intact sites had a lower Akaike information criterion (AIC, 134.8 versus 140.2), therefore we present this model.

For the second disturbance question, which aims to explore environmental correlates of disturbance; we treated the binary disturbance variable as the response in a mixed-effects logistic regression that included slope category as the only explanatory variable (fixed effect), and elevation zone included as a random effect.

Models of biomass potential

For the remaining analyses we excluded disturbed sites, as out interest was in the determinants of AGB potential along the altitudinal gradient. Our exploratory analyses revealed that AGB was not linearly related to altitude or any of the climate or edaphic variables (Appendix 3.4 Fig. S3.2). In fact these relationships tended to be bimodal. We therefore adopted a three-stage approach for analyzing AGB data.

First, we fitted a basic generalized additive model (GAM, without random effects) to the bimodal relationship between ln-transformed AGB and a key climate variable (moisture availability: MAP/MAPET). We then extracted the residuals from this model to test if the AGB variation, not explained by moisture availability, could be explained by edaphic variables (Appendix 3.5 Fig. S3.3). To do this, the residuals of this model were treated as the response variable in a linear mixed-effects model, with soil depth (plot-scale) and mean pH (per elevation zone) included as fixed effects and elevation zone included as a random effect. Second, we investigated how key forest structural attributes were related to AGB using linear mixed-effects models. Because large trees contribute substantially to AGB in our study, we modeled ln-transformed AGB as a function of the number of stems > 50 cm DBH per ha and the average height-diameter ratio (of stems >50 cm DBH). Again, elevation zone was included as a random effect to account for clustered sampling within zones. Third, we investigated how these key structural attributes were related to key climate and edaphic variables using GAMs. Because these environmental variables were so correlated we fitted them one at a time.

Statistical analyses were executed in R 3.0.3 (R Core Team 2013). The *nlme* package (Pinheiro et al. 2007) was used to fit linear mixed-effects models, the *lme4* package (Bates and Maechler 2009) was used to fit mixed effects logistic regressions and the *gamm4* package (Wood and Scheip 2012) was used to fit generalised additive mixed models. Post-*hoc* multiple comparisons were conducted for the model of disturbance probability using the *glht* function in the Multcomp package (Hothorn et al. 2008).

Comparison with other published biomass estimates from PNG

We compared our AGB values from Lowland (0 -1,000 m asl), Montane (1,000-2,800 m asl) and Upper-montane forests (>2,800 m asl) to those reported from other published field-based studies in PNG by conducting a search for available peer-reviewed journal articles.

3.4 Results

Exploratory results

AGB exhibited a bimodal relationship with altitude (Fig. 3.4). The first AGB peak was at elevation zone 50 (604 Mg ha^{-1} SD ± 258) and the second at elevation zone 2,200 to 2,400 (458 Mg ha^{-1} SD ± 140). These two peaks had significantly higher AGB than the other elevation zones (ANOVA $F_{8, 185} = 11.1$, $P < 0.0001$, Tukey Post-hoc). AGB also had a bi-modal relationship with climate and edaphic variables because these were highly correlated with altitude (Appendix 3.4). For this reason, there was no clear direct relationship between AGB and climatic/edaphic variables.

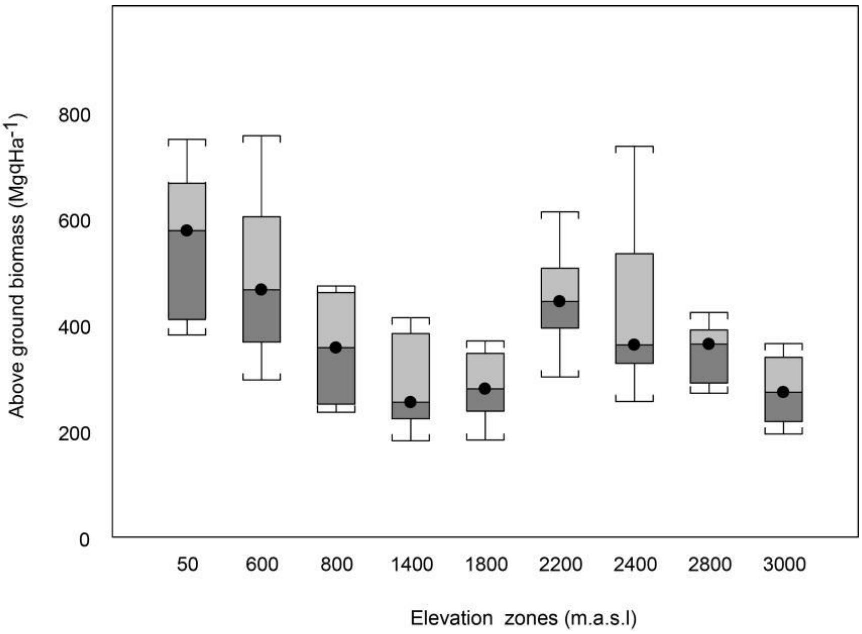

Figure 3.4. Above ground biomass estimates for nine elevation zones in primary forest of PNG (50m asl to 3,100m asl, N = 193). *The line across the middle of each box represents the median; the upper (light grey) and lower (dark grey) section of the boxes show the interquartile range around the median. The whiskers represent the 10th to the 90th percentile range of the spread.*

Disturbance effects on AGB

Natural disturbance, caused by landslides and windthrows, explained more variation in AGB than all other environmental variables obtained for this study (Appendix 3.6, Table S3.2). The model with same spline fitted to disturbed and intact sites, with an additive disturbance effect had the strongest explanatory power (Fig. 3.5). Despite the same spline function being fitted to disturbed and intact sites, the two fitted curves followed a similar bi-modal trend (Fig. 3.5), however they were not parallel; disturbance caused greater reductions in AGB at lower altitudes. On average, disturbed sites had around 20 % (75 Mg ha^{-1}) less AGB than intact sites.

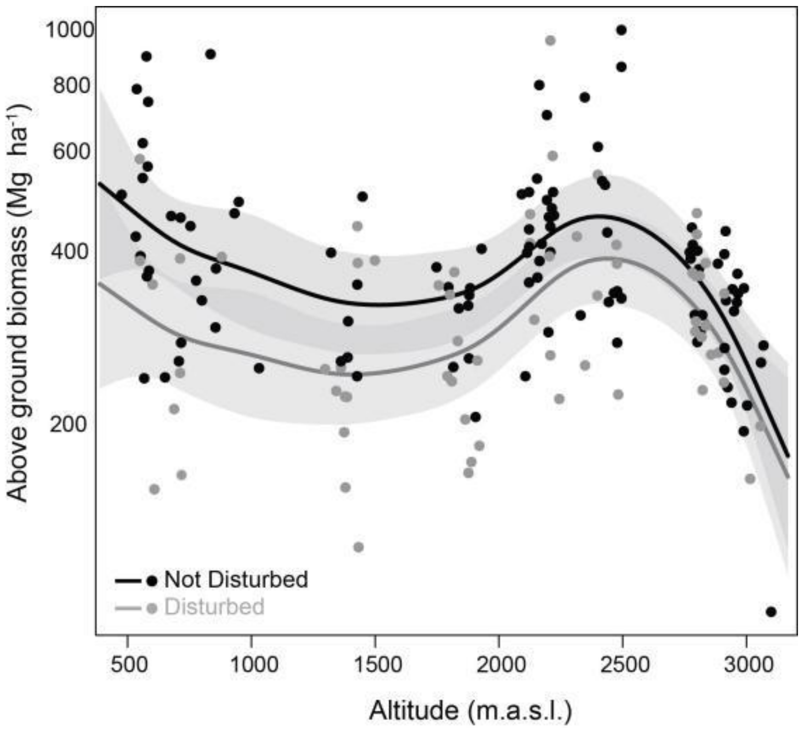

Figure 3.5. Relationship between AGB and altitude demonstrating the effect of natural disturbance in primary forests of the Huon Peninsula, PNG*. Sites with disturbance are shown as grey dots and without disturbance as black dots. Curves are fitted splines from GAMs for both undisturbed plots (black line) and disturbed plots (grey line). Shading indicates the 95 % confidence interval for each spline. We omitted the lowermost sites (50 m) from this analysis because of the high probability of human disturbance. Note the y-axis has been square root transformed.*

Probability of natural disturbance

The best environmental predictor of natural disturbances was the slope angle of terrain. The logistic model assessing how the probability of natural disturbance changed with slope (regardless of altitude) revealed strong positive effects of slope (Fig. 3.6, mixed-effects logistic regression, $Z = 2.35$, $P = 0.049$). Steeper sites (26º to 80º) were twice as likely to be disturbed as those on gentle terrain (disturbance probability of 0.27), but were as likely to be disturbed as intact (disturbance probability of 0.5).

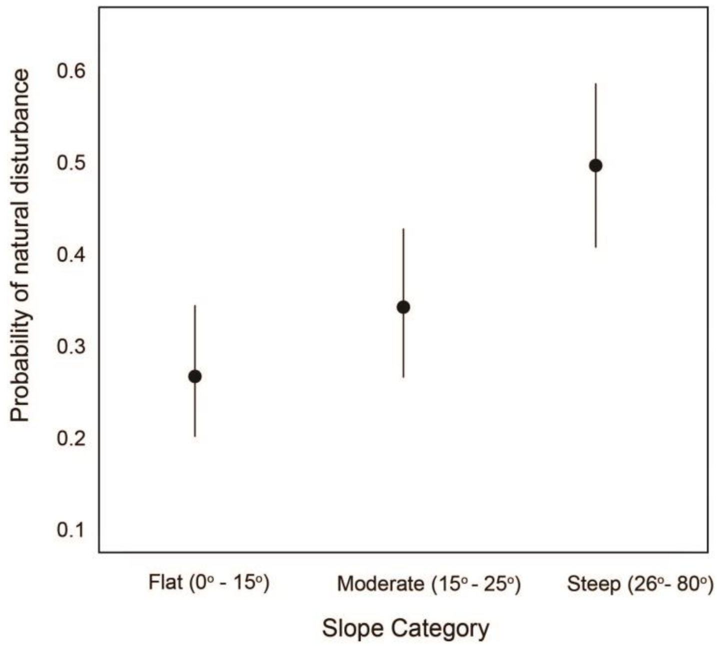

Figure 3.6. The effect of slope on the probability of natural disturbance in the study area. *Disturbance probability increased from 27 % on gentle terrain to 50 % on steep terrain. Analysis was by mixed-effects logistic regression with elevation zone included as a random effect, and pair wise differences amongst slope categories were tested using a post-hoc multiple comparison (Z = 2.35, P = 0.049). The error bars indicate the 95 % CI.*

Models of biomass potential

Edaphic factors

Soil depth and soil pH were poor predictors of AGB variation, even after accounting for the bimodal relationship between AGB and moisture availability (soil depth: $t = 1.62$, $P = 0.164$; mean soil pH: $t = 0.45$, $P = 0.671$, Appendix 3.6).

Forest structure and AGB

Forest structure attributes used to calculate AGB (e.g. DBH, height, WSG) had significant negative relationships with altitude, with the exception of stand density (Table 3.2). Mean canopy height decreased from 44 to 24 m from lowest to highest elevation zone, while WSG decreased from 0.58 to 0.48 g cm^{-3}, average DBH decreased from 37.3 to 28.5 cm and stem density increased from 914 to 1,703 ha^{-1} (Table 3.2). The strong relationship between altitude and stem density was caused by the increase in smaller stems (DBH < 50cm), as the number of large stems (DBH ≥ 50cm) was not related to altitude (Table 3.2).

Table 2.2 Mean forest structure attributes for the nine elevation zones ranging from 50m asl to 3,100m asl in primary forest of PNG (N = 193).

Elevation zone	AGB (Mg·ha^{-1})	Height (m)		WSG[b] (g·cm^{-3})	DBH (cm)	Stand density (ha^{-1})	
		canopy	tallest tree			all trees	DBH ≥ 50
50	604.1	44	64	0.58	37.3	914	62
500	451.3	37	49	0.62	30.1	1095	45
800	324.2	35	46	0.54	31.4	824	40
1400	268.7	30	48	0.54	35.8	736	57
1800	273.2	30	38	0.49	31.0	1194	55
2200	458.1	32	45	0.51	35.0	1382	86
2400	427.5	31	40	0.50	33.3	1612	83
2800	335.2	26	35	0.51	27.0	2205	58
3000	266.8	24	32	0.48	28.5	1703	52

[b]WSG means are weighted by basal-area.

To examine how forest structure attributes (plot-level) related to AGB, we modeled ln-transformed AGB as a function of these. This model revealed a strong, positive linear relationship between ln-transformed AGB and the number of large stems (Fig. 3.7, $t = 7.25$, $P < 0.001$). The model was improved when accounting for the additive effect of the average height:DBH ratio of large trees per plot, which was also significantly positively related to AGB (Fig. 3.7, t = 2.55, P = 0.012). This model explained 43 % of AGB variation among elevation zones and 39 % within. While it is are likely that taller trees for a given diameter have larger AGB, these results demonstrates the importance of tree height as a driver of AGB within and among elevation zones.

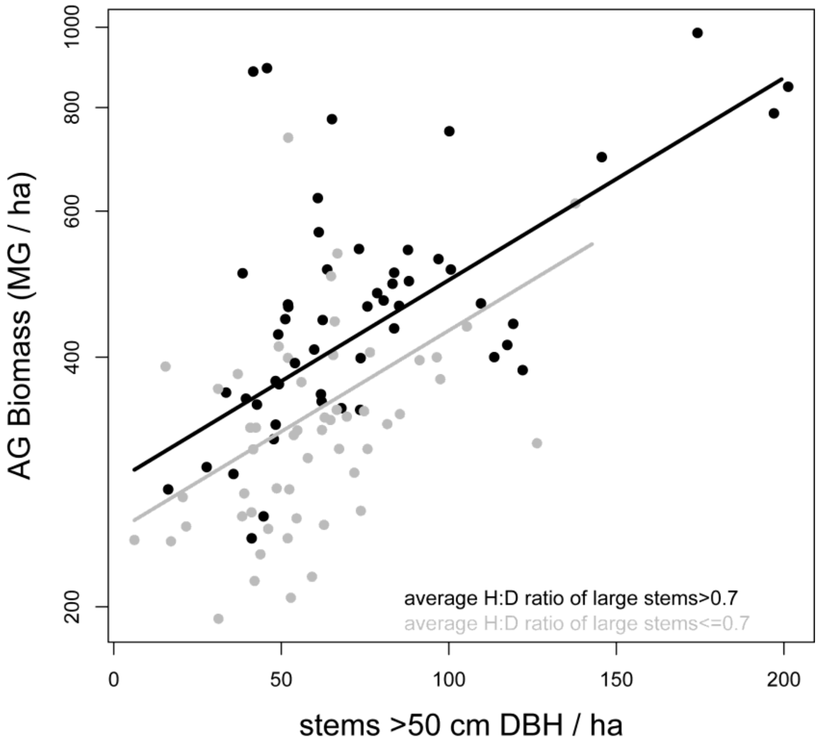

Figure 3.7. Relationship between AGB and the number of large trees in primary forest in the study area*. Black dots and the black fitted line represent sites with large trees having height to diameter ratio (HD) greater than the median (0.7). Grey dots and the grey fitted line represent sites with HD equal or lesser than the median. The model explains 43% of AGB variation among elevation zones and 39 % within elevation zones. Fitted lines are from a linear mixed effects model with elevation−site included as a random effect. Note the log-scale on the Y axis. Sites with signs of natural disturbances have been omitted.*

Incidence of large trees

We found a highest abundance of large trees in forests at 2,000-2,500m asl (Table 3.2). At high altitude, fifteen families were represented by trees > 70 cm DBH, and these ranged in height from 20 to 41 meters (Appendix 3.7, Table S3.3). Of these, the tallest angiosperms were *Dryadodaphne crassa* (40 m), *Nothofagus starkenborghii* (41 m), Elaeocarpus *sp.* (40 m), *Caldcluvia nymanii* (39 m), *Endospermum medullosum* (33 m) and *Saurauia capitulata* (30 m) while the tallest gymnoseperms were *Dacrydium nidulum* (35 m) and *Libocedrus papuana* (31m).

Climate and the density of large stems

The abundance of large trees followed similar hump-shaped relationships with moisture availability, MAT and intra-annual temperature range (not shown). Here, we report the results for moisture availability (Fig. 3.8) because these climate variables were strongly correlated (Appendix 3.4). The peak in large stem density occurred where moisture availability (P/E) was 2.8, and this coincided with MAT of 13.7ºC and an intra-annual temperature range of 7.5ºC. These conditions were found between 2,200 m asl and 2,500 m asl.

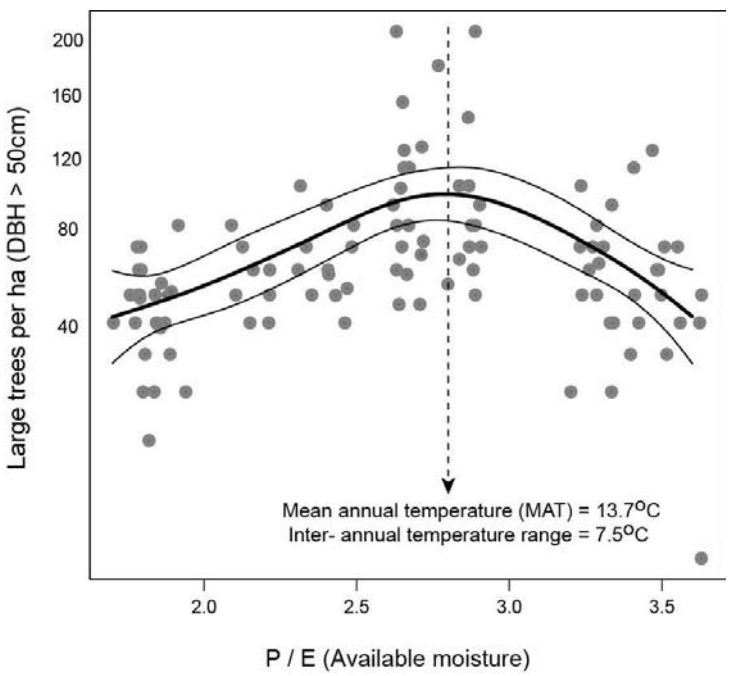

Figure 3.8. Relationship between the number of large stems and the ratio of precipitation to evapotranspiration (P/E), a measure of moisture available for trees. *Optimal MAT and inter-annual T in our study area. Note the square-root transformation of the y axis.*

Finally, we compared the AGB result from our study to those published for PNG. We found that AGB values for wet tropical Lowland forests in our study (496 Mg ha^{-1}) were more than double the values reported from other field assessments of PNG's Lowland forests and a third higher than the IPCC default values (Table 3.3). In Montane forest, the AGB value we report is 16-24 % higher than other PNG studies (Table 3.3).

Table 3.3 Comparison of primary forest above ground biomass from different studies in PNG.

Forest type (m asl)	Study	AGB Mg (ha^{-1})	Number of plots	Plot size (ha)
Lowland 0−1000	This study	496	56	0.1
	IPCC biome value	360	n/a	n/a
	Bryan et al 2010	252	6	n/a
	Fox et al 2010	212	10	1.0
	Bryan et al 2010	192	6	n/a
Montane 1000−2800	This study	368	90	0.1
	Edwards & Grubb 1977	310	1	0.24*
	Fox et al 2010	282	2	1.0
Upper - montane >2800	This study	310	47	0.1
	IPCC biome value	71$^{\beta}$		

*assessed by direct measures through destructive sampling. $^{\beta}$ (IPCC 2006b)

3.5 Discussion

In order to improve our understanding of the global carbon cycle and to predict how it will respond to global climatic and atmospheric change, it is important to reduce the uncertainty in estimates of globally important forest carbon pools, such as those in the forests of Papua New Guinea. In addition, this represents an important first step toward a country's participation in climate change mitigation initiatives under the UNFCCC. Our study presents the first assessment of above-ground biomass in PNG's high altitude forests, revealing that they are amongst the world's most carbon-rich at those altitudes. It would seem that the montane forests of PNG contain some of the tallest individual trees ever recorded globally at about 2,500-3,100 m asl. No direct relationship was found between AGB, climate and edaphic variables and the strongest of predictors of AGB were the occurrence of natural disturbance and the abundance of large trees (DBH > 50 cm). The abundance of large trees was related to a set of climatic variables, but the relationships were unimodal following an optimum curve, rather than linear.

The AGB trend we found with elevation was bi-modal, with two distinct peaks in Lowland and mid-Montane forests (2,200-2,500 m asl). The general trend reported for tropical forests worldwide is one of decreasing AGB and tree stature with increasing altitude (Kitayama and Aiba 2002, Leuschner et al. 2007, Girardin et al. 2010, Homeier et al. 2010, Moser et al. 2011, Girardin et al. 2013). However, a secondary peak in AGB occurring at high elevation has been observed in Southeast Asia and Africa where these altitudes coincide with the presence of large trees (Culmsee et al. 2010, Marshall et al. 2012).

One of our most striking observations was that large trees, the major driver of AGB patterns in our study, were associated with a set of optimal climatic conditions. These conditions, found in higher elevation forests in PNG, are remarkably similar to those which occur in regions identified as the most important for carbon storage in forests around the world, mostly found in temperate maritime regions (Keith et al. 2009). Optimal conditions were high water availability (P/E ratio of 2.8); moderate temperatures (MAT of 13.7ºC) and small intra-annual temperature variations (7.5ºC). These optimal climate conditions for the growth of large trees occur in maritime areas with substantial fog cover (Larjavaara 2014). For example, the foggy mid-west coast of the USA boasts the largest gymnosperm (*Sequoiadendron giganteum*) and the moist south-eastern coast Australia boasts the tallest angiosperm (*Eucalyptus regnans*) (Keith et al. 2009). This climate envelope has been identified outside of temperate regions in tropical montane areas covered in cloud. However, most studies report tropical montane forests having smaller trees than at lower elevations (Raich et al. 2006, Girardin et al. 2013).

Growing large and tall likely has many ecological advantages for trees, particularly where competing individuals are also large and tall (Becker et al. 2000), but it is energetically and resource intensive, thus requiring certain conditions to be met. The cloud emersion zones could also moderate intra-annual temperature leading to lower energy requirement for acclimating metabolism to varying thermal regimes, and with rainfall being more evenly spread throughout the year, thereby minimizing seasonal droughts that might lead to hydraulic failure in tall trees (Zhang et al., 2009; Larjavaara & Muller-Landau, 2012).

Constant cloud cover and reduce winds which cause windfalls and low temperature slow decomposition which allow for deeper soils.

Above 2,500 m asl, canopies are seldom higher than 15 m - 20 m (Raich et al. 2006). Our sites contained individuals reaching over 40m tall at 2,700 m asl (e.g. *Nothofagus starkenborghii, Dryadodaphne crassa, Caldcluvia nymanii*) and over 30 m tall at extreme altitudes of 3,000 m asl (e.g. *Dacrydium nidulum, Libocedrus papuana, Elaeocarpus sp and some Myrtaceae sp*). However, tropical montane forests are generally poorly represented in research (Malhi et al. 2010). This is especially true for PNG where montane forests have mostly been superficially described, with only some detailed ecological work provided by Wade and McVean (1969). Edwards and Grubb (1977) provided the only other account of forest biomass in high altitude forest at 2,400 m asl in PNG and although the plot was small (0.24ha) they remarked on the exceptional height and girth of trees at that altitude (Edwards and Grubb 1977). Therefore the general description of montane forest as being gnarly and squat may not always hold true and more field research is needed, particularly in areas where optimal climate conditions have been identified.

Reducing uncertainties from important forest carbon stores such as those in PNG is an important first step towards improving models of carbon stocks and fluxes (Mitchard et al. 2013). The most useful models are those with a parsimonious set of explanatory variables readily available and that have strong predictive power. However, weak relationships between AGB and most of the climatic and edaphic variables in our study suggests that simple AGB-climate-edaphic models may not be suitable, and process-based models that take into account tree size and disturbances may be more suitable (Stegen et al. 2011).

