CARBON REVENUES OF REDUCING EMISSIONS FROM DEFORESTATION AND DEGRADATION IN DIFFERENT FOREST ECOSYSTEMS, POPA MOUNTAIN PARK, MYANMAR

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ABSTRACT

Carbon emission reductions through the reduction in deforestation and forest degradation and through forest conservation, sustainable management of forests and enhancement of forest carbon stocks (REDD+) have been a critical mechanism of climate change mitigation since the Bali conference in 2007. Thus, this study attempts to analyze the potential carbon emission reductions under the REDD+ scheme in four natural forest types located in Popa Mountain Park in Myanmar (PMP); namely dry mixed deciduous forest (DMDF), dry dipterocarp forest (DDF), dry forest (DF) and dry hill/evergreen forest (DHEF). Based on data from 4-ha sample plots, average stem density ranges from 1293 trees ha^{-1} in DDF to 804 tree ha^{-1} in DHEF. According to the Jackknife estimator for species richness (trees with DBH $\geq$ 5 cm), the highest number of species was recorded in DMDF, 74 species ha^{-1}, and the lowest number of species was recorded in DF, 40 species ha^{-1}. DMDF occupied the highest value on the Shannon-Wiener index and Simpson diversity index while the lowest was in DF, indicating that DMDF is the most complex whereas DF is the simplest community. Results of the carbon estimation showed that dry evergreen forest had the highest aboveground carbon density, 200.22 $tCha^{-1}$, and the lowest in DDF, 91.28 $tCha^{-1}$. The total carbon (AGB, BGB, litter, deadwood and soil) stored by DHEF is the highest followed by DMDF, DDF and DF. In all forests in PMP, living trees (aboveground and belowground) accumulated 30 percent of total carbon. About 1% of total carbon is

stored in litter and about 10% in deadwood. Soil organic carbon stored 60% of total carbon.

Based on historic deforestation rates, this study found that about 256531.6 tCO_2 – 619460.5 tCO_2 would be lost between 2013 and 2043 without any conservation measures. By managing natural forest (7969 ha) in Popa Mountain Park for a 30-year period under the REDD+ project, carbon emissions could be reduced which account for 6390.6 tCO_2 – 14866.5 tCO_2 annually. If financial support is available to implement the REDD+ project, carbon credits from reducing deforestation are estimated at 4373.8 tCO_2 - 10406.6 tCO_2 or US$ 21881.2 - US$ 52032.9 annually over the 30 year period of the carbon project if carbon is priced at US$5 tCO_2. Therefore, reducing deforestation could result in huge emission reductions and substantial carbon-based revenues, while improving the livelihoods of forest-dependent communities. Actual emission reductions depend on the assumptions of the future implementation of the project actions considered in this study in order to reduce the drivers of deforestation. These assumptions will have to be revised based on the future experience in the project. Carbon revenues in this study will be affected by the future carbon price and therefore future adjustment of the revenue will be necessary. Nonetheless, carbon revenues from reducing deforestation could bring funds for forest conservation and livelihood improvement. As forest cover change, carbon stocks, and drivers of deforestation will affect the estimation of emission reductions. Further studies on these variables by district are important for reducing study biases or uncertainties.

This study mainly focused on the issue of avoided deforestation (RED). REDD+, however, also comprises other factors including avoided degradation, afforestation, forest management and other forestry activities. However, this study focused on the issue of avoided deforestation only because the study area is affected by human disturbance which is the main cause of deforestation and degradation, and is a cause to a limited extent only of the other activities under REDD+. As well, to include monitoring degradation as a separate indicator in the analysis, more detailed information is needed; for example, scale of degradation, availability of high resolution remote sensing data, availability of field data and other indicators (e.g., road networks) that indirectly refer to degraded areas. These aspects are not covered by this study, because the purpose of our study was to estimate carbon emission reductions attributable to forest cover loss and not to investigate the scale of degradation. This, however, would be an important subject for further research.

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CHAPTER I

INTRODUCTION

Background

The forest sector plays a key role in tackling climate change (Eliasch, 2008). Forests can be a sink or a source of atmospheric carbon depending on the management regimes designed to achieve one or more objectives (Sasaki, 2006). Forests play multiple roles in climate change mitigation because of their ability to absorb and store large quantities of atmospheric carbon through the process of photosynthesis. On the contrary when deforested or degraded, carbon emissions occur (Sathaya and Nilels, 2008). The loss of tropical forest is the major driver of carbon dioxide (CO_2) flux caused by land use changes during the past two decades (Lasco, et al., 2013). About 52% of the world's forests are concentrated in the tropics, where high rates of deforestation and land conversion have occurred (Brown, et al., 1996). The Food and Agriculture Organization of the United Nations (Food and Agriculture Organization of the United Nations, 2010) estimates that 13 million ha of tropical forest are lost annually. Annual carbon emissions due to deforestation in the tropics were estimated ranging from 1.1 PgC (Achard, et al., 2004) to 1.5 PgC (Gullison and Frumhoff, 2007), and up to 2.2 ± 0.6 PgC (Houghton, 2003) during 1990s (1 PgC = 1015 gC). These emissions account for about 13.7% to 27.5% of the 8.0 PgC of global emissions. Hence, tropical forest are critical role in global carbon cycle and climate change system. Tropical deforestation is a source of greenhouse gas (GHG) emissions, accounting for up to one-third of global emissions (Houghton, 2005). As well, significantly, deforestation and degradation in the tropics have caused the loss of biodiversity and ecosystem services (Costanza, et al., 1997; Foley, et al., 2007). These huge emissions and biodiversity loss prompted the Intergovernmental Panel on Climate Change (IPCC) to recognize the urgent need for preventing carbon emissions from tropical forests as the largest and most immediate carbon stock impact in the short term (Intergovernmental Panel on Climate Change, 2007). Recognizing the importance of tropical forests and the value of developing country participation in global climate change mitigation efforts, reducing deforestation in the tropics has again become a central theme of the United Nations Framework Convention on Climate Change (UNFCCC).

The Thirteen Conference of the Parties (COP13) of UNFCCC adopted the Bali Action Plan in 2007 (Decision 2) recognizing the increasingly important role of tropical forests in greenhouse gas emissions reductions through the reduced emissions from deforestation and forest degradation, conservation of forests, sustainable management of forests and enhancement of forest carbon stocks (REDD+) in developing countries. In Cancum (COP 16) agreements, the REDD mechanism as a means to reduce carbon emissions is based on the observation of and rewarding of actors for keeping or restoring forests (Karsenty, et al., 2012). This mechanism has potential to include developing countries more actively in international greenhouse gas mitigation and to address reducing emission which come from deforestation and degradation (Olander, et al., 2006). To measure carbon emission from deforestation and degradation, one possibility is to model a business-as-usual scenario for the emissions from deforestation and degradation based on the trend in emissions of a historical reference period, and to compare the business-as-usual scenario with the actual monitored emissions from deforestation and degradation (Förster, 2009). Identification of the drivers and management interventions for reducing deforestation and forest degradation and establishment of reference emission level (REL) are the fundamental activities for developing REDD+ projects in developing countries (United Nations Framework Convention on Climate Change, 2009). REL is the baseline emissions level in the absence of project activities against which, reduced emissions are compared. Thus, baseline is crucial to measure the emission reduction performance and it could subsequently lead to meaningful negotiations on emission reduction targets (Huettner, et al., 2009). A baseline for reducing emissions from deforestation could be based on historical emissions or could use historical emissions as input for business-as-usual (BAU) projections (Olander, et al., 2008).

In order to establish baselines, one must know how much carbon stock in the area. Especially, the developing countries should have well-authenticated estimates of forest carbon stocks for the successful implementation of mitigating policies to take advantage of the REDD programme of United Nations Framework Convention in Climate Change (UNFCCC) (Chaturvedi, et al., 2011; Miah, et al., 2011). REDD mechanism is based on the observation that developing countries have an opportunity cost if they decide to preserve their forests rather than convert them to other land uses (Karsenty, et al., 2012) and aim to make forests more valuable standing than they would be cut down, by creating a financial value for the carbon stored in trees (Hoang, et al., 2013). Estimation of accurate biomass and carbon of different forest components is important to estimate their contribution to total carbon stock (Chaturvedi and Raghubanshi, 2012). According to the intergovernmental Panel on Climate Change (IPCC), forest carbon should be estimate in five pools (aboveground, belowground, litter, deadwood and soil) (Intergovernmental Panel on Climate Change, 2006). REDD+ scheme focus not only on reducing carbon emissions from deforestation and degradation but also on safeguarding biodiversity and socioeconomics of forest-dependent communities. So, monitoring of tree diversity and forest structure is pre-requisites for understanding and managing forest. Knowledge on their structure and dynamics are necessary in order to understand how forest ecosystems organize and function (Nebel, et al., 2001). Beside, understanding the condition of species composition and diversity before any plan take a principle in stand is necessary (Farhadi, et al., 2013).

REDD scheme seeks to provide financial compensation to actors for their efforts to reducing deforestation and forest degradation or restoring forests as a means to reducing carbon emissions while improving local livelihood (Karsenty, et al., 2012). Meanwhile, developing countries have increasingly shown their interests in demonstrating and hosting REDD+ projects (Mattsson, et al., 2012). Nevertheless, developing countries (Non-Annex 1 countries of the Kyoto Protocol) are still lack of information on emission baseline or REL to enable them to benefit from the REDD+ scheme (Kerr, et al., 1999). To know how much deforestation emissions are truly prevent under REDD mechanism, one needs to know how much deforestation would be occurred in the business-as-usual without any climate change mitigation measure (Huettner, 2007). For that reason, establishment of baselines are crucial to measure the emission reduction performance and consequently to negotiate meaningful deforestation emission reduction target baseline (Huettner, et al., 2009). Thus, monitoring of forests is crucial in order to identify the historical and present changes in the extent of forest cover, its quality and carbon content, and to quantify related carbon dioxide emissions (Förster, 2009).

Statement of the problem

Myanmar has one of the highest proportions of forest cover in mainland South East Asia. Approximately 47% of Myanmar's total land area is forested (FD, 2010). The forests of Myanmar range from mangrove to alpine (Leimgruber, et al., 2005), and Myanmar's biodiversity is amongst the most diverse in the Indo-Pacific region (Myers, et al., 2000). However, the biodiversity in Myanmar has been under severe pressure due to population growth accompanied by increased resource use, as well as the ever-increasing demand for resources from neighboring countries (Aung, et al., 2004). According to the FAO, Myanmar is one of the ten countries with the largest annual net loss of forest area. Deforestation was 310,000 ha or 0.93% annually between 2000-2010 (Food and Agriculture Organization of the United Nation, 2010). Deforestation varies considerably between regions. The central and/or more populated States and Regions show the highest losses of forest resources, especially the Ayeyarwady Delta and Central Dry Zone (Leimgruberi, et al., 2005). Without reducing the current rate of deforestation and forest degradation, Myanmar is likely to face climate-driven food and water shortages and the continued loss of its valuable forests and biodiversity.

The government of Myanmar has been concerned about the extent of the forest degradation and deforestation in the country. Myanmar ratified the United Nations framework convention on climate change (UNFCCC) on November 1994 and ratified the Kyoto protocol as one of the Non-Annex I parties in 2003. The new policy, REDD+, is currently under discussion by Parties to the UNFCCC regarding crediting or otherwise rewarding reductions in carbon emission by reducing rates of deforestation and forest degradation. Under REDD, non-Annex 1 countries would, on a voluntary basis, reduce the rate at which their forests are being lost, and receive compensation in proportion to the carbon emissions saved as compared to a baseline which would represent the 'without intervention' case or some other agreed target (Moutinho and Schwartzman, 2005).

Although previous studies clarified the fundamental basis for understanding carbon storage in the plantations and natural forests in Myanmar, many of these studies failed to address the carbon emission from deforestation and potential reductions in carbon emissions by managing forests under the REDD+ mechanism. The REDD+ scheme of the United Nations Framework Convention on Climate Change is a scheme for carbon-based compensation for projects that have resulted in reducing carbon emissions or enhancing carbon sinks in tropical forests. However, estimating such emissions and sinks remains challenging, and without accurate estimates it is impossible to estimate carbon revenues from managing tropical forests. To better inform policy makers as well as negotiators of the REDD+ scheme, there is critical need for estimating baseline deforestation and emission values. Until recently, in Myanmar, there has been no research on estimation of baseline emissions and project emissions especially for REDD+ projects leading to reducing deforestation.

Estimating potential carbon emission reductions from managing natural forests is necessary when proposing appropriate intervention for managing tropical forests under the anticipated REDD+ scheme. Firstly, forest inventory and diversity assessments are essential on order to understand the structure and status of forests (Appiah, 2013) which will provide information for forest conservation and sustainable forest management. Knowledge of forest structures, composition and diversity of different species and different forest types would facilitate the creation and implementation of more effective conservation measures. Thus, the characterization of forest stand conditions becomes a fundamental tool in biodiversity conservation and sustainable forest management. Not only the investigation of forest structure and forest condition, but also the assessment of carbon in the forest areas is an important criterion of sustainable forest management (Brandeis, et al., 2006) and at the same time it is required for greenhouse gas inventories needed in the LULUCF sector (Land use, land use change and forestry) for the United Nations framework convention on climate change (UNFCCC) reporting (Fonseca, et al., 2011). Likewise, estimation on carbon emission from deforestation is required in order to provide the baseline information for future REDD+ readiness preparation. In Myanmar, it appears that only a handful of studies had been carried out and published to estimate carbon stocks in the natural forest (Myo, 2008; Oo, 2009). In addition, no studies have been done to estimate carbon stocks, emissions or emission reductions at the project level in Myanmar despite high deforestation rates in recent years. Without such estimation it is difficult for Myanmar to obtain incentives from the REDD+ scheme. It is timely and important that Myanmar develops methods to assess not only the current carbon stocks but also to establish emission baselines for various forest types at the project, subnational or national levels in Myanmar.

Therefore, the purpose of this study was to investigate four natural forest types to contribute to the REDD+ discussion by constructing baseline carbon emission estimates from deforestation, by estimating carbon storage, species diversity and stand structures. We acknowledge that forest degradation also needs to be addressed, but, nonetheless, we have omitted estimation of carbon emission from forest degradation for the reason that there is no standardized definition for forest degradation (Global Observation of forest and land cover Dynamics, 2009; Sasaki and Putz, 2008). Until recently, several organizations, such as the Food and Agricultural Organization of the United Nations (FAO), the International Tropical Timber Organization, the United Nations Environmental Program, and the Intergovernmental Panel on Climate Change have used different definitions for forest degradation (Schoene, et al., 2007).

Objectives

The main objective of this study was to measure baseline carbon stocks and set reference scenario of carbon emissions of the study area for supporting REDD readiness in Myanmar. The specific objectives are;

1. To investigate the species composition, stand structure and carbon storage capacity of the different forest types in order to support the country's sustainable forest management and

2. To construct a simple modelling tool capable of predicting and providing reliable information on emissions from deforestation.

Conceptual Framework

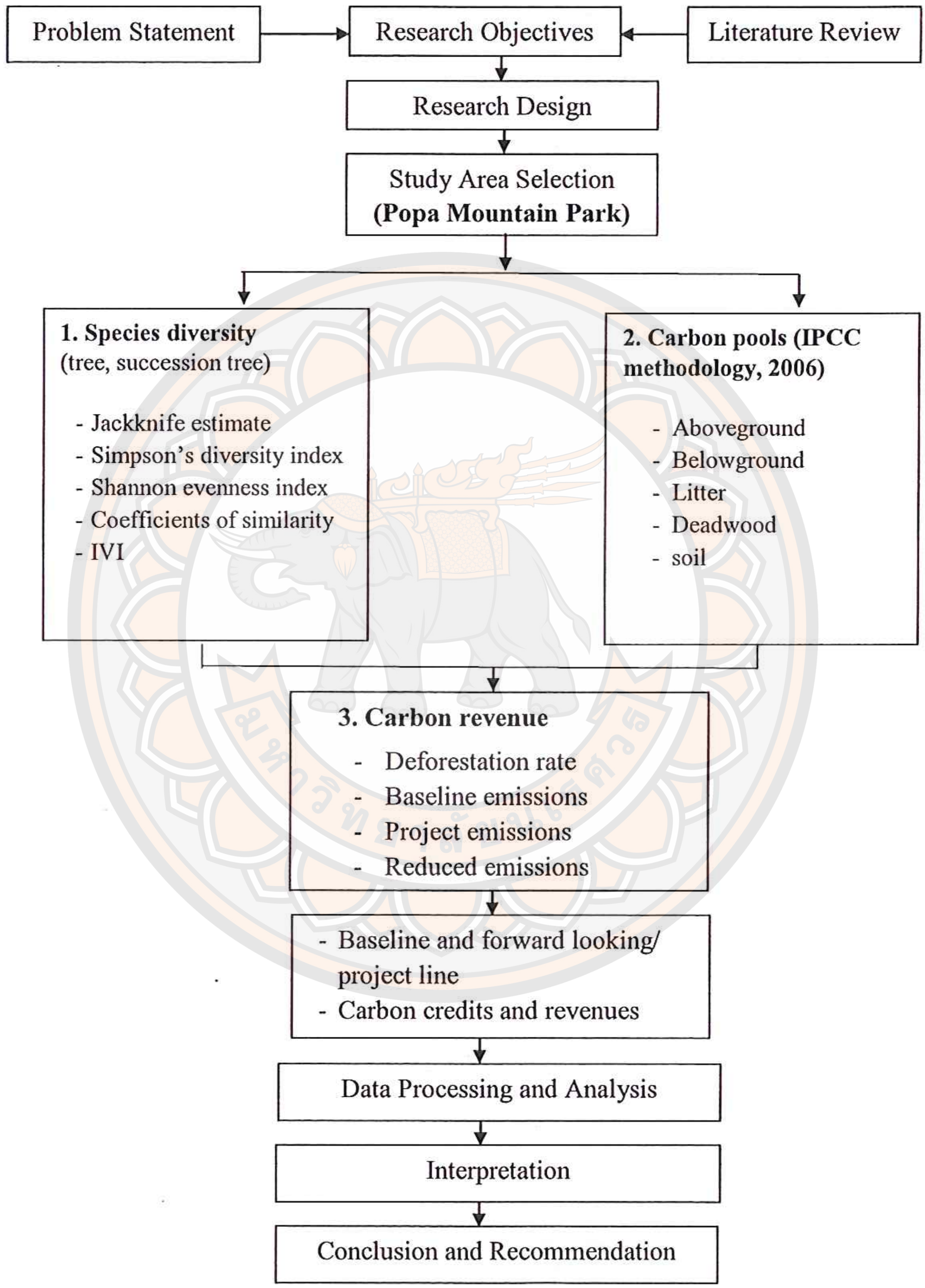

Figure 1 Conceptual Framework

Terms and Definitions

1. **Aboveground biomass:** All living biomass above the soil including stem, stump, branches, bark, seeds and foliage (Forest Resource Assessment, 2005, 2015; Intergovernmental Panel On Climate Change, 2006).

2. **Annex I Parties:** The industrialized countries listed in Annex I to the UNFCCC that were committed to return their greenhouse-gas emissions to 1990 levels by the year 2000 as per Article 4.2 (a) and (b). Annex I Parties have also accepted emissions targets for the period 2008–12 as per Article 3 and Annex B of the Kyoto Protocol.

3. **Below-ground biomass:** All living biomass of live roots. Fine roots of less than (suggested) 2mm diameter are sometimes excluded because these often cannot be distinguished empirically from soil organic matter or litter (Forest Resource Assessment, 2005, 2015; Intergovernmental Panel On Climate Change, 2006).

4. **Biomass expansion factor (BEF)**: A multiplication factor that expands the dry-weight of growing stock biomass, increment biomass, and biomass of wood- or fuelwood removals to account for non-merchantable or non-commercial biomass components, such as stump, branches, twigs, foliage, and, sometimes, non-commercial trees. Biomass expansion factors usually differ for growing stock (BEFS), net annual increment (BEFI) and wood- and fuelwood removals (BEFR). As used in these guidelines, biomass expansion factors account for above- ground components only (Intergovernmental Panel On Climate Change, 2006).

5. **Biomass**: Living plant and animal material both above-ground and below-ground usually expressed as dry weight (Intergovermental Panel on Climate Change, 2006).

6. **Business as Usual (BAU) baseline:** A BAU baseline represents a projection of what would happen without an intervention, and in this instance serves as a benchmark to measure the impact of REDD actions (Meridian Institute, 2009).

7. **Carbon credit:** In REDD+, a ton of CO_2 kept in a tree (not released into the atmosphere) is called a carbon credit. Though forestry, carbon credits are used to convert the carbon stored through forest conservation & management activities into money. If a tree is cut down and burned, one ton of carbon turns into a little more than 3 tons of CO_2 (1 ton carbon = 3.67 tons of CO_2) (Stone and Chacón León, 2010).

8. Carbon dioxide (CO_2): the result of joining carbon (C) with oxygen (O). It takes 1 part of carbon joining with 2 parts of oxygen to form the gas CO_2.

9. Carbon market: A market that creates and transfers emission units or rights.

10. Carbon pool: A reservoir that has the capacity to accumulate or release carbon. The Marrakech Accords provide that all changes in the following carbon pools shall be accounted for: aboveground biomass, below-ground biomass, litter, dead wood, and soil organic carbon; it also provides that a given pool may be ignored if transparent and verifiable information is provided that the pool is not a source (Meridian Institute, 2009).

11. Carbon revenues: The payments for carbon sequestration and is based on stand volume growth. These carbon revenues are assessed every 5 years based on the change in stem volume (Pohjola and Valsta, 2007).

12. Carbon sequestration: The removal of carbon from the atmosphere and long-term storage in sinks, such as marine or terrestrial ecosystems (Meridian Institute, 2009).

13. Carbon stock: The mass of carbon contained in a carbon pool (Intergovernmental Panel On Climate Change, 2006).The quantity of carbon in a "pool", meaning a reservoir or system which has the capacity to accumulate or release carbon (FRA, 2005)

14. Carbon: one of the most common elements in the universe, found in all living and non-living things.

15. Conference of the Parties (COP): The body under the UNFCCC expected to act as supreme authority over the REDD+ mechanism. Pending structural decisions made in Copenhagen, however, this role could be held by the Assembly of Parties to another Treaty as agreed by UNFCCC Parties (Meridian Institute, 2009).

16. Core idea of REDD: To create a multilevel (global-national-local) system of payment for environmental service (PES) that will reduce emission and increase forest carbon stocks (Angelsen, 2008).

17. Dead wood: All non-living woody biomass not contained in the litter, either standing, lying on the ground, or in the soil. Dead wood includes wood lying on the surface, dead roots, and stumps larger than or equal to 10 cm in diameter or any other diameter used by the country. (Forest Resource Assessment, 2015; Intergovernmental Panel On Climate Change, 2006).

18. Deforestation: the loss of forest and loss of carbon storage (Stone and Chacón León, 2010) ; direct human-induced conversion of forest land to non-forested (Meridian Institute, 2009). The conversion of forest to other land use or the permanent reduction of the tree canopy cover below the minimum 10 percent threshold (Forest Resource Assessment, 2015).

19. Forest Degradation: reduces the number of trees and the stock of carbon in a specific forest area (Stone and Chacón León, 2010); direct, human induced, long term loss or at least certain years of forest carbon stock since time T (Intergovernmental Panel on Climate Change, 2003). The reduction of the capacity of a forest to provide goods and services (Forest Resource Assessment, 2015).

20. Forest: an area of more than 0.5–1.0 ha with a minimum "tree" crown cover of 10–30%, with "tree" defined as a plant with the capability of growing to be more than 2–5 m tall (United Nations Framework Convention on Climate Change, 2002). FAO uses a minimum threshold of 40% tree crown cover to define "closed forest" and 10-40% for "open forest" (Food and Agricultural Organization of the United Nation, 2000).