Instead we found that results demonstrate that natural disturbance explained more variation in AGB than all other environmental factors. Papua New Guinea, and many tropical montane areas have episodic natural disturbances from landslides and windthrows that can cause substantial reductions in AGB (Dalling 1994). However, including natural disturbance in models that predict AGB can be difficult (Ozdogan 2014). We found that steeper sites (26º to 80º) were twice more likely to show signs of natural disturbance and when present, natural disturbance reduced AGB by 20%.We have demonstrated that the relationship between AGB and slope terrain could be incorporated into models using slope alone, a new insight gained from sampling on very steep slopes using rappelling equipment.

Our study had a number of limitations. First, field inventories were conducted on a single elevation transect and would have benefited from contrasts with climatic and edaphic relationships developed with AGB elsewhere in PNG. Also, obtaining data from weather stations, which are more accurate than global syntheses and do not conceal local variation, could provide better insights into the relationship between AGB and climate. This may become possible in the future as four weather stations have recently been installed in the area. Moreover, we suspect that soil depth and other soil attributes had significant impacts on AGB but we were not able to demonstrate the magnitude of these impacts, most likely because soil depths were measured to a maximum of 2.0 m deep and 80 % of the soil plots above 2,200 m asl where 2.0 m or deeper. Finally, our study design, which consisted of sampling many plots in order to capture spatial variation, may have led to heightened correlations of AGB to large trees (Stegen et al. 2011).

In conclusion, the high biomass of montane forest is most likely attributable to a conjunction or 'sweet spot' of environmental conditions. The island of New Guinea has one of the most reliably wet and least variable climate regimes on the planet (Hall 1984, McAlpine et al. 1983). Hence, the effects of predicted changes in temperature and rainfall could have severe consequences on cloud cover, the defining feature of montane cloud forest ecosystems and the large trees found in these regions (Karmalkar et al. 2008, Ponce-Reyes et al. 2012, Ponce-Reyes et al. 2013, Sheil 2014).

3.6 Summary of Chapter 3

- Our study revealed significantly higher forest carbon stocks than previously reported for PNG as prior inventories were mostly located in areas close to infrastructure.
- Though altitude was a reliable proxy for changes in climate and soil variables, no direct relationship was found between AGB and most of the climatic and soil variables.
- The two best predictors of AGB were the distribution of large trees (DBH>50 cm) and the occurrence of natural disturbances; new insights into the relationships between AGB, natural disturbance and topography were gained by sampling on slopes ranging from gentle to very steep (up to 80º).

- High-altitude forests, above 2,500 m asl, were among the world's most carbon-rich at those elevations (e.g. 313 Mg ha^{-1} at 3,000 m asl); this was caused by a high abundance of tall tree species, reaching to 40 m tall in high alpine forests
- Large trees and their distribution were most strongly associated with climatic conditions similar to those known to coincide with large carbon stocks in temperate regions. These conditions, high water availability (precipitation to evaporation ratio of 2.8) moderate temperatures (mean annual temperature of 13.7ºC) and small annual temperature variations of 7.5ºC, were found in the cloud emersion zone of the elevation transect.
- Finally, the predicted effects of changes in temperature and rainfall in montane forests could have severe consequences on the persistence of large, old trees found in these regions, potentially threatening important carbon stores and biodiversity.

-END OF CHAPTER 3-

Chapter 4

Validating Community-led Forest Biomass Assessments

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4.1 Overview

This Chapter evaluates the robustness of data collected by forest monitoring programs that employ local people. The study conducts a complete expert re-measurement of community-led biomass inventories in remote tropical forests of Papua New Guinea, including 4,481 experts' and non-experts' measurements of tree diameter, height, numbers of trees and plot surface area. The study design allowed for the detection of errors in the field and how these contribute to discrepancies in forest biomass estimates.

4.2 Introduction

Potential exists to mitigate anthropogenic climate change by reducing the rates of forest carbon loss and increasing forest carbon sequestration (Malhi 2010). Acknowledging this potential, the international community has developed mechanisms to reward efforts that halt and ultimately reverse forest carbon loss in developing countries (Venter and Koh 2011). To take part in these efforts, developing countries must first establish their own system for monitoring carbon stocks and fluxes from their forests (Asner 2011).

For important carbon pools, such as those of tropical forests, the Intergovernmental Panel on Climate Change (IPCC) recommends a high level of accuracy in monitoring (Tier 3), attained through a combination of field and remote-sensing inventories (IPCC 2006a). However, field inventories are resource intensive because they require teams of experts to work, often in remote locations, for extended periods. For this reason, the capacity of most developing countries falls short of the requirements needed to monitor forest carbon stocks. Only 7 of the 99 non-Annex I countries in tropical regions can perform field-based forest biomass-carbon inventories (Romijn et al. 2012). Clearly, additional resources and more effective methods are needed.

One option for increasing the capacity of developing countries is to engage local people in monitoring forest carbon stocks (Danielsen et al. 2011). Aside from contributing directly to the monitoring process, local participation can help improve natural resource management (Sheil and Lawrence 2004), the likelihood of permanence in emissions reductions and provide alternative livelihoods for those who forego destructive forest exploitation (Chhatre and Agrawal 2009). Additionally, these types of programmes could efficiently channel REDD+ (Reducing Emissions from Deforestation and Forest Degradation) incentives and rewards toward local communities (Chhatre and Agrawal 2009). For this reason, the potential for local people to contribute to the monitoring process has not gone unnoticed. Community forest monitoring was identified as a central component of national REDD+ readiness plans for multiple countries (Skutsch 2011). Fledgling programmes to harness this capacity are already in place in Tanzania, Nepal and Papua New Guinea (Skutsch 2011). Furthermore, the Subsidiary Body for Scientific and Technological Advice (SBSTA) recognises and promotes the engagement of Indigenous Peoples and local communities in monitoring and reporting activities (SBSTA 2009, Van Laake et al. 2011).

Given that community monitoring programmes are emerging as an important component of emissions reduction policies, there is a need to demonstrate whether data collected by these programmes are robust enough to comply with international monitoring requirements and to improve these programmes where possible (Palmer Fry 2011). Historically, most natural resource monitoring has been conduct by experts; though some studies in the field of conservation biology have shown that non-experts can monitor data as reliably as experts (Elbroch et al. 2011, Oldekop et al. 2011). However, few studies have looked specifically at the quality of community forest monitoring programmes.

Recently, three studies have quantitatively assessed the differences in experts' and non-experts' field measurements of forest biomass stocks. The first study by Danielsen et al.

(2011) involved a post-*hoc* comparison of existing community-based forest biomass inventory to that of experts in India and Tanzania. They found no significant differences in the forest biomass estimates, an important preliminary demonstration of the quality of non-experts' biomass surveys. The second study from Butt et al. (2013) involved a more detailed analysis and an expert data check from a subsample of non-experts' (volunteers) field measurements in the Oxford Forest, United Kingdom. Measurements were taken by both experts and non-experts of tree diameters and height, and from these results, a sampling error was produced. The study demonstrated that non-experts' measurements had a greater sampling error than experts' measurements. The third study by Danielsen et al. (2013) was a resampling campaign of biomass inventories that compared tree diameter measurements and tree counts from experts and non-experts in Southeast Asia. The researchers found that though biomass estimates were generally similar, the estimates differed significantly in one-third of the sites. They also found that tree diameters were significantly different for half the measurements. These three studies demonstrated that non-experts' biomass estimates and the field measurements required to calculate these estimates were generally of good quality. However, discrepancies did exist, and it remains unclear what types of error cause the largest discrepancies in forest biomass assessments.

Most of the countries participating in REDD+ activities are currently in the preparation phase, which involves the establishment of a national monitoring system (Maniatis et al. 2013). Therefore, refining the accuracy of the data produced by community monitoring programs is timely. Improving on the methodologies for data collection first requires the determination of the types of errors that are the most important, namely, which ones lead to the biggest discrepancies in forest biomass estimates. This study attempted to address this issue. We performed a full expert re-measurement campaign of community-led forest biomass inventories in the remote forests of Papua New Guinea. We compared 4,481 experts' and non-experts' measurements of tree diameter, height, numbers of trees and plot surface area. The study design allowed for the detection of errors in the field and how these contribute to discrepancies in forest biomass estimates. The study revealed unexpected results that could serve to improve forest biomass inventories and training protocols for experts and non-experts alike.

4.3 Methods

4.3.1 Ethics Statement

The study was approved by the James Cook University Human Research Ethics Committee (HREC). Written consent was not sought nor required by the Ethics Committee because the study did not have the aim to assess the capacity of 'individuals' but instead of 'programmes' that employ local people. Prior to our undertaking the work, community meetings with local landholders were held. We, the scientists, relied on customary decision-making by the landholders for determining the duration and location of, and participants in, the study. After a three-day training session with the participants, the participants were well informed and consented orally to the study. No minors participated in the study. The authority that issued the permit to work in the YUS (Yopno-Uruwa-Som) Conservation Area was the Tree Kangaroo Conservation Program (TKCP).

4.3.2 Study Area

(See Section 3.3.1)

4.3.3 Participants

The local participants of this study were landholders who pledged land to the YUS Conservation Area. The pledged land formed the first and only protected area under PNG's *Conservation Area Act 1978*, through the efforts of TKCP. About 30 villages with a population of ~12,000 are associated with the conservation area. The area features rugged topography and no road access, and the predominant livelihood is subsistence farming with a high dependence on forest resources. Only 1 % of households in the area earned a monetary wage. In 2011, 40 % of adults had never attended an educational institution, and those who did had attended for an average of six years (Cornelius and Murphy 2012). Local participants for this study were chosen by committees of local landholders and were paid a standard wage set by the TKCP. To promote the exchange of knowledge, we requested that the teams consist of at least one person with traditional knowledge of the forest and at least one young person (< 25 years old). We formed three teams from three different communities, each consisting of six people from different language groups.

4.3.4 Study Design

Each team individually took part in a three-day training session aimed at teaching self-directed forest biomass inventory techniques. The training consisted of knowledge exchange on the role of forests in climate change, drill exercises and games with compasses, diameter tapes, clinometers, survey tapes and GPS, and random systematic site-selection techniques. The final day was dedicated to a full inventory 'practice-site' followed by a discussion on adaptive teaching and learning to address the most common mistakes and challenges. A fourth day of training involved only the team leader and aimed at standardising data collection.

Self-directed inventories by non-experts took place two weeks after the training. Forty-one plots were established, 14 plots from two of the teams and 13 from the third. In total, the study took 80 field days to complete between November 2010 and April 2012. Unassisted local teams conducted plot establishment and recorded the number of trees in the sample and measured their diameter and height. Local teams also collected the altitude and site coordinates with a GPS unit. The angle of the predominant slope, necessary for calculating the horizontal plot area, was recorded with a hypsometer (Pearson et al. 2005). These slope angle values were classified in one of the following categories: (≤ 11˚ = gentle), (12˚ to 25˚ = medium), (26˚ to 45˚ = steep) or (45˚ to 90˚ = very steep).

Methods for plot establishment and estimating above-ground forest-biomass inventory were largely based on widely used LULUCF protocols (For more details see Section 3.3.2) (Pearson et al. 2005) and tree height measurement protocols from Feldpausch et al. (2012). The plots (20 m x 50 m) were delineated using compasses and survey tapes (Fig. 3.3, Chapter 3). Trees were counted in appropriate subplots and tagged with a unique ID. Diameters at breast height (DBH) were measured to the nearest millimetre with a diameter tape at 1.3m above the ground. Heights were measured by standing directly below the crown and measuring the highest point in the canopy with a rangefinder (LaserAce® hypsometer) multiple times until the highest point was reliably identified. Wood Specific Gravity (WSG) was from standard values from the Asian rain forest dataset (Chave et al. 2006, IPCC 2006b, Fox et al. 2010). Tree species identification to determine WSG was done on botanical specimens by experts and by DNA-barcoding analysis of leaf samples collected in the field (Appendix 3.1). Where possible, non-experts also recorded tree species' names in the experts' languages (Appendix 4.1). For the site to be located for remeasurement by experts, the non-expert teams pegged the four corners of the plot markers; a GPS coordinate and the location description were also recorded. The southwest corner plot was selected using a random systematic technique, and sites were expected to have a minimum 120 m distance from adjacent sites.

4.3.5 Validation Process

The 41 sites established and measured by communities (non-experts) were entirely re-measured by scientists (experts). Validation was performed in the same field-season after the community teams had finished their inventories, for a total of three field seasons. Experts worked in pairs, a total of six experts participated in this study, and all had completed a degree in the natural sciences.

The expert team verified the plot surface area set by the non-expert team by measuring the distance and angles of the four pegs demarcating the outer perimeters made by the non-experts with survey tapes and a compass. Additionally, the expert team verified the non-expert plot perimeter angle and distance with a laser hypsometer by shooting the laser from one peg to a receptor placed on another peg. Subplot areas were not verified. For the purpose of comparing expert and non-experts plot sizes, the expert team also delineated a new plot from the same southwest corner demarcated by the non-expert team, using compass and survey tapes; the experts verified the plot perimeter using a laser hypsometer. However, to avoid the discrepancies in tree count caused by different plot sizes of the experts and non-expert, we only compared tree counts from the overlapping expert and non-expert plots. The plot surface area was calculated using a trapezoid formula with the four distances and four angles measured from the non-expert plots.

Our sample area had a total of 1,433 trees; 1,364 of these were in both datasets (expert and non-expert) and of these, 1,281 tree height measurements were recorded. Some trees could not be assessed for height because of visual obstructions. DBH measurements were validated by experts using the same measurement point marked by the non-experts. Note that we did not validate WSG, the third parameter in the allometric model, as these values were obtained only by the expert team. For tree counts, trees were deemed to have been 'missed' by non-experts only when a tree within the designated plot had not been tagged and recorded by non-experts. However, trees were deemed 'extra' only if they had been tagged and recorded by non-experts but were deemed too small or too large for the subplot or if trees were recorded by non-experts but could not be located by the experts.

To estimate the dry biomass of trees, we used allometric equations for wet tropical forests formulated by Chave et al. (2005b). We chose this equation because it performs well across a broad range of wet tropical forests using DBH, height and WSG parameters (Eq. 3.1, Chapter 3). Carbon values from biomass estimate could be derived using a factor of 0.5, but we reported only values in biomass (in kg or tons) (IPCC Guidelines).

4.3.6 Statistical Analysis

We tested each of the pairs of measurements for deviation from normality using a Shapiro-Wilk test and found that the number of trees followed a normal distribution ($P > 0.5$), while the datasets for DBH ($W = 0.85$, $P = 0.0001$), height ($W = 0.87$, $P < 0.0001$), and plot surface area ($W = 0.56$, $P < 0.0001$) were not normally distributed.To test the effect of the different local teams on measurements, we used Kruskal-Wallis to compare DBH ($\chi^2 = 0.98$, df = 2, $P = 0.61$), height ($\chi^2 = 11.73$, df = 2, $P = 0.003$), plots surface area ($\chi^2 = 1.4$, df = 2, $P = 0.50$) and biomass (Fig. 4.1, $\chi^2 = 0.98$, df = 2, $P = 0.61$) estimates, and we used ANOVA to test the effect of teams on the numbers of trees ($F_{1,39} = 0.43$, $P = 0.51$). We considered the three non-expert teams as one group because our aim was to compare non-experts with experts, and not to compare non-experts with themselves (Fig. 4.1).

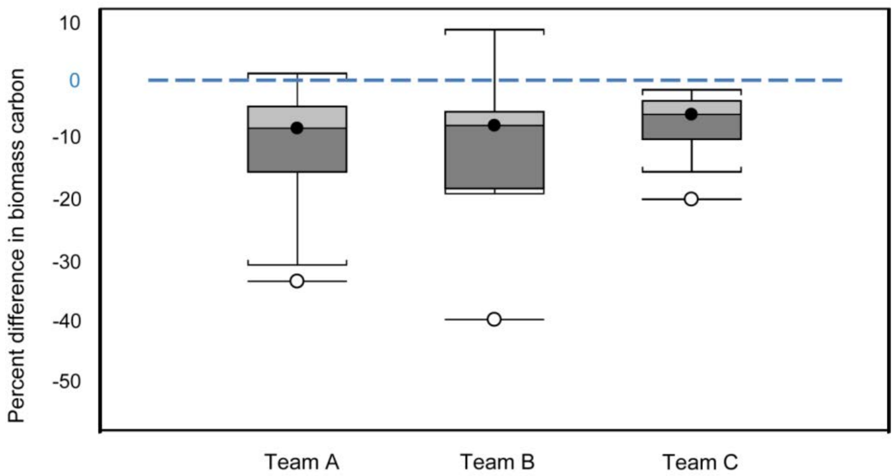

Figure 4.1. Percentage difference of biomass estimates of those produced by three non-expert teams (N = 41) and those produced by experts (N = 41)*. Team A and Team B had 14 participants, and Team C had 13 participants. The line across the middle of each box represents the median; the boxes show the interquartile range for around the median for half the data at the top and the other half at the bottom; the whiskers represent the 10^{th} to the 90^{th} percentile, and the outliers are demonstrated by the empty circles.*

To test for differences in the pairs of expert and non-expert values, we used the Wilcoxon rank-sum test for biomass estimates (number of pairs = 41), DBH measurements (number of pairs = 1,364), height measurements (number of pairs = 1,281) and plot surface area (number of pairs = 41). We used t-tests to compare the numbers of trees sampled at the site level (number of pairs = 41). To compare the expert and non-expert datasets, we used the Standard Major Axis (SMA) regression model because, as opposed to the Generalized Least Squares (GLS) regression, it does not assume that X is dependent on Y and that X is without error. However, we also report the analysis from the GLS regression as it is most commonly used in similar analyses (Sokal and Rohlf 1981).

To compare the DBH between expert and non-expert measurements, we used models that predicted the error of expert measurements from other studies (Condit 1998, Chave et al. 2004b). Condit (1998), evaluated 1,715 expert double-blind DHB measurements, and from these, Chave et al. (2004) built a model that described the expected measurement error of scientists. The first model from these studies described the smaller errors (the lower 95% of the error range), where the magnitude of the error was normally distributed and proportional to the DBH (Table S4.3, Appendix 4.3). The second model described the larger errors (the highest 5% of the error range) and the error had a fixed value. We used these two models to build the expected range of error using the expert DBH data from our study; we used a chi-square test to compare the range of errors observed in experts and non-experts to the range of error expected among experts (Table S4.3 Appendix 4.3).

We explored the influence of the discrepancy in DBH (% error) and height (% error) on discrepancy in tree biomass (% error) using a Generalised Linear Model (GLM) for trees that had both DBH and height measurements from expert and non-experts. We did not perform GLM at the landscape scale as assessing errors at this level required using plot averages for height and DBH which would have masked the variation from negative and positive values measured within the plot. Therefore, we took a hierarchical approach to determine what differences in measurement types from expert and non-expert measurements affected the overall differences observed in biomass (Fig. 4.2). First we compared all pairs of DBH and height (between teams at the tree level). Second, we calculated tree biomass (in kilograms) using expert derived measurement (DBH and H) and replaced only DBH with the value obtained by the non-expert and then calculated once more by replacing height only with the non-expert derived measurement, while always keeping WSG constant. In total, four biomass values were produced for each tree using Chave et al (2004) allometric equation, with the combination of the following sets: 1) $DBH_{(expert)}$ and $H_{(expert)}$; 2) $DBH_{(non\text{-}expert)}$ and $H_{(expert)}$; 3) $DBH_{(expert)}$ and $H_{(non\text{-}expert)}$; 4) $DBH_{(non\text{-}expert)}$ and $H_{(non\text{-}expert).}$ Third, the differences in biomass of the sum of each set (1-4) were calculated for the 41 sites to determine the difference in kilograms associated with DBH and height; we only included trees that had expert and non-expert measurements for height. Forth, we quantified errors introduced from missing/extra trees. Fifth, we quantified errors introduced by plot surface area (1000 m^2). Finally we extrapolated biomass errors from each measurement type at the landscape level and compared the final estimates.

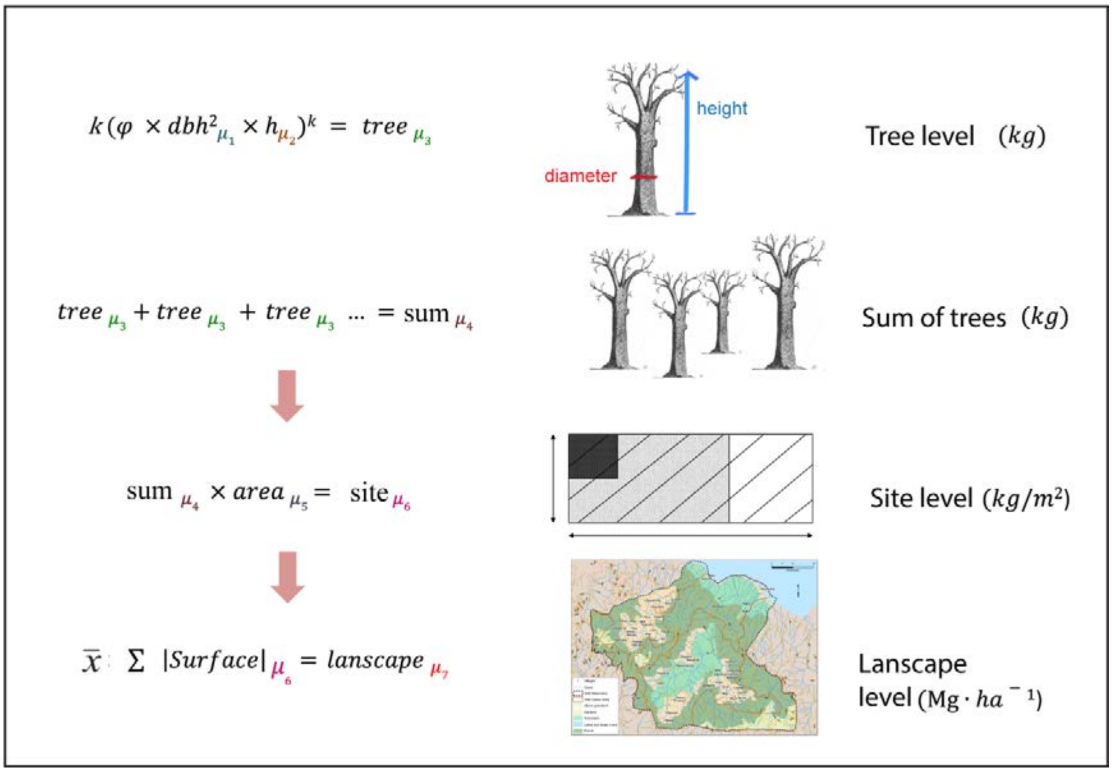

Figure 4.2. Schematic description of how errors (μ) propagate from the tree level to the landscape level in field biomass inventories.

All statistical analyses were conducted in the S-PLUS Enterprise Developer Version 8.0.4 for Microsoft Windows: 2007 Copyright (c) 1988, 2007 Insightful Corp and R version 2.15.1 (2012-06-22) 'Roasted Marshmallows' Copyright (c) 2012 the R Foundation for Statistical Computing.

4.4 Results

4.4.1 Biomass Discrepancies

First we present the final summary of the errors identified through our hierarchal approach; then, in the following section, we explore in details the different error types for each field measurements to further understand why non-experts produced generally lower biomass values.

For the 41 site pairs, experts' and non-experts' biomass estimates were not significantly different (Wilcoxon signed-rank: N = 41, W = 1.21, $P = 0.23$) and were strongly correlated (GLS: $F_{1,39} = 64.0$, $P < 0.001$ $R^2 = 0.62$). However, 78% of the non-experts' biomass estimates were lower than those of the experts (Fig. 4.1), resulting in a 9.1 % difference between experts' and non-experts' biomass estimates at the landscape level (Table 4.2).

Overall, we found that height measurements introduced the majority (71%) of tree level biomass discrepancies (GLM, F = 2691, df = 1, 1057, R^2= 0.714, $P < 0.0001$), while the combined DBH and H tree biomass error ranged from +15 % to –25 % (Fig. 4.3A). At the site level, discrepancies introduced from plot surface area and tree counts broaden the error range to +15 % to –74 % (Fig. 4.3B). In order of importance, total discrepancies in biomass at the landscape level were caused from height (41.7 %), missing trees (37.4 %), plot surface-area (12.1%) and DBH (8.8%) (Fig. 4.3C).

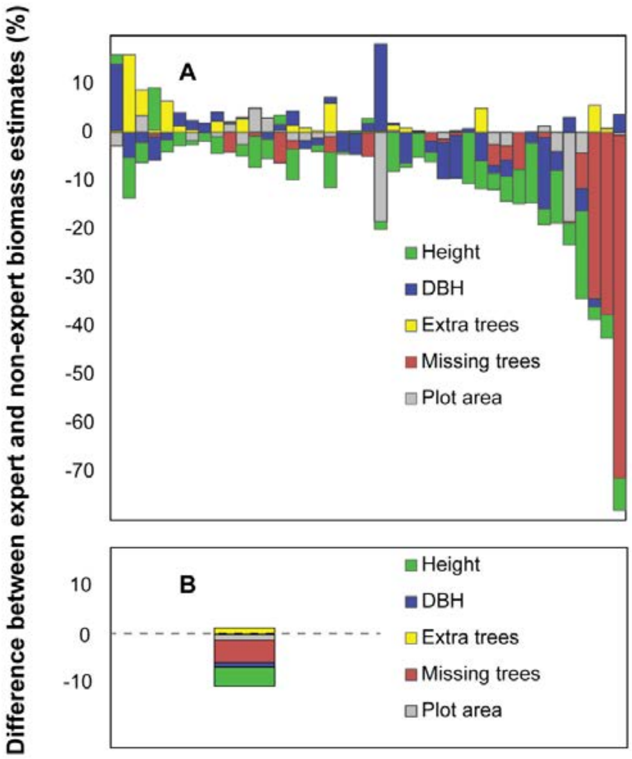

Figure 4.3. Difference in experts' and non-experts' biomass estimates due to measurement discrepancies. *A) At the 'tree level', each bar represents a site, B) at the 'landscape level', the bar represents the average of all site. Negative values represent smaller values of non-experts compared to experts.*

4.4.2 Tree-Level Measurement

There was no significant differences in experts' and non-experts' DBH measurements. More than a third of DBHs were exactly the same (Table 4.1, Wilcoxon signed-rank: number of pairs = 1,364, W = 2.1, P = 0.07). Experts' and non-experts' DBH values were strongly correlated (Table S4.2, Appendix 4.2, Fig. 4.4A, SMA R^2 = 0.99, GLS R^2 = 0.91) with a bias toward lower DBH in non-experts' surveys indicated by slopes of regressions having values significantly lower than one (Table S4.3, Appendix 4.3 Fig 4.4A, SMA and GLS, P < 0.0001). However, based on a model from Chave et al. (2004) and Condit (1998) that predicts the DBH error rate between experts (see the 'Methods' section and Table S4.1, APPENDIX 4.2), we showed that DBH error rates between experts and non-experts was significantly higher than the predicted error rate between expert measurements (small errors, χ^2 = 593.8, P < 0.0001; large errors, χ^2 = 13.4, P = 0.0002).

Table 3.1 Differences between expert and non-expert field measurements required their resulting discrepencies in forest biomass estimates.

Field measurements	**Mean percent difference$^{€}$ (95% CI)**	**Resulting error in biomass (%)$^{€}$**	**Total proportion of error (%)**
DBH*	0.23 (-0.1 to 0.6)	-0.8	8.8
Height	-1.7 (-3.0 to -0.4)	-3.8	41.7
Area*	-1.1 (-3.2 to 0.6)	-1.1	12.1
Missing trees*	32 of 1443$^{¥}$	-4.7	37.4§
Extra trees*	37 of 1443$^{¥}$	1.3	0§
Total forest-biomass discrepancy		-9.1%	100%

* Expert and non-expert measurement are not significantly different ($P \leq 0.05$). $^{€}$ Negative values means non-expert recorded lower values than experts. §Missing and extra trees are combined to demonstrate for the total discrepancies from number of trees sampled

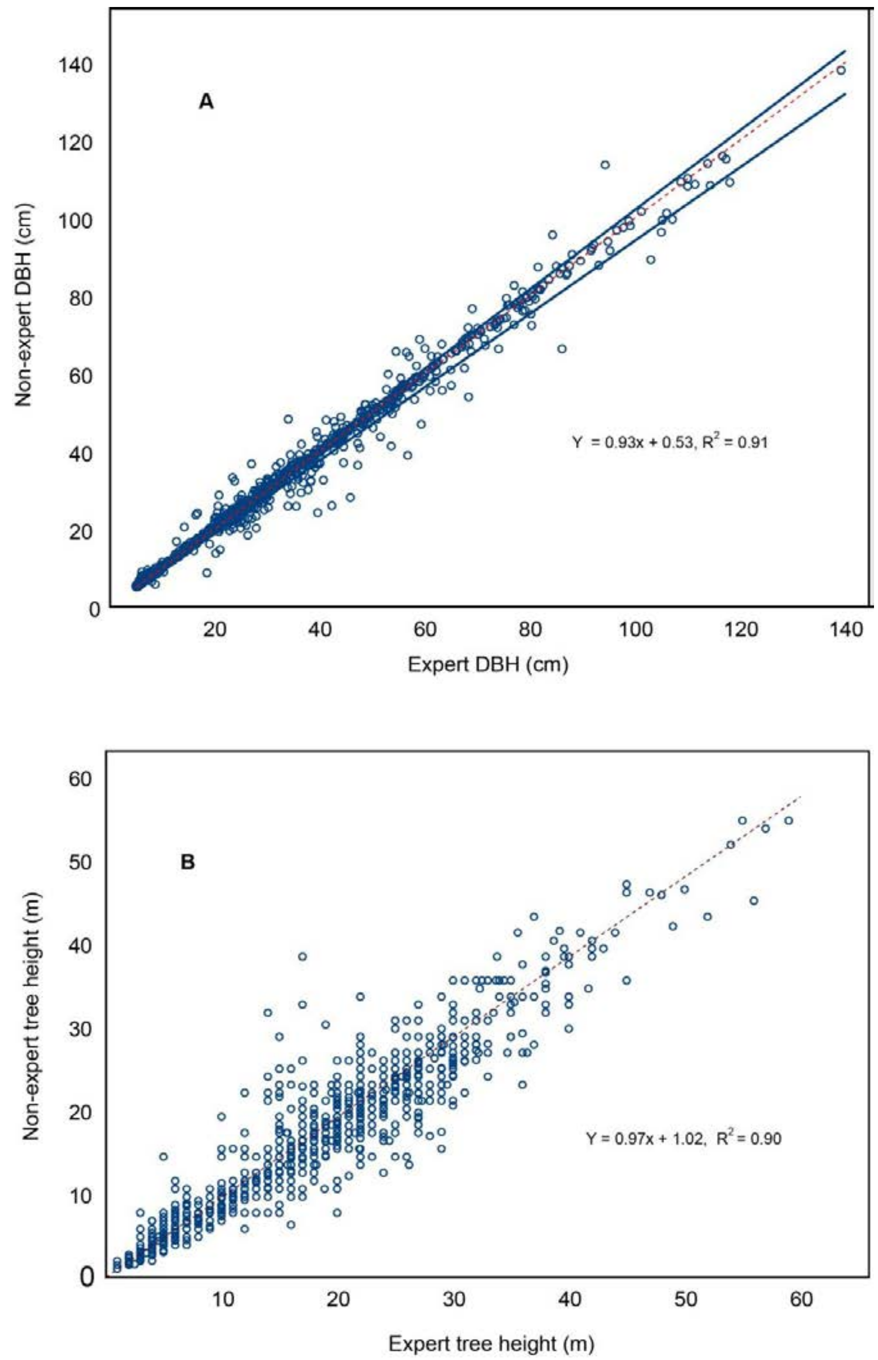

Figure 4.4. ***The relationship between expert and non-experts pairs of DBH (A; N = 1,364) pairs of height measurements (B; N = 1,281).*** *The dotted red lines represent 1:1 relationships, and the regression equation is from a GLM.*

Height measurements were significantly different between experts' and non-experts' surveys (Wilcoxon signed-rank: number of pairs = 1,281, W = –10.2, $P < 0.001$). In particular, the proportional error in height was significantly larger for big trees, with a 1.1% difference in measurement, which was about double the difference for smaller trees (Table 4.2, Kruskal-Wallis chi-square = 64.9, df = 2, $P < 0.001$). However, the relationships between height pairs were strong (Fig. 4.4B, SMA model: R^2 = 0.92 and GLS model R^2 = 0.90, Table S4.2), with most pairs (98.4%) varying by less than ± 10% (Fig.4.4B).

Table 4.2 Measurement errors for height and DBH and their contribution to discrepancies of forest biomass estimates.

DBH size category (cm)	Trees sampled	Percent of total biomass in study	Mean differences in DBH (cm) (95% CI)	Mean difference in height (m) (95% CI)	Percent difference in biomass
5.0 to 19.9	506	11.2	0.16 (0.09 to 0.24)	-0.33 (-0.45 to -0.21)	-0.9
20.0 to 49.9	711	27.5	-0.18 (-0.34 to -0.01)	-0.65 (-0.95 to -0.35)	-13.9
50.0 +	216	61.2	-0.73 (-1.34 to -0.15)	-1.15 (-1.64 to -0.65)	-85.2

Overall, most of tree level biomass discrepancies resulted from combined height and DBH measurement errors on large trees (DBH ≥ 50cm). Measurement errors on large trees caused 85% of the discrepancy; though these large trees consisted of only 14% of the stems (Table 4.2). Whereas, combined DBH and height measurement error on small trees (DBH <20cm) had a negligible effect on tree biomass estimates, causing less than 1% discrepancy, outlining the importance of properly measuring large trees.

4.4.3 Site-Level Measurement

When comparing the experts' and non-experts' datasets, there were 32 extra trees recorded and 37 trees missing in the non-experts' dataset. However, there was no statistical difference in the number of trees recoded in the experts' and non-experts' sample (t = 0.54, df = 40, P = 0.59). Because the missed trees were larger on average than the extra trees, DBH of 31.5cm ± SD 26.7 from the missing trees vs. 23.0 cm ± SD 17.2 for the trees recorded in extra, this resulted in biomass underestimates from non-expert surveys. We found that the size distribution of trees recorded as extra in the non-expert datasets were clumped and mostly within a few centimetres of the cut-off tree sizes for subplots (e.g. 5 cm, 20 cm and 50 cm, Fig. 4.5A and 4.5B), indicating a systematic bias toward recording trees in the subplot that were beyond the size-cut-off limit. Whereas, the size distribution of the missing trees was similar to that of the whole sample, indicating that missing trees were more likely a random error (Fig. 4.5A to 4.5C).

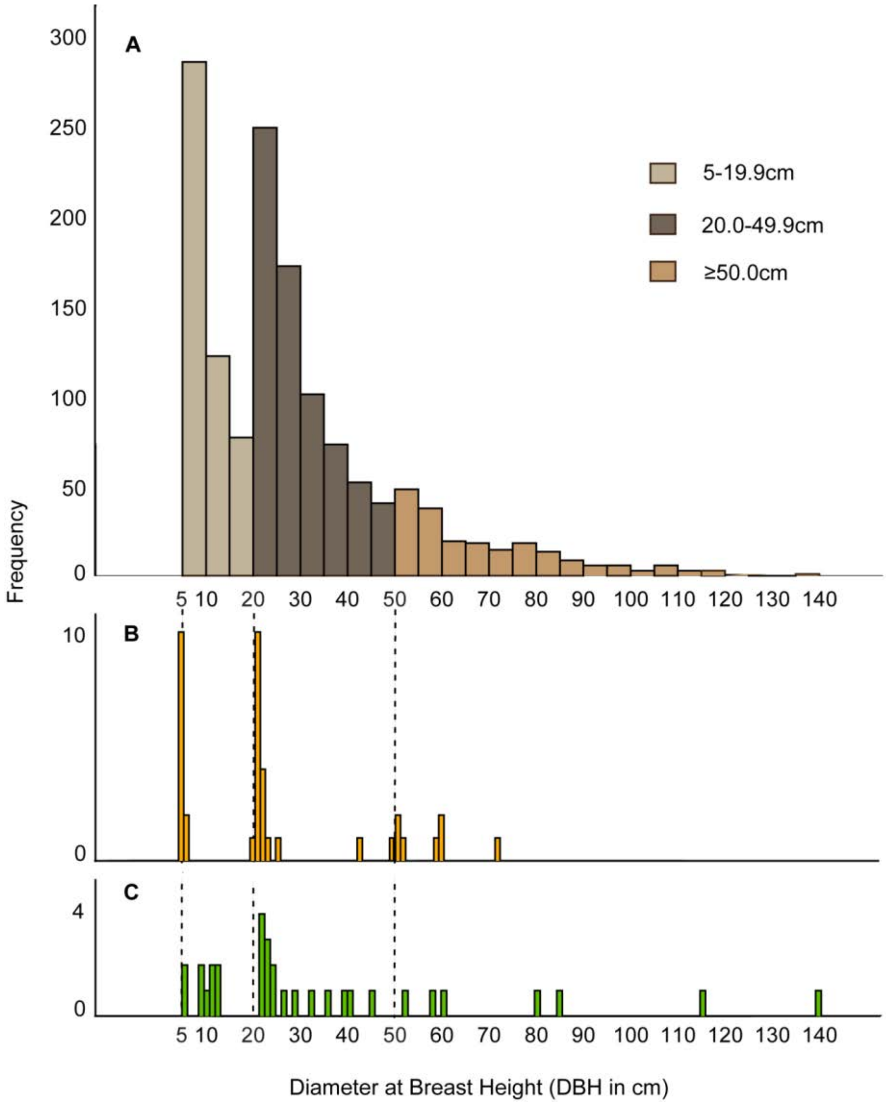

Figure 4.5. The size distribution of the DBH of A) all trees, B) trees missed and C) extra trees in non-experts' surveys. *The dotted lines represent cut-off DBH sizes for subplots (5.0cm, 20.0cm, and 50.0cm) for the three subplots (Fig. 4. 2).*

The surface area of the plots established by experts and non-experts was not significantly different ($Z = 5.6$, $df = 40$, $P < 0.0001$). Differences were small, with more than half the plots within ±1% of the standard area; well within the suggested accuracy for biomass inventories (Table 4.1) (MacDicken 1997). Furthermore, non-experts met all site selection criteria. They established sites with a minimum of 120 m distance from the nearest neighbouring site and recorded the GPS coordinates within the expected range of accuracy of the GPS instruments.

The slope angle classification in experts and non-experts were the exact same, except for one site, where non-experts reported a ‘medium slope’ whereas experts reported a ‘steep slope’.

Finally, non-expert survey recorded local names for 91 % of the trees, for a total of 552 different names; a clear illustration of the wealth of knowledge held by local landholders (Appendix 4.2). Because the local classification system did not completely overlap with Linnaeus species identification (e.g. the same tree species may have had two local names based on the location where it grew), drawing conclusions about the reliability of the traditional plant taxa identification would have required further investigations which went beyond the scope of this study.

4.5 Discussion

People who depend on forests have a great wealth of knowledge about how the ecosystem functions and often possess awesome skills for managing the forest resource. Harnessing this knowledge and the management skills of local people could greatly contribute to project success. Engaging local people in forest carbon schemes can take form of participation in forest carbon monitoring or participation in carbon management. Carbon management and locally based monitoring programs can provide payments for their management activities that can be directly measured; a feat that has shown to be a roadblock in many REDD+ activities. Finding strategies that work will not only achieve significant climate mitigation and livelihood outcomes, it could to help ensure the protection of biodiversity into the future.

By fully remeasuring and isolating the effects of 4,481 field measurements, we demonstrate that programs employing local people (non-experts) can produce forest monitoring data as reliable as data produced by scientists (experts). These findings corroborate those of Danielsen et al (2011, 2013), and then build on them by validating tree height measurement and plot surface-area measurements. Moreover, our study design enabled us to discover that missed trees in non-expert datasets are a large source of error in biomass estimates. Our results also demonstrated that the combination of errors in DBH and height measurements on large trees (DBH>50 cm) formed the bulk of errors at the tree level. Though larger trees may require more effort to measure precisely, they generally store most of the carbon in tropical rainforest (Slik et al. 2013) and training should emphasize techniques to accurately measure biomass in large trees.

In general, the non-experts produced lower biomass estimates than non-experts'. Non-expert surveys had on average 9.1 % lower biomass, equivalent to 55.2 (SE ± 24.0) fewer tons of biomass per hectare, for a total of 7.0 million tons fewer (SE ± 3.1) in the study area (unpublished data by the authors). These differences could have important financial repercussions for communities participating in carbon projects. Identifying what field parameters led to the discrepancies in biomass estimates was one of the aims of this study.

The lessons learn from our hierarchal approach to identifying sources of errors in biomass analysis were that the most frequent errors in the field may not necessarily led to the biggest discrepancy in forest biomass estimates (Molto et al. 2012). Partly because DBH and height measurements can interact in the allometric model and errors from plot size and discordant tree counts can be introduced at landscape scale. The most unexpected result in this study came from discrepancies caused by discordant tree counts. The missing trees caused 37.4 % of the differences between expert and non-expert biomass inventories. Because missing trees were on average smaller than extra trees, biomass values from 'missing trees' were more than compensated by the number of 'extra trees'.

At the tree level, measurement of heights caused more discrepancies than DBH measurements. A large proportion of DBH (37.1 %) matched exactly; this was similar to results reported by Danielsen et al (2013). These results are encouraging as DBH are almost always required for biomass inventories (MacDicken 1997). Also at the landscape scale, height measurements introduced the most discrepancy in biomass estimates (41.7 % of the error). Heights are difficult to measure accurately, even for trained experts under the best conditions, (Williams and Schreuder 2000). Based on findings from other studies and considering the rugged conditions of this study area, we believe that the discrepancy between expert and non-expert height in this study were within and acceptable range. The expected difference between the height measurements between two groups of experts is 10 % (Chave et al. 2004a) and our results showed that most of the pairs (98.4 %) varied by less than this. Furthermore, form Butt et al (2013) the average difference of two expert height measurements for the same tree was 2.8 m for tree < 35 m tall, while the difference in our study was only 1.6 m for trees < 35 m tall.

Because errors in height introduces important errors in biomass estimates and because height measurements are time consuming, monitoring programs may opt to forgo taking height measurements (Hunter et al. 2013). However, omitting height measurements from biomass inventories may result in a greater degree of inaccuracy. Studies have shown that omitting heights can cause discrepancies in biomass of 38 % in Lowland forests (Marshall et al. 2012) and 50 % at higher elevations (Girardin et al. 2010). Whereas, our results showed that height measurements caused only 3.8 % of the total errors in biomass estimates. Therefore, instead of omitting height altogether, alternative methods to estimate tree height should be considered. For example, height can be accurately estimated using deterministic models that predict height from the DBH or from LiDAR technologies (Feldpausch et al. 2012, Lines et al. 2012).

Our results showed that rectangular plots were established with a high level accuracy by non-experts, even in difficult terrain using previously unfamiliar equipment. Though many plot sizes and shapes exists for monitoring forest biomass surveys (Pearson et al. 2005), monitoring programmes may consider more rapid 'plotless' surveys. Because trees near size class limit were included, even if they were too small, careful consideration should be taken before adopting plotless methods. Studies have shown that as the lack of a sample perimeter in a dense forest may intensify biases toward excluding or including trees close to the boundaries (Hijbeek et al. 2013).

Our study had a number of limitations, notably our study would have benefitted from repeating the experts' and non-experts' measurements to create sampling error. This could have served as a reference to compare the difference we observed between the experts' and non-experts' surveys. However, logistic constraints associated with working in the remote forests of PNG for extended periods made this impossible. We partially addressed this by comparing the errors of experts and non-experts to those reported in other studies that have quantified sampling error in expert DBH and height measurements (MacDicken 1997, Chave et al. 2004b, Butt et al. 2013). Furthermore, this study did not validate wood-specific gravity (WSG), the third parameter used to estimate tree biomass. Many studies have shown that wood-specific gravity can be effectively and accurately predicted with taxonomic information in such a way to minimize the introduction of errors into biomass estimates (Molto et al. 2012).

Efforts to produce national forest biomass inventories for REDD+ programs could represent a unique opportunity to catalogue biodiversity and traditional knowledge. Although more than 500 local tree taxa were recorded in the study, assessing the reliability of these data went beyond the scope of this study. This study, however, introduces the potential for DNA-barcodes methodologies to be used with species identification in community-based forest monitoring as DNA-based approach requires only a small sample (1 cm^2) of leaf material preserved in silica, which can be shipped via regular mail for analysis. Using DNA-barcodes for tree species identification in areas where taxonomy is poorly known has been shown to be effective and would be particularly useful in areas where time and capacity are limited (Brofeldt et al. 2014).

Only 3 % of forest-rich developing countries have the expert capacity to monitor changes in forest biomass stocks (Romijn et al. 2012). Our results, together with other studies (Danielsen et al. 2011, Butt et al. 2013, Danielsen et al. 2013, Brofeldt et al. 2014) help demonstrate that programs that engage local people could address this capacity gap by providing quality data to support national monitoring programs. Though decades of scientific research has advanced and refined expert-led forest biomass inventories (Maniatis and Mollicone 2010, Wagner et al. 2010), the advent of community-based monitoring will most likely give rise to new challenges. Thus, developing reliable methodologies that remain flexible for local realities while meeting the needs of the participant will the key to the success of locally based monitoring programs.

Finally, the contribution of knowledge from local communities could go beyond collecting data for biomass estimates toward deepening our understanding of the changes accruing in tropical forests (Lewis et al. 2009a). Our finding that local communities were able to identify over 500 tree types is the type of ecological information iceberg held by these communities. The scale and the consequences of changes in tropical forests can be understood only through long-term monitoring studies, to which local communities could be valuable contributors (Phillips et al. 1998).

4.6 Summary of Chapter 4

- Involving local people in monitoring forest-carbon stocks could potentially address this capacity gap to monitor forest carbon stocks in developing countries.
- We demonstrate that programmes employing local people (non-experts) can produce forest monitoring data as reliable as those produced by scientists (experts).
- Overall, non-experts reported lower biomass estimates by an average of 9.1 %, equivalent to 55.2 fewer tons of biomass ha^{-1}, which could have important financial implications for communities.

- The study design allowed for the detection of errors in the field and how these contribute to discrepancies in forest biomass estimates. At the landscape level, the greatest biomass discrepancies resulted from height measurements (41 %) and, unexpectedly, a few large missing trees contributing to a third of the overall discrepancies.
- Community-based monitoring programmes should prioritise reducing errors in the field that lead to the most important discrepancies, notably; overcoming challenges to accurately measure large trees.

-END OF CHAPTER 4-

Chapter 5

Between Forests and People; Carbon and Conservation in Papua New Guinea

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5.1 Overview

This chapter tests an approach that integrates PEN socio-economic datasets into more traditional land-use management planning and compares restore and protect interventions for lowering forest carbon emissions. To do this, we integrated data from land-cover change analysis (1990-2010), field carbon assessments from eight land-cover types (263 forest plots and 115 soil sites) and socio-economic surveys (112 households in 9 communities) into a land-use planning model that explores management options for future emissions under local threat and industrial logging scenarios.

5.2 Introduction

The bulk of carbon emissions from tropical forests occur in areas where people interact with forests (Wright 2010). These boundaries are often shifting mosaics of land-uses with multiple functions (Lamb et al. 2005). Growing evidence reveals that people living in forest frontier regions in developing countries are some of the poorest and most vulnerable people on the planet (Angelsen et al. 2012), and that they rely on forests for many of their basic needs (Wunder et al. 2014). These needs include household energy, building materials, tools and food, in addition to cultural heritage (Sunderlin et al. 2008). Managing human-forest landscapes to lower emissions will invariably require alterations to the dynamics between individual communities and their forest resources.