21. Forward looking or projected baseline: A forward looking or projected baseline base on either a direct extrapolation of an historical reference level of actual emissions or on a modification of an historical reference level based on the implication of identified policy settings or any other political choice or construct that might be agreed by a Conference of the Parties (COP) (Meridian Institute, 2009).

22. Gross emissions: A method for estimating emissions from gross deforestation that does not include replacement vegetation (Meridian Institute, 2009).

23. Historical baseline: An historical baseline might be simply derived from an historical reference level by choosing a specific year or period of year or by discounting or inflating such amounts by an agreed factor (Meridian Institute, 2009).

24. Intergovernmental Panel On Climate Change 1996 GL: A methodological report published in 1996 by the Intergovernmental Panel on Climate Change (IPCC) that provides guidelines for national greenhouse gas inventories. In accordance with

Marrakech Accords, these methodologies shall be the basis for national GHG inventories prepared for the purpose of the Kyoto Protocol.

25. Leakage: GHG emissions displacement that occurs when interventions to reduce emissions in one geographical area (subnational or national) cause an increase in emissions in another area through the relocation of activities (Meridian Institute, 2009).

26. Litter: Includes all non-living biomass with a diameter less than a minimum diameter chosen by the country (for example 10 cm), lying dead, in various states of decomposition above the mineral or organic soil (Forest Resource Assessment, 2015). This includes litter, fumic, and humic layers. Live fine roots (of less than the suggested diameter limit for below-ground biomass) are included in litter where they cannot be distinguished from it empirically (Intergovernmental Panel On Climate Change, 2006).

27. Natural forest: A forest composed of indigenous trees and not classified as a forest plantation (Intergovernmental Panel On Climate Change, 2006)

28. Net emissions: For REDD, a method for estimating emissions from gross deforestation that considers both the carbon stocks of the forest being cleared and the carbon stock of the replacement land use (Meridian Institute, 2009).

29. Non-Annex I Parties: All countries that are not listed in Annex I to the UNFCCC or the Kyoto Protocol. Most developing countries are Non-Annex I Parties.

30. REDD Readiness activities: the actions that help countries 'get ready' for REDD+, including capacity building, scientific studies, and developing national strategies, with the goal of mitigating climate change (Meridian Institute, 2009).

31. REDD+ is described as policy approaches and positive incentives on issues relating to reducing emissions from deforestation and forest degradation in developing countries; and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries (United Nations Framework Convention on Climate Change, 2008). Benefiting from REDD+ depends on making agreements to prevent CO_2 emissions from deforestation and forest degradation and on conserving forests as storage places for carbon (Stone and Chacón León, 2010).

32. REDD+: Reducing Emissions from Deforestation and Forest Degradation. (+) is conservation, sustainable management of forests and enhancement of forest carbon stocks.

33. Reference emission levels (RELs): RELS to demonstrate reductions in emission from deforestation (Meridian Institute, 2009). Net source is RELs. RELs refers to the two activities which are going to reduce emissions from deforestation and degradation (United Nations Framework Convention on Climate Change, 2011).

34. Reference levels (RLs): A reference level is synonymous with a crediting baseline for providing incentives for a participating REDD country if emissions are below that level (Meridian Institute, 2009). Net sink is RL. RL provides us with carbon stock assessment (United Nations Framework Convention on Climate Change, 2011).

35. Sink (or carbon sink): A pool (reservoir) that absorbs or takes up carbon released from other components of the carbon cycle, with more carbon being absorbed than released (Meridian Institute, 2009).

36. Soil organic matter: Includes organic matter in mineral and organic soils (including peat) to a specified depth chosen by the country and applied consistently through the time series. Live fine roots (of less than the suggested diameter limit for below- ground biomass) are included with soil organic matter where they cannot be distinguished from it empirically (Intergovernmental Panel On Climate Change, 2006)

37. Source: A pool (reservoir) that absorbs or takes up carbon released from other components of the carbon cycle, with more carbon being released than absorbed (Intergovernmental Panel On Climate Change, 2006)

CHAPTER II

LITERATURE REVIEW

Global Warming and Climate Change

Climate change following global warming is one of the primary concerns of humanity today. Warming of the earth's climate system has led to increases in global average air and sea temperatures, rising global average sea levels and widespread melting of snow and ice (Eliasch, 2008). Over the last century, global temperatures have risen by 0.7°C. Sea levels are rising at three millimeters a year and Arctic sea ice is melting at almost three per cent a decade (Intergovernmental Panel on Climate Change, 2007b). Continued warming of the atmosphere at the same rate will result in substantial damage to water resources, ecosystems and coastlines, as well as having an impact on food supplies and health (Eliasch, 2008). The Intergovernmental Panel on Climate Change (IPCC) suggests that approximately 20-30 per cent of species assessed so far are likely to be at increased risk of extinction if increases in global average temperature exceed 1.5-2.5°C (Intergovernmental Panel On Climate Change, 2007b). Current greenhouse gas emissions are within the upper range of the emission scenarios projected by the Intergovernmental Panel on Climate Change (IPCC). Scenarios of the Intergovernmental Panel on Climate Change (IPCC) predict an increase in globally averaged surface temperature of 1.1 to 6.4° Celsius over the period 1990 to 2100 (Intergovernmental Panel on Climate Change, 2007). Consequently, emission reductions are urgently needed within all sectors (Förster, 2009).

It becomes obvious that there is a direct relationship between increased levels of greenhouse gases (GHGs) in the atmosphere and rising global temperature. GHGs include carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), chlorofluorocarbons (CFCs), tropospheric ozone (O_3), and stratospheric water vapor (H_2O) (Ledley, et al., 1999). The most important GHGs directly emitted by humans (United State Environmental Protection Agency, 2012) and have increased significantly during the industrial period (Ledley, et al., 1999). Of the several anthropogenic greenhouse gases, CO_2 is the most important agent of potential future climate warming (Houghton, J. T., et al., 1996). Global carbon emissions from tropical deforestation and land use change fall within the range of 0.8 to 2.4 GtC yr^{-1} for the 1990s (Houghton, R A., et al., 2000; Schimel, et al., 2001), accounting for 12–28% of the total annual anthropogenic greenhouse gas emissions (Achard, F., Eva, Mayaux, Gallego, and Richards, 2002). Other estimates for the same period range from 1.1 ± 0.3 GtC yr^{-1} (Achard, F. F., et al., 2004) and 1.6 ± 0.6 GtC yr^{-1} (Houghton, R A and Hole, 2005). Most recent studies therefore take the average of 1.4 GtC yr^{-1} (range: 0.9–2.2) (Achard, F. F., et al., 2004; Houghton, R A., 2003). Future carbon emissions from deforestation could rise due to an expected agriculture expansion, increasing from 2.5 million ha yr^{-1} in the 1990s to 4.1 million ha yr^{-1} in the 2000s and 6.1 million ha yr^{-1} in the 2010s (Mollicone, et al., 2007). Concerned about the potency of global climate change through GHG emission, especially carbon dioxide (CO_2), decision makers have explored the potential of using forests for mitigation (Pfaff, et al., 2000).

Global warming by 2 °C is likely to be inevitable (Richardson, K., et al., 2009) and such a warming would have serious negative impacts on ecosystems, their functions and society at large (Smith, et al., 2009). There is the risk that with continued climate change tropical forests could turn from a carbon sink into a carbon source (Fischlin, et al., 2007). Forest play a key role as part of the problem, but also an indispensable part for the future solution of the climate change challenge (Huettner, 2007). Therefore, the Kyoto Protocol of UNFCCC has included forestry aspects in its articles. In the Kyoto Protocol, the reduction of GHG emissions is emphasized for industrialized countries. But, this CO_2 emission from deforestation in tropical countries was account for 17% of total global emissions (Intergovernmental Panel on Climate Change, 2007a). Tropical deforestation was responsible for the annual release of about 5.5 (Gullison and Frumhoff, 2007; Intergovernmental Panel on Climate Change, 2007a) to 8.1 billion tonne CO_2 $year^{-1}$ (Houghton, R A., 2003) in the 1990s. Therefore, deforestation of tropical forests in developing countries is considered one of the major contributors to GHG emissions (Santilli, et al., 2005), and avoiding deforestation has been proposed as a method of mitigation.

Role of Tropical Forest in Climate Change

Forests play a twofold role in climate change by absorbing carbon dioxide from the air and store carbon through the process of photosynthesis; and release carbon dioxide when deforestation and degradation occurred (Sathaya and Nilels, 2008). About 30 percent of the earth's land surface, 3869 million hectares in 2000, was covered by the forest which represent the most significant terrestrial carbon store especially in vegetation (77 percent of all carbon) and soil (39 percent of all carbon) (Eliasch, 2008; Intergovernmental Panel on Climate Change, 2007a). Almost half of the total forest area, 1571 million hectares in 2000, consists of tropical forests (Huettner, 2007). While covering only 22% of potential vegetation by area, tropical forests have been estimated to account for 43% of the world's terrestrial net primary productivity (Melillo, et al., 1993). Tropical forests hold large stores of carbon and play a major role in the global carbon cycle (Dixon, et al., 1994; Houghton, R. A., Lawrence, Hackler and Brown, 2001; Phillips and Gentry, 1994). Because of higher net productivity, the tropical forests are more effective in carbon sequestration than any other forests (Brown, S., Gillespie and Lugu, 1989; Soni, 2003). The tropical forests store large quantities of carbon in vegetation and soil, exchange carbon with the atmosphere through photosynthesis and respiration. These forests account for 37% of the total 90% of the world's terrestrial C that is stored in forests (Houghton, J. T., et al., 1996). The terrestrial tropics absorbs 2 Pg C $year^{-1}$, equivalent to 1.1 Mg $ha^{-1}year^{-1}$ of carbon, or 2 Mg $ha^{-1}year^{-1}$of dry matter, of which 2.4 Pg C $year^{-1}$ is offset by deforestation (Yadvinder Malhi and Grace, 2000).

Despite increase in carbon sink by global forests, forest can become a major emissions source when the stored carbon is released into the atmosphere by means of forest degradation and deforestation activities (Sathaya and Nilels, 2008). Continuous loss of tropical forests was responsible for carbon emissions to the atmosphere. The loss of carbon to the atmosphere due to deforestation is estimated to contribute about 18% to the global anthropogenic greenhouse gas emissions (Gullison and Frumhoff, 2007; Stern, 2007). This is more than from the global transport sector and represents the largest single category of carbon emission within developing world (Förster, 2009). Carbon emission from deforestation and degradation of natural forest is 1300.4 to 1865.1 $TgCO_2year^{-1}$ in Southeast Asia (Khun and Sasaki, 2014b). The majority is caused by the conversion of tropical forests (Canadell, Raupach and Houghton, 2009). Much of the focus on anthropogenic CO_2 emissions has been on the developed world and emerging economies in Asia which together account for over 80% of the cumulative CO_2 emissions and growth (Raupach, et al., 2007). By recognizing the carbon storage potential of the forest and CO_2 emission from deforestation, forest become a central role in tackling climate change (Eliasch, 2008), thus, several recent advances have led to variety of estimates of carbon stocks and fluxes (Aye, et al., 2011; Glenday, 2008; Khun and Sasaki, 2014a; Lasco and Pulhin, 2003, 2009; Mattsson, et al., 2012; Nophea Sasaki, et al., 2013; Woo, Han and Lee, 2010).

Forest is not only important role in climate change mitigation but have a role in preservation global ecological system and for supporting the livelihoods of local population (Hufty and Haakenstad, 2011). Crucially, the forests are home to local and indigenous communities that rely on the forest to sustain their livelihoods, spirituality and welfare (Lambrecht, Wilkie, Rucevska and Sen, 2009). They provide important goods, such as timber, fuel wood, medicinal products and food, and also services which are of cultural, aesthetic and recreational value (Millennium Ecosystem Assessment, 200; Shvidenko, et al., 2005). Many of these services are sustained by the biodiversity of natural forests and tropical forests alone are a haven for at least half of the earth's species (Shvidenko, et al., 2005). Beyond these provisioning services forests also play a crucial role in regulating the water cycle and climate at regional to global scale (Bonan, 2008). The livelihood of millions of people depends on the rich array of ecosystem services tropical forest provide. According to a recent World Bank report (Chomitz, K. M., et al., 2006) almost 70 million people live in remote areas of closed tropical forests and another 735 million people live in or near tropical forests and savannas, relying on ecosystems services provided by them. But many of anthropogenic activities such as globalization and subsequent industrialization and urbanization has reportedly lead to a corresponding decline and degradation of tropical forest ecosystems of the world (Matricardi, et al., 2010; Vaidyanathan, et al., 2010; Zhang, et al., 2010).

Tropical forest are home to about half of the world's species and their continued loss creates large and potentially irreversible loss of biodiversity (Olander, et al., 2006). Assuming roughly 10 million species worldwide, the current deforestation rate leads to an extinction of 8,000 and 28,000 species per year, or 20-75 species per day (Reid, 1995). Millennium Ecosystem Assessment (MEA) expected that species extinction rates would be more than 10 times higher than current rate (Millennium Ecosystem Assessment, 2005). The protection of tropical forests conserves ecosystems of often high biological diversity, since an estimated 70% of the world's floristic and faunistic species occur in tropical forests (WRI, 2001). The combined effect of deforestation and climate change could accelerate the opposite effects, leading to a serious deterioration of ecosystem services for marginalized people in developing countries. Many attempts to conserve or sustainably manage tropical forests fail, since private benefits from deforestation are usually greater than benefits from forest protection (Chomitz, K., 2002). A REDD mechanism could yield higher opportunity costs for forest conversion through carbon credit payment schemes (Huettner, 2007). These carbon payments could contribute to the improvement of local livelihoods and indigenous rights (Mollicone, et al., 2007) and could constitute an important financing tool for developing countries with tropical forest (Convention on Biological Diversity, 2006; Ebeling, 2006). In response to the negotiations of a REDD strategy under the UNFCCC, countries and organizations are undertaking pilot activities in order to identify possible strategies for REDD and to inform the negotiation process (Förster, 2009). In accordance with the specific circumstances in each country and region appropriate incentives and mechanisms need to be developed and implemented (Förster, 2009).

Deforestation and forest Degradation

Deforestation is generally understood as the direct human-induced conversion of forest land to non-forest land (Intergovernmental Panel On Climate Change, 2003), while forest degradation is, according to Intergovernmental Panel on Climate Change (IPCC) (Intergovernmental Panel On Climate Change, 2003), the direct-human induced long-term loss of forest carbon stocks in areas which remain forest land. Through forest governance are applied in every country, deforesting still persist especially in developing countries. Sustainable forest management remains insufficiently competitive compared with more destructive uses of forests (van Dijk and Savenije, 2009); 13 million hectares are lost annually to deforestation (approximately the size of the United Kingdom) (Food and Agriculture Organization of the United Nations, 2010), 97% of which takes place in tropical countries (Nabuurs, et al., 2007). Current population growth and rapid infrastructure development are expected to put additional pressures on tropical forests leading to reduce forest area, rapid rise in the release of CO_2 into atmosphere and decline in atmospheric carbon sequestration (Mohanraj, Saravanan and Dhanakumar, 2011). Linking land conversion rates with an expected linear population growth for the period 1990- 2020, Mollicone (Mollicone, et al., 2007) assumes an annual net forest area loss of 9.4 million ha year^{-1} during the 1990s, 11 million ha year^{-1} during the 2000s and 13.1 million ha year^{-1} during 2010s.

Drivers of deforestation and degradation are diverse and complex and have their origin at the international, national and local level (Förster, 2009). It might be related to the forest dependent population. Nearly 90 per cent of the world's rural population are found in Africa and Asia (Department of Economic and Social Affiars, 2005), especially in developing countries. Individuals in developing countries become more dependent on natural resources for their survival as growing local communities continue to depend on the surrounding forest ecosystems for food, shelter and medicine (Parker, et al., 2008). The direct drivers of deforestation are often the expansion of agriculture due to the increasing demand for food by a growing population and the harvest of timber (Geist and Lambin, 2002) and there are also underlying drivers such as weak forest governance that is leading to illegal logging (Geist and Lambin, 2002). Moreover, these drivers are traditionally divided into proximate causes (conversion of land for agriculture, infrastructure expansion, logging, extractive industries, etc.) and underlying factors (economic aspects, such as the demand for timber and wood for fuel, as well as policy-related, institutional, technological, socio-cultural, etc…) (Geist and Lambin, 2002). These causes are interlinked each other; for example, a road built for logging will attract settlers, who may clear-cut forests to ready land for agriculture. By using satellite observations of gross forest cover loss and a map of forest C stocks, Harris (Harris, et al., 2012) estimated gross C emissions across tropical regions between 2000 and 2005 at 0.81 T GtC year^{-1} with a 90 % prediction interval of 0.57 to 1.22 GtC year^{-1}. According to Baccini (Baccini, et al., 2012), deforestation in the tropic emitted about 1.0 PgC year^{-1}

between 2000 and 2010. Carbon emission from deforestation and degradation of natural forest is 1300.4 to 1865.1 Tg CO_2 $year^{-1}$ ($1CO_2 = 3.67C$) in Southeast Asia (Khun and Sasaki, 2014b). Deforestation is responsible for 18% of global CO_2 emissions (Stern, 2007), adding as much carbon to the atmosphere as the transport sector (Hufty and Haakenstad, 2011). Therefore, reducing emissions from deforestation and forest degradation, conservation, enhancement of forest carbon stocks and sustainable management of forests (REDD+) has been one of the key issues in international climate negotiations within the United Framework Convention on Climate Change (UNFCCC) (Mattsson, et al., 2012).

Reducing Emission from Deforestation and Forest Degradation (REDD+)

1. **The Development of REDD+**

Due to the significance of global greenhouse gas emissions, the Kyoto Protocol under the United Nation Convention on Climate Change (UNFCCC) become the first international agreement for reducing greenhouse gas emissions for mitigation climate change (Förster, 2009). This agreement is an important step toward concerted action for mitigating dangerous climate change. Under the Kyoto Protocol's Clean Development Mechanism, more focus on energy efficient technologies and reforestation and afforestation project for carbon sequestration (United Nations Framework Convention on Climate Change, 2003). Even though deforestation emits 18 % of the global greenhouse gas (Stern, 2007), there are no incentives for reducing emission from deforestation under the Kyoto Protocol's Clean Development Mechanism (Förster, 2009). By recognizing the important of emissions from deforestation, the government of Papua New Guinea and Costa Rica proposed that "Reducing Emissions from Deforestation (RED) in developing countries" should be considered as a mechanism under a post-kyoto agreement (United Nations Framework Convention on Climate Change, 2005). With the support of other parties under UNFCCC, Reducing Emissions from Deforestation (RED) was taken into the agenda of the negotiations on climate change mitigation at the eleven session of the Conference of the Parties (COP 11) in 2005 (United Nations Framework Convention on Climate Change, 2005). According to FAO, degradation accounted for as much as one billion hectare globally in 2005 (Food and Agriculture Organization of the United Nations, 2005). Therefore, forest degradation had been added to the scope of the mechanism, and RED had turn into Reducing Emissions from Deforestation and Forest Degradation (REDD).

At COP 13 in December 2007, the UNFCCC officially recognized the need to take action on these matters, and REDD was included in the Bali Action Plan (Parker, et al., 2008; (United Nations Framework Convention on Climate Change, 2008b). This includes the support and development of institutional capacities, the transfer of technology for the monitoring and reporting of emissions from deforestation and forest degradation, and the demonstration of pilot activities (Förster, 2009). At COP14 in Poznan, three additional strategic areas were included in REDD apart from deforestation and forest degradation: 1) conservation of forest, 2) enhancement of carbon stocks and 3) sustainable forest management (United Nations Framework Convention on Climate Change, 2009b). Thus REDD evolved into REDD+. REDD+ is seen as the most comprehensive initiative in forestry to date, incentivizing countries at various stages in the forest transition curve (A. Angelsen and Wertz-Kanounnikoff, 2008). However, there is still uncertainty about whether afforestation and reforestation programmes under the ongoing Clean Development Mechanism activities will eligible for REDD+ (Wertz-Kanounnikoff and Kongphan-apirak, 2009). REDD+ remained the main focal point of negotiations in subsequent UNFCCC conferences. REDD+ is seen as the main winner of the COP 15 in Copenhagen, as the Parties recognized the need to provide positive incentives to REDD+ "through the immediate establishment of a mechanism" (Decision 2) and accepted to provide methodological guidance for activities related to REDD+ (Decision 4) (United Nations Framework Convention on Climate Change, 2009b).

Furthermore, a financial commitment of US $3.5 billion was made by Norway, Japan, the United States, Britain, France and Australia to finance REDD+ readiness activities, such as developing reference levels on forest cover, deforestation and forest degradation, Monitoring, Reporting and Verification (MRV) systems, and designing national strategies for REDD+ (Corbera and Schroeder, 2011). In the following COP-16 in Cancun (2010), the REDD+ text formally set the implementation guidelines for REDD+. It envisaged the implementation of REDD+ through a phased approach covering three main stages: (i) readiness phase, (ii) implementation of action plans and demonstration activities, and (iii) full scale implementation (United Nations Framework Convention on Climate Change, 2011b). The activities are - Phase 1: Development of national strategies or action plans, policies and measures, and capacity building, Phase 2: Implementation of national policies and measures and national strategies or action plans that could involve further capacity building, technology, development and transfer, and results-based demonstration activities, Phase 3: Results-based actions that should be fully measured, reported and verified (United Nations Framework Convention on Climate Change, 2011b). The choice of the starting phase of each country would depend on national circumstances and technical capacity (United Nations Framework Convention on Climate Change, 2011b). The agreement further clarified that, during the first and second stages, countries could take a sub-national approach to developing reference emission levels and an MRV system (Sumana Datta, 2013).

Beside, recognizing a risk of harming local forest dependent communities, such as indigenous peoples, for example by losing access to the forest area and resources or loosing hard-won rights to cultural integrity (Convention on Biological Diversity,2006; Keliman, et al., 2012). These discussions were taken into account at COP16 in Cancun. However, a few critical issues regarding safeguards remain unanswered, such as by whom and how will the information on safeguards be used to verify real change on the ground and what will be the monitoring mechanism of safeguards (Morgan, 2010). The subsequent negotiation in COP 17 at Durban (2011) continued the discussion on safeguards and reference levels. With no agreements at the REDD+ negotiations at COP18 in Doha (2012) on how exactly to Monitor, Report and Verify (MRV), on forest monitoring systems, on safeguard information systems or on which activities to include in REDD+ in order to address the drivers of deforestation (United Nations Framework Convention on Climate Change, 2013). It was concerned that how to measure results on forest, how to report on it and how to verify the results, which is linked to the discussions about finance (Baccini, et al., 2012). The international political REDD+ architecture is therefore not at all clear and will continue to change during future negotiations (Angelsen, 2012). This opens a space on ground for very different ways to finance, design and implement REDD+

(Skov, Andersen and Thomsen, 2013). Therefore, REDD+ can either be defined as an term for actions that reduce emissions from deforestation and forest degradation at all levels, or a mechanism for creating economic incentives by applying performance and results-based payments (Skov, et al., 2013). UNFCCC bring up the REDD+ as a mechanism to establish positive incentive for reducing developing country emissions through forest protection. The latter may again apply either to economic incentives at the national level paid to governments in developing countries or to payments to local communities and households (Sunderlin and Sills, 2012). In order to measure the emission reduction performance and consequently to negotiate meaningful deforestation emission reduction targets and payment, baseline are crucial (Huettner, et al., 2009).

2. Estimation of Baseline

A baseline is an essential precursor to a viable and robust international compensation scheme for reduces emissions from degradation and deforestation (REDD) (Olander, et al., 2006). Baseline provide a benchmark against which emissions reduction can be calculated. In most systems that credit carbon emissions reductions, a baseline is required against which the savings can be compared. However, the nature of the baseline depends on the accounting rules about what, exactly, can be credited. There is considerable uncertainty at the moment about how baselines may be determined for operationalization of UNFCCC policy on Reduced Emissions from Deforestation and Degradation (REDD), since it is not yet decided what will be included. The rewards (carbon credits) would also be issued centrally. The possible options include crediting: a) reduction in emissions from deforestation, b) reductions in emissions from degradation, that is to say reductions in biomass/carbon stock in the forest without loss of forest area based on comparisons of rates of loss over time, c) enhancement or increases of forest biomass within areas of existing forest (sometimes referred to as forest restoration), d) conservation (in this context this usually means crediting for maintenance of a steady level of forest area and biomass density i.e. not just for improvements in these values), e) carbon stock, under which all forest carbon stock receives some sort of credit. Among them, reduced emission from deforestation and forest degradation is the initial step in forest management.