Two basic land-management strategies are recognized under the REDD+ initiative: (i) to protect forest carbon stores from processes that threaten them and (ii) to restore forest carbon stores through active management. Though the benefits of restoring degraded areas through active management are widely recognized, deliberations about where and how to do so remain vague (Sasaki et al. 2011). Moreover, while most carbon projects have focussed either on protecting or restoring forest (Chazdon 2008), conservation practitioners have more readily adopted forest protection as the dominant REDD+ mechanism (Wendland et al. 2010). Often these efforts are concentrated in areas with industrial presence as these extractive activities causes substantial emissions that are relatively easy to quantify (Asner et al. 2009a). However, industrial activities are not the only sources of forest degradation and loss; community activities, such as small-scale agricultural expansion, fires and fuel-wood extraction can leave large tracts of forest with diminished carbon stores (Lamb 2011, Lun et al. 2014, Bailis et al. 2015).

Though forest carbon projects are relatively new, many lessons can be drawn from conservation initiatives, including the importance of considering local community expectations from these projects (Resosudarmo et al. 2012). For instance, meeting community needs is one of the key determinants and indicators of conservation success (Vejre et al. 2007, Sayer 2009, O'Farrell and Anderson 2010). Perhaps recognising the role of communities in project success, the main policy instrument for the implementation of forest carbon projects, Reducing Emissions from Deforestation and forest Degradation (REDD+) has shifted from narrow emissions reduction strategies to also including development objectives (Sutter and Parreño 2007, Linnér and Pahuja 2012). Therefore integrating societal needs has become integral to most carbon projects.

Land-use planning is at the heart of most environmental management decision making (de Groot 2006). Land-use planning recommendations for REDD+ typically use biophysical data, such as soil and biomass carbon surveys, historical land-use change and associated emissions and land-suitability analysis (Pearson et al. 2005, de Groot et al. 2010). Integrating societal needs within land-use planning is difficult, partly because of a lack of tools for linking community values, preferences and constraints into these models (O'Farrell and Anderson

2010). Existing land-use planning tools range from highly computational tools using biophysical data only (Tallis et al. 2008, Jenkins et al. 2015) to those that consider social-economic contexts by exploring the trade-offs imposed by different stakeholder expectations (Dewi et al. 2011, Venter et al. 2013) or the provision of guidelines for negotiating land-use plans with local communities (Sayer et al. 2013). The lack of proper integration of community values, preferences and constraints into land-use planning could undermine REDD+ program goals.

REDD+ approaches have also been widely criticized for lacking robust and replicable methodologies for measuring and monitoring both biophysical and socio-economic variables (Angelsen et al. 2012). The extent of local input into land-use planning processes is usually limited to community consultations and workshops (Dewi et al. 2011). Though community consultations are useful at setting targets and serving as platforms for knowledge exchange and collaborative learning (Arciniegas and Janssen 2012), they commonly do not provide measurable socio-economic variables that can be integrated into land use planning to enable them to be monitored through time (Caplow et al. 2011). These approaches also risk over-representing the interests of powerful community members that dominate in group settings (Agung 2011).

Some of the largest and most robust socio-economic datasets designed to monitor environmental resources in communities come from standardised household surveys designed for tropical rural ecosystems are the Poverty and Environment Network (PEN) surveys (Angelsen et al. 2011, Babigumira et al. 2014). Since PEN's inception, surveys have been conducted in over 25 developing countries; creating an extensive dataset on community values, preferences and constraints, thereby providing new insights into the relationship between communities, forests and poverty (Angelsen et al. 2014). However, these data have not been used to integrate local community needs into land-use planning tools.

Papua New Guinea (PNG) has the third largest expanse of tropical forest wilderness in the world. Yet what sets PNG apart from other tropical countries is the fact that 70-97% (court cases are underway to settle tenureship across large areas of PNG) of its forests are owned and managed by local people for millennia (Haberle 2007, Keenan et al. 2011, Hansen et al. 2013). The country holds enormous potential for reducing forest carbon emissions on locally owned lands and for improving the livelihoods of forest dependent communities. Like many other tropical countries, emissions from forest loss contributes to global climate change with rapid rates of deforestation and degradation over the past 30 years resulting in a net loss of 15% of its PNG's forests (Shearman et al. 2009, Christopher et al. 2012). Moreover, most rural Papua New Guineans earn less than US$1.25 per day and have limited access to medical, educational and other essential services (Rogers et al. 2012). With a global ranking of 154 out of 187 for poverty, PNG's population could clearly benefit from REDD+ initiatives.

Considerable investments into forest conservation for carbon sequestration are expected in PNG. Approved budgets of over 6 million $US by the UN-REDD Programme make PNG the second highest national investment from this programme (http://mptf.undp.org /factsheet/fund/CCF00, accessed April 8th 2015). However, if people and forests are to benefit from these investments, methods for integrating the needs of communities in decision making as well as quantifying the social impacts of these interventions are crucial (Mertz et al. 2009).

The objective of this study is to explore how best to manage forest carbon stores by taking into account community values and constraints. We compare interventions for restoring and protecting forests in a remote area of Papua New Guinea where 57 communities depend on forests for subsistence. More specifically we 1) determine carbon stores from seven different land-uses, 2) estimate carbon flux from past land-use change, local wood-harvesting and a proposed logging lease, 3) ask landholders to evaluate the importance of environmental products from these same land-uses 4) use a land-use planning exercise to compare climate mitigation outcomes from two basic management strategies, 'protect' and 'restore', 5) compare future emissions scenarios in a model that uses biophysical data only to one that also includes socio-economic data from household surveys Finally, we test how the Protect and Restore strategies might change in the face of existing local threats and planned industrial logging threats. The results from this study demonstrate the potential to tailor forest management in order to meet emissions reductions targets without compromising the needs and values of forest dependent communities.

5.3 Methods

5.3.1 Study Area

In addition to the primary tropical humid forest described in Section 3.3.1, this chapter also included the greater YUS landscape which constitute alpine grasslands and a variety of anthropogenic land uses, such as frequently burnt grasslands used for hunting, managed forests (disturbed and secondary forests) with a range of uses, mosaics of shade coffee plantations, cocoa plantations and swidden agriculture which are characterized by short periods of cropping and lengthy periods of fallow (Manner 1981).

The study area has no road or river access and has only limited coastal access (Fig. 5.1). An estimated population of 12,000 people live in the area, concentrated in 57 villages mostly in river valleys in anthropogenic grasslands between 1,000 and 2,300 m asl The area is accessible via ‘bush planes’ that land on grass airstrips, thereafter all travel to villages is by foot. With limited access to markets, the predominant livelihood for people is subsistence swidden farming with a high dependence on forest resources. In years 2010-2011, only 1 % of households in the area earned a monetary wage and only 60 % of adults had attended school, with an average attendance of 6 years for those who had attended (Cornelius and Murphy 2012).

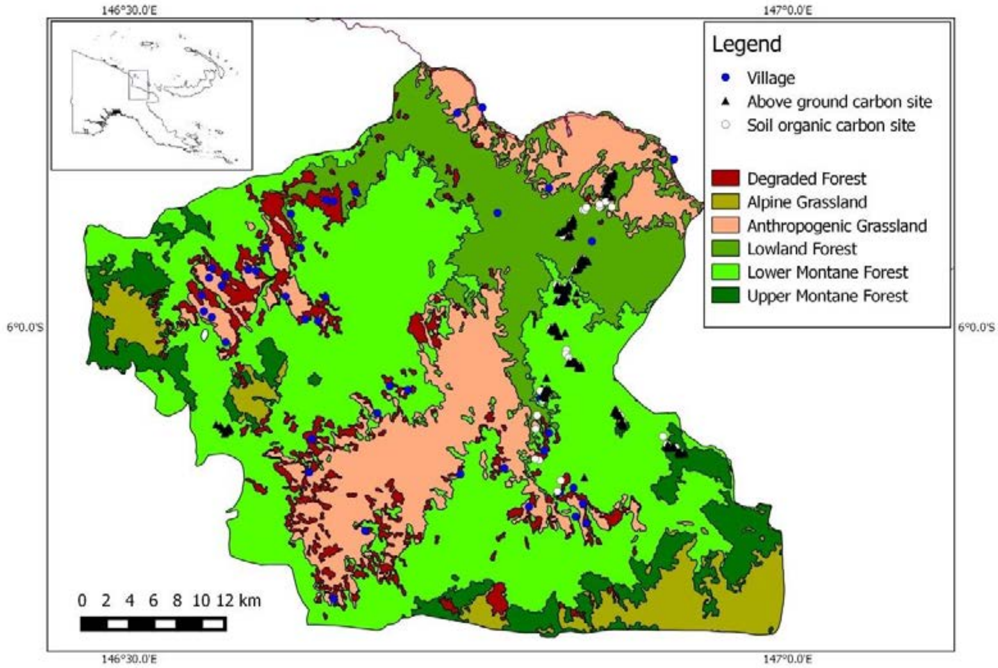

Figure 5.1. Map of land use cover produced through classification of medium-resolution multispectral images (Landsat 2010). *Villages are displaying by blue circle, Above Ground Carbon (AGC) sites by black triangles and Soil Organic Carbon (SOC) sites by white circles.*

In 2009, a milestone for conservation in PNG occurred in this area. A total of 74,000 ha were gazetted by the communities as the YUS conservation area, becoming the first area to gain protection under the *PNG Conservation Act of 1978.* This effort was initiated by a local non-government organization, The Tree Kangaroo Conservation Program (TKCP) and 35 communities from five language groups who each pledged a portion of their primary forests to protect the endangered Huon Tree Kangaroo (Dendrolagus matschiei) from overhunting.

5.3.2 Vegetation Mapping

Major vegetation types and anthropogenic land-use systems were identified using ENVI software for image classification, with primary inputs from Landsat 5 (TM5) and Landsat 7 (ETM+7) (USGS 2010). The classification system used a 5-ha minimum mapping unit raster format and 10-ha minimum mapping unit (mmu) polygons based on the PNG Government's Forestry Information Mapping System (Hammermaster and Saunders 1995). Because moderate resolution remote sensing lacks sufficient resolution to distinguish between certain vegetation covers (typically regrowth and garden areas), we also flew an unmanned aerial vehicle (UAV) to collect geo-tagged data which were mosaicked with Pix4D software (Fig. 5.2). The UAV photos, SPOT5 data (5 m resolution panchromatic and 10m pan-sharpened multispectral images), and 263 biomass field sites were used for ground truthing and as training regions for differentiation amongst vegetation types (Gillieson et al. 2011).

Figure 5.2. Arial photo of traditional houses with a mix of intensive agriculture and fallows in Nombo Village (2,353m asl). *This image was created by mosaicking photos taken by an unmanned aerial vehicle (UAV) and served as ground truthing and vegetation reference areas for remote sensing analyses.*

5.3.3 Above Ground Carbon Pool

Carbon stocks were assessed in above ground carbon (AGC) pools, which included live and dead standing trees, and coarse woody debris (CWD). For AGC, A total of 263 x 0.1 ha plots were censused using the widely used Land Use, Land-Use Change and Forestry (LULUCF) protocols (Pearson et al. 2005), with further methods described in Chapter 3. Plots were established in primary Lowland forest (N = 57), Montane forest (N = 101), Upper-montane forest (N = 57), shade coffee plantation (N = 12), fallowed swidden agriculture (N = 12), young secondary forest (21 yr (mean), N = 12) and mature secondary forest (> 50 yr, N = 12) (Fig. 5.3). Allometric equations used to derive biomass estimates from field measurements are given in Appendix 5.1. For anthropogenic and alpine grasslands we used values previously established for PNG and elsewhere (Hartemink 2001, Oliveras et al. 2014b). Field campaigns for AGC were carried out between August 2010 and May 2013. We compare AGC from different land-cover types using ANOVA with Tukey post-hoc tests. All statistical analyses were conducted in the S-PLUS Enterprise Developer Version 8.0.4 for Microsoft Windows.

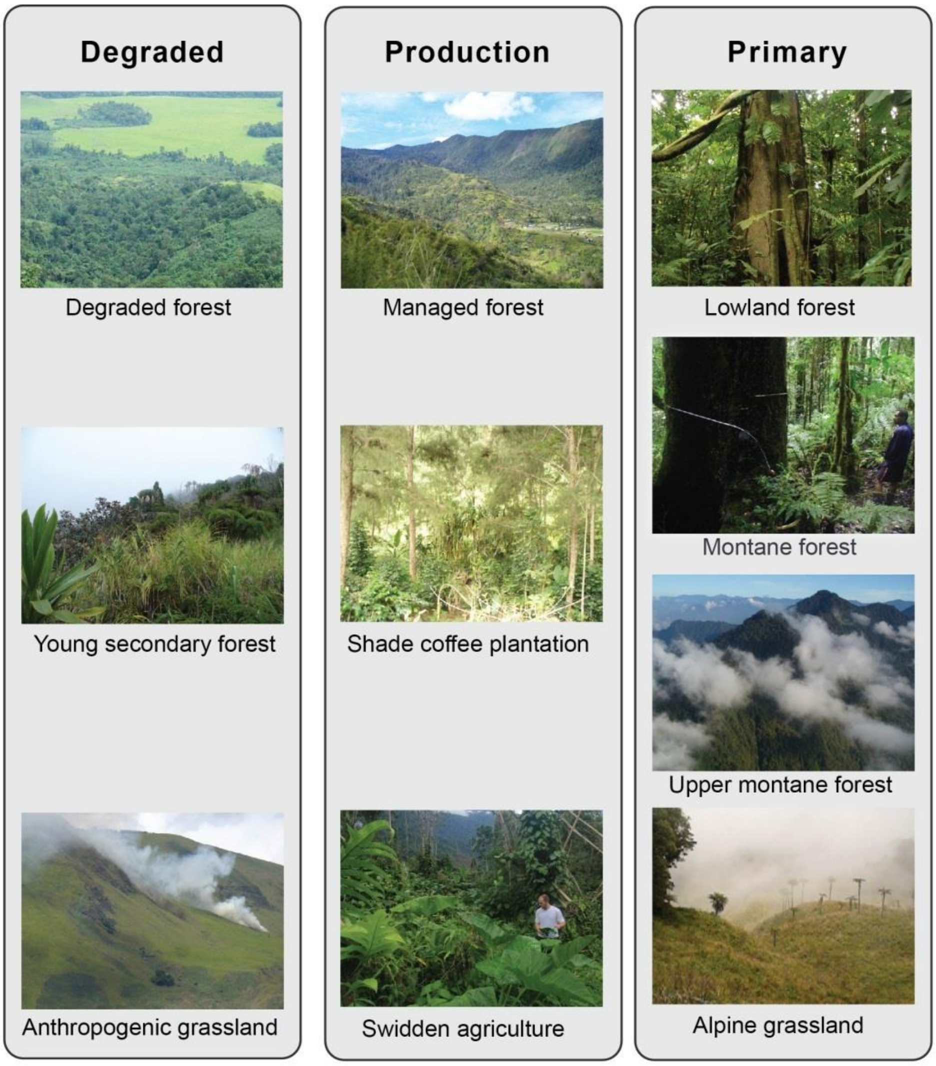

Figure 5.3. Representative photos of degraded, production and primary land-cover types assessed in the YUS catchments.

5.3.4 Soil Carbon Pool

Soil carbon was assessed in primary Lowland (50 - 1,000 m asl), montane (1,000 - 2,800 m asl) and Upper-montane forest (2,800 - 3,100 m asl) for a total of 91 sites and 497 soil samples; many of the soil and biomass forest sites were shared. Soil carbon was assessed in anthropogenic grasslands adjacent to forest sites, for a total of 14 sites and 70 soil samples between 50 and 3,100 m asl Soil carbon pools classes included Soil Organic Carbon (SOC), fine roots and litter. In each of the forest and grassland sites (0.1ha) sampled for soil; soil profiles were sampled at 3 locations. At each location, soil samples were taken at 0-10, 10-20 and 20-30cm depth. The three samples for the 0-10, 10-20 and 20-30cm interval were bulked, weighed and a subsample retained for analysis. For each sampling location, this resulted in a total of 3 (bulked to 1) replicates for the litter layer, 3 (bulked to 1) replicates for the 0-30cm layers and 1 replicate for the deeper layers of the soil. Further methods for soil sampling techniques and laboratory analysis are found in Dieleman et al. (2013). For SOC in swidden fallows, coffee plantations and degraded forest, we used a mean value of SOC from grasslands and forest from similar elevations, as complex agroforestry systems have shown little decline in soil fertility and carbon compared to forests (Albrecht and Kandji 2003, Schmitt-Harsh et al. 2012). Field campaigns for sampling SOC were carried out between August 2010 and November 2010. We compared SOC from grassland and forest from the same elevation clusters using a paired t-test.

5.3.5 Carbon Flux from Land-use Change Analysis

Total carbon (million tons of carbon, Mt C) at the landscape scale in 2010 was calculated by summing the product of each land-cover type (ha) by their respective carbon stocks (Mg C ha^{-1}), including above ground living and dead biomass, and soil pools (Pearson et al 2005). Land cover was determined from the vegetation mapping analysis and carbon stocks were from field inventories, as described above. Land-cover change analyses were based on methods described by Sader et al. (2001) using Landsat-5 imagery dating from 1990 to 2010 (US Geological Survey, http://landsat.usgs.gov), co-registered to ortho-rectified scenes from the Global Land Cover Facility (www.glcf.umiacs.umd.edu). The spectral properties from different land-use cover changes were classified using a decision-tree algorithm trained to detect the *various changes in land-cover types* for three time periods (1990 - 2000, 2000 - 2004 and 2004 - 2009) using a 0.5 ha mmu. We were able to distinguish spectral properties of intact forest from burnt vegetation as well as living vegetation and bare soil but were unable to distinguish degraded forest from regrowth forest; hence these were lumped into a single degraded forest category. The carbon flux (Mg C yr^{-1}) from land cover change was determined for each land-cover type described in the vegetation mapping exercise between for the period between 1990 and 2010. This carbon flux can be considered to be a historical emissions baseline for the area commonly known as the 'business as usual' or reference period (Dewi et al. 2011). It is important to note that no commercial extractive activities were conducted during this period, and all carbon flux was from 'local threats' only.

5.3.6 Carbon Flux from Wood Harvesting

Carbon flux from wood extraction was based on a field study in the YUS area conducted by Page et al. (2011) (Fig. 5.4). The two main sources of wood extraction evaluated were fuelwood (Mg) and household timber (m^3). Weight of weekly fuelwood consumption was obtained from housesold survey questionaires adjusted by weighing bundles of wood in the field (Annual survey Section E, Appendix 5.2). Weekly houshold consumption of fuelwood estimates were multipled by the number of housholds in the study area and converted to yearly consumption rates. The volume of timber required to build traditional houses was estimated by measuring the diameter and length of poles, with a mean house storing 5.9 tonnes of carbon with each person consuming on average 0.62 houses in 30 years (Page et al. 2012). To estimate the mass of timber, we used a mean wood density of 0.62 g cm^{-3} the average of the main species used to build houses (Zanne et al. 2009) and coverted to carbon using a factor of 0.5 (for more detail on wood extraction estimate, see Page et al 2012). It is important to note that we assumed that carbon flux from wood harvesting and from land-use change were mutually exclusive as small-scale wood extraction was unlikely to be detected by remote sensing analysis (Ramankutty et al. 2007).

***Figure 5.4. Photos of wood used as main source of fuel and construction material**. Photos by Michelle Venter.*

5.3.7 Carbon Flux from Proposed Industrial Logging

An industrial logging lease, *Timbe Kwama,* has been approved to extract 9,040 ha of Lowland forest in the study area (Brooks and Ramachandra 2012). According to the *PNG Logging Code* of practice (PNGFA 1996) yearly harvests of 700 ha yr^{-1} are allowable on terrain that does not exceed 25° slope. To estimates potential future rates of carbon loss from *Timbe Kwama*, we used a digital elevation model to identify the loggable areas within the lease zone (< 25°) and calculated the yearly carbon flux based on conversion of Lowland forest to Lowland grassland at a rate of 700 ha yr^{-1} during the projection period (2010-2040). Carbon stocks were converted to CO_2 emissions using a factor of 3.667.

5.3.8 Socio-Economic Surveys

Socio economic data were obtained by conducting Poverty and Environment Network (PEN) standardised household and village surveys following protocols from the Centre for International Forestry Research (CIFOR) (PEN 2007, Cornelius and Murphy 2012). A total of 112 surveys were carried out in nine villages selected through community consultation with the aim of having a representative sample of elevations and distances from forests (Table 5.1). All, villages surveyed, except one, had customary rights and access to primary and degraded forests, grasslands and swidden fallows (Appendix 5.2, Annual Survey Section C). For more details on methods, see Cornelius and Murphy (2012). The PEN surveys were conducted in 2010 and 2011.

Table 5.1 Characteristics of villages chosen to conduct socio-economic household surveys.

Village name	Elevation (m asl)	Watershed	Proximity to forest (hours walk)	N^o household surveyed
Ronji	79	Yopno	0.9	20
Mumanarang	1,459	Som	1.7	7
Boit	1,488	Uruwa	3.2	13
Yawan	1,526	Uruwa	3.9	20
Torik	1,707	Som	1.3	4
Bungawat	1,821	Som	1.4	5
Gogiok	1,825	Som	0.7	7
Nombo	2,352	Yopno	4.5	20
Kumbul	2,624	Yopno	2.3	20

PEN surveys aim to monitor changes in demographic, land-tenure, service infrastructure, poverty, and forest use in forest-dependent communities. For our purposes, we only used data pertaining to land management practices (PEN, Quarterly household survey section D and E), use of environmental products (PEN, Village survey section D), income derived from environmental products (PEN, Quarterly Household Survey, Sections B to D), and community perception of conservation, development, poverty (Cornelius and Murphy (2012) Adapted village survey). The Annual Survey was adapted to collect additional details specific to land-types and land-use management in PNG, but otherwise followed the template and the technical guidelines of PEN and complies with conventions for data entry into the common data bank (version 7.1, April 2009).

Introducing the value of environmental products for local communities into a land-use planning exercise requires identification of products associated each land-use, how these products are valued by communities and how much land must be allocated to ensure the provisioning of essential and desired products. Another way of estimating values of environmental products is by the level of financial income they generate (Nielsen et al. 2012). However, because almost no products were sold either locally or regionally (data not shown), we used a scoring technique to value enviromental products (Cornelius and Murphy 2012). The respondents were asked to identify and rank the three most important environmental products from primary forests, degraded/managed forests, shade coffee plantations, swidden fallow and grasslands (Annual Survey under section C, question 1 A to C1). Each product was given a score according to its rank (most important = 4, second most important = 2 and third most important = 1). The scores of the products were tallied for each land-cover type and were transformed to a relative scale with a minimum of 0 and maximum of 1 of the maximum score obtainable based on the number of responses (Cornelius and Murphy 2012). The products were categorized into food crops, cash crops, fuelwood, wild meat and eggs, wild vegetables and nuts, and traditional garments and instruments.

5.3.9 Land-use planning exercise

We built a simple land-use planning model that compares the projected carbon flux from years 2010 to 2040 from two basic climate mitigation strategies; forest protection and forest restoration. The management actions we chose to include in the land-use planning exercise were identified *a priori,* in a report presented to the *KFW Bankengruppe* (Venter et al. 2012a), which used bio-physical constraints only. Then, we modified this model to include constraints and opportunities from the information gained from the PEN survey. The decision rules used to select land areas for different strategies are outlined below, summarized in Table 5.2 and further elaborated in Appendix 5.2 to include rates of carbon loss or accumulation from AGB and SOC pools, and the rates of land-use change for the restoration actions.

Table 5.2 Summary of biophysical and socio-economic constraints in the land-use planning model.

	Management action	Biophysical constraints	Biophysical and socio-economic constraints
Protect	Protected area	All primary forests are protected	60% of primary forests are protected from local and industrial logging, but not from fire.
	Shade coffee plantation	Grassland at 1,100-2,000 m asl, <4km of village, <45°slope	Grassland at 1,100-2,000 m asl, <4km of village, <45°slope, with a 5ha maximum per household
Restore	Accelerated natural regeneration	Alpine grasslands from 2800-3,800 m asl	Alpine grasslands from 2800-3,800 m asl Lowland grasslands from 1,000-2,800 m asl, < 10km of village, max 50% of this area
	Enrichment planting	n/a	Degraded forest, slope <25° slope, <10km of village

The 'Protect' intervention using biophysical constraints only assumed effective protection of all primary forest, thus no carbon flux from wood harvesting, nor local or industrial logging were included in the model. However, business as usual (BAU) carbon flux from fire was included, with specific rates for each forest types as determined by land-use change analyses. Adding socio-economic constraints reduced the protection to 60 % of primary forests; in our model, the land allocated for local use (40 % of primary forest) underwent a business as usual (BAU) carbon flux. The amount of primary forest required for local use was determined by the amount of area communities were willing to set aside for conservation (Brooks 2011). All land-cover types that are not primary forest undergo BAU rates of land-use change.