Baseline of deforestation and degradation can be determined at a number of special scales such as local, National and global (Olander, et al., 2006). Local baselines, such as those used for individual projects at the sub-national level, focus on activities at smaller scales and can have relatively high accuracy. However, project level assessment does not take into account emission leakage caused by the movement of deforestation to other areas. National level accounting and baseline determination for deforestation are technically feasible and will avoid undetected sub-national leakage. Local reductions that shift emissions elsewhere will be captured in the national accounting. A global or pan-tropical deforestation baseline could be used as a point of reference for all countries in the system. This has been proposed as one means for differentiating between countries with high versus low deforestation relative to the global average.

Baseline can refers to the historical baseline, that is, the rate of deforestation and degradation (DD) and the resulting CO_2e emissions over the past years. And baseline can refer to the projected BAU scenario; how would emission from deforestation and degradation evolve without the REDD activity (Arild Angelsen, 2009). Likewise, Olander (Olander, et al., 2006) stated that Historical or "reference period" baseline refers to activity and emissions in a defined period as they existed prior to a policy taking place. Business-as-usual baseline (BAU) refers to activities or emissions that might otherwise occur were a policy not put into place. For REDD, a historical baseline would determine the extent of deforestation and degradation over a predetermined period (i.e a 5 to 10 year period) before any policy is put into place. And, for REDD, a BAU baseline would estimate the extent to which deforestation and degradation would occur over time without a policy (or project) intervention (Olander, et al., 2006). BAU is appropriate for local project but less applicable to national-scale GHG accounting inherent in the REDD proposal. Because BAU baseline depend critically on behavioral assumptions to model BAU activity. BAU baseline, while not a perfect approach, has greater certainty and transparency for REDD. For setting BAUs baseline, historic deforestation data is the main variable. Because historic deforestation (the past 10 years) helps to predict future deforestation, taking into account both the rate of deforestation and trends in deforestation rate (increasing or decreasing) (Institute, 2011).

The past deforestation and related emissions need to be quantified to assess possible future emissions under business as usual (Förster, 2009). The establishment of baseline that allows the demonstration of reductions in emissions from deforestation is one of the issue in UNFCCC (United Nation Department of Economic and Social Affairs, 2014). Therefore, carbon sequestration projects need to develop a baseline of carbon emissions or removals by projecting the rate of land-use change over a given time period combined with carbon stock data (Brown, S., et al., 2007). When converting land use change (forest area (ha) change) to carbon density (tCO_2) change, forest inventory work is necessary to establish what the natural biomass/ carbon stock in different ecotypes is. In some countries there may be sufficient data on this to be multiplied by forest areas of different types to obtain the total carbon stock at any one point in time. Most countries have no data on this at all. Ninety three out of 147 countries reporting to FAO in 2005 register no change in biomass density of their standing forest form 1990 to 2005, although it is well known that all but the most remote tropical forest have been partly to severely degraded during this period (Forest Resource Assessment, 2005). The use of remote sensing is the most convenient method since satellites have been recording the earth's land cover over the past decades and the archived data allow the analysis of past changes in forest cover (Global Observation of Forest and land Cover Dynamics, 2009). While it is possible to identify deforestation with satellite images the monitoring and quantification of emissions from forest degradation with remote sensing remains to be a challenge (Förster, 2009).

Group of countries led by Papua New Guinea and Costa Rica, claimed that emission reductions from deforestation must be estimated against a national baseline of GHG emissions (Potvin and Bovarnick, 2008). National baselines are presented by their proponents as the only way to control leakage, or displacement of deforestation activities within a country. On the contrary, a group of Spanish- speaking Latin American countries led by Columbia argues that national baselines are currently inapplicable because many countries lack the capacity and the necessary information to determine a national baseline for GHGs or do not fully control their territory (Pelletier, Kirby and Potvin, 2012). Therefore, in Bali, countries agreed that demonstration activities could be done at both the national and the sub-national level.

Furthermore, a technical paper of the UNFCCC on the cost of implementing methodologies and monitoring systems for REDD signals that the majority of non-Annex I countries have limited capacity in providing complete and accurate estimates of GHG emissions and removals from forests (United Nations Framework Convention on Climate Change, 2009b). Regardless of the scale at which baseline emissions are estimated, accuracy and precision are needed to ensure that the reductions compensated for in a hypothetical REDD mechanism are properly quantified (Mollicone, et al., 2007a). The evaluation of baselines could rest on: 1) the assessment of changes in land use/land cover (activity data) and 2) the associated carbon stock change (Emission factor) (Gofc-Gold, 2009). Therefore changes in forest area over time and changes in the average carbon stock per unit area over time should be monitored.

3. Monitoring, Reporting and Verification (MRV) systems for REDD+

To comply with the accounting and reporting guidelines issued by UNFCCC (United Nations Framework Convention on Climate Change, 2011a), five major carbon pools in forests have to be considered. The IPCC (2003) defines five carbon pools to be monitored for deforestation and degradation: aboveground biomass, below-ground biomass, litter, dead wood and soil organic carbon. A MRV system for REDD+ needs to focus on two components (Intergovernmental Panel On Climate Change, 2003): i) assessing changes in forest area over time (activity data) and, ii) assessing changes in the average carbon stock per unit area over time (emission factors). Changes in forest area can be monitored by remote sensing or by systematic forest inventories with a sample size large enough to detect significant changes in forest area by forest type. Because deforestation is easily detected from space, particularly when it occurs on a large scale. But monitoring changes in areas subject to forest degradation is much more challenging for remote sensing than monitoring deforestation. Forest degradation affect the canopy cover only minimally but can affect the forest stock (Defries, et al., 2007). Murdiyarso (Murdiyarso, Skutsch, Guariguata and Kanninen, 2008) recommended that probabilistic approach should be used to investigate the risk of degradation since it is difficult to observe remotely. This involves stratifying forest by risk of degradation, based on observation of past trends and related to proxy variables such as accessibility (e.g., density of road networks, distance from settlements) (Schelhas and Sánchez-Azofeifa, 2006). The parameters in this kind of modelling would be different for different types of degradation processes (e.g., selective logging, fuelwood collection) (Iskandar, et al., 2006).

Changes average carbon stocks per unit area per forest type can be monitored using various methods, including secondary datasets and estimations from IPCC (Intergovernmental Panel On Climate Change, 2003), as well as in situ forest inventories and sampling using permanent plots. To measure changes in carbon stocks for deforestation and degradation, (United Nations Framework Convention on Climate Change, 2006) recommends two options: the stock-difference method and the gain-loss method. The stock-difference method builds on traditional forest inventories to estimate sequestration or emissions. Carbon stocks in each carbon pool are estimated by measuring the actual stock of biomass at the beginning and end of the accounting period. The gain-loss method is built upon an ecological understanding of how forests grow and upon information on natural or anthropogenic processes producing carbon losses. Biomass gains are estimated on the basis of typical growth rates in terms of mean annual increment (MAI) minus biomass losses estimated from activities such as timber harvesting, logging damage, collection wood for fuel, fire and overgrazing.

The Intergovernmental Panel On Climate Change (2003) also provides three tiers for estimating emissions, with increasing levels of data requirements and analytical complexity and therefore increasing accuracy:

3.1 Tier 1 uses default emission factors (indirect estimation of the emissions based on canopy cover reduction) for forest activities ('activity data') that are collected nationally or globally.

3.2 Tier 2 applies emission factors and activity data from country-specific data.

3.3 Tier 3 uses methods, models and inventory measurement systems that are repeated over time, driven by high-resolution activity data and disaggregated sub-nationally at a finer scale.

The choice of method will depend largely on the availability of data and resources to collect additional data (GOFC- GOLD, 2008). According to the technical capacity and resources of each country to monitor REDD, different reporting levels could be chosen in order to allow a broad participation of countries with different capacities. Tier 1 is the simplest method with average national estimates for forest carbon content and the assumption that the entire forest carbon is emitted when deforestation occurs. And it mainly utilizes default values and is therefore subject to high uncertainties (Plugge and Köhl, 2012). Tier 2 includes regional specific and up-to date estimates of the forest carbon content. Tier 3 requires more detailed carbon measurements on the ground, which need to be repeatedly monitored in permanent plots. Tier 2 and Tier 3 require increasing efforts and improve the reliability of estimates (Plugge and Köhl, 2012). For most of the countries involved in REDD+ demonstration and readiness activities, deforestation and forest degradation can be assumed to be a key category (Maniatis and Mollicone, 2010) requiring Tier 2 or Tier 3 (Plugge and Köhl, 2012). This demands a thorough evaluation of alternatives for a MRV system to reduce uncertainties by choosing either the best reliability for a given budget or a given reliability for the least costs (Köhl, et al., 2011).

4. Assessment of Carbon Stock

The report on COP-14 meeting identifies outstanding issues and highlights the presence of gaps in data and data quality including standing stocks per hectare and biomass density of forest type and forest ecosystem (United Nations Framework Convention on Climate Change, 2009b). In order to know the carbon storage potential of different forest ecosystem, estimation of different carbon pools in the forests is necessary. Forests cover approximately 65% of the total land surface and play a vital, yet complex role in the global carbon cycle (Newell and Vos, 2012). Holding 90% of the plant biomass carbon and 80% of soil carbon found in all terrestrial ecosystems, they also assimilate 67% of the total carbon dioxide removed from the atmosphere by these ecosystems (Landsberg and Gower, S. T., 1997). Annual loss of forests due to disturbance (harvesting, conversion, fire, insects, pathogens, and wind) contributes as much as 20% of total global greenhouse gas (GHG) emissions each year—rivaling emissions from the global transportation sector (Denman, et al., 2007). Gower (Gower, 2003) divides the carbon cycle into two interrelated phases: 1) initial disturbance effects on carbon pools and 2) changes in carbon cycle processes during forest ecosystem recovery or succession. These can be 'natural' disturbance events (e.g., fire, pest outbreak, etc.) or anthropogenic events (e.g., timber harvest, road construction, mining, etc.).

Estimating tree and forest biomass is essential for assessing ecosystem yield and carbon stock in compliance with the Kyoto Protocol on GHG reduction (Korner, 2003). Forests have five primary carbon pools–above-ground biomass, below-ground biomass, deadwood, litter, and soils–that simultaneously accumulate and release carbon (Intergovernmental Panel On Climate Change, 2006). Among them, carbon stored in the aboveground biomass is the most important (Gibbs, et al., 2007). Beside, Coomes and Grubb (Coomes and Grubb, 2000) found that in the forest and woodland 9% to 33% of aboveground carbon was allocated in the belowground (root). Likewise, deadwood comprised 1-17% of aboveground carbon (Houghton, R. A., et al., 2001), while 2% of aboveground carbon was allocated in litter layer (Aye, et al., 2011). Soil organic carbon also need to account in total forest carbon because more than 50 % of total forest carbon are stored in the soil organic carbon (Aye, et al., 2011). Estimates of total global soil prganic C are converging on about 1500 PgC in top 1 m soil (Eswaran, Van Den Berg and Reich, 1993). And IPCC (Intergovernmental Panel On Climate Change, 2000) estimated the total soil C pool in top 1 m as 2011 PgC. To comply with the accounting and reporting guidelines issued by UNFCCC (United Nations Framework Convention on Climate Change, 2011a), five major carbon pools in forests have to be considered (Intergovernmental Panel On Climate Change, 2003): 1) aboveground biomass, 2) belowground biomass, 3) dead wood, 4) litter, and 5) soil organic matter.

5. REDD+ Funding and Cost

The REDD proposal from Papua New Guinea and Costa Rica recommended that carbon credits should eventually be traded in a carbon market. After countries have sufficient capacity building funded through voluntary contributions, they can initiate a pilot market system or continue developing national level REDD activities using voluntary funds, until each country can participate in a carbon trading market. International funds, such as those obtained through the United Nations REDD program (UN-REDD), World Bank Forest Carbon Partnership Facility (WB FCPF), the Global Environment Facility (GEF) Tropical Forest Account, and others, support certain developing nations in readiness and capacity building for REDD. Money is awarded to stakeholders in the beginning of the project, while the payments from emissions reductions are to be awarded at the end of the commitment period. (Acosta-Morel, 2011).

The costs associated with REDD policies are likely to affect its degree of implementation. Some countries have begun capacity building and infrastructure development to monitor and reduce deforestation. This implies that the costs for implementing the REDD mechanism differ among nations. Costs also vary by the quantity of emission reductions available, which depends on the actual amount of carbon stored through avoided deforestation and degradation (Acosta-Morel, 2011). And there is no single cost per tonne (Boucher and Street, 2008), but a range of costs; some emissions reductions are cheaper to achieve when the opportunity costs of the alternative land use is low while others are more expensive because the benefit from the deforesting activity is greater. Moreover, the marginal costs of REDD are increasing, implying that as more carbon emission reductions are pursued, additional abatement becomes more expensive.

Forest Cover in Myanmar

According to the Forest Department of Myanmar (FD, 2010), forest area in Myanmar is estimated to be 31,773,000 ha (during, 2010), making up 46.9 % if the total geographical area of the country. Forest definition in Myanmar follows the international FAO definition "Land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use." Closed forest has a crown cover of over 40%. FAO defines deforestation as "change of land cover with depletion of tree crown cover to less than 10%." And degradation is defined as "changes within the forest class (e.g., from closed to open forest) that negatively affect the stand or site, but leave the tree canopy cover above 10%, in particular, lower the production capacity." In Myanmar, the progressive forest policies and programmes have contributed to reduce rates of deforestation, increase afforestation and overall stabilization of area under forest. Myanmar has one of the greatest biodiversity in the Indo-Pacific region (Myers, et al., 2000) and have a variety of forest types (Table 1).

Table 1 Main forest types in Myanmar

Forest type	Sub-categories	% of total forest area (2000)
Dry Forest		10
Coastal Forest	Mangrove or Tidal Forest, Beach and Dune Forest, Swamp Forest	4
Tropical Evergreen Forest		16
Mixed Deciduous Forest	Moist Upper Mixed Deciduous Forest (MUMD) Lower Mixed Deciduous Forest (LMD), Dry Upper Mixed Deciduous Forest (DUMD)	39
Hill and Temperate Evergreen Forest	Includes Pine Forest	26
Indaing Forest	Deciduous Dipterocarp Forest	5

Source: Forest Department, 2010

Deforestation in Myanmar

Myanmar has high forest cover with high deforestation rate (Table 2). In 1925, the forest cover was 65.8 % of total land area and decreased to 50.8% of total land area in 1989. This corresponds to a forest cover decrease of 15% of total land area in 65 years. From 1989 onwards, forest cover started to increase to reach 52.4% in 2004 (United Nations Reducing Emissions from Deforestation and forest Degradation, 2013). According to the FAO (Food and Agriculture Organization of the United Nations, 2010), deforestation was 0.93% annually between 2000-2010. Myanmar is one of the ten countries with the largest annual net loss of forest area. Myanmar is still endowed with a substantial forest area covering 47% of the country's total land area of 67,658,000 ha (Food and Agriculture Organization of the United Nations, 2010; FD, 2010). The loss of forest cover was high in the Dry Forest, the Mixed-deciduous and Deciduous (Indaing) Forests located in and around the central region or the Coastal Forest in the delta and coastal regions (United Nationand Reducing Emissions from Deforestation and forest Degradation, 2013) (Table 3).

Leimgruber (Leimgruberi, et al., 2005) stated that the major reasons for forest losses in these hotspots stemmed from increased agricultural conversion, fuelwood consumption, charcoal production, commercial logging and plantation development. While Myanmar continues to be a stronghold for closed canopy forests, several areas have been experiencing serious deforestation.

Table 2 Forest cover change at different periods

Year of appraisal	Forest cover (ha)	% of total land area	% of closed forest (crown cover>40%)
1925	44,518,700	65.8	
1955	38,700,300	57.2	57.0
1975	35,665,600	52.7	47.8
1989	34,370,100	50.8	43.2
1997	35,374,700	52.3	37.4
2004	35,478,000	52.4	27.3
2010	31,773,000	46.9	19.9
Total land area	67.658,000	100,0	

Source: The United Nations Reducing Emissions from Deforestation and forest Degradation, 2013.

Table 3 Forest cover change in different forest types

Forest type	**1995**		**2000**		**2010**	
	ha	**% of total forest area (2010)**	**ha**	**% of total forest area (2010)**	**ha**	**% of total forest area (2010)**
Dry Forest	3,442,400	10	3,455,400	10	3,114,710	10
Coastal Forest	1,376,900	4	1,382,160	4	467,330 (only mangrove)	4
Tropical Evergreen Forest	5,507,800	16	5,528,640	16	5,470,600	17.22
Mixed Deciduous Forest	13,425,300	39	13,476,060	39	12,157,300	38.26
Hill and Temperate Evergreen Forest	8,950,100	26	8,984,040	26	8,541,190	26.88

Table 4 (cont.)

Forest type	1995		2000		2010	
	ha	% of total forest area (2010)	ha	% of total forest area (2010)	ha	% of total forest area (2010)
Indaing Forest	1,721,200	5	1,727,700	5	1,321,870	4.16
Scrub land	-	-	-	-	700,000	2.21
Total	34,423,700	100	34,554,000	100	31,773,000	100

Source: Planning and Statistic Division, Forest Department, Myanmar; Forestry in Myanmar (FD, 2010); Myanmar REDD+ roadmap (The United Nations Reducing Emissions from Deforestation and forest Degradation, 2013.)

The government of Myanmar has been concerned about the extent of the forest degradation and deforestation in the country. For this reason, efforts are being made to combat deforestation through sustainable management and rehabilitation measures, including tree planting on degraded forest lands. The declaration of protected areas (PAs) is a major conservation measure designed to reduce deforestation (Andam, et al., 2008). Understanding the conditions under which PAs deliver conservation benefits for habitats and species is essential for policy makers, managers and conservation advocates (Brooks, et al., 2004; Kleiman, et al., 2000; Margules and Pressey, 2000). Various studies suggest that protected areas (PAs) can play a crucial role in conserving the biodiversity of Myanmar (Allendorf, et al., 2006) and are an effective approach for reducing deforestation and forest degradation (Andam, et al., 2008). Fully protected areas are often assumed to be the best way to conserve plant diversity and maintain intact forest composition and structure (Banda, Schwartz and Caro, 2006). Protected Areas (PAs) have long been regarded as an important tool for maintaining habitat integrity and species diversity (Bakarr, et al., 2004; Brooks, et al., 2004; Butchart, et al., 2010; Coad, et al., 2009) and for biodiversity safeguard in National starting point approach (Swan, et al., 2012). In Myanmar, 6.67 % of total land area is manage under the Protected Area and National

Park (Forest Department, 2010). Among them, Popa Mountain Park is famous for high biodiversity and species diversity. Due to population pressure, forest degradation and deforestation are found in PAs despite it is designated as a Protected Area. The Popa Mountain Park, located in central dry zone, covers 10, 000ha and consist of a protected area and buffer zone. Deforestation in the protected area is widespread, despite its formally protected status (Htun, et al., 2010). This is believed to have severe ecological effect on not only flora but also fauna species. It is important to curb deforestation in PMP with an effective conservation measure.

CHAPTER III

METHOD

Study area

The research was carried out at Popa Mountain Park (PMP), the only prominent extinct volcano in central Myanmar and famous for high plant diversity, medicinal plant. PMP is located in the central dry zone of Myanmar between 25° 56′ N and 95° 16′ E (Figure 2). The peak of Mount Popa is in the north of the park and reaches about 1,500 m above sea level. A range of small hills gradually declines in altitude for a distance of about 12 km to the south. The peaks of the small hills are range from 450 to 600 m above sea level. The surrounding plains are about 300 m above the sea level. Due to its comparatively high elevation, PMP has lower temperatures and higher rainfall than the rest of the central dry zone of Myanmar. Mean maximum and minimum monthly temperatures were 31.30°C and 8.52°C, respectively, and the mean annual rainfall was about 1170 mm (Popa Forest Department office, 2013).

The area was legally classified as a forest reserve in 1902. During the Second World War, villagers in the area surrounding the reserve gradually encroached and cleared large areas for cultivation. Within 1955-1972, the Forest Department conducted extensive reforestation and illegal cultivators were removed from the reserve. Reforestation work consisted of establishing plantations and encouraging natural regeneration of the native tree species. The vegetation gradually recovered in reclaimed areas (Htun, 2011). Between 1981 and 1984, the Nature Conservation National Park Project was conducted and Popa Reserve was proposed as a protected area (PAs). Afterward, the area was declared a Protected Area in 1989. The park covers an area of about 100 km^2 of which 88.7 % is covered by forest (Htun, et al., 2011). All forests in PMP are second or third growth after cutting and clearing for agriculture in the early twentieth century (Forest Department, 1981). Elevation ranges from about 300 to 1500 m asl. The PMP includes a diverse range of vegetation types such as dry mixed deciduous forest, and dry dipterocarp forest (scrub indaing forest), dry forest (Than-dahat forest) and dry hill or evergreen forest (Htun, N Z, Mizoue and Yoshida, 2011) (Figure 3). Moreover, in some areas there are *Pinus insularis* plantation, Eucalypus (*Eucalyptus camadulensis*) and *Xylia xylocarpa* plantations. These plantations were established during the period 1955–1972.