The 'Restore' intervention included a variety of management actions, such as shade coffee plantations in anthropogenic grasslands and accelerated natural regeneration through fire control. Enrichment planting in degraded forest emerged as a third restoration action through the analysis of the PEN surveys. The lands allocated for restoration actions (Table 5.2) were afforded no protection against by local or industrial threats, thus underwent business as usual (BAU) rates of forest loss pertinent to each land use; except that areas deemed suitable for fire control were omitted from BAU rate of loss attributable to fire (Table 5.2). To evaluate the effect of including socio-economic data into a land-use planning model, we first ran the model using bio-physical data only and then ran the model again using both socio-economic and biophysical constraints (Table 5.2).

Finally, to explore the outcomes under planned industrial logging impacts, we compare the carbon flux from the land-use planning model of BAU land-use change and wood extractions from local threats to one that also includes carbon flux from the proposed industrial logging lease (see methods above). We conducted spatial analyses in ArcGIS v10.1 comparing a 1990 land cover map created using the Forestry Information Mapping System (Hammermaster and Saunders 1995) and an updated vegetation cover map from our remote sensing analyses. For selecting criteria of spatial suitability, we used a Digital Elevation Model (METI-NASA 2011) and BIOCLIM data (Kriticos et al. 2012) to gain data on slope and aspect. Each management action was spatially explicit, if more than one action was suitable in the same area; the action with the greatest carbon benefit was selected.

5.4 Results

5.4.1 Carbon stocks and vegetation cover

Primary forest was the main land cover type in the study area (68.9 %) and with 74.8 Mt of C, stored more than 85 % of the total carbon in the study area for 2010 (AGB and SOC; Fig. 5.5). Lowland forests had particularly high carbon density with significantly more above ground carbon (496.1 Mg·ha^{-1} ± 219 1σ) than the other seven land-use types evaluated (Fig. 5.6, ANOVA $F_{6, 224} = 17.18$, $P < 0.0001$, Tukey *post-hoc*). Montane forests and Upper-montane forests also had high AGC (Fig. 5.6; 368 ± 158 Mg C ha^{-1} and 313 ± 102 Mg C ha^{-1}, respectively), with Montane forests being the most extensive land-cover (Fig. 5.1). The ratio of AGC to SOC changed with elevation, with an increasing contribution of C from soil with increasing elevation (8:1 in Lowland, 3:1 in Montane forest to 1.5:1 in Upper-montane forest (Fig. 5.6).

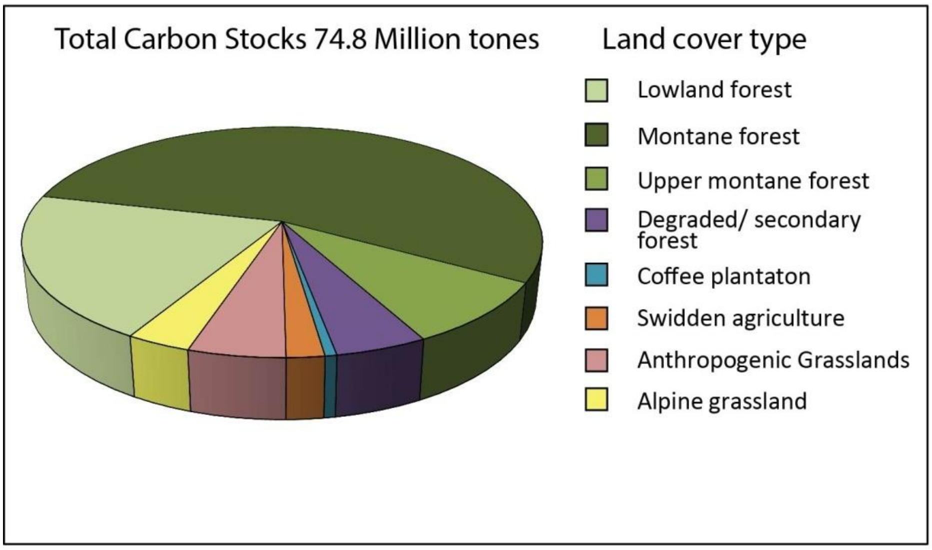

Figure 5.5. Percent contribution of each land-use type to total carbon stocks (AGB + SOC) estimated at 74.8 M tonnes in the YUS catchment.

Degraded landscapes consisted of degraded forests and anthropogenic grasslands and covered one quarter of the study area. Degraded forests were mostly located near villages (Fig. 5.1) and had surprisingly high AGC with 265.6 ± 166 Mg C ha^{-1} (Fig. 5.6), while young regenerating forest (16 - 25yrs) contained 97 ± 43 Mg C ha^{-1}. Anthropogenic grasslands were extensive (~20 % of land area) but had significantly lower SOC than primary forests at similar elevations and negligible AGC (Fig. 5.6; Paired t-test, $t_{(2)\,9} = -2.51$, $P = 0.029$).

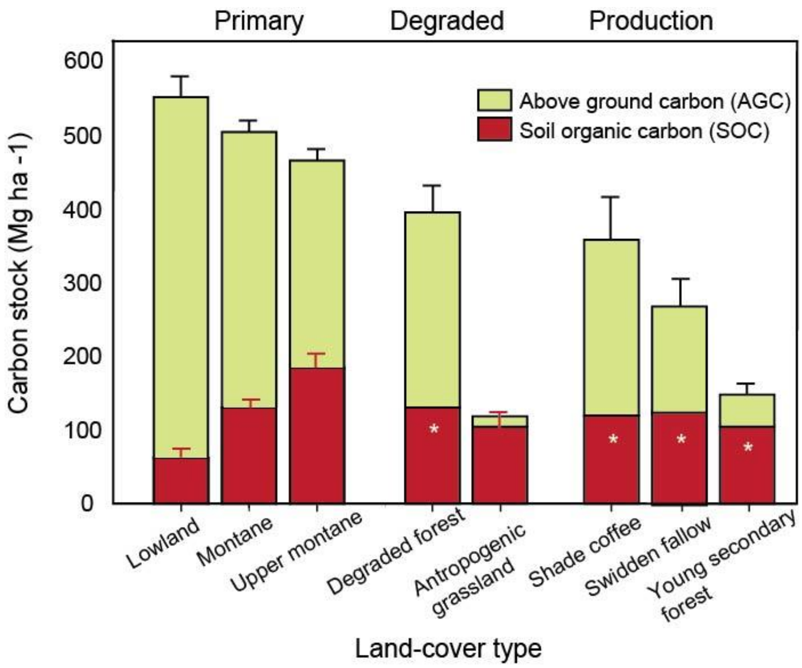

Figure 5.6. Carbon stock (Mg C ha^{-1}) in above ground pool (yellow) and soil pool (red) evaluated in eight common land-cover types of PNG. **Mean SOC values from nearby grasslands and forests sites. Error bars represent Standard Deviation of the mean.*

Production landscapes, consisting mainly of swidden agriculture and shade coffee plantations, covered less than 5 % of the YUS area. Swidden fallows stored 148 Mg C ha^{-1}, while shade coffee plantation had much higher AGC stocks at 238 ± 191 Mg C ha^{-1}. The unusually high carbon stocks in shade coffee plantations were attributable to the use of the native *Casuarina oligodon* as the dominant shade tree, a fast growing species reaching 35 m tall with high wood density (0.83 g cm^{-3}) (Zanne et al. 2009).

5.4.2 Carbon Flux

Anthropogenic fires were the leading cause of forest loss, with 17 % of Upper-montane destroyed by 1997 El Niño fires which had not yet recovered by 2010 (Fig. 5.7). It should be noted that in common with other subalpine areas of PNG, many fires at higher elevations were the result of lightning strikes and not human actions. Lowland deforestation occurred at a low rate of −0.03 % yr^{-1} and was driven by local 'walkabout' sawmills (Brooks and Ramachandra 2012). Montane deforestation occurred at similar rates to Lowland forest but was caused by agricultural or village expansion, and was therefore concentrated near villages.

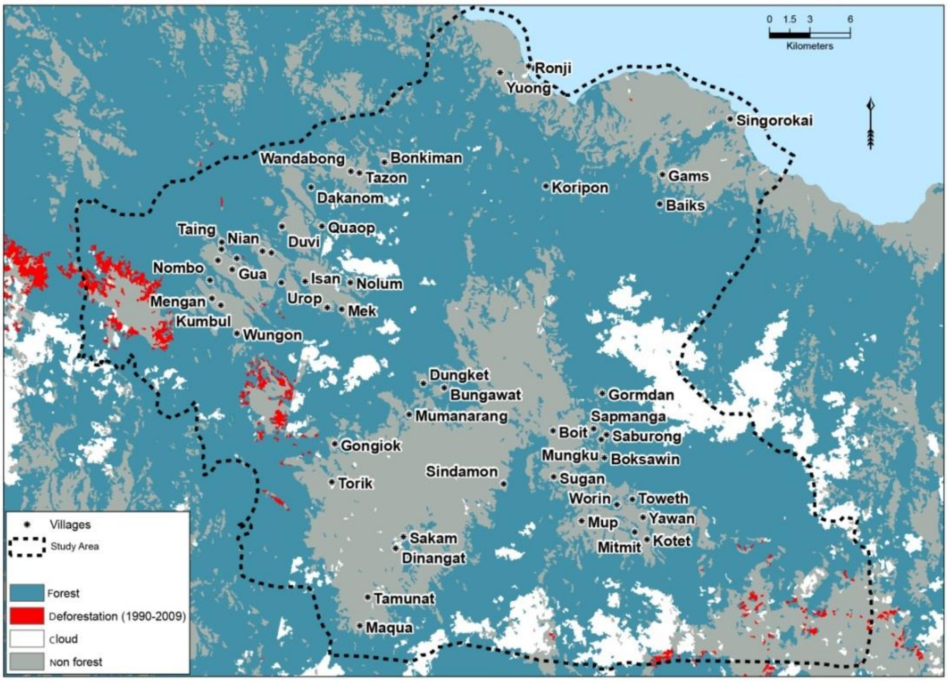

Figure 5.7. Deforestation between year 1990 and 2009 in the YUS catchments of the Morobe province, PNG. Most of the forest loss depicted in red occurred at the margins of Montane forest and alpine grasslands caused by 1997 El-Nino fires.

Net forest loss from land cover change analysis over the 1999 – 2010 period was −0.3 % per annum loss, for a total loss of 2,615 ha in the YUS catchment, equivalent to per capita emissions of 14.0 CO_2e Mt yr^{-1} (Fig. 5.8).The average personal consumption of traditional fuelwood and construction was 8.1 Mg CO_2e yr^{-1} and 0.4 Mg CO_2e yr^{-1}, respectively. Forest loss in the area was estimated to caused 14.0 Mt CO_2e yr^{-1} and forest degradation estimated at 8.5 Mt CO_2e yr^{-1}. The estimated potential yearly per capita emissions from the proposed industrial logging concession were 25.8 Mt CO_2e yr^{-1}, which would result from the loss of 6,750 ha of Lowland forests (Fig. 5.8).

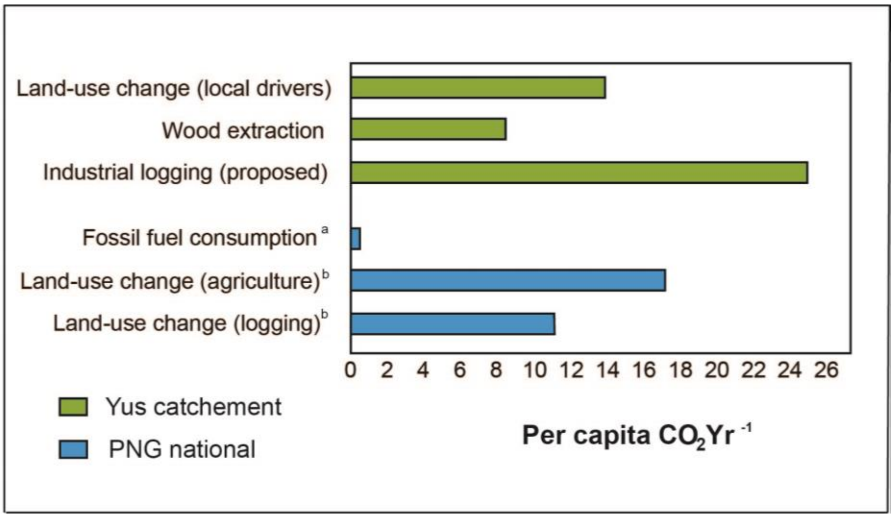

Figure 5.8. Local (green) per capita CO_2 emissions from land-use change and wood-harvesting compared to national values for PNG (blue). [a]*World Bank (http://data.worldbank.org/ country/papua-new-guinea accessed May 1 2015);* [b]*Bryan et al. (2011).*

5.4.3 Local Forest-use

Subsistence activities generated 99 % of the household income in YUS, with only two of 112 households earning any form of wage during the survey period. The primary source of subsistence income was environmental products sourced from nearby landscapes. However, environmental products were rarely traded for cash, with the exception of coffee, betel nut and tobacco.

Land cover types provided a varied range of environmental products (Fig. 5.9). The environmental product scoring the highest in importance overall was construction material, obtained from degraded forests, primary forests, and swidden fallows. Food scored the third highest in importance with different food types provided by different land-covers. However, most of the dietary protein was sourced from wild product (meat, nuts and eggs) in primary and degraded forests rather than in production landscapes. The primary energy source for 100% of surveyed households was fuelwood, used for cooking, heating and light. Fuelwood was obtained from nearby swidden agriculture and shade coffee plantations with negligible amounts collected from primary forests. Products that served in traditional practices, such as dress, ceremony, and musical instruments were found in all land-cover types.

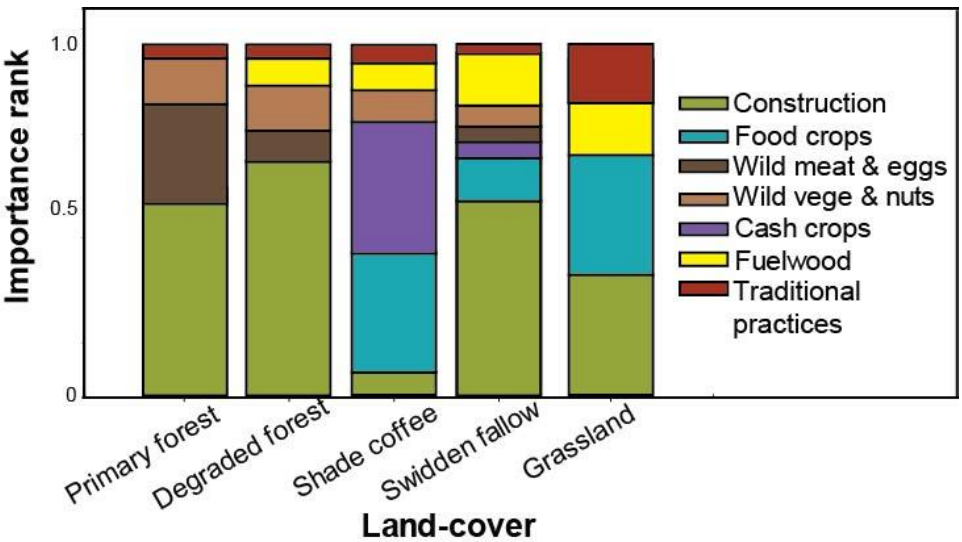

Figure 5.9. Relative importance of environmental products from different land-use types in YUS catchments, PNG.

5.4.4 Land-use Planning Outcomes

Scenario including local drivers of forest loss and degradation

We compared climate mitigation outcomes for two basic management strategies, 'Protect' and 'Restore', and found that the magnitude of carbon flux changed whether biophysical constraints only (Fig. 5.10A) vs biophysical and socio-economic constraints combined (Fig. 5.10B) were considered in the land-use planning exercise. In particular, protecting forest could result in an estimated net carbon loss from −1.5 Mt CO_2e over the 2010 to 2040 period when only considering biophysical information in the land use planning (Fig. 5.10A). However, when socio-economic data from PEN surveys are considered alongside the biophysical data, the net flux after planned land management is estimated at −7.3 Mt CO_2e over the 2010 to 2040 period (Fig. 5.1B). The loss of AGC in the Protect intervention using socio-economic information was caused by limitations imposed on forest protection in order to allow for the provisioning of environmental products for communities, which was an important priority revealed through the PEN surveys (Fig. 5.10). In contrast, restoring forests could result in net carbon sequestration increase of 10.8 Mt CO_2e using biophysical data to plan land management. This compares to potential net sequestration of 28.2 Mt CO_2e in 30 years when including socio-economic constraints and opportunities, a threefold increase in carbon benefit relative to only using biophysical data in the Restore strategy (Fig. 5.10). Most of these gains in the Restore intervention when including of socio-economic data were from new opportunities identified during surveys, such in the potential to accelerate reforestation in grasslands near villages and enrichment plantings in degraded forest.

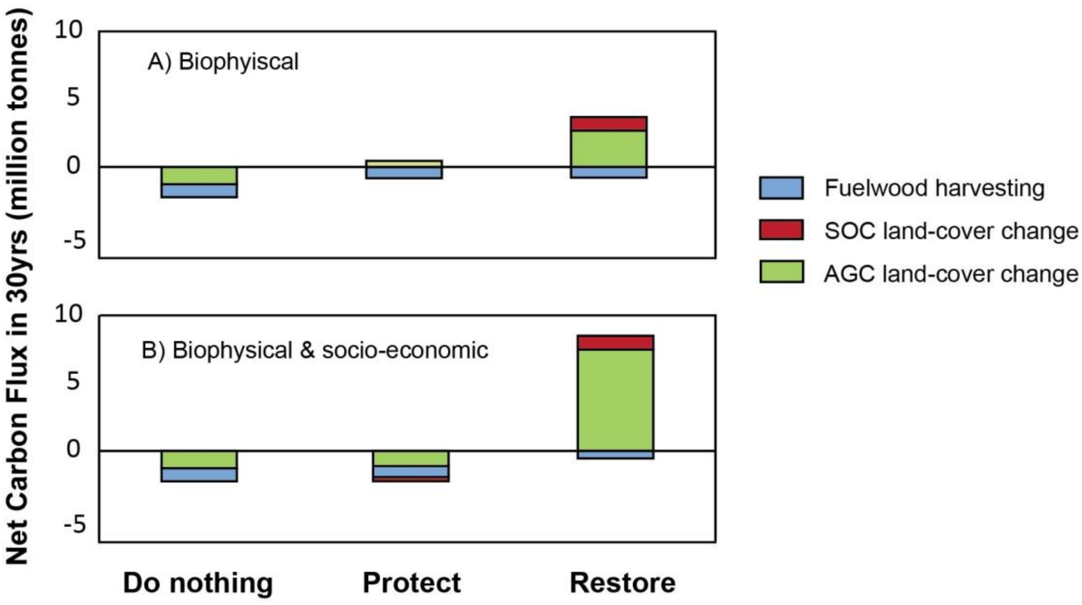

Figure 5.10. Carbon flux from land-use planning model that uses only biophysical data (A) and a model that also includes socio-economic data (B) in scenarios of no action, forest protection and forest restoration.

Scenario including local and industrial logging drivers of forest loss and degradation

We explored whether the outcomes of our models as presented above would change if the proposed *Timbe Kwama* logging lease eventuates (Fig. 5.8). Because of the large emissions from industrial logging, the Protect scenario was favoured over the Restore scenario when using biophysical data only (Fig. 5.11B). However, when incorporating communities' needs and constraints, the intervention with the greatest climate mitigation outcomes changed from Protect to Restore (Figs. 5.11A & B). These results differ from the local threats only results presented above, where the Restore strategy was always favoured over he Protect strategy.

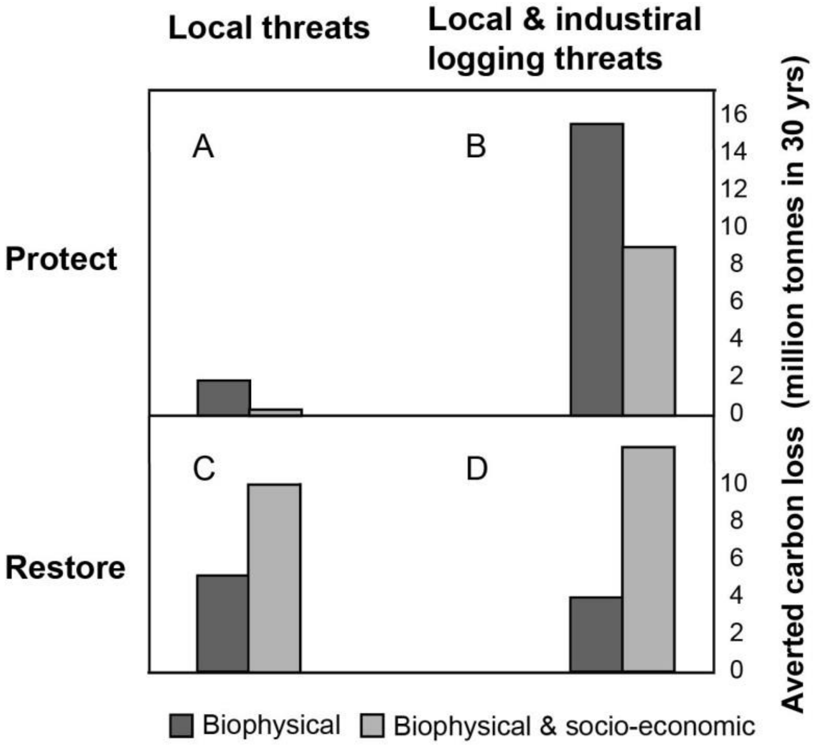

Figure 5.11. Averted carbon loss from protect and restore strategies under existing local threats and planned industrial logging threats using two land-use models, on that includes biophysical and socio-economic variables and one that omits socio-economic variables.

5.5 Discussion

Though the desires and constraints of forest-using communities are important drivers of local environmental change and are recognized as such under REDD+ policy, they are often missing from land-use planning tools (Angelsen et al. 2012). In this study, I integrated socio-economic variables from household surveys with more traditional biophysical parameters into a land-use planning framework. In the seven land-use types analysed, biomass and soil carbon surveys revealed significant variation in carbon stocks between land-use types (Fig. 5.5), with most carbon stored in primary forests. However, some degraded and production landscapes stored a surprisingly large amount of carbon and offered significant carbon sequestration potential (Figs. 5.5 & 5.6). From 1990 to 2010, all GHG emissions were from local drivers of forest loss and degradation, produced mostly by fires and fuel wood extraction (Fig. 5.7). The rates of forest carbon emissions were high (22.8 Mt CO_2e yr^{-1}); double the per capita national rates of emissions from industrial logging (Fig. 5.8) (Shearman and Bryan 2011). Results from household surveys showed that each land-use type hosted a unique suite of essential environmental products which participants were almost entirely dependent on for subsistence (Fig. 5.9), highlighting the importance of considering these environmental products in land-use planning exercises for REDD+ type projects (Fig. 5.9). The results of our land use planning model suggest that restoration activities are likely to hold greater climate mitigation potential than protection in most cases, as restoration tends to increase the land base that provides important resources to local communities (Fig. 5.10).

The value of forests for the rural poor is becoming increasingly recognized in international forums; reinforced by recent estimates of 1.2 - 1.5 billion people depending directly on forests (FAO 2014), including 60 million indigenous people who are almost wholly dependent on forests (WorldBank 2002). In a recent pan-tropical study, socio-economic data from household surveys in 24 developing countries revealed that household incomes from forests where on par with the income derived from cropping (Angelsen et al. 2014). Our results also show the indispensable values of primary forests to subsistence communities, with most dietary protein sourced from forest areas. Deficiencies in protein are a national health concern in PNG, inhibiting child growth (Bryant 2009).

The recognition of the importance of forests to the functioning of our societies has been widely used as a call to action for forest protection; e.g. if forests are threatened and communities depend on forest, then we should protect forests from the activities that threaten them (Chhatre and Agrawal 2009). Our findings indicate that forest protection could actually be less favourable when people's needs are taken into account. Though seemingly contradictory, the logic of these results are revealed in the details. First, natural forests were most often located at long walking distances from villages (Table 5.1); therefore increasing the footprint of forests near villages through activities that promote growth was more valuable than restricting the access to forests through protected areas. Second, forests were only one land cover type valued by communities and there was a strong desire to increase the services provided by these other land covers, moreover some of these other land covers had high carbon stores. Third, all forest loss over the historical period 1999-2010 was caused by local drivers, but only about half of the emissions came from natural forests with the rest deriving from degraded or production landscapes.

Though our study used Protect vs Restore to contrast the two main climate mitigation strategies in tropical forests, forest management to enhance livelihoods and carbon should ideally transcend simple dichotomies such as only ‘preserving natural forest’ or only ‘restoring degraded lands’ (Leach and Leach 2004). In reality, the benefits of each strategy are seldom mutually exclusive. To manage forest for climate mitigation, it is useful to use a model where forest can act as sink or a source of carbon dioxide. In the absence of logging, landscapes could be managed as a significant carbon sink. In this case restoration actively sequesters carbon and becomes a form of insurance against depleting carbon stores (Dixon et al. 1994). For instance, we found that restoration resulted in landscapes operating as a net carbon sink for all scenarios that excluded industrial logging but that protection never resulted in a net carbon sink as it was not able to overcome carbon losses from local drivers (Fig. 5.10).

Though protecting forests remains one of the most efficient and powerful ways to manage forest for climate change mitigation (Thomson et al. 2010), taking full advantage of benefits provided by forests will require actively restoring degraded lands (Houghton 2014). Because most international discussions surrounding climate mitigation have focused on forest protection to avoid deforestation (Gullison et al. 2007, Kindermann et al. 2008) rather than restoration to halt or revers forest degradation (Thomas et al. 2010), interventions for increasing carbon sinks through restoration are poorly represented in major environmental non-governmental as well as national and international strategies (Sasaki et al. 2011). In PNG the scope for restoration activities have not been explored.

Our results indicate that in the case of emerging industrial logging, it is almost impossible to manage forests as carbon sinks. Instead, benefits come from avoiding some of those emissions through protection (Houghton 2014). To implement appropriate climate change mitigation interventions, it is clearly important to identify and address the drivers of deforestation and forest degradation (Chan and Sasaki 2014). Across PNG, forest loss and degradation are caused by industrial logging (~40 %) subsistence agriculture (~40 %) and fires (~15 %), with no assessments accounting for the role of fuelwood apart the work presented here (Bryan et al. 2010b, Fox et al. 2010, Fox and Keenan 2011, Shearman and Bryan 2011). Our results revealed per capita emissions from fuelwood extraction in the YUS catchment surpassing national per capita emissions from fossil fuels and cement (Fig. 5.8 (CDIAC 2011). Moreover, anthropogenic fires destroyed Upper-montane forests at a faster rate than industrial logging of Lowland forest at a national scale (Shearman and Bryan 2011). Upper-montane forests are also being lost to fire at a rampant rate across PNG (Bryan et al.

2010). Because these forests store more carbon than previously thought, fires may also be causing higher carbon emissions than previously thought.

This chapter explores four restoration interventions, two of which directly address the main causes of carbon emissions while improving livelihood benefits. The first example is the restoration of degraded grassland through the expansion of shade coffee plantations. The establishment of shade coffee could help offset emissions from household fuelwood consumption as well as provide cash income for communities with few other income options. Shade coffee plantations in YUS stored more than twice the amount of carbon than those in other surveyed tropical regions (Schmitt-Harsh et al. 2012). The high carbon stocks are largely attributable to the type of shade-tree planted; a native *Casuarina sp*. having three times the wood density of the *Leucaena sp.* shade tree commonly used in other regions, meaning it stores three times the carbon for the same tree volume (Zanne et al. 2009). The use of native casuarina could be implemented at a national scale as coffee is a thriving export market for PNG, and the market could be extended to more remote areas with subsidies for transport provided by premiums paid for 'climate smart coffee'.

The second intervention explored in this chapter is fire control in high altitude forests. Accelerated natural regeneration by fire control is inexpensive, and thus represents an attractive restoration alternative to forest protection (Durst et al. 2010). Recent programs to reduce carbon emissions in northern Australian savannahs rely on extensive measurements of gas emissions during fires; these data underpin both public and private carbon farming schemes which are providing substantial income to Indigenous communities (Douglass et al. 2011). PNG could also be a suitable place to test this strategy because large areas, covering approximately 5 Million ha of degraded grassland that were once forest, are now maintained as grasslands through frequent anthropogenic fires (McAlpine and Freyne 2001). Because fires can be relatively easily monitored by satellite, it possible that a novel approach that sought to minimize fire could be pursued through payment for a reduction in fires scar area in a region (Oliveras et al. 2014a). The local participants in this study exhibited significant support for fire control as a means to restore forests, as fires also cause significant decreases in the habitat of the Huon Tree Kangaroo (Dendrolagus matschiei), the largest mammal in these forests and an important cultural icon (Brooks 2011).

Research that quantifies the provision of multiple services associated with different interventions and the trade-offs and synergies between them will lead to climate mitigation schemes that are more likely to be successful (Bennett et al. 2009). The landscape approach to environmental management, a framework within which to negotiate conflicting objectives with multiple stakeholders in order to reach a common vision for a landscape (Sayer et al. 2013), recognizes that maintaining multi-function landscapes has benefits to society and that trade-offs exist between different land-use strategies (Nelson et al., 2009). However, such an approach is not a land-use planning tool and is more akin to a set of guidelines. If the landscape approach adopts a land-use planning framework that integrates both biophysical and socio-economic variables, it could become a powerful approach to explore land-use strategies for multiple objectives, including livelihood benefits and carbon sequestration potential.

Though this study benefited from an extensive biophysical dataset and socio-economic PEN surveys, it had a number of limitations. Although we explored the potential impact of industrial logging, we did not explore the risk of alternative scenarios such as increased fire incidence or ingress of industrial mining or roads. The land-use planning results in this chapter should therefore not be considered prescriptive. Nonetheless, they do demonstrate that significant potential exists in combining household survey data with more traditional biophysical land-use planning models. However, further research is necessary to develop a more standard approach and a user-friendly tool for merging information gathered in questionnaires into spatial modelling tools.

Part of what makes REDD+ and other forest carbon projects different from most conservation or development initiatives is the accuracy to which outcomes must be quantified in order to reward successful efforts with potentially significant income streams. Additional complexities arise in countries where a significant fraction of the economy is subsistence-based, such as PNG. This is because most non-market environmental products provided directly to communities by natural ecosystems are not represented in national balance sheets and thus represent underestimates of national outputs and underestimates of poverty (Sunderlin et al. 2008). For this reason, forest-dependent people have may have lots to gain from REDD+, but it remains unclear how much they may lose in the face of new forest management systems (Angelsen et al. 2012). Though methods for quantifying the social impacts of different land use regimes exist, they have not been linked to carbon projects at a significant scale (Angelsen et al. 2012). This study provides a starting point for exploring alternative use of PEN surveys as a means to incorporate community values and constraints on land-use planning. The use of PEN surveys could potentially be extended to monitor the impact of climate mitigation strategies on forest resources and forest dependent communities.

In conclusion, by measuring carbon stocks, emissions and community values in a dynamic forest-human system this study shows that meeting climate mitigation objective does not necessarily require trade-offs against community needs. However, for these synergies to operate, broadening the focus from forest protection to including restoration in remote forested areas is required (Babon and Gowae 2013).

5.6 Summary of Chapter 5

- Land-use types not only ranged considerably in their carbon stocks, but also in the essential environmental products they provided to local communities.
- The potential for emissions reduction was significantly altered by including socio-economic considerations.
- Including socio-economic data resulted in a switch towards favouring restoration over protection under local threats and threats of industrial logging and novel opportunities for carbon sequestration were revealed.
- Shifting from the current focus for lowering forest emissions through protection near deforestation frontiers, towards restoring degraded lands could have multiple benefits, including meeting international emissions reduction targets, adhering to local values and improving livelihoods.

-END OF CHAPTER 5-

Chapter 6

General Discussion

Author: Michelle Venter

6.1 Main Contributions of Thesis to Research in My Field

Significant investments into tropical forest conservation for carbon purposes are now demanding an improved understanding of the state of these forests as well as new ways to manage tropical forests into the future. The forests in Papua New Guinea (PNG) are of considerable cultural, biological and economic value (Sillitoe 2013). In Chapter 2, I performed a literature review to explore options for managing tropical forests globally for carbon retention. In Chapter 3, I explored environmental drivers of forest biomass variations along an environmental gradient in Papua New Guinea, where remote indigenous people depend on forests for subsistence. In Chapter 4, I explored how these forest dependant people can contribute their knowledge and skills to monitor the carbon stored in their forests. In Chapter 5, I integrated biophysical and socio-economic data into a land use planning model to predict flux in biomass across a broad landscape, to help prioritize management actions for lowering greenhouse gas (GHG) emissions.

The research conducted for this thesis represents a comprehensive inventory of forest carbon across all three main forest types of PNG (Lowland, Montane and Upper-montane), and found overall higher carbon stores than others have reported for the country (Chapters 3 & 5). Presenting some of the first accounts of Upper-montane forest biomass for PNG, this thesis has measured some of the highest above ground biomass (AGB) at elevations of 2,200 to 3,100 m asl worldwide. I showed that these high biomass estimates were the result of a diverse range of tree trees growing to exceptional height and girth in a cloud immersion zone that is part of a similar climate envelope found in temperate maritime areas where some of the world's largest trees grow (Chapter 3).

To further our understanding of the environmental drivers of forest biomass in Papua New Guinea, I conducted a large field campaign that sampled primary forest biomass across a broad environmental gradient, including climate, topography and edaphic factors along a 3,100m elevation transect. Through this work, natural disturbance from landslides and windthrows was identified as the main driver of variation in forest biomass (Chapter 3). By sampling across varied topography, including very steep slopes which have not previously been extensively studied in biomass inventories; new insights into the relationships between AGB and natural disturbance were gained. These data allowed for the development of a simple model that quantifies the effect of natural disturbance on AGB using slope of terrain alone. Since PNG is typified by rugged topography, and slope angle can be easily obtained from digital elevation models, findings from this research could potentially help improve predictions of forest biomass across PNG's forest estate.

In addition to being carbon-rich, I showed through a land-use change analysis that Upper-montane forests in our study area underwent the highest deforestation rates due to El Niño fires (Allen and Bourke 2009)(Chapter 5). These rates of loss were similar to national rates of deforestation found in those forests (Shearman and Bryan 2011). However, previously most discussions surrounding REDD+ projects have focused on Lowland forests. Because of their high carbon values than expected and their high deforestation rates, my findings show the importance of including fire control in high altitude forests as an essential climate mitigation strategy in PNG.

To help prioritize management actions for lowering greenhouse gas (GHG) emissions in forest-dependent communities, the thesis used an approach to land-use planning that integrates both carbon and community values for different land-use types. This approach revealed new opportunities and constraints (Chapters 2 & 5); notably, community desires altered the choices for mitigation strategies without compromising the scale of GHG mitigation outcomes. In particular, carbon restoration in human-modified landscapes produced the greatest synergies between climate mitigation and community needs. A deeper understanding of the trade-offs between the value of forests provided to the local community allows tailoring management actions to achieve local and global objectives.

To assess whether communities monitoring programs collect forest-carbon data with the potential to increase PNG forest monitoring capacity, I conducted a complete re-measurement campaign of community-based forest inventories. The results from this Chapter demonstrated that quantitative data produced by these programs are as reliable as those produced by scientists. Therefore, community-based monitoring can serve to overcome an important roadblock for PNG's participation in global climate mitigation strategies. Furthermore, by tracking the measurements in the field that led to the largest discrepancies in measured forest biomass between the two surveys, my findings highlight the importance of refining existing biomass survey protocols to improve techniques for accurately measuring and accounting for large trees.

6.2 Forest Carbon in Remote PNG

PNG has had relatively little investigation of their carbon stores partly because of the high cost and time constraints associated with sampling in roadless areas with no infrastructure or means of communication (Bryan et al. 2011, Martin et al. 2012). As a result, the lack of forest-carbon inventories introduces significant uncertainty into global carbon stock and flux models as well as impedes the countries' participation in REDD+ activities (Mitchard et al. 2013).

Because PNG's vegetation varies considerably with elevation (McAlpine et al. 1983), the most commonly used vegetation classification uses Lowland, Montane and Upper-montane elevation zones for forest mapping (Paijmans 1976). The largest effort to quantify above ground carbon (AGC) stocks in PNG to date is from a country-wide network of permanent forestry plots (Fox et al. 2010). Though these plots are representative in geographical extent, they are not representative of all forest types. Most of the plots are in Lowland forests, with only two plots in Montane forests and none in Upper-montane forests. PNG has 28 Million ha of forests, of which 65 % are in Lowland, 32 % in Montane forest and 3 % in Upper-montane forests (Shearman and Bryan 2011). Another substantial effort by Bryan et al. (2010a), used field data from various sources to model biomass using high resolution mapping and contributed the first estimate of national carbon stocks in logged and unlogged forests. From these sources, mostly located in Lowland and Montane forests, they estimated the average biomass of unlogged forest to be 358 Mg ha^{-1} (or roughly 179 Mg C ha^{-1}).

The above ground carbon (AGC) inventories reported in Chapter 3 were higher than those reported by Fox et al. (2010), and substantially higher those used as IPCC default values for each forest type (IPCC 2006). Aside from plot location and size, the field methodologies and allometric equations used here were similar to those employed by Fox et al. (2010). Thus, the differences in AGC between these two studies are likely to be genuine. Fox et al. (2010) acknowledge that their AGC values were low for moist tropical forests and the low values were probably linked to plots being located near roads or populated areas, and thus likely to be degraded.

Significant resources were required to complete the field work for this thesis. For instance, I was required to spend 250 days living and working in the forest with the contributions of over 70 field assistants from a dozen villages. The cost of bush flights and provision of food and salaries for the field assistants and carriers exceeded US$70,000. To make the most of this effort, the research prioritized gathering a representative sample across environmental gradients that are known to affect biomass (Clark and Clark 2000). This approach was valuable for generating an assessment of carbon inventories both for the main vegetation types assessed during a parallel remote sensing analysis (Gillieson et al. 2011) as well as allowing an integrated measure of the carbon inventory for the area (Venter et al. 2012b) and estimating the carbon emissions from land-use change between 1990 and 2010 (Chapter 5).

However, the approach to prioritize gathering a representative sample also had a number of drawbacks. First, to adequately stratify primary forests required the establishment of many small plots of 0.1 ha as opposed to larger 1.0 ha plots and small plots have been shown to heighten correlations between AGB and large trees (Stegen et al. 2011). Secondly, the intensive nature of the campaign meant that only one elevation gradient could be sampled within a three year time frame. The lack of replication limited the scope for the interpretation of my results as no other similar studies have been carried out in PNG and the results diverged from those obtained from other tropical elevation gradients. Thirdly, because the time constraints, tree measurements could only be recorded at one point in time. Thus my results represent a snap shot of biomass in the forest, with no real indication of primary productivity. Therefore, the lack of information about forest growth and carbon accumulation over time limits our ability to assess climate mitigation efforts in the area.

Moreover, the environmental variables collected in the field, including forest structure, soil characteristics and topographic variables as well as a large number of variables from various global datasets on climate and topography did not reveal the significant direct relationships with AGC that were expected (Selmants et al. 2014). Many studies have measured AGC in the tropics across a range of elevations and most of these have demonstrated clear and direct relationship between AGC and either climate (e.g. solar radiation, rainfall, temperature), edaphic variables (e.g. soil depth, soil type, nutrients), topographic variables (slope angle, aspect) or altitude (Vázquez and Givnish 1998, Kitayama and Aiba 2002, Girardin et al. 2010, Homeier et al. 2010, Malhi et al. 2010, Girardin et al. 2013). In contrast, my findings were defying the quasi pan-tropical trend of significant decline in AGB with increasing elevation, for exceptions see Selmants et al. (2014) and Culmsee et al. (2010). It is possible that PNG's forest carbon stores follow different elevation trends from other tropical forests, with a peak at higher elevations. However, drawing a firm conclusion from the findings of this thesis will not be possible until other forests are measured across other environmental and altitudinal gradients in PNG.

Nevertheless, the large dataset collated for this work served to uncover other important drivers of forest biomass. For instance, by sampling on steep terrain, I was able to demonstrate that slope alone could be used as a reliable predictor of natural disturbance, the strongest driver of AGC in the study area. Though LIDAR and RADAR and other remote sensing technologies are promising technologies for providing surrogate measurements of forest structure, insights into small-scale forest dynamics have been limited to ground-based forest inventories (Antonarakis et al. 2010). However the difficulty in accessing steeply sloped areas and the dangers associated with working on them means that still little is known about tropical forest dynamics on steep slopes (Southworth and Tucker 2001, Lu 2006). Overcoming this challenge required the use of rappelling equipment and led to novel insights about these forest which could help describe potentially generic trends in tropical Montane forests (Spracklen and Righelato 2014).

It is common for ecological studies to be located in easily accessible areas, as such studies are time and resource efficient (Reddy and Dávalos 2003). However, such geographically-biased sampling has strongly skewed our understanding of ecological trends (Costa et al. 2010, Martin et al. 2012, Varela et al. 2014). A national forest inventory of PNG forest, led by UN-REDD is due to commence in 2016 (http://www.unredd.net). The national forest inventory should include remote areas; as the results from this thesis have showed conducting forest near human settlements could seriously underestimate PNG's carbon stores. Likewise, any effort to generate a representative sample set should include inventories in montane and Upper-montane forests, which form 68 % of PNG's forest estate (Shearman and Bryan 2011).

6.3 Large Trees

Improving our understanding of the factors that drive the distribution of large trees is a necessary step towards prioritizing their conservation. Papua New Guinea is rich in flora with high endemism and may host about 6 % of the world's flora, but current estimates remain highly imprecise, ranging from 11,000 to 25,000 species (Sekhran and Miller 1994, Bishop_Museum 2015). The uncertainty in the knowledge of the taxonomy and extent of botanical diversity, forest structure attributes that may drive patterns in forest carbon will impede the development of biodiversity and forest carbon conservation plans (Ashton 2003). The collection of botanical specimens required by IUCN species listing has not been met for most of PNG's tree species, with most sampling having occurred only in road-accessible areas, along major rivers, the coast and around large communities (Bishop_Museum 2015).

The importance of large trees in forest carbon inventories emerged as a theme in several chapters of this thesis; their disproportionate contribution to AGC stocks meant that they governed trends in forest biomass across broad environmental gradients (Chapter 3) and introduced important errors if not properly measured (Chapter 4). Moreover, a high density of large trees was found at high altitude, correlated with optimal climatic conditions, and resulting in unusually high carbon stores for these elevations (Chapter 3). Rapid loss of these forests are causing higher GHG emissions than previously thought (Chapter 5). A number of recent global studies (Goldsmith et al. 2013) have highlighted the role of large trees in the global carbon cycle (Keith et al. 2009, Slik et al. 2013, Stephenson et al. 2014). For example, large old trees were previously thought have lower sequestration rates than smaller trees in young fast-growing forests, but a study of 403 tree species demonstrate that growth rate increases with size, meaning that older, larger trees are not only valuable carbon stores but important carbon sinks (Stephenson et al. 2014). Moreover, local and pan-tropical patterns in forest carbon have been shown to be driven by the distribution of large trees (Laumonier et al. 2010, Slik et al. 2013).

Despite recognizing the function of large trees in the carbon cycle, little has been done to improve how large trees are accounted for in biomass surveys in tropical forests (Brown 2002, Masera et al. 2003, Pearson et al. 2005, Qureshi et al. 2012). The importance of properly measuring large trees was apparent in Chapter 4, where large trees constituted only 14 % of the stems in the study area, but errors on these trees caused 85 % of the biomass discrepancies. Often, forest inventory protocols suggest taking less accurate 'eyeballing' estimates for large trees in order to overcome the significant challenges in directly measuring large trees. Improvements in the field could be as simple as attributing proportionally more time and resources to large trees in acknowledgement of their proportionately larger carbon stocks. For example, building bush ladders when required to measure diameter above buttresses and taking multiple canopy height readings from different locations until the highest point is recorded could substantially improve large tree measurements and in turn forest biomass estimates.

Reducing error in estimating the carbon stocks of large trees could also be achieved through tailoring allometric equations specifically to large trees, because generic equations are most often constructed from tree parameters measured on smaller trees (Chave et al. 2005a) and thus could be less applicable to larger trees. For example, the heaviest tropical tree ever recorded, estimated at 76.1 tonnes, was measured directly by weighing it piece by piece. It did not have exceptional diameter (158 centimetres) or height (44 metres) (Goodman et al. 2012) and the weight of this same tree was significantly underestimated when using a number of commonly used allometric equations. This clearly demonstrates the difficulty of predicting the weight of large trees. Considering that the majority of forest carbon is in large trees, it is possible that we have been underestimating the terrestrial carbon sink to a significant degree.

Compared to temperate regions, very little is known about what drives the presence of large trees in the tropics (Larjavaara 2014). One of the most enigmatic issues concerning large tropical trees is why mature Lowland forest have larger trees than upland forest, if higher elevation forest have the more temperate climates that are known to promote the growth of large trees in other parts of the world (Rumney 1968, Larjavaara 2014)? Clearly, rainfall and temperature are only two of the factors that affect the ability of trees to grow tall; other factors which can affect tree growth in Montane forests are the presence of strong winds, landslides on steep slopes, shallow soils and climatic isolation (Malhi et al. 2010). Another factor that could influence the ability of a species to grow tall is their genetic ability to do so (Givnish et al. 2014). The traits that shape plant strategies and influence community assembly processes are strongly linked to the phylogenetic history and the relatedness of co-occurring species and their traits (Kembel 2009).

Currently, no empirical analysis has explored the biogeographical origin of tree lineages in PNG (Sniderman and Jordan 2011). The country has an interesting assortment of natural histories; though the island of New Guinea lies on the eastern side of the Wallace Line and shares many faunal characteristics with Australia, the flora is most closely related that of East Asia. The only exception to this is in pockets of forest in higher elevation that have only been superficially described as having Gondwanan ancestry (Linder and Crisp 1995). Further research could explore whether traits for large trees are found in thermally similar regions of PNG and if spatial sorting of tree lineages has occurred, using elevation as a proxy for climate. It is also possible that elevation acted to sort lineage ancestry (Costion et al. 2011) with temporally conserved habitats of upland moist forest retaining tall statured species from ancestral southern temperate rainforest lineages (Kooyman et al. 2012, Givnish et al. 2014).

Cloud forests are known for their climatic stability (Oliveira et al. 2014); the sheer diversity of large trees found in the study area could be an indication that these communities are well adapted to a climate that could have originated from more ancient and temperate region, such as Gondwana (Sniderman and Jordan 2011). If this is the case, the predicted effects of climate change on Montane cloud forests could upset the climate stability required by these forests, threatening their biodiversity and carbon values (Oliveira et al. 2014). It is unclear how changes in traditional burning practices as well as climate change will affect PNG's biodiversity. However, alpine plant species are known to be affected by changes in fire regimes and climate change (Kirkpatrick and Bridle 2013). Because Upper-montane forests are under the greatest threat in PNG, due to fire and potentially due to climate change, future research should further our understanding of how to secure these forests into the future.