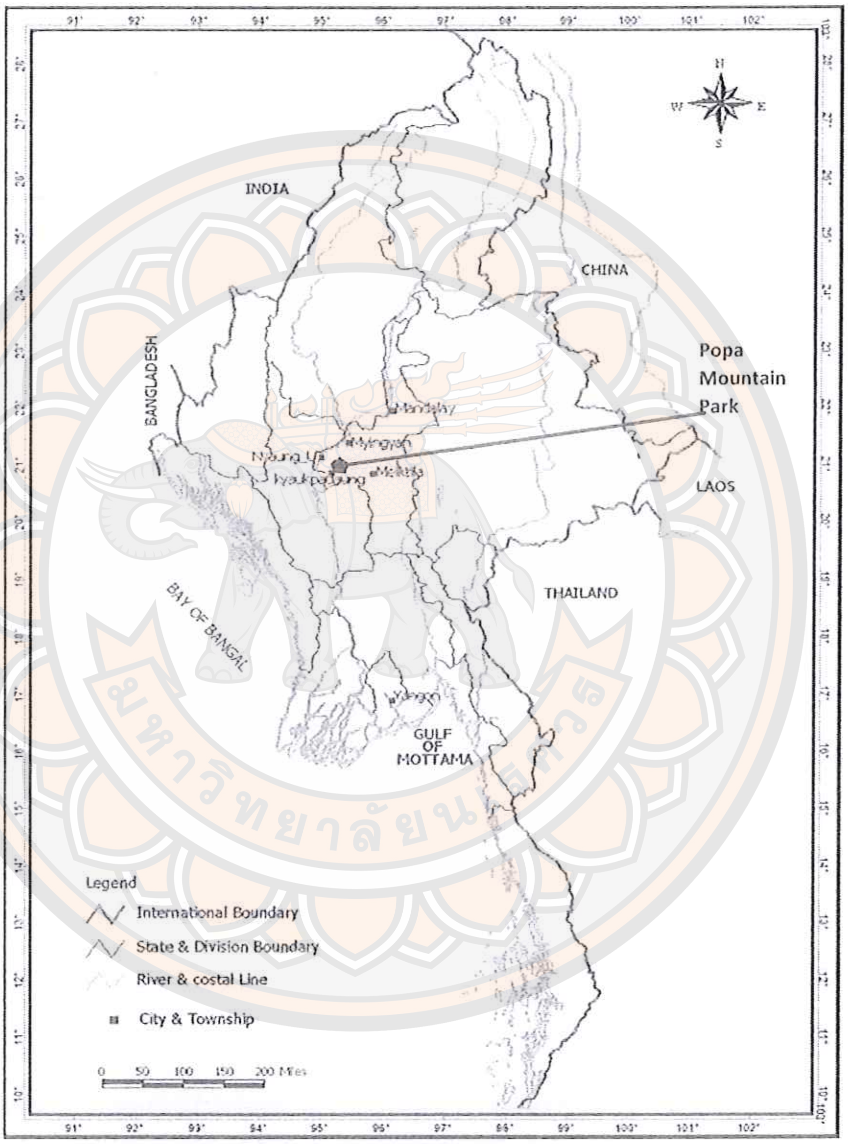

Figure 2 Location of Popa Mountain Park

Source: GIS section, Planning and Statistic Department, Forest Department, 2013

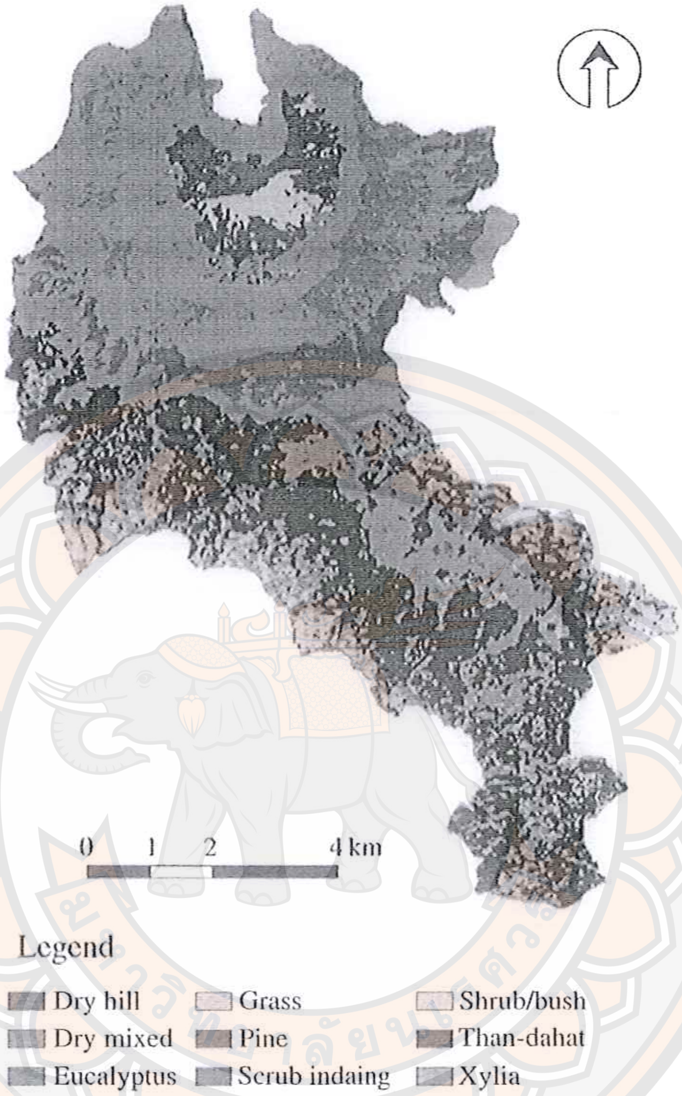

Figure 3 Land Cover Status map of Popa Mountain Park

Source: Htun, et al., 2011

A volcanic plug at the western foot of Mount Popa, locally called Taung-kalat, is a prominent landmark and famous for religious site in Myanmar and several thousand people visit it each year for religious and tourism purposes. Moreover, Popa Mountain Park is an important watershed for surrounding area. More than 100 springs in PMP supply drinking and irrigation water to the surrounding area. Therefore Popa

Mountain Park was designated as protected area for conservation of forest, protection of the watershed of Kyet-mauk-taung dam located at the southern edge of the park, conservation of medicinal plants for sustainable use, preservation of existing religious sites and ensuring sustainability of water sources, including natural springs (Htun, et al., 2010). To provide better protection for the forest within, and to regulate the local use of vegetation outside the boundary, an area of 103.6 km^2 is to be established as a buffer zone surrounding the Park in 2010. There are 45 villages are located surrounding the park. The major agricultural practices on the eastern side are cultivation of bananas, fruits and other seasonal crops, while on the west side the main activities are cultivation of rain-fed rice paddies, palm-sugar production, small-scale fisheries, and seasonal crops (Htun, et al., 2013).

Forest inventory for vegetation analysis

The vegetative data for this study were collected from four natural forest in PMP namely dry mixed deciduous forest, dry dipterocarp forest, dry forest and dry hill evergreen forest. Total 4 ha (for 4 forest types), 25 square plots with an area of 400 m^2 (20m x 20m) each were laid out in each forest type. Height (Ht) and diameter at breast height (DBH) of all trees (DBH $\geq$ 5 cm, Ht $\geq$ 1.3m) were measured in each plot. Plant species identification was achieved by using the "A Checklist of the Trees, Shrubs, Herbs and Climbers of Myanmar" (Kress, et al., 2003), local name and taxonomic experts.

Analysis of species composition and stand structure

1. **Important Value Index (IVI)**

To access the ecological important or significance of a species, important value index was used in this study. This is well-known method for the comparison of the ecological significance of species in a given forest type. Surveys that yield more or less the same important value index for the characteristic species should indicate the same or at least similar stand composition and structure, site requirements and comparable dynamics (Lamprecht, 1989).

Important value index (IVI) was calculated for each species by adding up relative density (RD) + relative frequency (RF) + relative coverage (RC) or relative basal area (RBA), thus permitting a comparison of the ecological significance of species in a given forest types. Density is the number of individuals per species, frequency is the occurrence or absence of a given species in a sample plot. Coverage is considered as an equivalent of the space a tree is occupying in the stand, which is calculated as the basal area of a species. The IVI was calculated as:

$$\text{Importance value (IV)(\%)} = \text{RD} + \text{RF} + \text{RBA}$$

$$\text{Relative density (RD) (\%)} = \frac{\text{Number of individuals of a species}}{\text{Total number of individuals}} \times 100$$

$$\text{Relative frequency (RF) (\%)} = \frac{\text{Frequency of a species}}{\text{Frequency of all species}} \times 100$$

$$\text{Relative basal area (RBA) (\%)} = \frac{\text{Basal area of a species}}{\text{basal area of all species}} \times 100$$

To find out the taxonomic and structural composition between the forest types, the IVI value was calculated on the level of families. Family important values were computed as the average of the relative basal area, density and frequency.

2. **Jackknife estimate of species**

The species richness was estimated using the Jackknife estimator (Heltshe and Forrester, 1983), which is based on the observed frequency of rare species in the community. Species richness is commonly expressed the number of species (tree species over a specified minimum diameter at breast height) per ha, and is also referred to as species density.

$$\hat{S} = s + \left[\frac{n-1}{n}\right]^{k}$$

where,

$\hat{S}$ = the Jackknife estimate of species richness,

s = the observed total number of species in " n " sample plots,

n = the total number of plots sampled, and

k = the number of unique species.

3. **Simpson's diversity index (D)**

Qualification of tree species diversity is an important aspect as it provides resources for many species (Cannon, 1998). The formula measuring the species diversity is as follows:

$$D = \sum_{i=1}^{s} \left[\frac{n_i (n_i - 1)}{N (N - 1)} \right]$$

where,

D = the Simpson's index of diversity,

n_i = the number of individuals of species "i" in the sample,

s = the number of species in the sample, and

N = the total number of individuals in the sample.

Simpson's index ranges from 0 to 1. The closer it is to 1, the less diverse the community. The index is usually expresses as **1 – D** because diversity decrease as D increase. Simpson's diversity index (1-D) gives more weight to those species which occur more frequently (Lamprecht, 1989).

4. **Shannon diversity index (H')**

Tree diversity inclines provide important information about rarity and commonness of species in a community (Suratman, 2012). Shannon's diversity index (H') places more weight on the rare species (Magurran Anne E., 1988). Shannon diversity index was used to provide the quantitative estimates of plant diversity.

$$H' = \sum_{i=1}^{s} -(P_i)(\ln P_i)$$

where,

H' = the Shannon diversity index, S is the number of species

P_i = the proportion of total sample belonging to "ith" species, and

ln = the natural logarithm of a number.

5. **Shannon evenness index (E)**

Evenness indices, which are a structural composition index reflecting the dominance of species were calculated using the following formula:

$$E\,(\%) = 100\left(\frac{H'}{\ln H_{max}}\right)$$

where,

E is the Shannon's evenness (evenness measure, range 0-1),

H' is the Shannon diversity index,

H_{max} is the ln (S), and

S is the number of total species found in the sample plot.

6. **Floristic heterogeneity**

An approximate indication of the homogeneity of a stand and of high diversity of tree species can be expressed by frequencies (Lamprecht, 1989). Species were assigned the frequency classes I, II, III, IV and V using their absolute frequencies:

Frequency class	Absolute Frequency
A = I	1 – 20 %
B = II	20 – 40 %
C = III	40 – 60 %
D = IV	60 – 80 %
E = V	80 – 100 %

If the diagram had a high values in frequency classes I/II and a low values in frequency classes IV/V, it indicates a high degree of floristic heterogeneity. If a high values was found in frequency classes IV/V and a low values in I/II, it indicates constant or similar tree species composition in the area. Frequencies depend on the size of the subplots. The larger the subplots, the high number of species were found in the higher frequency class. Therefore, comparison of frequency diagrams is possible based on the areas with the same subplot sizes.

7. Sorenson's quantitative index and Jaccard index

The Sorenson's quantitative index and Jaccard's index which accounts for the relative abundance of shared species was used to assess the degree of floristic similarity within and between the forests stands (Magurran Anne E., 1988). These indices are based on the presence or absence of species. If both forest stands are floristically identical, the value of indices are 100 and if they are completely different, the value of indices are zero. Sorenson's similarity and Jaccard's similarity were calculated using the following equations:

$$\text{Sorenson } (C_s) = \frac{2j}{a+b} \text{ X } 100$$

$$\text{Jaccard } (C_j) = \frac{2j}{a+b-j} \text{ X } 100$$

where,

Cs = coefficient of similarity,

a = number of species in one site,

b = number of species in another site, and

j = number of species common to both sites.

These measures are simplicity and all species count equally in the equation irrespective of whether they are abundant or rare (Magurran Anne, E., 1988).

Carbon measurement

1. Measurement of aboveground carbon

Data from 100 sample plots of 400 m^2 (20 m × 20 m) were collected in dry mixed deciduous forest, dry dipterocap forest, dry forest and dry hill forest in Popa Mountain Park (Korea Forest Service, 2007). Diameter at breast height (DBH) and height of all trees (DBH ≥ 5 cm) were measured in each plot. Aboveground carbon (AGC) (tonne $C{\cdot}ha^{-1}$ or tC hereafter) was estimated using the equation below (Brown, S., 1997).

$$AGC = VOB \times WD \times BEF \times CC \quad (1)$$

where, VOB ($m^3 \cdot ha^{-1}$) is stand volume over bark. WD ($Mg \cdot m^{-3}$) is wood density. WD for all tree species were taken from Zanne, et al. (2009; ICRAF, 2010), using genus level averages where species specific data were not available following Chave, et al. (Jerome Chave, et al., 2006) and Bryan, et al. (Bryan, et al., 2010) and plot level averages for the cases where species could not identified following (Mattsson, et al., 2012). Carbon content default value (CC) 0.47 was used (Intergovernment Panel on Climate Change, 2006). BEF is the biomass expansion factor, determined from S. Brown (1997).

$$BEF = e^{\left[3.213\text{-}506 \times Ln(BV)\right]} \quad (2)$$

where, BV is the biomass of inventoried volume in $ton \cdot ha^{-1}$, calculated as the product of stand VOB ($m^3 \cdot ha^{-1}$) and wood density, WD (WD = 0.57 $Mg \cdot m^{-3}$) for tropical forests (S. Brown, 1997).

2. **Measurement of belowground carbon**

For this study, an allometric equation developed by Carins (Cairns, et al., 1997) was used to calculate belowground carbon (BGC),

$$BGC = e^{[-1.0587 + 0.8836 \times Ln(AGB)]} \times CC \quad (3)$$

where, AGB = aboveground biomass in ton.

Carbon content default value (CC) 0.47 was used to estimate the carbon content of tree biomass as proposed by the IPCC (IPCC, 2006). The root/shoot ratios of the dry biomass (as well as carbon) were calculated for each species.

3. **Measurement of litter layer carbon**

Litter layer were collected from each plot and calculated as part of the aboveground biomass. For this purpose, 4 subplots having size of 30 cm x 30cm for litter layer (Korea Forest Service, 2007) were randomly established in each plot. All litter layer inside the frame were collected and weighed. The litters were mixed thoroughly and a sample that is representative of the materials found in the litter were collected. Samples (approximately 200 to 300 g) were collected in each plot. Collected samples were oven-dried (80°C) until constant weights were obtained (Kenzo, et al., 2009). The biomass of each sub-plot was calculated by the ratio of dry and fresh weight of the sample. A carbon content default value of 0.47 was used to estimate the carbon content of tree biomass as proposed by the IPCC (IPCC, 2006).

$$\text{Biomass (kg)} = \frac{\text{Sample dry weight}}{\text{Sample fresh weight}} \text{ X Total fresh weight} \quad (4)$$

4. Measurement of deadwood carbon

Dead wood (DBH > 10 cm) were measured in the 25 square plots (20m × 20m) where the length and two orthogonal diameters at each end were taken and classified decay stages according to decomposition classes by following (Coomes, et al., 2002): 1) Stage I, bark largely intact; 2) Stage II, bark and twig lost, but shape of trunk remaining intact; and 3) Stage III, shape no longer maintained, and trunk sinking into the ground. The carbon stock of standing dead trees and coarse wood debris (CWD) was estimated by multiplying the log volume by the deadwood density (fresh-wood density × decay stage modifier) according to Coomes (Coomes, et al., 2002). As it was difficult to identify these logs to species a mean fresh-wood density (490 kgm^{-3}) and decay stage modifiers: Stage I, 0.82; Stage II, 0.66; Stage III, 0.47 were used (Coomes, et al., 2002). The biomass of all CWD biomass (CWD-B) in the plot was derived using Equation (4) and carbon content of each CWD was calculated as 47% of dry weight (biomass).

$$\text{CWD-B} = \sum \text{LV} \times \text{FWD} \times \text{DSM} \quad (5)$$

where, LV is log volume of dead tree (m^3), FWD is fresh-wood density (kgm^{-3}), DSM is decay-stage modifier (DSM is defined as a deadwood density as a proportion of fresh-wood density). The volume of each log (LV) was calculated as:

$$LV = \frac{\pi \times l}{32}\left[(a+b)^2 + (c+d)^2\right] \quad (6)$$

where, l is the length of the log, a and b are orthogonal diameters at one end, and c and d at the other.

5. Soil organic carbon

Soil organic carbon (SOC) of 1 m depth was estimated in each forest type (Intergovernmental Panel on Climate Change, 2006; Batjes, 1996). A 100 cm^3 of soil samples were taken from each different layer (0-10 cm, 10-20cm, 20-40cm, 40-60cm, 60-100cm). The total soil samples were 60 cans (4 forest types x 3 points x 5 layers). Intergovernmental Panel on Climate Change (2006) recommended that the minimum sampling depth should be 30 cm depth. But the calculations of global SOC traditionally report results to a 1 m depth (Eswaran, Van Den Berg and Reich 1993; Batjes, 1996). Likewise, this is the most common reference depth used in related studies (Han, et al., 2010; Sleutel, Neve and Hofman, 2003; Lettens, et al., 2006). Therefore, 1 m depth was selected in order to facilitate comparison with international literatures. Each soil sample was placed in a plastic bag and sealed in the field. Then all samples were taken to the Forest Research Institute, Myanmar for testing.

Soil bulk density (g cm^{-3}) was determined by getting the quotient of the dry weight of the soil (gram) and bulk volume of the soil (cm^3). The weight of soil (Ws) for each soil layer was calculated by multiplying the volume (m^3) of soil per hectare (Vs) and the soil bulk density (BD) (g cm^{-3}).

$$\mathbf{Ws = Vs \times BD} \quad (7)$$

To estimate the soil organic matter (OM) content, a soil sample were analyzed by loss on ignition (LOI) method, and at conversion factor of 0.58 was used to convert OM to SOC (Konen, et al., 2002)

$$\text{Organic matter content (\%)} = \frac{\text{weight before ignition} - \text{weight after ignition}}{\text{weight before ignition}} \times 100 \quad (8)$$

Per ha SOC was estimated by multiplying the weight of soil per ha (1 m in depth) in metric ton (Ws) and the content of SOC in percent (% SOC) (Batjes, 1996;

Lal, 2000).

$$\mathbf{CD = Ws \times \%SOC} \tag{9}$$

where: *CD* is the carbon density (Mg ha^{-1} or tonne ha^{-1} or t ha^{-1} hereafter), *Ws* is the weight of soil (Mg) and *SOC* is the soil organic carbon (%).

$$Vs = l \times w \times d \tag{10}$$

where: *Vs* = the volume of soil (m^3), *l* = the length of soil equivalent to 100 m, *w*= the width of the soil equivalent to 100 m and *d* = the depth of the soil equivalent to 1 m.

6. **Statistical analysis**

One way analysis of variance (ANOVA) was used to test the different among carbon content of various issues (aboveground, belowground, Litter layer, undergrowth, deadwood), and various forest types. Post-hoc test was used to know the significant of one way ANOVA. Duncan's multiple range test (DMRT) was used to examine the difference in soil carbon accumulation within the soil depths and among different forest types.

Estimation of carbon revenue

To estimate carbon revenues from reducing deforestation and forest degradation, one must understand baseline deforestation and project deforestation (the latter is deforestation when carbon project is implemented) and respective carbon emissions.

1. **Baseline Deforestation**

Baseline deforestation was determined using data analyzed by Htun, et al. (2010) who based their study on remote sensing data in 1989, 2000 and 2005. In his study, the images were captured at Path 133 and Row 46 at three different times (landsat 5 TM, 16 January 1989; landsat 7 ETM, 4 April 2000; and landsat 7 ETM, 13 February 2005) and processed with ERDAS IMAGINE software. To provide a range of possible deforestation, three rates of deforestation were used to predict future deforestation, namely low, average, and high rates. Baseline Deforestation (BD) was derived from the forest cover change between Time 2 and Time 1. Forest cover of each forest type in Time t was estimated by

$$FA(t) = FA(0) \times e^{-k \times t} \quad (6)$$

where, FA(t) is the area of forest at time t (ha), k is the rate of deforestation.

$$BD(t) = \Delta FA(t) = FA(2) - FA(1) \quad (7)$$

Where, BD(t) is Baseline deforestation of the forest at a year t (deforestation between time t_2 and time t_1). Htun, et al. (Htun, et al., 2010) found that the deforestation rate in PMP (park area and surrounding area, 3 km) is 0.09% to 4.42% based on remote sensing analysis on Landsat image in 1989, 2000 and 2005 (Htun, et al., 2010). Therefore, 0.09%, 2.25%, and 4.42% were respectively used as low, mean, and high deforestation rates in our analysis.

2. **Project Deforestation**

This is the deforestation when forest carbon project is implemented. There are various methods to predict forest cover change once project is implemented. For simplicity, we followed methods developed by Ty, et al. (Ty, et al., 2011) to predict the change of forest covers in the PMP;

$$PD(t) = RPI(t) \times BD(t) \quad (8)$$

where, PD(t) is Project deforestation at year t (ha yr^{-1}), RPI(t) is the relative impact of all project activities on deforestation at Time t in (%). RPI depends on activities undertaken to reduce the drivers of deforestation and degradation.

Htun, et al. (2010) identified the deforestation rates by analyzing increments of non-forest areas; individual tree, shrub land, grassland, regeneration, non-vegetative area, agriculture, water bodies and villages. He found that the deforestation rate in PMP is 0.09% to 4.42% (in the Park and the surrounding 3 km). Htun, et al. (2013) also stated that fuelwood and forest products are illegally collected for income and for producing sugar from palm sap, a major source of income for some people living in the western side of PMP. Likewise, it was personally observed that most of the households harvest wood for fuel in the summer season (March-May) in order to keep for the whole year. Mass extractions of wood for fuel, household and farm materials followed by annual fires lead to deforestation within a few years. Accordingly, we would conclude that the drivers of deforestation and forest degradation in PMP were conversion of agriculture land, conversion to settlement, annual fire, illegal logging, fuelwood collection, extraction of wood for household consumption and use as farm material. Similarly, Ty, et al. (2011) identified drivers of deforestation and forest degradation in a northern province of Cambodia as conversion to cropland, conversion to settlements and migrant encroachment, forest fire due to land clearing, hunter inducing forest fire, illegal logging for commercial sale and timber harvesting for local use. Therefore, the project activities proposed by Ty, et al., (2011) could also be introduced to PMP. These project activities include strengthening land-tenure, land-use plans, forest protection, assisted natural regeneration and establish fuelwood plantation, introduction of fuel-efficient stoves, introduction of mosquito nets, agricultural intensification, water resource development projects or maintenance of watershed area, non-wood forest product (NTFP) development and fire prevention. For simplicity, RPI estimated by Ty, et al. (2011) was also used in our study. It is suggested that this RPI should be revised when more data become available.

3. Baseline and Project carbon emissions

Estimation of baseline emissions is a prediction of what would have happened without the policy in place. And against which, reduce emissions are compared after the policy was enacted. In this study, carbon emissions due to deforestation can be therefore estimated by

$$CE_{baseline}(t) = BD(t) \times CS \times \frac{44}{12} \tag{9}$$

$$CE_{project}(t) = PD(t) \times CS \times \frac{44}{12} \tag{10}$$

where, $CE_{baseline}(t)$ is Baseline emissions or emissions without REDD + project ($tCO_2 \cdot yr^{-1}$), $CE_{project}(t)$ is carbon emissions under the project ($tCO_2 \cdot yr^{-1}$), CS is carbon stocks ($tC \cdot ha^{-1}$) and 44/12 is the ratio of the molecular weight of CO_2 (44) to the molecular weight of carbon (12).

Reduced emissions ($tCO_2 \cdot yr^{-1}$) when REDD+ project is implemented were estimated by.

$$RE(t) = CE_{baseline}(t) - CE_{project}(t) \tag{11}$$

4. Carbon credit and revenues

Carbon credits and carbon revenues at Year t were estimated by

$$CC(t) = RE(t) \times (1 - lk) \quad (12)$$

$$CR(t) = CC(t) \times \$ \quad (13)$$

where, CC(t) is Carbon Credits at time t ($tCO_2 \cdot yr^{-1}$), lk is the leakage of carbon emissions outside the project boundary ($tCO_2 \cdot year^{-1}$). Leakage refers to direct emissions elsewhere caused by the emission reduction in the project/ program. Example, protection of a forest area in one place may lead to deforestation another place (Makundi, 2009). Leakage is difficult to estimate, and Murray, et al., (2002) found that it varies greatly from one location to another. In this study, we assumed 30% leakage (lk = 0.30) following (Ty, et al., 2011).

CR(t) is carbon revenues at time t (US\$ yr^{-1}). The price of US \$5 per tCO_2 was used because it similar to the current price for forestry offsets in the voluntary market (Diaz, Hamilton and Johnson, 2011; Busch, et al., 2009). Price of US \$5.00 per tCO_2 was used in the bilateral REDD+ agreement between the governments of Guyana and Norway (Guyana and Norway, 2009) and in the Brazilian Amazon fund MMA (2008).

CHAPTER IV

RESULTS

Species composition, diversity and stand structure of different forest in PMP

Species composition, richness and diversity

1. Tree species composition

To assess the species composition and stand structure, the important value index (IVI) was used. No single species clearly dominated in the dry mixed deciduous forest (DMDF) (Table 4). In the DMDF, the most abundant species were *Shorea obtusa* (8.58% important value index, IVI), *Croton roxburghianus* (7.34% IVI) and *Pittosporum napaulensis* (4.83% IVI). The highest IVI value belonged to the species *Shorea obtusa* in the dry dipterocarp forest (DDF), *Tectona hamiltoniana* in the dry forest (DF) and *Vitex canescens* in the Dry hill or evergreen forest (DHEF).

Among all forest types, the most frequently occurring species is *Techtona hamiltoniana* (in dry forest), with 610 individuals recorded with a relative frequency of 17.12%. Due almost solely to its high relative frequency, *Techtona hamiltoniana* also has the highest species importance value with 54.25. The next most frequently occurring and most important species are *Vitex canescens* (in the DHEF) with important values of 29.13 represented by 263 individual trees with relative frequencies of 12.72 %. The result of our study suggests that *Techtona hamiltoniana* is an ecologically important species in Popa Mountain Park. In the DMDF, 52.59% of the relative abundance included 10 common species (13% of total in DMDF) while 55.45% of RD (Relative Density) included five common species (9% of total species) in the DDF. In the DF only one species made up 70.03% of RD whereas two species made up 52.61% of RD in the DHEF. These findings indicate that the number of species per unit area were high in all investigated forests.

At the family level, it was found that the taxonomic composition of the forests in PMP are different. The DMDF was dominated by Dipterocarpaceae, Euphorbiaceae and Combretaceae. Dipterocarpaceae is also dominant family in the DDF, followed by Verbenaceae, Combreaceae. While, Verbenaceae (mainly *Techtona* *hamiltoniana*), Combretaceae and Rhamnaceae were the most common forest tree families in the DF. In the DHEF, Verbenaceae (mostly *Vitec canescens*), Myrsinaceae and Bixaceae are the dominant family (Table 5).

The Dipterocarpaceae family was the family with the highest ecological importance given the IVI value in the DMDF. The Dipterocarpaceae with only 3 species ranking 4th in the species-rich family in the DMDF, is the most important family based on the IVI values. This highlights that the most species-rich families are not necessarily the most important families based on the IVI values. For example, in the DDF, the Dipterocarpaceae with only 3 species ranking 3rd in the species-rich family was the most important family based on the results on IVI. This is due to the high number of individuals and high frequencies. In DF, the highest important value was found in the Verbenaceae family, which has 4 species, ranking it as the 2nd species-rich family. In the DHEF, the Verbenaceae family, although with only one species, was the highest ecologically important family according to the IVI value. This is because each species was represented with many individuals.

Verbenaceae, Caesalpiniaceae and Moraceae were observed as species rich families for the DMDF, representing 5 species (8% of the total species) in each family (Table 5). For the DDF, the species rich families were Verbenaceae, which possessed 5 species with 9% of the total species, Anacardiaceae and Rubiaceae which possessed 4 species with 8% of the total species. The species rich families for DF were found to be Fabaceae and Mimosaceae which were represented by 5 species with 11% of the total species. In the case of DHEF, the species rich families were Fabaceae and Mimosaceae which processed 3 species with 8% of the total species.

Table 4 Ten highest species important value index (IVI) of forests in Popa Mountain Park

	Scientific Name	SD (tree ha^{-1})	BA (m^2 ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Dry Mixed deciduous forest	*Shorea obtusa Wall.*	103	3.78	9.71	2.25	13.76	8.58
	Croton roxburghianus N.P.Balakr	100	1.91	9.43	5.63	6.95	7.34
	Pittosporum napaulensis (DG) Rehder Wilson	76	0.93	7.16	3.94	3.39	4.83
	Bixa orellana L.	64	1.08	6.03	3.10	3.95	4.36
	Terminalia crenulata (Heyne) Roth	41	1.83	3.86	2.25	6.66	4.26
	Flacourtia cataphracta Roxb.	45	0.97	4.24	3.10	3.53	3.62
	Litsaea glutinosa (Lour) C. B. Cl.	43	0.94	4.05	3.38	3.43	3.62
	Strychnos potatorum L.f	40	0.52	3.77	3.38	1.89	3.01
	Dipterocarpus tuberculatus Roxb.	17	1.327	1.60	1.69	4.82	2.70
	Diospyros spp.	29	0.47	2.73	3.38	1.71	2.61
	Others	204	5.12	47.41	30.99	18.60	22.94
	Total	**1061**	**27.52**	**100**	**100**	**100**	**100**
Dry Dipterocarp forest	*Shorea obtusa* Wall.	251	6.63	19.41	6.19	28.91	18.17
	Dipterocarpus tuberculatus Roxb.	172	5.19	13.30	5.93	22.65	13.96
	Shorea siamensis (Kurz) Miq.	121	2.16	9.36	5.67	9.43	8.15
	Terminalia crenulata (Heyne) Roth	93	1.42	7.19	6.19	6.20	6.53
	Dalbergia oliveri Gamble	80	0.97	6.19	5.67	4.23	5.36
	Buchanania lanzan Spreng.	49	0.99	3.79	5.93	4.33	4.68
	Premna pyramidata Wall.	47	1.05	3.63	4.64	4.58	4.29
	Chionanthus ramiflora Roxb.	54	0.41	4.18	4.12	1.79	3.36
	Diospyros burmanica Kurz	49	0.59	3.79	3.61	2.56	3.32
	Wendlandia tinctoria DC.	44	0.34	3.40	3.87	1.47	2.91
	Others	333	3.18	25.75	48.20	13.85	29.27
	Total	**1293**	**22.93**	**100**	**100**	**100**	**100**

Table 4 (cont.)

	Scientific Name	SD (tree ha^{-1})	BA (m^2 ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Dry Forest	*Tectona hamiltoniana* Wall.	610	20.32	70.03	17.12	75.59	54.25
	Terminalia oliveri Brandis	75	3.48	8.61	13.01	12.94	11.52
	Tectona grandis L.f	42	0.50	4.82	6.16	1.88	4.29
	Lannea coromandelica (Houtt). Merr.	18	0.28	2.07	8.22	1.03	3.77
	Dalbergia oliveri Gamble	18	0.56	2.07	6.16	2.10	3.44
	Diospyros burmanica Kurz	10	0.12	1.15	2.74	0.45	1.44
	Morinda tinctoria Roxb.	8	0.07	0.92	2.74	0.24	1.30
	Acacia catechu Willd.	6	0.12	0.69	2.74	0.43	1.29
	Dalbergia cultrata Grah.	4	0.09	0.46	2.74	0.32	1.17
	Albizzia chinensis (Osbeck) Merr.	3	0.25	0.34	2.05	0.94	1.11
	Others	77	1.10	8.84	36.30	4.09	16.41
	Total	**871**	**26.88**	**100**	**100**	**100**	**100**
Dry hill/ evergreen forest	*Vitex canescens* Kurz	263	20.07	32.71	12.72	41.96	29.13
	Rapanea af. Neriifolia (Seib & Zucc) Mez.	160	5.47	19.90	8.09	11.43	13.14
	Bixa orellana L.	63	1.95	7.84	9.25	4.09	7.06
	Eriobotrya bengalensis (Roxb.) Hook. f.	32	3.17	3.98	5.78	6.63	5.46
	Syzygium cumini (L.) Skeels.	22	2.23	2.74	8.09	4.65	5.16
	Wendlandia tinctoria DC.	43	1.31	5.35	6.36	2.75	4.82
	Croton roxburghianus N.P.Balakr	44	1.48	5.47	4.62	3.10	4.40
	Litsaea glutino (Lour)C.B.Cl.	30	2.62	3.73	3.47	5.48	4.23
	Cinnamomum obtusifolium (Roxb.) Nees	27	0.62	3.36	4.62	1.29	3.09
	Cissus discolor Blume	20	0.19	2.49	5.20	0.40	2.70
	Others	100	8.71	12.44	31.79	18.22	20.82
	Total	**804**	**47.83**	**100**	**100**	**100**	**100**

*SD = stand density, BA = basal area RD = relative density, RF = relative frequency, RBA = relative basal area, IVI = important value index

Table 5 Ten highest family important value index (IVI) of forest in Popa Mountain Park

	Family Name	NS	SD (n/ha)	RD (%)	RF (%)	RBA (%)	IVI (%)
Dry Mixed deciduous forest	Dipterocarpaceae	3	146	13.76	3.00	21.72	12.83
	Euphorbiaceae	5	118	11.12	8.00	8.41	9.18
	Combretaceae	4	58	5.47	5.00	10.47	6.98
	Fabaceae	3	52	4.90	6.33	5.11	5.45
	Pittosporaceae	1	76	7.16	4.67	3.39	5.07
	Verbenaceae	6	46	4.34	5.33	4.92	4.86
	Bixaceae	1	64	6.03	3.67	3.95	4.55
	Caesalpiniaceae	6	39	3.68	4.67	4.64	4.33
	Lauraceae	1	43	4.05	4.00	3.43	3.83
	Flacourtiaceae	1	45	4.24	3.67	3.53	3.81
	Others	43	374	35.25	51.67	30.42	39.11
	Total	**74**	**1061**	**100**	**100**	**100**	**100**
Dry Dipterocarp forest	Dipterocarpaceae	3	544	42.07	9.26	60.98	37.44
	Verbenaceae	5	101	7.81	8.52	6.75	7.69
	Combretaceae	3	98	7.58	8.89	6.37	7.61
	Anacardiaceae	4	95	7.35	8.89	5.66	7.30
	Fabaceae	2	89	6.88	8.52	4.67	6.69
	Rubiaceae	4	64	4.95	7.78	2.15	4.96
	Ebenaceae	2	51	3.94	5.56	2.59	4.03
	Oleaceae	1	54	4.18	5.93	1.79	3.96
	Loganiaceae	2	30	2.32	5.19	2.16	3.22
	Mimosaceae	3	24	1.86	4.44	1.37	2.56
	Others	24	143	11.06	27.04	5.50	14.53
		53	**1293**	**100**	**100**	**100**	**100**

Table 5 (cont.)

	Family Name	NS	SD (n/ha)	RD (%)	RF (%)	RBA (%)	IVI (%)
Dry Forest	Verbenaceae	4	657	75.43	10.66	77.57	54.55
	Combretaceae	4	83	9.53	0.82	13.56	7.97
	Rhamnaceae	1	1	0.11	20.49	0.01	6.87
	Rubiaceae	2	9	1.03	17.21	0.24	6.16
	Dipterocarpaceae	2	3	0.34	9.02	0.22	3.19
	Flacourtiaceae	1	2	0.23	9.02	0.25	3.16
	Fabaceae	5	27	3.10	2.46	2.49	2.68
	Mimosaceae	5	17	1.95	1.64	1.57	1.72
	Sapindaceae	1	6	0.69	4.10	0.22	1.67
	Burseraceae	1	2	0.23	4.10	0.29	1.54
	Others	19	64	7.35	20.49	3.58	10.47
		45	**871**	**100**	**100**	**100**	**100**
Dry hill/ evergreen forest	Verbenaceae	1	263	32.71	13.25	41.96	29.31
	Myrsinaceae	1	160	19.90	8.43	11.43	13.25
	Bixaceae	1	63	7.84	9.04	4.09	6.99
	Euphorbiaceae	2	48	5.97	7.23	6.74	6.65
	Rosaceae	1	32	3.98	6.02	6.63	5.54
	Myrtaceae	1	22	2.74	8.43	4.65	5.27
	Rubiaceae	2	45	5.60	7.23	2.82	5.21
	Fabaceae	3	39	4.85	4.22	6.39	5.15
	Lauraceae	1	27	3.36	4.82	1.29	3.16
	Vitaceae	1	20	2.49	5.42	0.40	2.77
	Others	26	85	10.57	25.90	13.60	16.69
		40	**804**	**100**	**100**	**100**	**100**

*NS= number of species, SD = stand density, RD = relative density, RF = relative frequency, RBA = relative basal area

2. Tree species richness

Species richness is the basic component of diversity of any community (De, 2007). The Jackknife estimator for species richness of trees (DBH $\geq$ 5cm) showed that the DMDF occupied the greatest number of species (74 species) followed by the DDF (53 species), DF (45 species) and DHEF (40 species) (Table 6). Our findings of tree species are in the range of that found in mature tropical forest, 56-282 species ha^{-1} (DBH $\geq$ 10cm) (Phillips and Gentry, 1994). Our findings are also in line with other study (Gillespie, et al., 2013) in five tropical dry forests on Oceanic islands. Their study found 242 species ha^{-1} (DBH $\geq$ 2.5 cm) in New Caledonia, 112 species ha^{-1} in Fiji, 39 species ha^{-1} in the Mariannas, 24 speciesha^{-1} in the Marquesas and 96 species ha^{-1} in Hawaii. But our findings are lower than that found in the dry tropical forests of India, where 190 species ha^{-1} (DBH $\geq$ 5 cm) were recorded (Homeier, et al., 2010) and 105 species ha^{-1} (DBH $\geq$ 10cm) were found in the tropical seasonal forest in Khao Yao National Park in Thailand (Kitamura, et al., 2005). The difference may be due to the forest locations and tree size (i.e. DBH) recorded during forest inventory. We enumerated trees with only having above 5 cm DBH. The difference in terrain, gradient and slope direction causes differences in the soils, water and microclimate which causes differences in species adaptability (Suratman, 2012).

Table 6 Species richness and diversity in Popa Mountain Park

Parameter	DMDF	DDF	DF	DHEF
Species richness (Jackknife estimator)	74.00	53.61	45.54	40.75
Shannon-Wiener Function (H')	3.61	2.96	1.45	2.41
Simpson's diversity index (1-D)	0.96	0.92	0.50	0.84
Shannon evenness (j') (%)	83.95	74.62	38.08	65.36

*DMDF = dry mixed deciduous forest, DDF = dry dipterocarp forest, DF = dry forest, DHEF = dry hill/evergreen forest.

3. Tree species diversity

Tree diversity inclines provide important information about rarity and commonness of species in a community (Suratman, 2012). Shannon's diversity index (H') places more weight on the rare species while Simpson's diversity index (1-D) gives more weight to those species which occur more frequently (Lamprecht, 1989; Magurran Anne, E., 1988). In Popa Mountain Park, the DMDF occupied the highest value of Shannon-Wiener index and Simpson diversity index, 3.61 and 0.96, respectively. Both diversity indexes indicated that the species diversity of the DMDF is the highest among all forests in PMP (Table 6). The larger the value of H', the greater the species diversity and vice versa (Kessler, et al., 2005). The Shannon-Weiner diversity Index (H') value stood at 3.61 in DMDF, 2.96 in DDF, 1.45 in DF and 2.41 in DHEF, indicating that among forest types, DMDF was the most complex in species diversity whereas DF was the simplest community in term of species composition. This implies that elevation and aspect favors diversity and may be partly responsible for the diversity index obtained at Popa Mountain Park.

Our study findings are in line with the findings of Bhat (Bhat and Kaveriappa, 2009) in the six fresh water swamp forests of Karnataka in India, with H' of 2.53, 3.69, 2.46, 4.04, 3.25 and 4.90. Similarly, the northern forest-savana ecotone of Ghana (Attua and Pabi, 2013) obtained H' values of 3.02, 0.04 and 0.39 at near-forest ecotone, near-savana ecotone and mid-ecotone. However, when compared with the findings of Kumar (Kumar, et al., 2010) in three sites of tropical dry deciduous forest in western India, where H' values of 0.67 - 0.79 were recorded, the H' value of the DDF (2.96) in PMP is significantly higher. This indicates that diversity and richness in terms of species, and their distribution, is largely dictated by climate and ecological conditions, a view supported by Bello (Bello, Isah and Ahmad, 2013).

Species evenness (E) is a measure of equitability of spread. Values obtained were 0.83 in DMDF, 0.74 in DDF, 0.38 in DF and 0.65 in DHEF. The species in DMDF were more abundant, and the percentage of evenness j (%) was close to 1.0. Therefore, Shannon's evenness (j) shows that DMDF have highest species diversity and DF have the lowest species diversity in PMP. The slope direction influences tree species diversity at different altitudes (Changcheng, et al., 2007).

4. Floristic heterogeneity and similarity

An approximate indication of the homogeneity of a stand and of high diversity of tree species can be expressed by frequencies (Lamprecht, 1989). Species were assigned the frequency classes I, II, III, IV and V using their absolute frequencies 0%–20%, 20%–40%, 40%-60%, 60%-80% and 80%-100%, respectively. If high values are found in frequency classes I/II and low values in frequency classes IV/V, it would indicate a high degree of floristic heterogeneity. If high values were found in frequency classes IV/V and low values in I/II, it indicates constant or similar tree species composition in the area. Frequencies depend on the size of the subplots. The larger the subplots, the higher the number of species that will be found in the higher frequency class. Comparison of frequency diagrams is therefore possible when based on the areas with the same subplot sizes. As shown in Figure 4, frequency values in I/II classes are higher than in IV/V classes indicating that all forests in PMP have a high degree of floristic heterogeneity and high diversity of species.

The similarities in species composition between the forests are presented in Table 7. Similarity indices range from 0.0-1.0, corresponding to a 0%-100% similarity between any two plant communities. Coefficients of similarity of species composition (Sorensen's index) between the DMDF and DDF showed higher values. Their respective vegetation communities share 86% of the species between the DMDF and DDF. When the forest stands were paired and compared against each other, only DMDF and DDF showed a highly similar species composition. It was found that 46 species were common between the DMDF and DDF, 27 species between the DMDF and DF, 30 species between the DMDF and DHEF, 23 species between the DDF and TDF, 20 species between the DDF and DHEF and 11 species between the DF and DHEF. Likewise, high similarity values of the family were found in the DMDF and DDF.

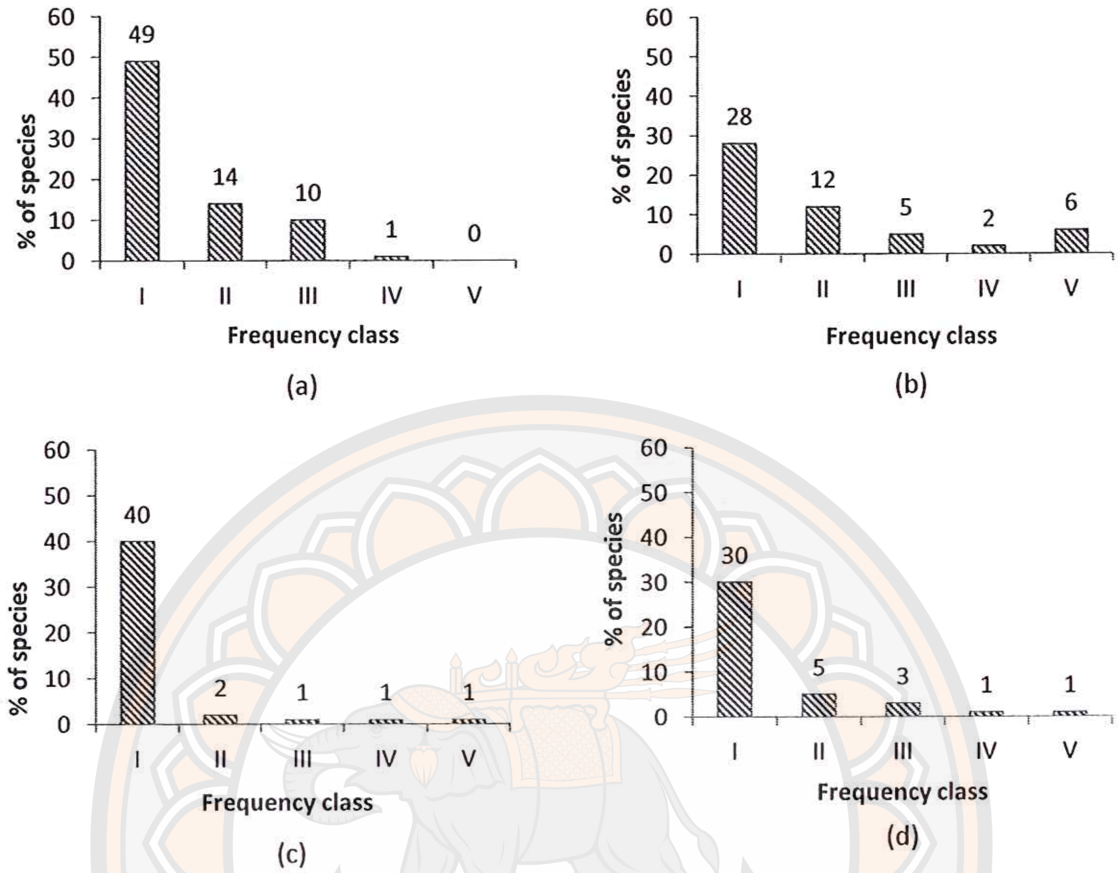

Figure 4 requency diagram of the dry mixed deciduous forest (a), dry dipterocarp forest (b), dry forest (c) and dry hill or evergreen forest (d)

Table 7 Floristic similarity of forest types in Popa Mountain Park

Forest stand	Families similarity		Species similarity	
	Sorencen's index (%)	Jaccard index (%)	Sorencen's index (%)	Jaccard index (%)
DMDF and DDF	86.67	79.79	64.57	67.41
DMDF and DF	64.29	59.55	45.38	54.49
DMDF and DHEF	77.42	77.73	52.63	50.54
DDF and DF	72.00	71.67	46.94	65.40
DDF and DHEF	64.29	77.67	43.01	57.74
DF and DHEF	53.85	75.87	25.88	53.44

Stand Structure

1. Characteristic of investigated forests in Popa Mountain Park

Based on the data from 4 ha sample plots in PMP , average stem density ranges from 804 tree ha^{-1} to 1293 tree ha^{-1} which were represented by 40 – 74 tree species belonging to 23 – 33 families for ≥ 5cm DBH threshold (Table 8).

In the DMDF, 1061 stems with DBH ≥ 5cm were recorded. All of the stem are lower than 70cm DBH. There are no large tree (DBH ≥70 cm as mentioned in Clark (Clark and Clark, 1996)) in the DMDF. The mean DBH of all stems was 17.50 cm and the largest stem was 68 cm belonging to *Terminalia crenulata.* All trees belonged to 33 families, 74 species.

The DDF had 1293 stems, which was the highest stems density in PMP. The total basal of the forest was 22.93 m^2 ha^{-1}. Although the stems density was high, all of the trees were lower than 55 cm DBH. All trees (DBH ≥ 5 cm) belonged to 27 families and 53 species.

The DF recorded 871 stems belonging to 45 species and 23 families. Total basal area was 26.87 m^2 ha^{-1}. The density of stems, was also much lower than that of other forest. Mean DBH of all stems was 13.34 cm and the largest stem was 68.5 cm belonging to *Terminalia oliveri.* There are 45 species belonging to 23 families.

The DHEF had 804 stems, which was the lowest stems density in PMP. The total basal of the forest was 47.80 m^2 ha^{-1}, which was the highest among forests in PMP. The large tree (DBH ≥ 70 cm) was found in the DHEF, accounting for 10 tree ha^{-1} (Table 8). There are 40 species belonging to 29 families.

Table 8 Characteristics of different forest types in Popa Mountain Park

Parameter	DMDF	DDF	DF	DHEF
Mean DBH (cm)	17.50	11.00	13.34	24.03
Mean Ht (m)	13.80	6.69	7.38	13.20
BA (m^2ha^{-1})	27.50	22.93	26.87	47.80
Vol (m^3ha^{-1})	214.69	126.80	220.33	610.71
Stand Density (trees ha^{-1})	1061	1293	871	804

Table 8 (cont.)

Parameter	DMDF	DDF	DF	DHEF
No. of families	33	27	23	29
No. of Species	74	53	45	40

*DMDF= dry mixed deciduous forest, DDF= dry dipterocarp forest, DF= dry forest, DHEF = dry hill/evergreen forest, DBH = diameter at breast height, Ht = height, BA = basal area, Vol = volume.