Given the opportunity, I would pursue this research using a biogeographic approach to study forest community assembly processes, and test whether large trees in Montane forests are described as having predominantly 'southern' (Gondwanan) origins, compared to Lowland species using molecular phylogeny. The first phase of this project would involve collating a species list for PNG's tree species that have a spatial or elevation information through a literature review from various sources including unpublished botanical surveys, whilst collaborating closely with the Lae Herbarium and the Australian Tropical Herbarium. Quantifying rain forest assemblage-level phylogenetic structure and trait variation across gradients has been used to elucidate the link between functional diversity and community assembly processes through time (Kooyman et al. 2012). This phase would allow me to answer broad questions about linage of ancestry with elevation at a minimal cost. The second phase would involve using the more traditional approaches of climate modelling and population genetics to explore the distribution of large trees and whether these are associated with particular climates or geographical features. The third phase would collect DNA samples from trees in areas identified as most likely to harbour Gondwanan relics. This integrated approach would be used to provide support for the existence of rainforest refugia that could potential boast some of the largest trees in high elevation forests worldwide.

6.4 Restoration of Degraded Lands

The benefits of restoring degraded forest lands for climate mitigation strategies was first highlighted through a literature review in Chapter 2, which demonstrated that the wide range of restoration strategies having various costs, benefits and risks. From forest inventories in PNG, human modified landscapes were revealed as having high potential for carbon sequestration in Chapters 3 & 5. Finally, in Chapter 5, a land-use change analysis and assessment of fuelwood extraction revealed that local drivers of deforestation and forest degradation resulted in substantial carbon emissions. In Chapter 5 I went on to perform a land-use planning exercise that included data from household surveys to discover that the benefits of restoration actions could go beyond carbon sequestration to provide near-term livelihood benefits for local communities. Taken together, these findings provide support for restoration actions that directly address drivers of forest loss and degradation, such as fuelwood extraction and anthropogenic fires. These actions can provide win-win solutions for climate mitigation and development goals.

An estimated 1.2-1.5 billion people depend to some extent on forests for their livelihoods, including 60 million indigenous people who are almost wholly dependent on forests for their basic needs (WorldBank 2002). More specifically, many studies have identified the set of ecosystem services and direct benefits, such as food, fibre and energy, provided by forests to people (Parrotta et al. 2015). These arguments have often been used to support the adoption of strategies to protect forests (Zenteno et al. 2013, Wunder et al. 2014, Agung 2011). In agreement with this narrative, the results of household surveys in in the YUS Conservation Area presented in Chapter 5 also showed that forests provide essential products to remote communities, including their primary source of dietary protein and building material (Fig. 5.9). However, once community forest values were added to a land-use planning exercise, I found that the possibility for forest protection was diminished as these communities needed extractive access to forests on their lands. This finding calls into question the current view that sees the implementation of forest protection as the primary means to accommodate forest-dependent communities (Pistorius 2012).

Instead, Chapter 5 indicates that pursuing forest carbon benefits through restoration may in cases be preferential over forest protection. In contrast to forest protection, restoration can increase net forest carbon stores while simultaneously increasing the provision of forest services to communities. Given community's values of forest resources, forest restoration may be met with greater local participation than forest protection Most natural forests are located long distances from villages (Table 5.1) and require many hours to reach by foot. Thus, increasing the extent of these valuable resources rather than restricting access to them is valued by these communities. Also, many daily essential environmental products, such as fuelwood and food are provided by nearby degraded or production landscapes, not primary forests (Fig. 5.9). However, this strategy does depend on a supply of degraded lands near villages that does not supply other important values. In the case of the YUS area, and potentially other areas of PNG, this supply of degraded lands is available from the extensive anthropogenic grasslands adjacent to villages.

Despite the potential benefits of forest restoration, most international discussion surrounding climate mitigation action in the tropics have focused on forest protection to avoid deforestation (Gullison et al. 2007, Kindermann et al. 2008, Sasaki et al. 2011). Moreover, much of the debate about land-use in the tropics adopts the 'land sparing vs sharing' approach, which aims to either minimize forest loss by intensifying agriculture or extraction practices (sparing) or promoting 'forest friendly' industrial or extraction practices (sharing) (Ewers et al. 2009). Though these approaches have made a significant contribution to land-use science (DeFries and Rosenzweig 2010); they largely ignore restoration options and overlook scenarios involving degraded areas (Law and Wilson 2015) . In PNG, the potential for restoration in the context of climate mitigation strategies and community benefits seems strong.

In PNG, forest loss and degradation are caused by industrial logging (~ 40 %) subsistence agriculture (~ 40 %) and fires (~ 15 %), with no assessments accounting for the role of fuelwood extraction on forest degradation or carbon emissions aside the work provided in this thesis (Bryan et al. 2010a, Fox et al. 2011). Results from Chapter 5 revealed per capita emissions from fuelwood extraction in the YUS catchment surpassed national per capita emissions from fossil fuels and cement (Fig. 5.8)(CDIAC 2011). Moreover, fires between

1990 and 2010 destroyed Upper-montane forests at a faster rate than national rates of Lowland deforestation from industrial logging. With Upper-montane forests being lost to fire at a rampant rate across PNG (Bryan et al. 2010a) and because these forests store more carbon than previously thought (Chapter 3), fires may be causing higher carbon emissions than previously thought. Therefore, the implementation of successful carbon management strategies will need to address these drivers through appropriate interventions (Chan and Sasaki 2014).

Accelerated natural regeneration could address two main drivers of carbon emission is the use of fire control to promote natural growth. In particular, accelerated natural regeneration by fire management is inexpensive, and thus represents an attractive restoration alternative to forest protection (Durst and Spirovska-Kono 2010). In Australian savannahs, manipulating fire regimes in order to reduce carbon emissions is providing substantial income to Indigenous communities (Murphy et al. 2015, Scheiter et al. 2015). PNG could be a suitable place to further test this type of strategy, as more than 10 % of its surface area is covered by degraded grasslands which are maintained by frequent anthropogenic fires (McAlpine and Freyne 2001). Payments for reduction in fires scars can be relatively easily monitored by satellite, and could promote the return of these grasslands to their naturally forested state (Oliveras et al. 2014b).

Future investigation into fire management in PNG could include identifying areas suitable for accelerated natural regeneration. As a preliminary exercise, grassland that has naturally reverted to forest from grassland during the last 30 years could be identified using satellite imagery. These areas would be assessed for environmental correlates (such as slope, aspect, soil type, altitude) and socio-economic correlates (distance to village, income generation). These correlates could then be used identify grassland areas that have the highest potential for natural regeneration. Once these areas have been identified, rough estimates of carbon sequestration rates could be achieved using carbon stock and flux estimates from different land-use types (Chapters 3 & 5) and forest growth projections developed by Keenen and Fox (2011). The feasibility of using accelerated natural regeneration by fire control would be contingent on conducting controlled field experiments to measure the efficiency of carbon sequestration through fires and the recovery rate of Montane forests, as most of our knowledge about such restoration activities have been garnered in Lowland regions (Lippok et al. 2014).

6.6 General Conclusion

In Papua New Guinea, large tracts of pristine, carbon rich-forests interspersed with forest-dependent communities make it a unique and challenging place to implement climate mitigation strategies. Though many roadblocks have hampered the implementation of climate mitigation strategies in PNG, the findings of this thesis provide many reasons to remain hopeful. For instance, it is possible that more carbon is stored in remote areas of the country than previously thought, particularly in montane forests. Also, locally-based monitoring programs could produce robust data on changes in forest carbon; improving the countries monitoring capacity. Moreover, local needs align with land management practices that sequester carbon through forest restoration. The contribution of knowledge and skills by local communities to climate mitigation actions is an essential component of any REDD+ strategy (Angelsen et al. 2012). PNG's population possess immense knowledge of their forests. With locally relevant management practices and engagement of local communities, PNG could become an exemplar where both global needs of climate change alleviation and local livelihood are improved.

-END OF CHAPTER 6-

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Appendices

Appendix 3.0

Methodology for altitude−specific and height−diameter models taxa−specific

Height−diameter (HD) relationships have been shown to vary for different species (King 1996) and different altitudes [Marshall et al 2012] in tropical rainforest. Fox et al. (2010) fitted several non−linear models that performed well for tropical forests and found that the hyperbolic model by Fang & Bailey 1998 best described the HD relationship for tree species in PNG. We applied the same model which computes HD relationships for different species. However to account for the change in HD relationship with altitude, we obtained parameters of the model (a, b and c) for five elevation categories (0−500; 500−1500; 1500−2500, 2500−2800 and 2800+)

$$H_{altitude(i)} = a_i \ + b_i(1 - e^{-c(D_i - Dmin_i)})$$

Equation 3.3 Height predicted from diameter and altitude. Where: a, b and c are parameters estimated for each of the tree species for an altitude category ($\boldsymbol{i}$); and D_{min} is the minimum observed diameter for the species at that altitude. The model was computed for five altitude categories (0−500m; 500−1500m; 1500−2500m, 2500−2800m and 2800m+)

Appendix 3.1

DNA−barcode methodology and results from the analysis

DNA−barcodes are effective at providing tree species identification in areas where taxonomy is poorly known and which has cryptic populations (Costion et al. 2011). DNA−barcoding analysis was carried out on leaf samples from 50 common trees in the study area. The analysis required a sample of only a small piece ($1cm^2$) of fresh leaf material preserved in silica beads. The DNA was extracted from the samples using a Qiagen DNeasy mini kit and quantified using the naDrop 200, then amplified using standard polymerase chain reaction techniques. Successful amplification is determined by agarose gel electrophoresis and the samples are sequenced using Applied Biosystems BigDye Terminator v. 3.1. The samples were amplified using a generic plastid rbcL marker. Sequence results were compared to the GenBank database (National Centre for Biotechlogy Information) using the nucleotide BLAST function (Basic Local Alignment Search Tool) and a phylogenetic distance tree is created using the neighbor joining method.

The distance tree places the unknown sample within a broad phylogenetic framework, thus providing information on the identity of the unknown. Analysis of the distance tree allows us to evaluate to what taxonomic level we have identified the specimens reliably. The molecular technique reliably identified trees without any information about the individual, where 100% were reliably identified to Family, 78% to Genus and 2% to species. Local names from five languages were recorded for over 300 tree taxa. A sample of the local tree taxa identification is found below in Table S3.1.

Table S3.1 DNA−barcoding from 50 trees without any information about the individual except the local name

Sample number	Family	Species	Local name
1	Apocynaceae	Ochrosia sp	Otigot
2	Apocynanceae	Cerbera sp	Singlabana
3	Cunoniaceae	n/a	Unknown

4	Cunoniaceae	Davidsonia sp	Moun
5	Cunoniaceae	Ceratopetalum sp	Wapmang
6	Elaeocarpaceae	Elaeocarpus sp	Koyo
7	Elaeocarpaceae	Sloanea sp	Yat
8	Euphorbiaceae	Macaranga sp	Bon
9	Euphorbiaceae	Endospermum sp	Ipa ipa
10	Euphorbiaceae	Acalypha sp	Modung
11	Euphorbiaceae	Endospermum sp	Oromop
12	Fagaceae	Lithocarpus sp	Domong
13	Himantandraceae	Galbulimima belgraveana	Sogung
14	Icacinaceae	*Possibly undescribed species	Debat
15	Lauraceae	n/a	Bahoho Ongam
16	Lauraceae	n/a	Ferom Siup
17	Lauraceae	Cryptocarya sp	Moyup
18	Malvaceae	Trichospermum sp	Bakup
19	Malvaceae	Sterculia sp	Sebong
20	Meliaceae	Synoum sp	Dimunu
21	Meliaceae	Aglaia sp	Sopon
22	Monimiacea	*Not in database	Somamon
23	Moraceae	Artocarpus sp	Bonbon
24	Moraceae	Ficus sp	Sicum
25	Moraceae	Maclura clade	Uping tap tap
26	Myrsticiaceae	Horsfeldia sp	Nayac
27	Myrtaceae	n/a	Diwa
28	Myrtaceae	Eugenia sp	Gip
29	Myrtaceae	Myrcianthes sp	Kokec Kokec
30	Myrtaceae	Syzygium sp	Manung
31	Myrtaceae	Decaspermum	Viroc
32	Myrtaceae	Decaspermum	Songamon
33	Ochnaceae	Brackenridgea sp	Farot
34	Oleaceae	Olea sp	Bot Nongun
35	Paracryphiaceae	Quintinia sp	Busic
36	Phyllanthaceae	Glochidion sp	Tet
37	Pittosporaceae	Pittosporum sp	Wakumbong
38	Podocarpaceae	Prumnopitys sp	Tatong Kombut
39	Proteaceae	Helicia sp	Dumang simup
40	Proteaceae	Helicia sp	Kunoring
41	Rousseaceae	n/a	Tom
42	Rubiaceae	Wendlandia sp	Fopun
43	Rutaceae	Acronychia sp	Unknown 2
44	Rutaceae	Acronychia sp	Yararip
45	Sapotaceae	Pouteria sp	Bisic
46	Staphyleaceae	Staphylea sp	Unknown 3
47	Symplocaceae	Symplocos sp	Dum simup
48	Winteraceae	Tasmannia sp	Sumbirin
49	Winteraceae	Zygogynum sp	Tongo tongo

[1]n/a means the analysis was not conclusive to that level of taxa and '*' means the DNA−from that sample was not in the database or that the species is possibly undescribed

Appendix 3.2

Estimating biomass of trees

Biomass of dead trees without obvious signs of decomposition was estimated using the same equation as live trees but used a reduction factor for WSG of −6% for broadleaf species and −3% for conifers (Harmon et al. 2011). Biomass of dead standing trees with obvious signs of decomposition was estimated using the same equation for live trees with a WSG value of 0.30g cm^{-3} (Harmon et al. 2011). Lying dead biomass was estimated using protocols by Pearson et al. (2005) but WSG values of lying dead trees were determined using the density reduction values for the different decay classes as described by Harmon et al. (2010) (e.g. using Hardwood: −1 %, −49 % and −67 % and softwood: −3 %, −8 %, −45 %) (Harmon et al. 2011)

Table S3.1 Allometric equation used for estimating dry above ground biomass (AGB)

Tree Type	Allometric equation[§]	Source
Tree (wet tropical forest)	$AGB = 0.0776 \times (\rho\, D^2 H)^{0.940}$	Chave et al. (2005b)
Tree (moist tropical forest)	$AGB = 0.0509 \times \rho\, D^2 H$	Chave et al. (2005b)
Palm, tree ferns, Pandanus	$AGB = 6.67 + 12.826\ (H)^{0.05} ln(H)$	Pearson et al. (2005)
Liana	$AGB = exp^{(0.12+0.91[logBA])}$	Pearson et al (2005)
Dead standing broadleaf	$DSB = [Eq.1] \times 0.94$	Pearson et al. (2005)
Dead standing conifer	$DSB = [Eq.1] \times 0.97$	Pearson et al. (2005)

Dead standing bole	$DSB = [0.33\pi \times 0.00008(D^2)\ H\ \rho]100$	Harmon et al. (2011).
Lying dead wood*	$LDW = \left(\frac{\pi^2[D_1 + D_2 + D_n]}{8\ \times l}\right)\rho$	Pearson et al. (2005)

where AGB above ground dry biomass is in Kg, 'D' is DBH in centimeter, 'H' is height in meters, and 'ρ' is wood specific gravity (WSG) in grams per cubic centimeters, BA is basal area in Kg·m^{-2} measured at DBH. *Note: the formula in the bracket is volume in m3 ha-1

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Appendix 3.3

Exploratory analyses for sample design of Chapter 3

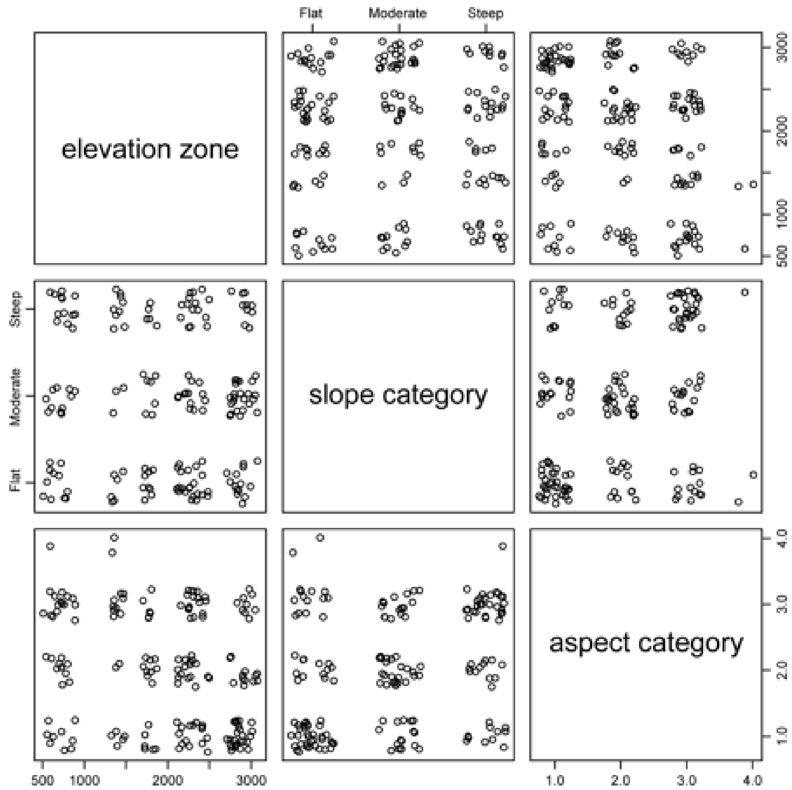

Figure S3.1. Pairwise plots of various slope and aspect categories were well represented in each of the nine elevation zones along the altitudinal gradient. Elevation zone 50 was omitted from this test as sites in this zone had zero aspect and slope.

Appendix 3.4

Pairwise relationship between AGB and a set of environmental variables

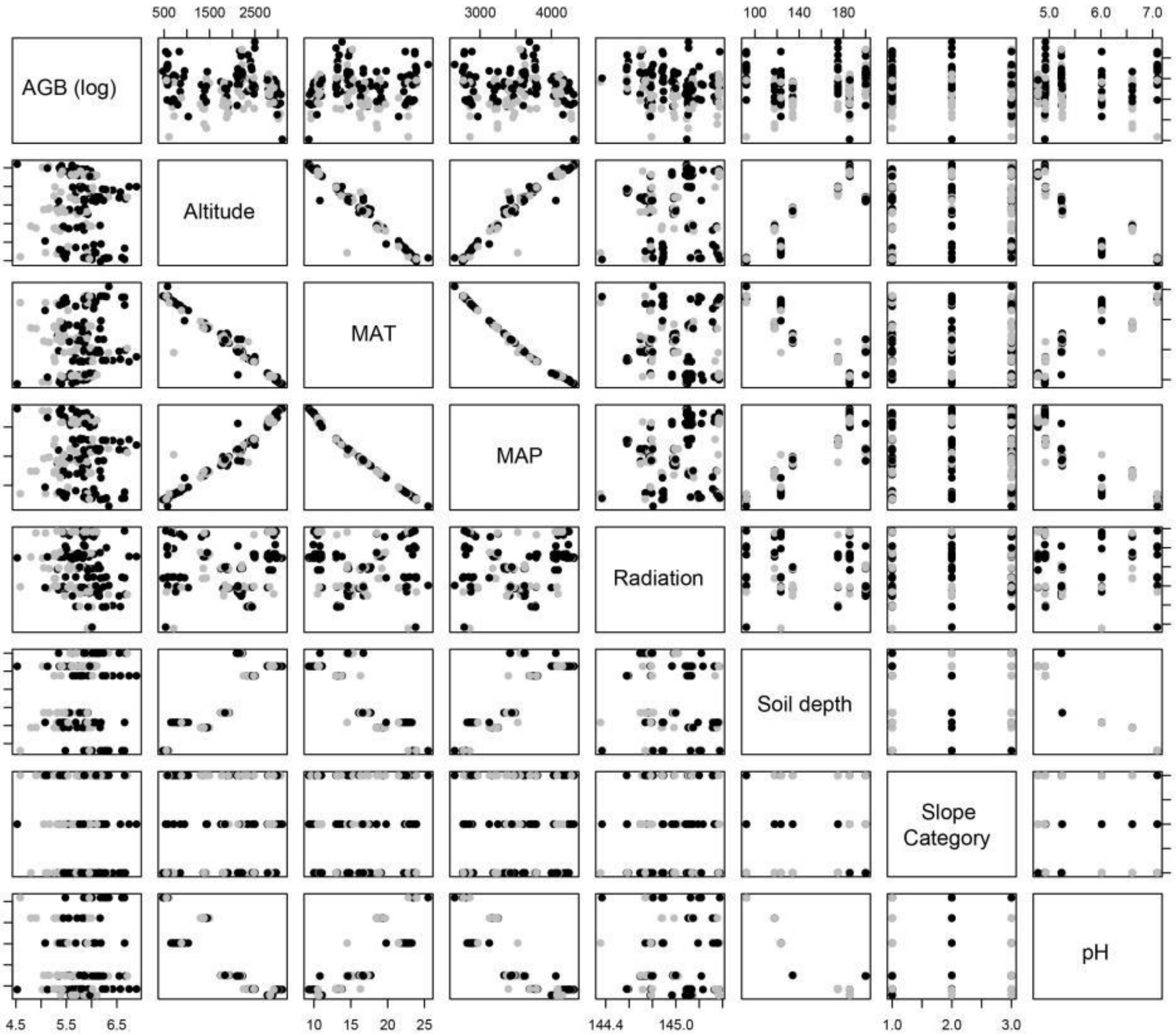

Figure S3.2. Pair plot for AGB (log scale), altitude and six of the environmental variable in examined our study. Black dot represent a plot without disturbance and grey plot represent a plot with natural disturbance.

Appendix 3.5

Residuals of MAP/MAPET analysis

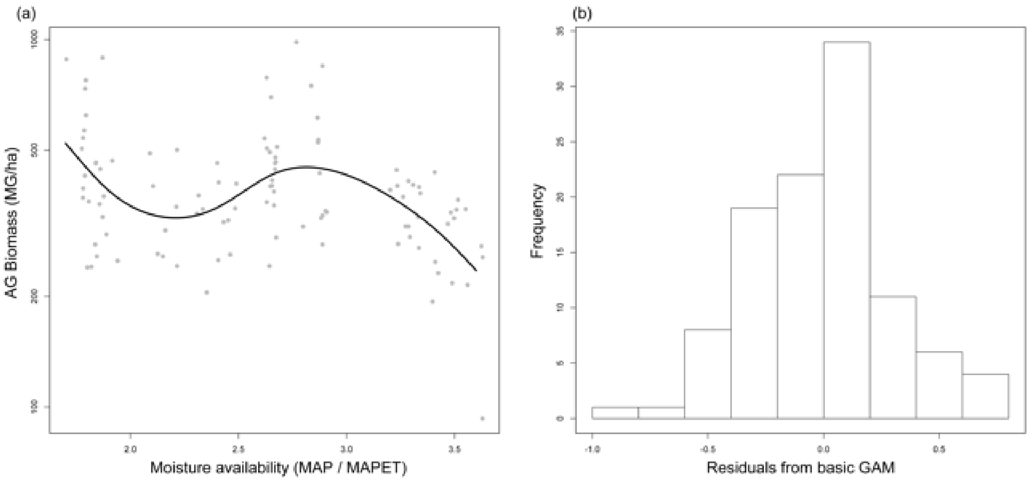

Figure S3.3. GAM Relationship of AGB with climatic, edaphic and topographic variables.