2. **Stand density**

Mean tree density (DBH≥5cm) of the various forest stands was DMDF, 1061 trees ha^{-1}, DDF, 1293 trees ha^{-1}, DF, 871 trees ha^{-1} and DHEF, 804 trees ha^{-1}. In all forests, ten highest species IVI contributed the bulk of tree density, and basal area. These 10 species contributed 81%, 74%, 91% and 88% of total stand density for forest DMDF, DDF, DF and DHEF, respectively. Tree abundance patterns varied among forest types. The abundance of two species in the DDF, one species in the DF and one species in the DHEF is pronounced. No single species is dominant in the DMDF.

Stand density are decreasing with increasing DBH class (Table 9). Many trees are found in the DBH range of less than 20 cm, 75% of total stand density for DMDF, 84% of total stand density for DDF, 72% of total stand density for DF and 58% of total stand density for DHEF. In DBH ≥ 60 cm, only one stem in the DMDF, 3 stems in the DF and 29 stems in the DHEF are found. But there are no trees in the DMDF, DDF and DF with DBH larger than 80 cm. There were nine stems in PMP with DBH ≥ 80 cm which was found only in the DHEF. Most of the forest in PMP could not reach mature stage probably due to resource collection from local communities.

The density of 804 – 1293 trees ha^{-1} for the diameter at breast height (DBH) threshold ≥ 5cm DBH obtain in our study is comparable to that of rainforests in Malaysia, Borneo (1091 trees ha^{-1}) (Small, et al., 2004). The tree density in the present study is lower than the dry tropical forest, India wherein an average of 1347 trees ha^{-1} was enumerated (Homeier, et al., 2010) for the same DBH threshold. But this finding is higher than the stand density of dry mixed deciduous forest, 560 trees ha^{-1} and evergreen forest, trees ha^{-1}, in Thailand (Terakunpisut, et al., 2007).

Table 9 Distribution of stem and its percentage in investigated forests in Popa Mountain Park

DBH class (cm)	DMDF		DDF		DF		DHEF	
	SD (trees ha^{-1})	SD (%)	SD (trees ha^{-1})	SD (%)	SD (trees ha^{-1})	SD (%)	SD (trees ha^{-1})	SD (%)
≥ 5 - 20	798	75.2	1093	84.5	634	72.8	469	58.3
> 20 - 40	245	23.1	197	15.2	200	23.0	241	30.0
> 40 - 60	17	1.6	3	0.2	34	3.9	65	8.1
> 60 - 80	1	0.1	0	0.0	3	0.3	20	2.5
> 80 - 100	0	0.0	0	0.0	0	0.0	9	1.1

3. Basal area and Volume

Although the stand density is lower in the DHEF, it was twice as voluminous, with a basal area of 47.80 m^2 ha^{-1}, than the DMDF (basal area 27.50 m^2 ha^{-1}), the DDF (basal area 22.93 m^2 ha^{-1}) and the DF (basal area 26.87 m^2 ha^{-1}). This is because many large trees were found in the DHEF and the mean DBH is the highest among other forest in PMP. The basal area contribution by 10 highest species IVI was 81%, 86%, 96% and 82% for forest DMDF, DDF, DF and DHEF, respectively.

The stand density is highest in small DBH, ≥ 5cm – 10cm, but basal area is highest in medium DBH class of 15.1-20 cm in the DMDF, 10.1-15 cm in the DDF, 15.1-20 cm in the DF and 25.1-30cm in the DHEF. The basal area range 22.93 m^2ha^{-1} to 47.83 m^2ha^{-1} for ≥ 5cm DBH threshold is well within the range of 17 – 40 m^2ha^{-1} reported for dry tropic (Murphy and Lugo, 1986) and four sites of Karnataka, India, 33.74 to 48.60 m^2ha^{-1} (Rai and Proctor, 1986). But higher than the range of dry evergreen forests in India 15.4 to 29.5 m^2ha^{-1} (Parthasarathy and Karthikeyan, 1997).

The standing volumes were 214.69 m^3 ha^{-1} in the DMDF, 126.81 m^3 ha^{-1} in the DDF, 220.33 m^3 ha^{-1} in the DF and 610.72 m^3 ha^{-1} in the DHEF. The mean volume of the DHEF was significantly different ($p>0.01$) among the forests in PMP.

However, the stand density was the lowest in the DHEF while the standing volume strongly suggests that the trees in the DHEF are more mature than other forests. In terms of species, *Shorea obtusa* made up the highest standing volume (29.48 m^3ha^{-1}) in the DMDF and (34.82 m^3ha^{-1}) in the DDF while *Tectona hamiltoniana* was the highest (160.97 m^3ha^{-1}) in the DF and *Vites canescen* (284.52 m^3ha^{-1}) in the DHEF.

4. Diameter distribution

Diameter frequency distributions provide a useful substitute for development trends of the stands (Lamprecht, 1989) and help to evaluate potential forest sustainability (Rubin, Manion and Faber-Langendoen, 2006). For all forest types in PMP, it was found that increasing tree size classes results in drastically decreased species richness (SR), density (D) and diversity (H') (Table 10). Shannon-Wiener diversity Index (H') showed that the smaller DBH size classes have the higher diversity. The study found that lower size classes; 5-10 cm and 10-15 cm, contributed more than 50% of the total tree density in the investigated forests. As well, the lowest size class, DBH 5-10 cm, possessed the highest species richness in all of the forests. In the 5-10cm DBH size class, we found 55 out of 74 species in the DMDF, 47 out of 53 species in the DDF, 32 out of 45 species in the DF and 20 out of 40 species in the DH. The higher numbers of species were found in the lower size class of all forest types.

The findings in this study indicate that, where tree density was generally higher in small DBH classes compared to large DBH classes, this is a secondary forest characteristic. In all the forest stands, the greater numbers of trees were observed in the lowest diameter class (5-10cm). This indicates that the density of smaller trees in a stand is sufficient to replace the current population of larger trees. The diameter distribution of the trees followed the inverse J-shape pattern (Figure 5). This pattern indicated that stands are developing and regeneration is occurring in the forest indicating a high potential for species substitution when mature trees in the dominant species die.

The diameter classes 15.1-20 cm, 10.1-15 cm, 15.1-20 cm and 25.1-30cm occupied the largest basal area per ha in DMDF, DDF, DF and DHEF, respectively (Figure 6). No stems in diameter classes 55.1cm were found in the DDF and no 70.1 cm diameter stems were found in the DMDF and DF. Large diameter size classes of tree were rarely found in most of the forests in PMP. This might be due to the over-cutting of trees (for firewood) from the surrounding area. In the DHEF, large trees with stem diameters up to the 95.1 – 100cm DBH class. It is therefore probable that less human disturbance in the DHEF occurred above 1000m, being less accessible.

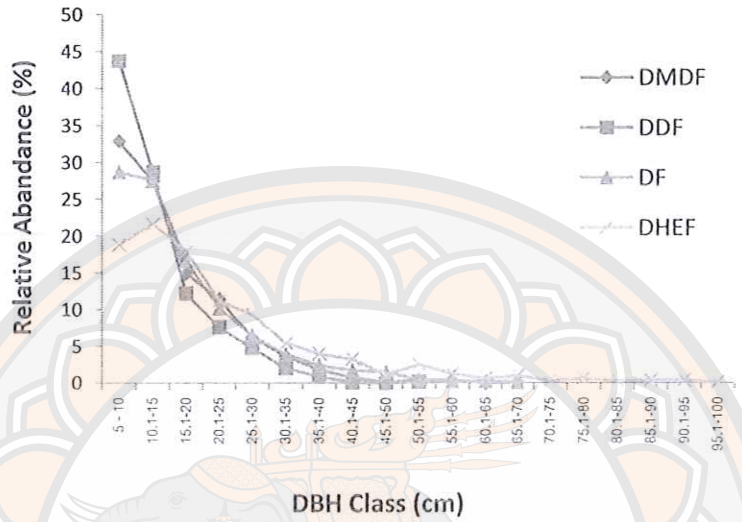

Figure 5 Relative abundance in relation to diameter classes in the investigated forest in Popa Mountain Park

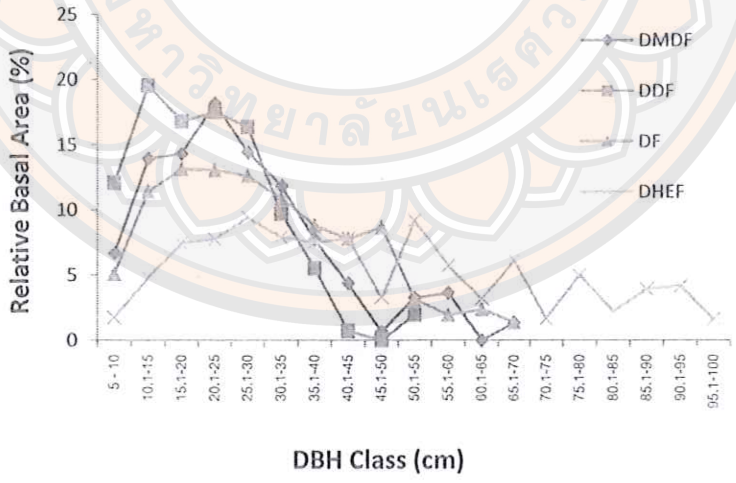

Figure 6 Relative basal area in relation to diameter classes in the investigated forests

Table 10 Frequency of species distributed DBH class and the Shannon-Wiener (H') of investigated forests in Popa Mountain Park

DBH class (cm)	DMDF			DDF			DF			DHEF		
	SR	D	H'	SR	D	H'	SR	D	H'	SR	D	H'
5 - 10	55	348	0.37	47	565	0.36	32	250	0.36	20	151	0.31
10.1-15	52	292	0.36	40	371	0.36	22	240	0.36	19	174	0.33
15.1-20	40	158	0.28	20	157	0.26	18	144	0.30	21	144	0.31
20.1-25	44	121	0.25	15	98	0.20	8	89	0.23	16	90	0.25
25.1-30	29	66	0.17	10	62	0.15	9	56	0.18	16	75	0.22
30.1-35	20	39	0.12	4	26	0.08	6	34	0.13	11	44	0.16
35.1-40	13	19	0.07	5	11	0.04	3	21	0.09	10	32	0.13
40.1-45	7	8	0.04	1	1	0.01	3	15	0.07	11	26	0.11
45.1-50	1	1	0.01	0	0	0.00	2	13	0.06	4	9	0.05
50.1-55	4	4	0.02	2	2	0.01	2	4	0.02	6	20	0.09
55.1-60	2	4	0.02	-	-	-	1	2	0.01	5	10	0.05
60.1-65	-	-	-	-	-	-	1	2	0.01	3	5	0.03
65.1-70	1	1	0.01	-	-	-	1	1	0.01	3	8	0.05
70.1-75	-	-	-	-	-	-	-	-	-	2	2	0.01
75.1-80	-	-	-	-	-	-	-	-	-	2	5	0.03
80.1-85	-	-	-	-	-	-	-	-	-	2	2	0.01
85.1-90	-	-	-	-	-	-	-	-	-	1	3	0.02
90.1-95	-	-	-	-	-	-	-	-	-	2	3	0.02
95.1-100	-	-	-	-	-	-	-	-	-	1	1	0.01

*SR = species richness, D = density, H’ = Shannon-Wiener diversity index

5. Species-area curve

Figure 7 shows species-area-curves. It can be seen the DMDF has a greater richness of trees of stem diameter ≥ 5 cm than the other forest types while the DDF has intermediate richness levels. However, the DF and DHEF has similar species richness but lower than the DMDF and DDF. In this study, the species-area curves were drawn based on trees with height ≥ 1.3m and DBH ≥ 5cm, and the total survey areas were one hectare in each forest. Lamprecht (Lamprecht, 1989) suggested that a stock with DBH > 10cm is generally adequate to draw species-area curves. He also suggested that the species-area curve is the best criterion for determining a minimal plot size needed to survey a community adequately. The pattern of the DMDF curve increases significantly up to the point of 0.40 ha, after which the increase is much more gradual with the increment of the number of species remaining below 10%. Similarly, in the DDF new species were found in areas up 0.48 ha after which the increment remained below 10% from 0.48 ha up. Likewise, the pattern of DF curve and DHEF curve went up gradually and then became constant from 0.64 ha and 0.48, respectively. The minimum representative area would be reached if the number of species increases by less than 10% when the expansion of sampling area is 10% (Cain and Castro, 1959). According to the species-area-curve, one ha sample size represented the minimum area for each forest type since there were only minimal numbers of new species discovered.

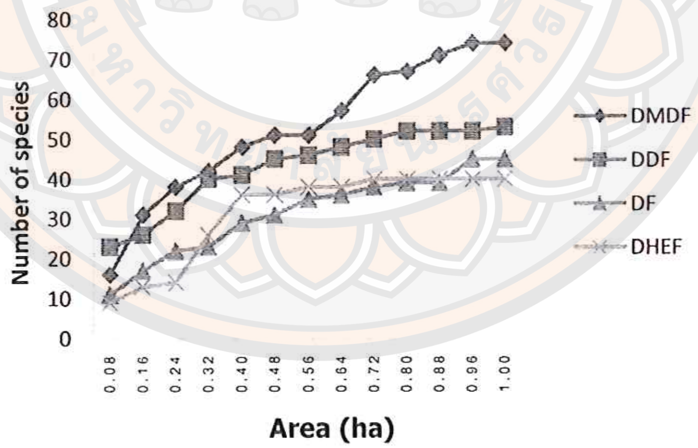

Figure 7 Species area curve for all trees (DBH ≥ 5cm) in Popa Mountain Park

Carbon stock of different forests in Popa Mountain Park

Carbon stock in each pools

1. Aboveground Carbon

1.1 Carbon stock in dominant species

In the DMDF, the high stand density (103 tree ha^{-1}) and aboveground carbon (20.22 $tCha^{-1}$) was found in *Shorea obtusa* while *Dipterocarpus tuberculatus* and *Terminalia crenulata* occupied 8.52 $tCha^{-1}$ and 8.43 $tCha^{-1}$. But no single species was highly dominant in the DMDF. All of the species possess less than 10 % IVI value. The high IVI value are found in *Shorea obtus*a, 8.58% IVI, *Croton roburghianus* (7.34% IVI), *Pittosporum napaulensis* (4.83% IVI), *Bixa orellanel* (4.36% IVI) and *Terminalia crenulata* (4.26% IVI). After *Shorea obtuse*, stand density was higher in *Croton roburghianus* (100 $treeha^{-1}$). Carbon stock was higher than *Croton roburghianus* (6.14 t Cha^{-1}) in both *Terminalia crenulata* (8.43 $tCha^{-1}$) and *Dipterocarpus tuberculatus* (8.52 $tCha^{-1}$) (Table 11).

In the DF, the dominant species is *Tectona hamiltoniana* which occupied the highest IVI value (54.25%) and highest carbon stock (93.68 $tCha^{-1}$). Followed by *Terminalia oliveri* with an IVI value of 11.53% and carbon stock 21.70 $tCha^{-1}$. *Dalbergia oliveri* store 3.65 $tCha^{-1}$ and *Albizzia chinesis* stored 2.79 $tCha^{-1}$. Though mean DBH of *Albizzia chinesis* (32.2 cm) is larger than *Dalbergia oliveri* (17.3 cm), the stand density of *Albizzia chinesis* (3 $treeha^{-1}$) is lower than *Dalbergia oliveri* (18 tree ha^{-1}). The stand density influences on the carbon storage of the tree (Perea Cordero and Kanninen, 2003) as well as the forest. But the third highest stand density, 42 tree ha^{-1}, was found in *Tectona grandis* which mean DBH size 11.1 cm and WD 0.70 Mgm^{-3}. In term of DBH and wood density, *Tectona grandis* was lower than *Dalbergia oliveri* (DBH 17.3 cm, WD 1.02 Mgm^{-3}) and *Albizzia chinesis* (DBH 32.2 cm, WD 0.82 Mgm^{-3}). Therefore, the results indicat that tree size and wood density effect the carbon storage of forests.

In the DDF, the highest AGC was found in *Shorea obtusa*, 28.30 $tCha^{-1}$, followed by *Dipterocarpus tuberculatus*, 21.29 $tCha^{-1}$ and *Shorea siamensis* 9.64 $tCha^{-1}$. The pattern of AGC allocation is similar to the important value index (IVI%). *Shorea obtusa* occupied 34.81 %IVI, *Dipterocarpus tuberculatus* occupied 34.20% IVI and *Shorea siamensis* with 12.38 % IVI. Almost half of the carbon stock in the DDF was observed in two dominant species. *Shorea obtusa* and *Dipterocarpus tuberculatus* occupied carbon stock amounted to 91.28 $tCha^{-1}$ in the DDF. Because of hight stand density, large DBH size and high wood density, the high aboveground carbon was observed in *Shorea obtusa* and *Dipterocarpus tuberculatus*.

The most dominant species in the DHEF are *Vitex canescens* (29.13% IVI), *Rapanea af. Nerrifolia* (13.14 % IVI) and *Bixa orellana* (7.06% IVI). These three species occupies 49.33 % IVI of total species. In terms of carbon stock, *Vitex Canescens* accumulate the highest carbon stock, 82.21 tC ha^{-1}, followed by *Eriobotrya bengalensis* 22.76 tC ha^{-1} and *Rapanea af Nerrifolia* 19.52 tC ha^{-1}. Therefore, in the DHEF *Vitex canescens* is the dominant species and occupied the highest IVI value and the highest carbon stock. Though *Rapanea af Nerrifolia* is the second highest IVI, *Eriobotrya bengalensis,* the fourth highest IVI, occupied more carbon stock. It may be due to mean DBH and the wood density effect on the carbon storage capacity. The mean DBH of *Eriobotrya bengalensis* is 27.84 cm and WD is 0.73 Mgm^{-3} while the mean DBH of *Rapanea af Nerrifolia* is 18.63 am and WD is 0.71 Mgm^{-3}. The accumulation of biomass and carbon related to different wood density (Elias and Potvin, 2003) and tree diameter (Brown, S., 1997). Biomass and carbon per tree increase geometrically with increasing diameter (Brown, S., 1997).

Table 11 Ten highest carbon storage species in the investigated forests in Popa Mountain Park

Forest	Scientific Name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C
Dry mixed deciduous forest	*Shorea obtusa Wall.*	20.11	103	3.79	29.49	8.58	20.21
	Dipterocarpus tuberculatus Roxb.	29.41	17	1.33	16.14	2.70	8.52
	Terminalia crenulata (Heyne) Roth	20.85	41	1.83	14.07	4.26	8.43
	Croton roxburghianus N.P.Balakr	14.31	100	1.91	11.96	7.34	6.14
	Anogeissus acuminata Wall.	24.33	9	0.63	9.71	1.42	6.05
	Shorea siamensis (Kurz) Miq.	18.17	26	0.86	7.06	2.52	4.64
	Senna siamea (Lam.) Irwin & Barneby	26.67	12	0.71	8.35	1.52	4.16
	Flacourtia cataphracta Roxb.	14.71	45	0.97	5.71	3.62	3.77
	Litsaea glutinosa (Lour) C. B. Cl.	15.67	43	0.95	6.15	3.62	2.66
	Pittosporum napaulensis (DG) Rehder Wilson	11.68	76	0.93	5.15	4.83	2.68
	Others		589	13.61	100.91	59.59	49.39
	Total		**1061**	**27.52**	**214.69**	**100**	**116.65**

Table 11 (cont.)

Forest	Scientific Name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C
Dry dipterocarp forest	*Shorea obtusa* Wall.	16.00	251	6.63	34.82	0.25	28.30
	Dipterocarpus tuberculatus Roxb.	17.50	172	5.19	34.21	0.12	21.30
	Shorea siamensis (Kurz) Miq.	13.90	121	2.16	12.38	0.12	9.64
	Terminalia crenulata (Heyne) Roth	12.50	93	1.42	6.86	0.24	4.92
	Dalbergia oliveri Gamble	11.50	80	0.97	5.93	0.12	5.23
	Diospyros burmanica Kurz	11.70	49	0.59	2.93	0.15	4.84
	Premna pyramidata Wall.	15.70	47	1.05	6.33	0.12	3.89
	Buchanania lanzan Spreng.	14.60	49	0.99	6.03	1.02	2.04
	Xylia xylocarpa (Roxb.) Toub.	12.40	21	0.29	1.87	4.68	1.38
	Strychnos nux-blanda A.W. Hill	13.90	17	0.34	1.55	2.17	1.29
	Others		393	3.29	13.91	91.00	8.47
	Total		**1293**	**22.93**	**126.81**	**100.00**	**91.28**

Table 11 (cont.)

Forest	Scientific Name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C
Dry Forest	*Tectona hamiltoniana* Wall.	18.00	610	0.01	0.05	0.27	93.68
	Terminalia oliveri Brandis	19.90	75	0.02	0.19	0.27	21.70
	Dalbergia oliveri Gamble	17.30	18	0.01	0.06	0.70	3.65
	Albizzia chinensis (Osbeck) Merr.	32.20	3	0.13	1.57	3.44	1.63
	Tectona grandis L.f	11.10	42	0.01	0.01	0.28	1.25
	Diospyros burmanica Kurz	11.70	10	0.02	0.11	0.59	0.70
	Lannea coromandelica (Houtt). Merr.	12.90	18	0.01	0.03	0.40	0.53
	Acacia catechu Willd.	14.70	6	0.01	0.03	54.25	0.50
	Boscia variabilis Collett & Hemsl.	18.50	5	0.06	0.35	1.11	0.47
	Miliusa velutina Hook.f. & Thoms	32.00	1	0.08	0.88	0.38	0.39
	Others		**83**	**0.51**	**2.54**	**38.31**	**2.37**
	Total		**871**	**0.86**	**5.83**	**100.00**	**126.87**

Table 11 (cont.)

Forest	Scientific Name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C
Dry hill or evergreen forest	*Vitex canescens* Kurz	24.93	263	20.07	284.52	29.13	82.21
	Eriobotrya bengalensis (Roxb.) Hook. f.	27.84	32	3.17	57.19	5.46	22.76
	Rapanea af. Neriifolia (Seib & Zucc) Mez.	18.63	160	5.47	50.42	13.14	19.52
	Syzygium cumini (L.) Skeels.	32.36	22	2.23	24.49	5.16	10.68
	Flacourtia cataphracta Roxb.	52.80	5	1.14	20.36	1.58	9.99
	Litsaea glutino (Lour)C.B.Cl.	30.25	30	2.62	25.79	4.23	8.30
	Holarrhena pubescens Wall. ex G. Don	55.00	3	0.79	14.88	1.25	5.76
	Wendlandia tinctoria DC.	16.74	43	1.31	13.11	4.82	5.36
	Sapium baccatum Roxb.	74.00	4	1.74	24.22	2.15	4.75
	Croton roxburghianus N.P.Balakr	19.75	44	1.48	11.18	4.40	4.27
	Others		**198**	**7.80**	**84.56**	**28.67**	**26.62**
	Total		804	47.83	610.72	99.99	200.22

SD = Stand Density, BA = Basal Area, Vol = Volume, IVI = Important Value Index, AG-C = Aboveground Carbon

1.2 Carbon allocation in diameter class

Aboveground carbon accumulation was highest in the DHEF (200.22 ± 31.23 $tCha^{-1}$) followed by DF (126.87 ± 12.06 $tCha^{-1}$), DMDF (116.64 ± 11.34 $tCha^{-1}$) and DDF (91.28 ± 5.0 $tCha^{-1}$). While mean tree density (DBH ≥ 5cm) of the DHEF, DF, DMDF and DDF amount to 804 trees ha^{-1}, 871 trees ha^{-1}, 1061 trees ha^{-1} and 1293 trees ha^{-1}, respectively. Although the DMDF and DDF have high numbers of trees and species, most of the trees were smaller than 20 cm (Figure 8). Therefore, the DMDF and DDF have lower individual volume and carbon storage than DHEF and DF. Moreover, it may be that the mean DBH of the forest also has an effect on forest carbon storage. This is because the mean DBH of the DHEF (24.03 cm) is larger than DMDF (17.50 cm), DDF (11.00 cm) and DF (13.34 cm). The trees in DHEF are larger than the other forests in PMP. So, the high carbon was accumulated in DHDF because the tree biomass and carbon storage increased geometrically with increasing diameter (Brown, S., 1997).

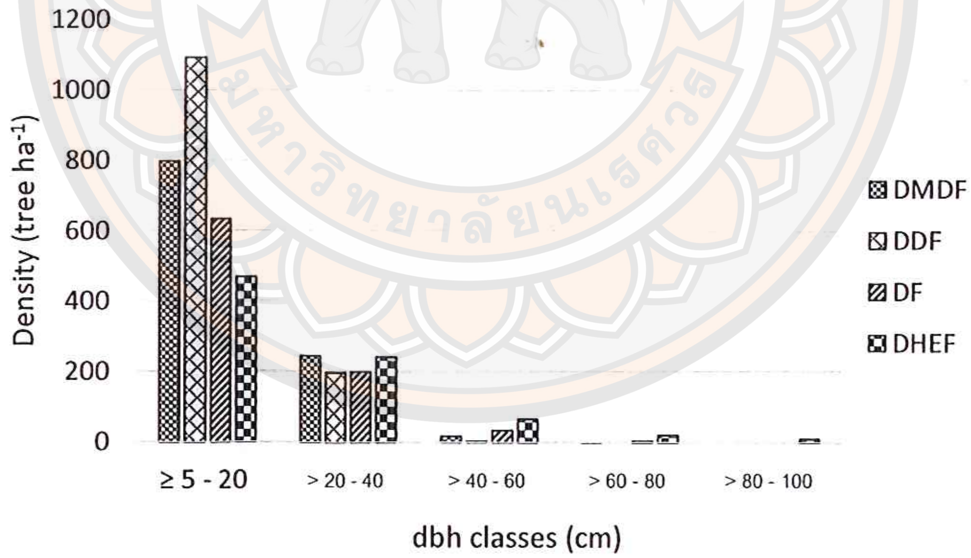

Figure 8 Tree Density in different size classes at DMDF, DDF, DF and DHEF

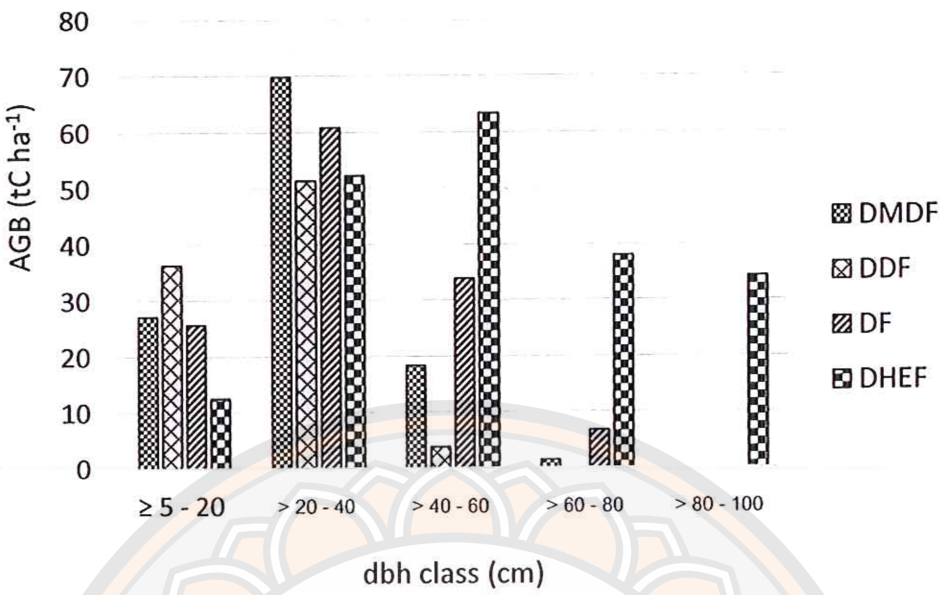

Figure 9 Carbon allocation in different DBH classes of DMDF, DDF, DF and DHF

The result shows opposite relationship between carbon stock and tree size class (Figure 8 and Figure 9). The most aboveground carbon accumulation was found in the tree size class at ≥ 20 – 40 cm in the DMDF, DDF and DF while the highest aboveground carbon was found at ≥ 40 – 60 cm size class in the DHEF (Figure 9). This is because these tree size classes had the highest stem volume and basal area. The high stand density was found in small DBH classes and a few trees in the large DBH class. In the DMDF, DDF and DF, there were no trees in the largest DBH class >80-100cm, for this study. Only a few trees occurred starting from the >40-60 DBH class. The largest trees were found only in the DHEF. Therefore this study found that high stand density with small trees (>5 -10cm) do not have much effect on the carbon stock of the forest if compared to low stand density with large trees. It may be due to the fact that carbon accumulation is related to DBH size. It supports the statement of Brown (Brown, S., 1997), he mentioned that the accumulation of the biomass and carbon per tee is geometrically related with DBH size.

Aboveground carbon (AG-C) increment was a steadily decreasing function of diameter, from a maximum of around 70 Mg ha^{-1} year^{-1} in ≥ 20 – 40 cm trees (Figure 9). Most of the AG-C increment was in the diameter class ≥ 5 – 40 cm and as much as 30% was in the ≥ 5 – 20 cm class. AGB was more evenly distributed, with a median in ≥ 20 – 40 cm trees. AGB change was positive in the ≥ 5 – 40 cm diameter class but negative in the ≥ 40 – 100 cm class. In the DHEF, aboveground carbon is well allocated in all DBH classes.

The percentage data of tree density and aboveground carbon were presented in Table 12 and showed the similar pattern of tree density and aboveground cabon in each forest type. In all forests, the largest number of stem were found in lowest DBH class ≥5 – 20 cm, 75.2 % in the DMDF, 84.5 % in the DDF, 72.8% in the DF and 58.3 % in the DHEF. But the highest carbon storage was found in the >20 – 40 cm in the DMDF, DDF, DF and >40 – 60 cm in the DHF. It may be due to high basal area and volume in the >20 – 40 cm class for the DMDF, DDF, DF and in the >40 – 60 class for the DHEF.

Table 12 A Comparison of the percentage of tree density and carbon sequestration potential in each size class in the different forest types

Size class (cm)	DMDF		DDF		DF		DHEF	
	Density (%)	AG-C storage (%)	Density (%)	AG-C storage (%)	Density (%)	AG-C storage (%)	Density (%)	AG-C storage e (%)
≥ 5 - 20	75.2	23.3	84.5	39.7	72.8	20.2	58.3	6.2
> 20 - 40	23.1	59.8	15.2	56.2	23.0	47.9	30.0	26.1
> 40 - 60	1.6	15.7	0.2	4.0	3.9	26.7	8.1	31.6
> 60 - 80	0.1	1.2	0.0	0.0	0.3	5.2	2.5	19.0
> 80 - 100	0.0	0.0	0.0	0.0	0.0	0.0	1.1	17.1

*DMDF = dry mixed deciduous forest, DDF = dry dipterocarp forest, DF = dry forest, DHF = dry hill/ evergreen forest

In most of the forests in PMP, more than 50% of carbon was accumulated in the tree DBH ≤ 40 cm (≥ 5 – 40 cm) , 83.10% in the DMDF, 95.9% in the DDF and 68.1% in the DF. But in the DHEF, 67.7% of total carbon was accumulated in the DBH > 40 cm class (> 40 -100cm). Therefore DHEF is more mature than other forests in PMP. In DDF, 99.8% of stand density or 95.9% of total carbon was found in the trees ≥5-40cm. It may be due to human disturbance or edge effect on the DDF since the DDF is close to park circular road and the park boundary and easily accessible by local communities. This finding was comparable with Htun's (2013) statement. He stated that more forest cover loss was found near the road and park boundary. The DDF have the highest stand density among forests in PMP with the lowest mean DBH and the lowest carbon stock.

1.3 Carbon storage in different forest types

The aboveground carbon storage of the four natural forests in PMP ranging from 91.28 $tCha^{-1}$ in DDF to 200.22 $tCha^{-1}$ DHEF. This finding was in line with carbon sequestration in Southeast Asia including India, Thailand, Cambodia, Malaysia and Indonesia ranging from 17.5 $tCha^{-1}$ or less in severely degraded tropical dry forest to almost 350 $tCha^{-1}$ in relatively undisturbed mature tropical rain forest (Flint and Richards, 1994). Moreover these findings are compatible with the carbon storage in some major forest types, India ranging from 59.0 $tCha^{-1}$ to 245.0 $tCha^{-1}$ (Sharma, et al., 2010). But the carbon storage range in this study was lower than the secondary forest carbon stock in the Philippines ranging from 118 to 306 $tCha^{-1}$ (Lasco and Pulhin, 2003). The accumulation of carbon may be related to stand density (S. Brown, 1997), wood density (Kenzo, et al., 2009) and growth pattern of fast and slow growing species.

In terms of the carbon stock in each forest, the findings of aboveground carbon in dry dipterocarp forest (DDF), 91.28 $tCha^{-1}$, was in the range of the carbon storage in dry dipterocarp forest, Vietnam, 85.0 – 138.0 $tCha^{-1}$ (Con, et al., 2013) and in dipterocarp forest, Philippine ranging from 90 - 184 $tCha^{-1}$ (Lasco and Pulhin, 2003). But this finding was lower than natural dipterocarp forest in Philippine, 119.4 $tCha^{-1}$ (Lasco and Pulhin, 2009).

The aboveground carbon storage of dry mixed deciduous forest (DMDF) (116.64 ± 11.34 $tCha^{-1}$) was higher than that in the mixed deciduous forest in Thailand, 60.06 $tCha^{-1}$ (Petsri and Pumijumnong, 2007). This is because stand density of the mixed deciduous forest in Thailand (644.72 trees ha^{-1}) was lower than this study (1061 trees ha^{-1}). But carbon stock in the DMDF of this study was lower than deciduous forest in AlaungdawKathapa National Park, Myanmar (227.7 $tCha^{-1}$). This is because the basal area of the deciduous forest in AlaungdawKathapa National Park (60.0 m^2ha^{-1}) was larger than the DMDF in Popa Mountain Park (27.5 m^2ha^{-1}).

Carbon storage of Dry hill/evergreen forest (DHEF) in this study, 200.22 $tCha^{-1}$, was higher than dry evergreen forest in Thong Phum national forest, Thailand,137.73 $tCha^{-1}$, and the natural semi evergreen forest in India, aboveground biomass 323.8 ton ha^{-1} (carbon stock 161.9 tC ha^{-1}) (Baishya, Barik and Upadhaya, 2009). It may be due to the fact that tree density, DBH size and dominant species are different in different locations. The carbon storage was highest in the 60-80 cm classes in the natural forest (Baishya, et al., 2009).

Tropical dry forest lands in the PMP have a wide range of carbon stocks. The highest stocks can be found in the DHEF (200.22 tC ha^{-1}) while the lowest are in DDF (91.28 tC ha^{-1}). This finding was in line with aboveground carbon in different forest types ranged between 7.81 tC ha^{-1} to 298.57 $tCha^{-1}$ (biomass 15.61 to 597.13 t ha^{-1}) from open scrub to evergreen forest in India (Mohanraj, et al., 2011). This may be different soil type, climate, disturbance regime, succession status, topography and human impacts. Furthermore, the aboveground carbon stock will be affected by the diameter class distribution throughout the forest (Terakunpisut, et al., 2007). Moreover, carbon stock may be affected by species composition in each forest type, especially the wood density of tree species (Kenzo, et al., 2009).

2. Belowground carbon (Root Carbon)

Estimates of belowground biomass or carbon are fundamental to understanding carbon and the biogeochemical dynamics of forest ecosystems (Cairns, et al., 1997). Belowground carbon was calculated based on aboveground carbon by using Cairn equation (Cairns, et al., 1997). Bray (Bray, 1963) and Cairn (Cairns, et al., 1997) suggested that biomass allocation to roots can be estimated based on aboveground allometries. The highest belowground carbon is allocated in the DHEF, 36.45±3.80 $tCha^{-1}$, followed by the DMDF, 18.5±1.3 $tCha^{-1}$, DF, 18.55±1.54 $tCha^{-1}$ and DDF, 14.67±0.68 $tCha^{-1}$ (Figure 10).

In DHEF, 18 % of aboveground carbon was allocated in the root. Carbon allocation in the belowground (root) was 15.87% of aboveground carbon in DMDF, 16.07% of aboveground carbon in DDF and 14.62 % of aboveground carbon in DF.

The root to shoot ration obtained in the present study was within the range of 9% to 33% for forest and woodland (Coomes and Grubb, 2000). And the ratio was also close to those reported by Birdsey (Birdsey, 1992) for the hardwood forest in the

United States ranging from 18% to 24% R/S ratio. As well, this study's findings (R/S ratio range 14.62% - 18%) were comparable to other estimates ranging from 13%-26% for the forest in amazon forest (Houghton, R. A., et al., 2001). Mean root to shoot ratio in PMP, 16% (0.16), was close to the mean root to shoot ratio in Malaysia, 18% (0.18) (Niiyama, et al., 2010). It may be due to the available soil moisture being strongly correlated with root biomass allocation, with water stress causing greater biomass allocation to roots (Murphy and Lugo, 1986; Sanford and Cuevas, 1996).

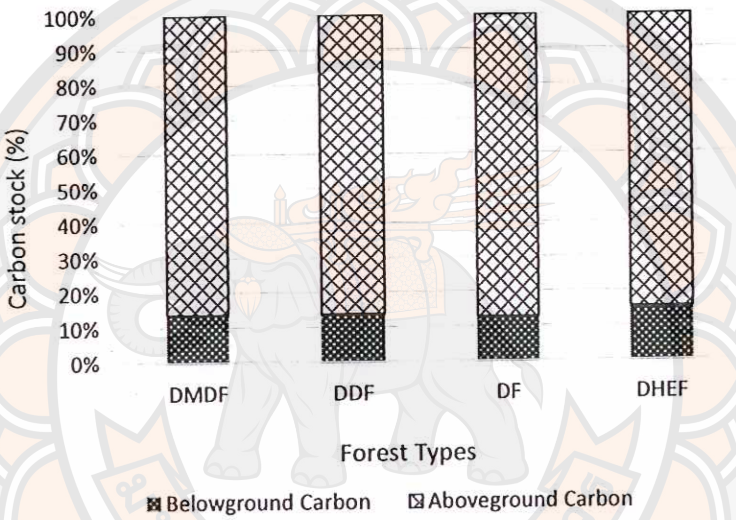

Figure 10 Aboveground carbon (AG-C) and belowground carbon (BG-C) allocations in DMDF, DDF, DF and DEF

3. Litter Carbon

Litter is defined as all detached and dead leaves, flowers, fruits, seeds, bark fragments, and deadwood less than 2.5 cm in diameter (Jaramilo, 2003). Based on 100 sub-sample plots in each forest, it was found that litter layer carbon in the DMDF, 16.6 ± 2.8 t Cha^{-1}, was higher than in other forests in PMP. The litter layer carbon stock was amount to 8.62±1.18 $tCha^{-1}$ in the DDF, 2.15±0.77 $tCha^{-1}$ in the DF and 10.77±1.34 $tCha^{-1}$ in the DHEF (Table 14). The carbon accumulation in the DMDF litter layer was significantly higher than in the DDF and the DF ($P<0.05$). However, litter layer carbon in the DHEF was not significantly different to the DMDF and DDF. Among the four forest types, the litter carbon in the DF was lowest in PMP.

The DF litter carbon accumulation was significantly different to litter carbon of the DMDF, DDF and DHF (P<0.05).

The DMDF was the highest followed by the DHEF, however, there is no significant differences between these two forests. It may be due to high species diversity in the DMDF and no single species dominance, thus, the rate of litter production might be different among different tree species. Likewise, the DHEF was found in high elevation and have many large trees. It might be that less light penetration to forest floor stimulates decomposition. Therefore, a lot of litter is accumulated on the forest floor due to less decomposition in the DHEF.

The carbon accumulation of litter in the DF was the lowest among all forest in PMP. It might be the DF is affected by annual fire, because the dominant species in the DF was *Tectona hanatoniani* (more than 50% of total stand density). The leaves of *Tectona hanatoniani* are large and the decomposition rate is slower than for small leaves, thus, they compile on the forest floor. Because of the strong heat in the DF area, it is easy for there to be forest fire. When annual fires happen, most of the area of the DF litter layer are destroyed. Therefore, the DF accumulates the lowest amount of carbon in the litter layer while the DMDF has the highest.

In terms of percentage, litter layer occupied 3.14% of total aboveground carbon (aboveground, Litter, deadwood) in the DMDF, 1.79% in the DDF, 0.53% in the DF and 1.21% in the DHEF (Table 13).

Table 13 Litter Carbon in DMDF, DDF, DF and DHEF

	DMDF	DDF	DF	DHEF
Mean ($tCha^{-1}$)	16.58	8.16	2.15	10.77
Percentage of total carbon (aboveground, Litter, deadwood) **in the respective forests**	3.14	1.79	0.53	1.21

4. Deadwood Carbon

Deadwood is an important store of aboveground biomass, carbon and nutrients (Turner, et al., 1995). Full enumeration (>10cm diameter of standing and fallen deadwood) was done within the 20m x 20m plot and these measurements converted to per-hectare C-stocks. Large (>10 cm diameter) pieces of standing dead trees, stumps and fallen deadwood (coarse woody debris, CWD) comprise a significant fraction of the total stocks (Baker, et al., 2007).

Based on the 25 sample plots in each forest, the trend of deadwood carbon storage in PMP is DDF > DMDF > DHEF > DF, 47.46 ± 7.52 $tCha^{-1}$, 28.3 ± 5.3 $tCha^{-1}$, 27.21 ± 6.23 $tCha^{-1}$ and 19.15 ± 5.62 $tCha^{-1}$, respectively. Deadwood carbon stock in this study was close to the deadwood of the watershed area in New Zealand, 30.2 ± 27.6$tCha^{-1}$, (Staley, 2010) and similar with South Island forests in New Zealand, 29.0 $tCha^{-1}$ (Coomes, et al., 2002) and indigenous forest in New Zealand, 27.00 6$tCha^{-1}$ (S. J. Richardson, et al., 2009). It is because stand age and disturbance history strongly affect the amount of coarse wood debris in a forest (Harmon, et al., 1986). Also, variation among forest types is determined by ecological factors that include rates of tree mortality and the effects of natural disturbance (Carmona, et al., 2002; Harmon, Franklin, et al., 1986), management history and depletion of the live tree biomass pool (Jönsson and Jonsson, 2007) and climate (Kennedy, Spies and Gregory, 2008). This finding was close with CWD carbon, 25.4 $tCha^{-1}$, in the tropical rain forest at la Selva (Clark, et al., 2002).

In PMP, the range of deadwood carbon was very wide from 19.15 $tCha^{-1}$ to 47.46 Cha^{-1}. This finding was in line with Clark (Clark, et al., 2002) who mentioned that the estimates of CWD stocks in tropical forest were from 0 to > 60 tha^{-1} in term of biomass, 0 to > 30 $tCha^{-1}$ in term of carbon. In addition, these findings were comparable to other estimates from the forest in Oregon (Nonaka, et al., 2007) ranging from 50 – 200 tha^{-1} in biomass, 25 – 100 $tCha^{-1}$. This finding was within the range of 18-413 tha^{-1} in biomass, 9-206.5 $tCha^{-1}$ in carbon, with a recently disturbed and old growth forest having the largest value (Carmona, et al., 2002). Dead wood patterns and dynamics vary with biophysical factors, disturbance history and management practices (Kennedy, et al., 2008). Microclimate conditions also influence decomposition rates of dead wood (Harmon, et al., 1986). The amount of deadwood in forests varies greatly around the world most range from 30 to 200 Mg ha^{-1} (Baker, et al., 2007; Carmona, et al., 2002; Harmon, et al., 1986; Jönsson and Jonsson, 2007; Tyrrell and Crow, 1994).

The proportion of total aboveground carbon (AGB, Litter and CWD) as deadwood varied from 11% to 32% across all forest. These results were higher than Houghton who found that deadwood biomass averaged 9% of aboveground biomass (range 1% - 17%) (R. A. Houghton, et al., 2001) and 10.4 % in Nascimento's study (Nascimento and Laurance, 2002). Jaramillo reported that deadwood (standing + downed combined) comprised 27% - 29% of aboveground in tropical dry forest (Jaramillo, et al., 2003) which was close to this study's result. But lower than Clark's findings in the fallen and standing CWD of the tropical rain forest at La Selva, 3.3% of ABG live woody biomass or carbon. This is because CWD carbon allocation was affected by rising global temperature (Cochrane, 1999) and fire frequency (Clark, et al., 2002). CWD pool is more labile than the live wood pool (Clark, et al., 2002) and the magnitude of the CWD pool would vary as a function of the relationship between inputs and outputs (Clark, et al., 2002).

5. **Soil Organic Carbon**

According to IPCC guideline, SOC was estimated to 100 cm depth in this study and matched with others studies (Gallali, Brahim and Bernoux, 2010; Han, et al., 2010; Lettens, et al., 2006; Sleutel, et al., 2003). Likewise, calculations of global SOC traditionally report results to a 1 m depth (Batjes, 1996; Eswaran, et al., 1993). Arrouays and Pel issier stated that extrapolating to a depth greater than 1m would not significantly change estimation of total carbon amounts (Arrouays and Pelissier, 1994).

In PMP, the total soil organic carbon (up to 100 cm) accumulated by the DMDF is 348.40 ± 32.5 ton ha^{-1}, 318.20 ± 23.38 ton ha^{-1} in the DDF, 238.70 ± 34.8 ton ha^{-1} in the DF and 610.30 ± 14.60 ton ha^{-1} in the DHEF. The mean SOC accumulations in 100 m depths were not significantly different in all forests except the DHEF. The DHEF was significantly higher than other forest in PMP ($p < 0.001$). SOC density varied with different land use and terrain (Han, 2009).

Within the 100 cm soil depth, SOC varied from 29.00 to 46.20 ton ha^{-1} in the DMDF, 26.70 to 41.10 ton ha^{-1} in the DDF, 19.63 to 36.40 ton ha^{-1} in the DF and 55.63 to 77.87 ton ha^{-1} in the DHEF. Soil organic carbon accumulations decreased in increased soil depth. The top soil layers were higher in SOC than other layers but not significantly different within 0-40 cm. In most of the forests except the DDF, the top soil layers, 0-10cm, were significantly different in the SOC content between 40-60 cm and 60-100cm as shown in Figure 11.

Mean SOC in different forests was compared in Table 14. SOC accumulation in each layer was compared by forest types. As shown in the table, the DHEF had the highest SOC density in all 10 cm depths, and next came the DMDF. The DDF and DF soil had the lowest SOC density (Table 14). In all investigated forests, SOC decreased with the increasing soil depths but the surface SOC values varied depending on the forest. This is because the surface layer accumulated more soil organic carbon due to litter fall, the humus layer being decomposed and going into the top soil layer, 0-10cm. As evaluated by Ducan multiple range test (DMRD), mean SOC content (each 10cm depth) in the DHEF showed significant difference ($P < 0.05$) among the different forest types for the layer of 0-10cm, 20-40 cm, 40-60cm and 60-100cm. But in the 10-20 cm layer, significant differences were found between the DHEF and DF ($P < 0.05$).