Appendix 3.6

Natural disturbance, caused by landslides and windthrows, explained more variation in AGB all other environmental variables obtained for this study

Table S3.2 Output of relationship AGB and edaphic variables from GAMs

Variable	F Value	SE	df	t-value	P-value
MAT	0.039	0.024	191	1.590	0.113
MAP	0.046	0.049	191	1.687	0.098
Solar radiation	-0.060	0.131	191	-0.459	0.646
Soil depth	0.006	0.004	8	1.627	0.164
Soil pH	0.088	0.195	8	0.450	0.671
SOC	0.082	0.214	8	0.517	0.623
Disturbance	-0.236	0.057	159	-1.153	0.516
Intercept	12.52	19.26	191	0.658	0.5146

Appendix 3.7

List of Large Trees

Table S3.3 Trees height of tropical montane of the Huon Peninsula, PGN of families with trees of DBH > 70cm$^{\beta}$

Family	Genus	500-1,000m asl	1,000-2,000m asl	2,000-2,700m asl	2,700-3,100m asl
		Tree height (m)			
Fabaceae	Pterocarpus	64			
Hernandiaceae	Hernandia	56			
Tetramelaceae	Octomeles	55			
Elaeocarpaceae	Sloanea, Elaeocarpus	49		24	40
Sapindaceae	Pometia	47	26		
Melastomataceae	Melastoma	46	29	32	
Cunoniaceae	Caldcluvia, Davidsonia Schizomeria,	45	33	37	39
Moraceae	Ficus	43	35	36	
Escalloniaceae	Quintinia, Carpodetus	41	18	29	28
Anacardiaceae	Dracontomelon	40			
Meliaceae	Chisocheton	37			
Achariaceae	Pangium	36			

Myrtaceae	Syzygium	33	40		27
Nothofagaceae	Nothofagus, Lithocarpus	30	33	30	41
Myristicaceae	Myristica, Horsfeldia	36	33	36	30
Rubiaceae	Neonauclea, Timonius	37	46	15	20
Euphorbiaceae	Endosperum, Cleidion Homalanthus, Macaranga	31	33	39	29
Lauraceae	Cryptocarya		34	32	27
Sapotaceae	Pouteria		33	31	
Urticaceae	Dendrocnide		32	30	30
Cupressaceae	Libocedrus		30	27	31
Monimiaceae	Dryadodaphne			40	30
Apocynaceae	Alstonia			39	
Podocarpaceae	Dacridium, Podocarpus		28	36	31
Staphyleaceae	Staphylea			31.0	
Actinidiaceae	Saurauia			28.0	30
Icacinaceae	Platea			27.0	
Proteaceae	Helicia				29
Theaceae	Eurya				26
Ruteacea	Zanthoxylum, Melicole	42	32	36	27

Rosaceae	Prunus	25

[β] Here we choose to select trees >70cmDBH rather than 50CmDBH as a means to filter the list of species

Appendix 3.8

Other studies in high altitude forest that report tree height

Table S3.4. Studies in high altitude forest that report tree heights rarely surpass 15m

Country	Altitude range (m asl)	Max tree height (m)	MAT (°C)	MAP (mm)	Source
Ecuador	2,100-2,600	10	13.3	4,743	(Homeier et al. 2010)
Ecuador	2,700-2,900	12	11.0	3,000	(Madsen and Øllgaard 1994)
Bolivia	2,800	15	n/a	n/a	(Girardin et al. 2013)
Peru	3,020	11	12.5	1,705	(Girardin et al. 2010)
Borneo	2,800-3100	11	n/a	3,285	(Aiba and Kitayama 1999, Kitayama and Aiba 2002)
Costa Rica	2,600	22	10.5	5,600	(Lieberman et al. 1996)
Columbia	2,700	15	n/a	n/a	(Cleef et al. 1984)
Costa Rica	3,100−3,400	20	n/a	n/a	(Kappelle et al. 1989)

Appendix 4.1

Table S2. A Sample of 190 local tree names from a total of 505 recorded in this study.

N°	Local Tree name	Family	Genus	Forest type	Language
1	Matip	Achariaceae	Pangium	Montane	Mato
2	Jamba	Actinidiaceae	Saurauia	Montane– Upper-montane	Ngun
3	Tom Tom	Actinidiaceae	Saurauia	Montane– Upper-montane	Yao/Ngnun
4	Eminamminam	Anacardiaceae	Dracontomelon	Upper-montane	Ngun
5	Wonim	Anacardiaceae	Mangifera	Montane	Yao
6	Otigot	Apocynaceae	Alstonia	Montane– Upper-montane	Ngun/yao
7	Singlabana	Apocynaceae	Cerbera	Montane– Upper-montane	Yopno gen
8	biambo	Bambusoideae	Bamboo	Montane	Yao
9	Trema sp	Cannabaceae	Trema	Montane	Yao
10	Mindang	Caricaceae	Carica	Montane	Yao
11	Yaro	Casuarinaceae	Casuarina	Montane	Yao
12	Rilorilo	Combretaceae	Terminalia	Lowland	Mato
13	Bop ep (1)	Cunoniaceae	Caldcluvia	Montane	Yao
14	Bop ep (2)	Cunoniaceae	Davidsonia	Montane	Yao
15	Dinixa	Cunoniaceae	Caldcluvia	Montane	Yao
16	Mupmo ep	Cunoniaceae	Caldcluvia	Montane– Upper-montane	Yopno gen
17	Nia	Cunoniaceae	Caldcluvia	Montane– Upper-montane	Yopno gen
18	Talison	Cunoniaceae	Caldcluvia	Lowland	Mato
19	Wagudi	Cunoniaceae	Caldcluvia	Lowland– Montane	Mato
20	Moun	Cunoniaceae	Davidsonia	Montane	Yao
21	Wapmang	Cunoniaceae	Davidsonia	Montane– Upper-montane	Ngun
22	Igot	Cunoniaceae	Schizomeria	Montane	Yao
23	Saxep	Cunoniaceae	Schizomeria	Montane– Upper-montane	Yopno gen
24	Irararoc	Cupressaceae	Dacrydium	Upper-montane	Ngun
25	Sombe	Cupressaceae	Libocedrus	Montane	Yao/Ngnun
26	Ing ing	Cyatheaceae	Cyathea	Montane	Yao

27	Kandurum	Cyatheaceae	Cyathea	Montane	Yopno gen
28	Kondorum	Cyatheaceae	Cyathea	Montane	Yao/Ngnun
29	Sirimbing	Cyatheaceae	Cyathea	Montane	Yao
30	Sum sum	Cyatheaceae	Cyathea	Montane– Upper-montane	Yao/Ngnun
31	Rixang	Datiscaceae	Octomeles	Lowland	Mato
32	Amigomon	Dicksoniaceae	Dicksonia	Upper-montane	Ngun
33	Sop sop	Dicksoniaceae	Dicksonia	Montane	Yao
34	Abal	Elaeocarpaceae	Elaeocarpus	Montane– Upper-montane	Yopno gen
35	Koyo (1)	Elaeocarpaceae	Elaeocarpus	Montane	Yopno gen
36	Malu malu	Elaeocarpaceae	Sloanea	Lowland	Mato
37	Yat	Elaeocarpaceae	Sloanea	Montane	Yopno gen
38	Buya buya	Escalloniaceae	Quintinia	Lowland	Mato
39	Fia fia/ Huya huya	Escalloniaceae	Quintinia	Montane– Upper-montane	Yao/Ngnun/mato
40	Gabap	Euphorbiaceae	Cleidion	Montane	Yao
41	Gahac	Euphorbiaceae	Cleidion	Montane	Yopno gen
42	Ibaba	Euphorbiaceae	Endosperum	Upper-montane	Ngun
43	Ipa ipa	Euphorbiaceae	Endosperum	Montane	Yao
44	Gau	Euphorbiaceae	Homalanthus	Lowland	Mato
45	Gusong dong	Euphorbiaceae	Homalanthus	Lowland– Montane	Mato
46	Osu Osu	Euphorbiaceae	Homalanthus	Montane	Mato
47	Udan/udong	Euphorbiaceae	Homalanthus	Montane	Yao
48	Bom	Euphorbiaceae	Macaranga	Montane	Yao
49	Gereng	Euphorbiaceae	Macaranga	Montane	Yao/Ngnun
50	Tapioc	Euphorbiaceae	Manihot	Lowland	Mato
51	Sum	Fabaceae	Crotalaria	Montane	Yao
52	Yandro	Fabaceae	Leucaena	Montane	Yao
53	Taik	Fabaceae	Pterocarpus	Lowland	Mato
54	Domung	Fagaceae	Lithocarpus	Lowland– Montane	Yao
55	Gaum	Fagaceae	Lithocarpus	Montane	Yao
56	Koroc Koroc	Fagaceae	Nothofagus	Montane	Yopno gen
57	Bee	Gentianaceae	Fagraea	Montane– Upper-montane	Yopno gen

58	Yonggam	Gnetaceae	Gnetum	Lowland	Mato
59	Gebu gebu	Hernandiaceae	Hernandia	Lowland	Mato
60	Sogun	Himantandraceae	Galbulimima	Montane	Yao
61	Debat	Icacinaceae	Platea	Lowland– Montane	Yao
62	Moyup	Lauraceae	Cryptocarya	Montane	Mato
63	Muk muk	Lauraceae	Cryptocarya	Montane	Yao
64	Bingboc	Lauraceae	Litsea	Upper-montane	Ngun
65	Boboc	Lauraceae	Litsea	Upper-montane	Ngun
66	Ogoc	Lauraceae	Litsea	Lowland	Mato
67	Bata	Lauraceae	Persea	Montane	Yao
68	Bahohoc ongam	Lauraceae	n/a	Montane	Yao
69	Ferong Siup	Lauraceae	n/a	Montane	Mato
70	SeBon	Malvaceae	Sterculia	Montane	Mato
71	Bagop	Malvaceae	Trichospermum	Montane	Yao
72	Mangoc	Melastomataceae	Melastoma	Lowland	Mato
73	Sabon	Meliaceae	Aglaia	Montane– Upper-montane	Yopno gen
74	Sonurung	Meliaceae	Aglaia	Upper-montane	Ngun
75	Sopon	Meliaceae	Aglaia	Upper-montane	Ngun
76	Sunbun	Meliaceae	Algaia	Montane	Yao
77	Buxu buxu	Meliaceae	Chisocheton	Montane	Mato
78	Buxu Sambexa	Meliaceae	Chisocheton	Lowland	Mato
79	Dimumuc	Meliaceae	Synoum	Montane	Yao
80	Sori	Monimiaceae	Dryadodaphne	Montane	Yao
81	Sorin	Monimiaceae	Dryadodaphne	Upper-montane	Ngun
82	Manambung	Monimiaceae	Palmeria	Lowland	Mato
83	Somamon	Monimiaceae	n/a	Montane	Yao
84	Bon	Moraceae	Artocarpus	Montane	Yao
85	Uping taptap	Moraceae	Artocarpus	Lowland– Upper-montane	Yao/Ngnun
86	Abak	Moraceae	Ficus	Lowland	Mato
87	Bontup	Moraceae	ficus	Montane	Yao
88	Borup	Moraceae	Ficus	Montane	Yao

89	Damuč	Moraceae	Ficus	Lowland– Montane	Mato
90	Don	Moraceae	Ficus	Montane– Upper-montane	Yao/Ngnun
91	Goxombian	Moraceae	Ficus	Montane	Yao/Ngnun
92	Kawat	Moraceae	Ficus	Montane	Yopno gen
93	Kodamat	Moraceae	Ficus	Montane– Upper-montane	Ngun
94	Kotaoac	Moraceae	Ficus	Lowland– Montane	Mato
95	Langlang	Moraceae	Ficus	Montane– Upper-montane	Yopno gen
96	Nawan	Moraceae	Ficus	Montane– Upper-montane	Yopno gen
97	Ohum	Moraceae	Ficus	Upper-montane	Ngun
98	Oxim	Moraceae	Ficus	Lowland	Mato
99	Siwit	Moraceae	Ficus	Montane	Yao
100	Sobon	Moraceae	Ficus	Lowland	Mato
101	Sokung	Moraceae	Ficus	Montane	Yao
102	Suamsuam	Moraceae	Ficus	Upper-montane	Ngun
103	Tang tang	Moraceae	Ficus	Montane	Yao
104	Uxu	Moraceae	Ficus	Montane	Mato
105	Waden	Moraceae	Ficus	Montane	Yao
106	Yang yang	Moraceae	Ficus	Montane	Yao
107	Figet	Musaceae	Musa	Upper-montane	Ngun
108	Ongam	Musaceae	Musa	Montane	Yao
109	Nayac	Myristicaceae	Horsfeldia	Lowland	Mato
110	Stop	Myrsinaceae	Ardisia	Lowland– Montane	Yao
111	Stot	Myrsinaceae	Ardisia	Lowland– Upper-montane	Yao/Ngnun
112	Nasi nasi	Myrtaceae	Decaspermum	Montane– Upper-montane	Yopno gen
113	Songamon	Myrtaceae	Decaspermum	Montane	Yao
114	Sunggamang	Myrtaceae	Decaspermum	Lowland– Montane	Mato
115	Gip	Myrtaceae	Eugenia	Lowland	Mato
116	Nim	Myrtaceae	Eugenia	Montane	Yao
117	Viroc	Myrtaceae	Eugenia	Montane	Ngun
118	Kokekokeč	Myrtaceae	Myrcianthes	Montane– Upper-montane	Yopno gen
119	Aimela	Myrtaceae	Syzygium	Lowland– Montane	Mato

120	Manong	Myrtaceae	Syzygium	Lowland– Montane	Mato
121	Map	Myrtaceae	Syzygium	Lowland– Upper-montane	Yao
122	Wuniong	Myrtaceae	Syzygium	Montane– Upper-montane	Yopno gen
123	Giwa	Myrtaceae	n/a	Upper-montane	Ngun
124	Farot	Ochnaceae	Brackenridgea	Montane	Yao
125	Fandot	Ochnaceae	Schuurmansia	Montane	Mato
126	Handot	Ochnaceae	Schuurmansia	Lowland	Mato
127	Ipmoroc	Ochnaceae	Schuurmansia	Montane	Ngun
128	Bot Nongon	Oleaceae	Olea	Montane	Yao
129	Marita	Pandanaceae	Pandanus	Montane– Upper-montane	Yopno gen
130	Tet	Phyllanthaceae	Glochidion	Montane	Yao
131	Kowoc	Piperaceae	Piper	Montane	Yopno gen
132	Wakonbona	Pittosporaceae	Pittosporum	Montane	Yao
133	Wakumbong	Pittosporaceae	Pittosporum	Lowland– Montane	Yao
134	Mogum	Poaceae	Bamboo	Montane	Yopno gen
135	Kadimut	Podocarpaceae	Dacrydium	Montane– Upper-montane	Yopno gen
136	Quedumak	Podocarpaceae	Dacrydium	Montane– Upper-montane	Yopno gen
137	Yorum	Podocarpaceae	Dacrydium	Montane	Yao
138	Geng	Podocarpaceae	Podocarpus	Lowland	Mato
139	Tatong	Podocarpaceae	Podocarpus	Montane	Yao/Ngnun
140	Tatong Kombut	Podocarpaceae	Prumnopitys	Montane	Yao
141	Dumang Simup	Proteaceae	Helicia	Lowland– Montane	Yao
142	Guaman	Proteaceae	Helicia	Lowland	Mato
143	Kunoring	Proteaceae	Helicia	Upper-montane	Ngun
144	Boram/Borum	Rosaceae	Prunus	Montane	Yao
145	Buram	Rosaceae	Prunus	Lowland– Montane	Mato
146	Yorip	Rosaceae	Prunus	Upper-montane	Ngun
147	Tom	Rousseaceae	n/a	Montane	Yao
148	Batot/Batup	Rubiaceae	Neonauclea	Montane	Yao
149	Kokekokeč	Rubiaceae	Phalaria	Montane	Yao
150	Tongo tongo	Rubiaceae	Phalaria	Upper-montane	Ngun

151	Katang	Rubiaceae	Psychotria	Montane– Upper-montane	Yopno gen
152	Kofing	Rubiaceae	Psychotria	Montane	Yopno gen
153	Dandu dandu	Rubiaceae	Timonius	Montane	Yao
154	Noya	Rubiaceae	n/a	Montane	Yao
155	Fotom	Rubiaceae	Wendlandia	Montane	Yao
156	Fundong	Rubiaceae	Wendlandia	Montane	Yopno gen
157	Egec	Rutaceae	Acronychia	Lowland– Montane	Yao
158	Fiup	Rutaceae	Acronychia	Montane	Yao
159	Kaweng	Rutaceae	Acronychia	Montane	Yopno gen
160	Yararip	Rutaceae	Acronychia	Upper-montane	Ngun
161	Gobak	Rutaceae	Melicope	Upper-montane	Ngun
162	Gogondi	Rutaceae	Melicope	Montane	Yao
163	Kupam fium	Rutaceae	Melicope	Montane	Yao
164	Dinom	Rutaceae	Zanthoxylum	Lowland– Montane	Mato
165	Bop ep (3)	Sapindaceae	Ceratopetalum	Montane	Yao
166	Kohi	Sapindaceae	Dodonaea	Montane	Yao
167	Kayol	Sapindaceae	Dodonaea	Montane	Yao
168	Kokbawok	Sapindaceae	Dodonaea	Upper-montane	Ngun
169	Koyo (2)	Sapindaceae	Dodonaea	Upper-montane	Ngun
170	Daxanang	Sapindaceae	Pometia	Montane	Mato
171	Daxum	Sapindaceae	Pometia	Lowland	Mato
172	Rube	Sapindaceae	Pometia	Lowland	Mato
173	Kanokim matno	Sapindaceae	n/a	Upper-montane	Ngun
174	Biric/Bisič	Sapotaceae	Pouteria	Montane	Yao
175	Yorowang	Sapotaceae	Pouteria	Montane	Yao
176	Bilic	Paracryphiaceae	Manilkara	Montane	Yao
177	Gofing/Going	Sphenostemonaceae	sphenostemon	Lowland– Upper-montane	Mato/Yao/Ngnun
178	Imuč	Sphenostemonaceae	sphenostemon	Montane	Yao
179	Bonom	Staphyleaceae	Turpinia	Montane	Yao
180	Suloriu	Tetramelaceae	Tetrameles	Lowland– Montane	Mato
181	Uria	Theaceae	Eurya	Upper-montane	Ngun

182	Bamak	Urticaceae	Dendrocnide	Lowland– Upper-montane	Yao/Yopno gen
183	Bomot	Urticaceae	Dendrocnide	Montane	Yao
184	Bumak	Urticaceae	Dendrocnide	Lowland	Mato
185	Dendrocnide	Urticaceae	Dendrocnide	Montane– Upper-montane	Yopno gen
186	Salat	Urticaceae	Dendrocnide	Montane– Upper-montane	Yopno gen
187	Guram	Urticaceae	Elatostema	Montane	Yopno gen
188	Gopi dudu	Verbenaceae	Vitex	Upper-montane	Ngun
189	Gerewon	Winteraceae	Bubbia	Montane	Yao/Ngnun
190	Sumbiri	Winteraceae	Tasmannia	Upper-montane	Ngun

Appendix 4.2

Expected errors from experts

Table S4.2. The distribution of expected errors from experts using Chave et al. (2005) and Condit (1998) compared to those observed in non-experts and experts in our study for both the distributions of smaller errors (95% of the smaller error range) and the expected distribution of larger errors (5% of the larger error range).

Results within one standard deviation

	# samples expected	**# samples observed**	**expected error (prop)**	**observed error (prop)**
Typical small error	886	640	0.68	0.49
Large-uncommon error	47	23	0.68	0.33

The rates of error were determined by fitting the discrepancies with a sum of two normal distributions (SD1 and SD2). The first described small common errors with a Standard Deviation (SD) proportional to the trunk diameter; the second has a fixed larger S.D., with 5% of the trees subject to the larger error. We applied the two standard deviations to build a distribution of expected errors using the expert sample as the mean, and using standard deviation that describe the small error and large errors where [1]SD_{Small} = 0.0062*Diameter + 0.0904 (95% probability); [2]SD_{large} = 4.64cm (5% probability).

Appendix 4.3

Relationship between expert and non-expert measurements

Table S4.3 **Two models (GLS and SMA) that describe the relationship between experts' and non-experts' DBH and height measurements.**

Measurement	Regression model	Slope of best fit Ð	Coefficient of correlation	S.E. for slope and intercept	Test against slope = 1
DBH	GLS	(Eq. 4.3) Y = 0.93x + 0.53	$R^2 = 0.91$	± 0.008; ± 0.15	$p < 0.001$
	SMA	(Eq. 4.4) Y= 0.99x + 0.01	$R^2 = 0.99$	± 0.003; ± 0.001	$p < 0.001$
Height	GLS	(Eq. 4.5) Y = 0.97x + 1.02	$R^2 = 0.90$	± 0.009; ± 0.17	$p < 0.001$
	SMA	(Eq. 4.6) Y = 1.06x - 0.035	$R^2 = 0.92$	± 0.003; ± 0.008	$p < 0.001$

(Ð) X = expert values. Y = non expert values. GLS and SMA used the same data but estimated different slopes and slope intercepts because GLS slope is fit to minimise the vertical residuals (all error in the y axis) whereas in the SMS the slope is fit to minimise the perpendicular distances from each point to the line, accounting for error in both y and x dimensions.

Appendix 5.1

Allometric equation used for estimating dry above ground biomass in managed landscapes (AGB)

Stem Type	Eq. #	Allometric equation§	Source
Tree (wet tropical forest)	Eq.1	$AGB = 0.0776 \times (\rho\, D^2 H)^{0.940}$	Chave et al. (2005b)
Tree (moist tropical forest)	Eq. 2	$AGB = 0.0509 \times \rho\, D^2$	Chave et al. (2005b)
Palm, tree ferns, Pandanus	Eq.3	$AGB = 6.67 + 12.826\, (H)^{0.05} ln(H)$	Pearson et al. (2005)
Liana	Eq.4	$AGB = exp^{(0.12+0.91[logBA])}$	Pearson et al (2005)
Dead standing Softwood	Eq.5	$AGB = [Eq. 1] \times 0.94$	Pearson et al. (2005)
Dead standing hardwood	Eq.6	$AGB = [Eq. 1] \times 0.97$	Harmon et al. (2011).
Dead standing bole	Eq.7	$AGB = [0.33\pi \times 0.00008(D^2)\, H\, \rho]100$	Harmon et al. (2011).
Banana	Eq.8	$AGB = 0.03(D^{2.13})$	Van Noordwijk et al. (2002)
Coffee	Eq.8	$AGB = 0.281(D^{2.06})$	Van Noordwijk et al. (2002)
Bamboo	Eq.9	$AGB = 0.131(D^{2.28})$	Van Noordwijk et al. (2002)

where AGB above ground dry biomass is in Kg, 'D' is DBH in centimeter, 'H' is height in meters, and 'ρ' is wood specific gravity (WSG) in grams per cubic centimeters, BA is basal area in Kg·m^{-2} measured at DBH

Appendix 5.2

Assumptions used to produce the land-use planning models

Action	**Decision rules**
Forest protection	The amount of primary forest required for local use was determined by the amount of area communities were willing to set aside for conservation (Brooks 2011).
Shade coffee plantation criteria	For the establishment of shade coffee plantation, our biophysical model selected areas limited to grasslands in the range of elevations that produce commercially viable coffee crops and can support the native shade tree *Casuarina oligodon* (1,100-2,100 m asl, < 45° slope) that were in proximity to villages (< 4km) (Bourke 1985). The suitable areas were managed to convert the land cover from anthropogenic grassland to shade coffee plantations, as 30 years is a suitable time from mature establishment (Vergara and Nair 1985). When adding the socio-economic variables the model, the suitable area was further constrained to a maximum of 5ha per household to match labour constraints identified in the survey.
Accelerated natural regeneration criteria	For accelerated natural regeneration by fire control, our biophysical model selected areas limited to alpine grasslands below the tree line (< 3,800 m asl,) but within the Upper-montane forest zone (> 2,800 m asl), as these are considered to be maintained by human fires (Wade and McVean 1969). The suitable areas were converted from anthropogenic grasslands to young secondary forests. Adding socio-economic constraints to the model increased the XX by including grassland between 50 m asl and 2,800 m asl within 4km from any village; where 50% of the suitable grasslands were converted to young secondary forest and 50% were retained as grassland for local use. We included this action in response to the large proportion of participant (62%) in favour of implementing fire

	control near villages to increase the availability of fuelwood.
Enrichment planting criteria	Enrichment planting was only identified through the analysis of PEN surveys and thus was not part of the land-use planning model based on biophysical information only. Enrichment plantings (Chapter 2.5.3) were desired by 96% of the respondents as a means to increase construction material availability. Our biophysical model selected areas limited to swidden fallows and degraded forests < 10km from a village and with terrain < 25° slope. Areas suitable for enrichment plantings were not converted to another land-use, instead carbon stocks in those areas were increased by 20% (Keefe et al. 2009).