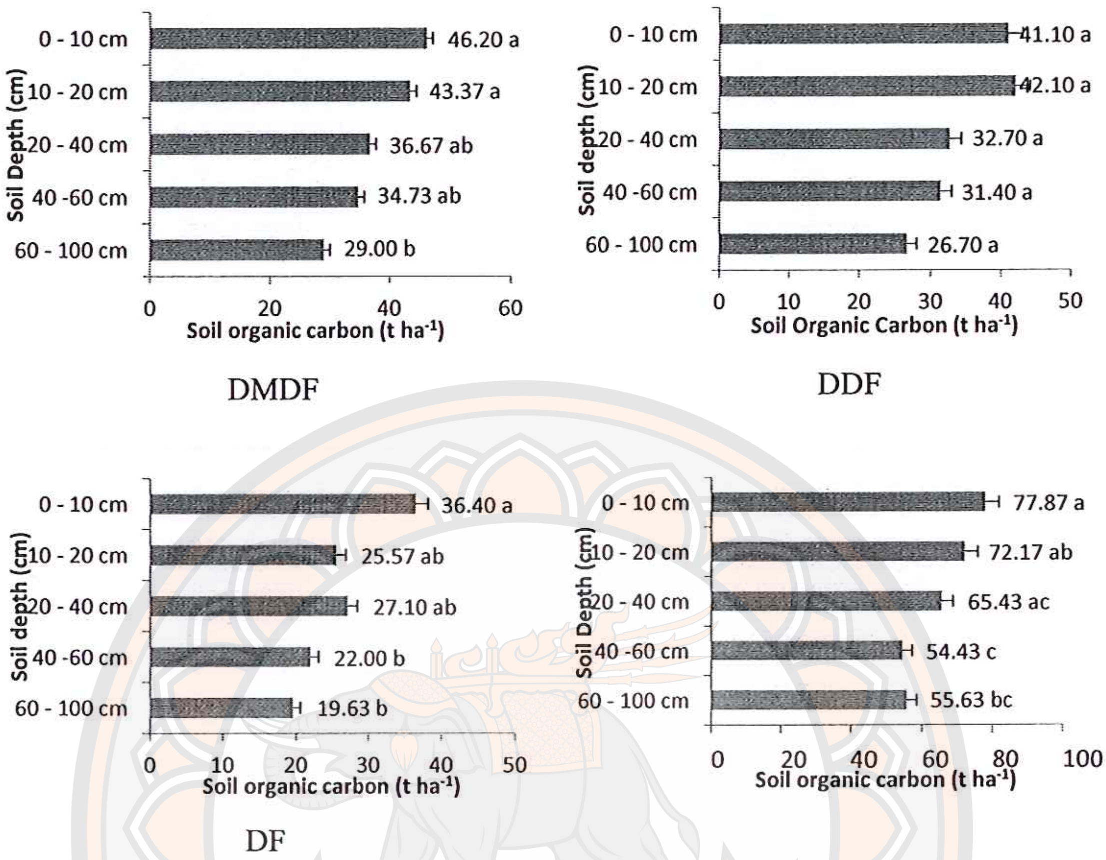

Figure 11 Soil organic carbon by soil depths (up to 100 cm)

Note: Different letters indicate the significant differences in SOC among the soil depths according to Ducan's multiple range tests at 5% level of probability. The letters are the rank order from highest to lowest value (alphabetically).

Table 14 Comparison of mean SOC of 10 cm depth in DMDF, DDF, DF and DHEF

	DMDF	DDF	DF	DHEF
0-10 cm	46.20[a]	41.10[a]	36.40[a]	77.87[b]
10-20 cm	43.37[ab]	42.10[ab]	25.57[a]	72.17[b]
20-40cm	36.67[a]	31.70[a]	27.10[a]	65.43[b]
40-60cm	34.73[a]	31.40[a]	22.00[a]	54.43[b]
60-100 cm	29.00[a]	26.70[a]	19.63[a]	55.63[b]

Note: Different letters indicate the significant differences in SOC among the forest types according to Ducan's multiple range tests at 5% level of probability

Table 15 Soil organic carbon by different depth in DMDF, DDF, DF and DEF

	DMDF	%	DDF	%	DF	%	DHEF	%
0-30 cm	124.87[a]	36.35	115.90[a]	36.42	89.07[a]	37.31	215.47[b]	35.19
0-50 cm	194.53[a]	56.63	180.00[a]	56.57	135.17[a]	57.88	335.33[b]	54.77
0-100 cm	343.53[a]		318.20[a]		235.70[a]		612.30[b]	

Table 15 showed that in 0-30cm, 0-50cm and 0-100cm depth, the DHEF had the highest SOC 215.47 $tCha^{-1}$, 335.33 $tCha^{-1}$ and 612.30 $tCha^{-1}$, respectively. Soil organic carbon stocks were higher in the DMDF 124.87 $tCha^{-1}$, 194.53 $tCha^{-1}$ and 343.53 $tCha^{-1}$ in 0-30cm, 0-50cm and 0-100cm soil depth, respectively. In the DDF, there were 115.90 $tCha^{-1}$ in 0-30cm, 180.00 $tCha^{-1}$ in 0-50cm and 318.20 $tCha^{-1}$ in 0-100cm. But the DF had the lowest SOC densities, at 0-30cm, 0-50cm and 0-100cm it had 89.07 $tCha^{-1}$, 125.17 $tCha^{-1}$ and 235.70 $tCha^{-1}$ correspondingly.

The average SOC results per hectare for the upper 30 cm of DMDF (124.87 $tCha^{-1}$), DDF (115.90 $tCha^{-1}$), DF (89.07 $tCha^{-1}$) and DHEF (215.47 $tCha^{-1}$) agreed well with the results obtained by Mohanraj who estimated SOC in mixed forest, 148.86 $tCha^{-1}$, deciduous forest, 246.02 $tCha^{-1}$ and Evergreen forest, 209.54 $tCha^{-1}$ in the upper 30 cm of Kolliforest eastern Ghats, India (Mohanraj et al., 2011).

Moreover, this findings correspond well with the estimated values of Oo (Oo, 2009) **for 0-30 cm** of 181.0 $tCha^{-1}$ in Oktwin Teak bearing forest, 192.4 $tCha^{-1}$ in Latpanpin Community forest and 195.2 $tCha^{-1}$ in Alaungdaw Kathapa national Park forest.

Accumulation of SOC in 0-50 cm was 135.17 $tCha^{-1}$ in DF to 335.33 $tCha^{-1}$ in the DHEF (Table 15). This corresponds with the estimated values by Zhou (Zhou, Yu and Zhao, 2000) for 0-50 cm of 193.55 $tCha^{-1}$ for major forest type of China. As well as this forest at the 0-50 cm depth, Park (Park, et al., 2012) found a SOC content of 178.1 $tCha^{-1}$ in dipterocarp forests, Philippines. The SOC may be influenced by forest density and altitude. The soil organic carbon content may depend upon physiography or location of the study's soil profile (Dadhwal, Palria and Chhabra, 2003) and land use change (Post and Kwon, 2000), conversion of closed forests to open forests (Jose, Koshy and Joes, 1972). Moreover, root biomass and litter production constitute important factors affecting the soil organic carbon (Zheng, et al., 2007).

In terms of SOC stock ($tCha^{-1}$), dry evergreen was significantly higher than other natural forest in PMP ($P< 0.01$). But the ratio of SOC accumulation by depth was similar. The proportion of the total organic carbon in the upper 100 cm held in the first 30 cm, that was 36.35% in the DMDF, 36.42 in the DDF, 37.31% in the DF and 35.19% in the DHEF on average, can be deduced from Table 15. The upper 50 cm (of the total organic carbon) holds 56.63 % in DMDF, 56.57% in DDF, 57.88% in DF and 54.77% in DHEF. These values were well within the estimates of Batjes (Batjes, 1996), who calculated for the world on average that 39-70% of the total organic carbon was found in the upper 30 cm and 58-81% in the upper 50cm.

The SOC allocation in upper 0-30 cm of this study was close to the other's studies. Gallali observed that the surface horizon (0-30 cm) stored 40% of the total soil organic carbon stock in the upper 100 cm (Gallali, et al., 2010). Lugo and Brown found that the amount of SOC to 40 cm depth varied from 46 to 80% of that to 100 cm depth, with generally lower percentages in the wetter life zone than in dry ones (Lugo and Brown, 1993). The percentage of SOC content in the different layer (0-30 cm, 0-50 cm) were similar with other studies. But SOC density ($tCha^{-1}$) were differed in different forest types.

Total Carbon pools in different forest types

The total carbon stock (aboveground, belowground, litter and soil) was significantly greater ($p<0.001$) in the DHEF (274.63 tC ha^{-1}) than other forests in PMP. But carbon stocks in the DMDF (180.00 tC ha^{-1}), DDF (162.03 tC ha^{-1}) and DF (166.71 tC ha^{-1}) were not significant different from each other. It may be due to the living tree carbon accumulation which was significantly higher in the DHEF than any other forest in PMP. In litter and deadwood carbon, DHEF was not significantly different from other forests. The carbon allocation of aboveground carbon pools by sample plots was as shown in Figure 12.

Table 16 Comparison of carbon stock in each pool of 4 forest types

	DMDF	DDF	DF	DHEF
AG-C (tC ha^{-1})	116.6 ± 11.3^{a}	91.28 ± 5.40^{a}	126.87 ± 12.06^{a}	200.22 ± 31.23^{b}
BG-C (tC ha^{-1})	18.5 ± 1.3^{a}	14.67 ± 0.68^{a}	18.55 ± 1.54^{a}	36.45 ± 3.80^{b}
Litter (tC ha^{-1})	16.6 ± 2.8^{a}	8.62 ± 1.18^{b}	2.15 ± 0.77^{c}	10.77 ± 1.34^{ab}
Deadwood (tC ha^{-1})	28.3 ± 5.3^{ab}	47.46 ± 7.52^{a}	19.15 ± 5.62^{b}	27.21 ± 6.23^{ab}
SOC (tC ha^{-1})	384.40 ± 32.5	318.20 ± 23.38	238.70 ± 34.8	612.30 ± 14.60

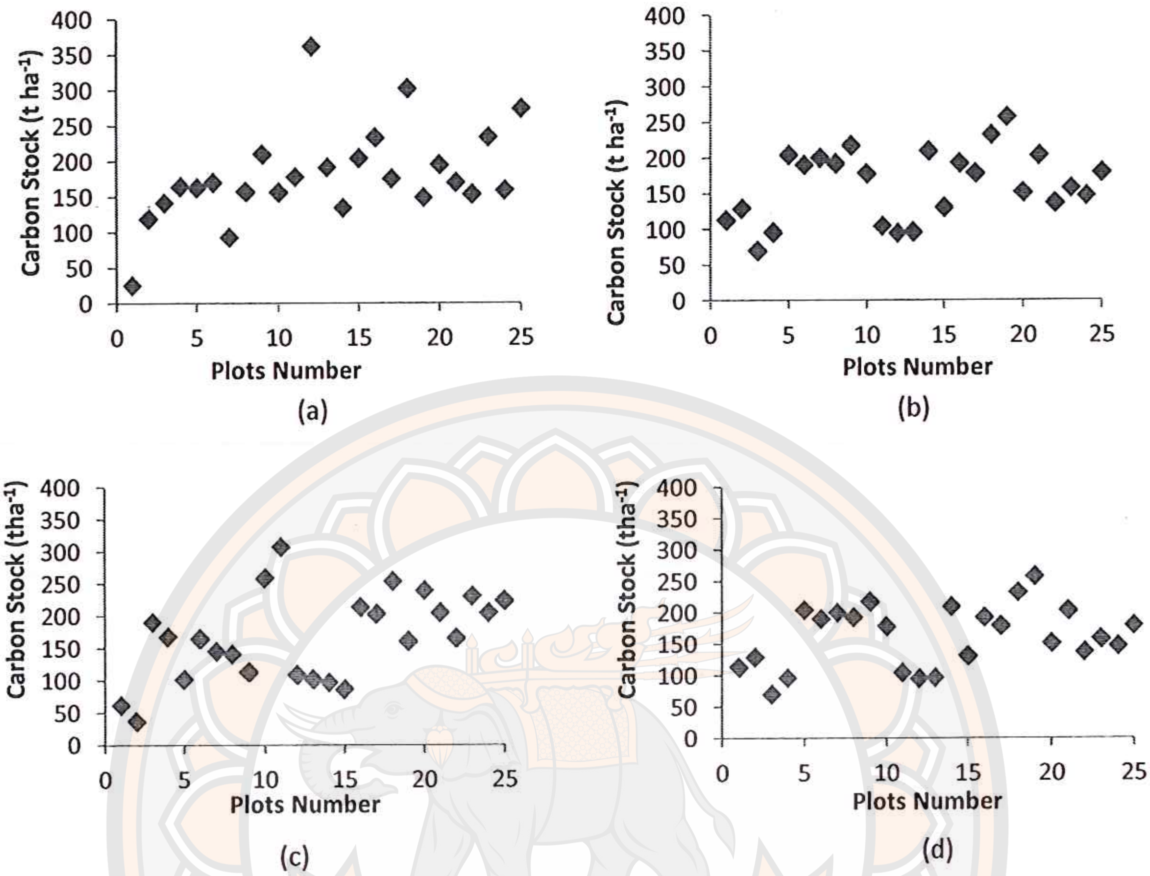

Figure 12 Total aboveground carbon allocation within 25 sample plots in DMDF (a), DDF (b), DF (c) and DHEF (d)

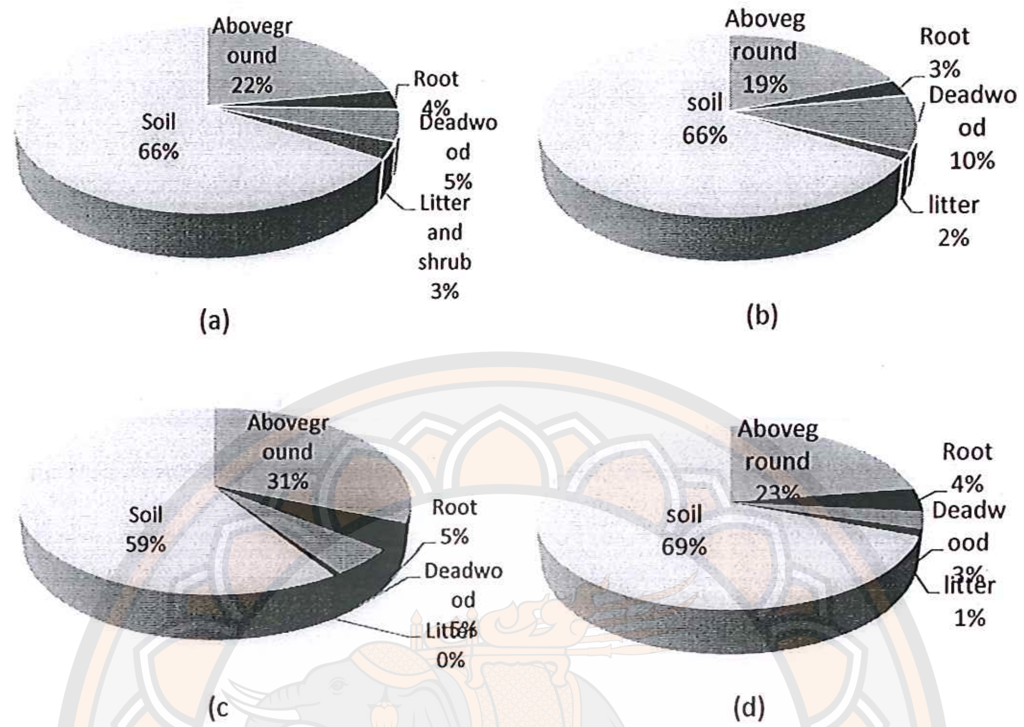

Figure 13 Carbon content in each component of 4 forest types (a) DMDF (b) DDF (c) DF and (d) DHEF

The carbon stock in each carbon pools and different forest types in PMP can be summarized as follows:

Living tree (aboveground and belowground) carbon stock: DHEF > DF > DMDF >DF.

Litter carbon: DMDF > DHEF > DDF > DF

Deadwood carbon: DDF > DMDF > DHEF > DF

Soil organic carbon: DHEF > DMDF > DDF > DF.

Based on data from 100 sample plots, total average carbon stocks in dry mixed deciduous forest, dry dipterocarp forest, dry forest and dry hill/ evergreen forest were estimated. In each forest, 5 carbon pools were estimated. The proportion of carbon storage for DMDF, DDF was shown in Figure 13. In all investigated forests, more than 60 percent of the carbon was accumulated in the soil. Living tree carbon allocation (aboveground and belowground) was around 30 percent of the total carbon in all investigated forests in PMP. Deadwood (coarse wood debris) stores less than 10 percent of total carbon stock in each forest. The proportion of carbon accumulation in carbon pools was similar among investigated forest types as shown in figure 13, but the amount of carbon stock in the respective pools was different in different forest types (Table 16). The largest SOC stocks were found in the soil, as expected.

Estimation of Carbon credits and revenue from reducing deforestation

Forest area change

The trend of forest cover change is important for defining baseline deforestation (Aye, et al., 2014). Historic (e.g. the past 10 years) deforestation helps to predict future deforestation, taking into account both the rates of deforestation and trends in deforestation rates (increasing or decreasing) (Arild Angelsen, et al., 2011). In this study, forest area change was estimated by using the past deforestation rate which was analyzed by Htun (Htun, et al., 2010) from the remote sensing data of forest cover changes in 1989, 2000 and 2005. Therefore the historic deforestation rate such as 0.09% low deforestation rate (LDR), 2.25% average deforestation rate (ADR), and 4.42% high deforestation rate (HDR) was used to project future forest cover change between 2013 and 2043.

By using the past deforestation rate, forest cover changes for each year in each forest type were estimated. In the dry mixed deciduous forest (DMDF), forest area decreased to 3033.9 ha, 2181.1 ha and 1127.3 ha in 2043 from 4095.3 ha, 3974.3 ha and 3742.8 ha in 2013 (forest cover was 4220 ha in 2010 (Htun, et al., 2011)) for low, average and high deforestation rate, respectively. According to different deforestation rate, 36.4 ha to 88.7 ha of forest area will be lost annually in the DMDF. In term of percentage, about 25% - 63% of forest area will be lost between 2013 and 2043.

In dry dipterocarp forest (DDF), forest area (forest cover was 882 ha in 2010) 855.9 ha, 830.6 ha and 782.3 ha in 2013 will be reduced to 634.1 ha, 455.86 ha and 235.61 ha in 2043 for LDR, ADR and HDR. With Low, average and high deforestation rate, 21.96 ha, 36.74 ha and 52.52 ha of the forest area will be lost annually in DDF. Within 2013 and 2043, 25 % - 60 % of today forest cover will be lost at different deforestation rates.

In dry forest (DF), forest area will be decrease from (2685 ha in 2010) (Htun, et al., 2011) 2605.6 ha, 2528.6 ha and 2381.4 ha in 2013 to 1930.3 ha, 1387.7 ha and 717.2 ha in 2043 losing 22.2 ha, 37.3 ha and 53.3 ha for low, average and high deforestation rate, respectively. Due to past deforestation rate, 20 % to 60 % of forest area will be lost within 30 years (2013-2043).

There are 182 ha of dry hill evergreen forest (DHEF) in 2010 (Htun, et al., 2011). According to low, average and high deforestation rates, 176.6 ha, 171.4 ha and 161.4 ha in 2013 would be decreased to 130.8 ha, 94.1 ha and 48.6 ha in 2043. Within 2013 – 2043, annual forest area lost about 75.8 ha (45.3 ha for low rate, 108.4 ha for high rate). Today's forest cover will be decreased by 25% - 60% in the next 30 years.

The trends of forest area change in all investigated forests were similar (Figure 14) but the amount of forest area loss was different according to each forest area. There are 4220 ha in DMDF, 882 ha in DDF, 2685 ha in DF and 182 ha in DHEF in 2010 (Htun, et al., 2011). According to historic deforestation trend, approximately 20% - 60% of today forest cover will be lost in the investigated forest of Popa Mountain Park.

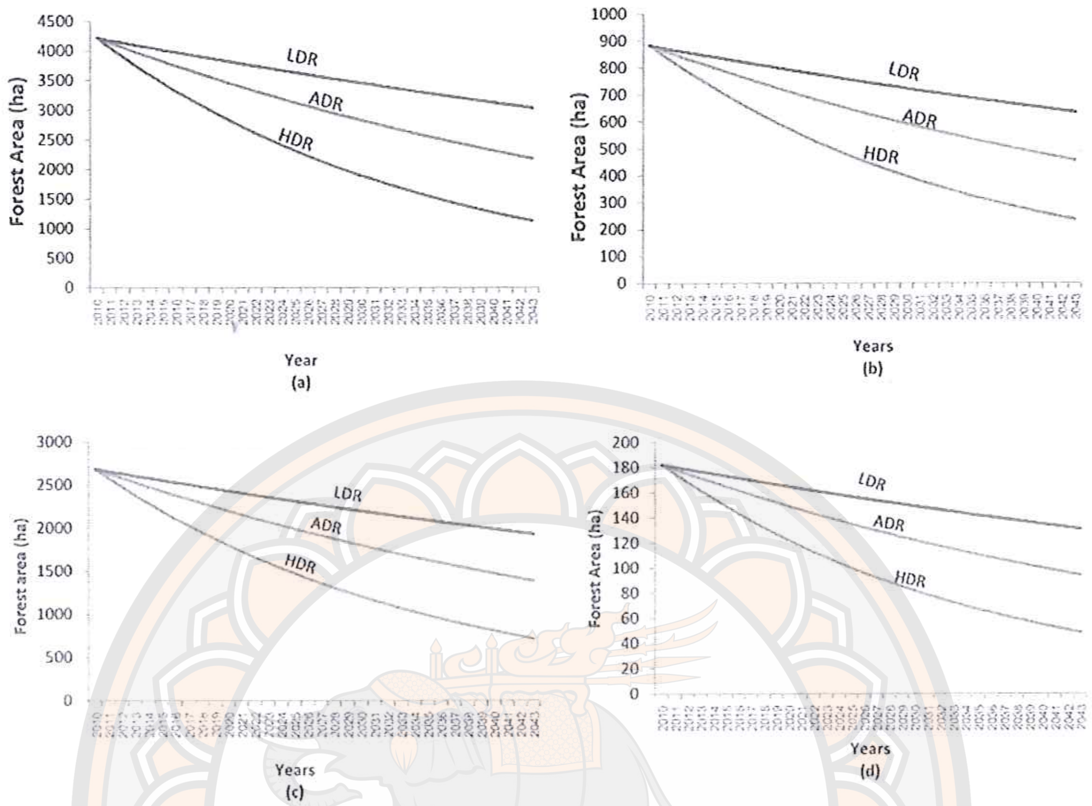

Figure 14 Forest cover change without REDD project in (a) DMDF, (b) DDF, (c) DF and (d) DHEF

Baseline and project deforestation

1. **Baseline deforestation**

The term "baseline" refers to a situation without a particular policy in place and is used as a reference case for quantifying policy performance (Ȯlander, et al., 2006). Baseline Deforestation (BD) was derived from the forest cover change between Time 2 and Time 1. To provide a range of possible deforestation, three rates of deforestation (low, average and high) were used to predict future deforestation. Total deforestation (without REDD project) over a 30-year timeframe is estimated at 1091.6 ha, 1836.3 ha and 2659.7 ha in the DMDF, 219.63 ha, 367.35 ha and 525.22 ha in the DDF, 668.6 ha, 1118.3 ha and 1598.8 ha in the DF and 45.3 ha, 75.8 ha and 108.3 ha in the DHEF for low deforestation rate, average deforestation rate and high deforestation rate, respectively. This deforestation was considered baseline, upon which baseline emissions or reference emission level were calculated.

2. Deforestation during the project implementation

The REDD + project is designed to reduce or stop deforestation and forest degradation. Various management interventions were introduced in the investigated area in order to reduces drivers of deforestation and forest degradation. Each intervention affects drivers and thus reducing deforestation and forest degradation. Over a 30-years project period, the forest area will lose annually about 529.3 ha (303.7 ha for low rate, 826.8 ha for high rate) in the DMDF, 273.2 ha (164.6 ha for low rate, 383.1 ha for high rate) in the DDF, 831.6 ha (501.3 ha for low rate, 1166.2 ha for high rate) in the DF and 56.4 ha (33.9 ha for low rate, 79.1 ha for high rate) in the DHEF (Figure 15).

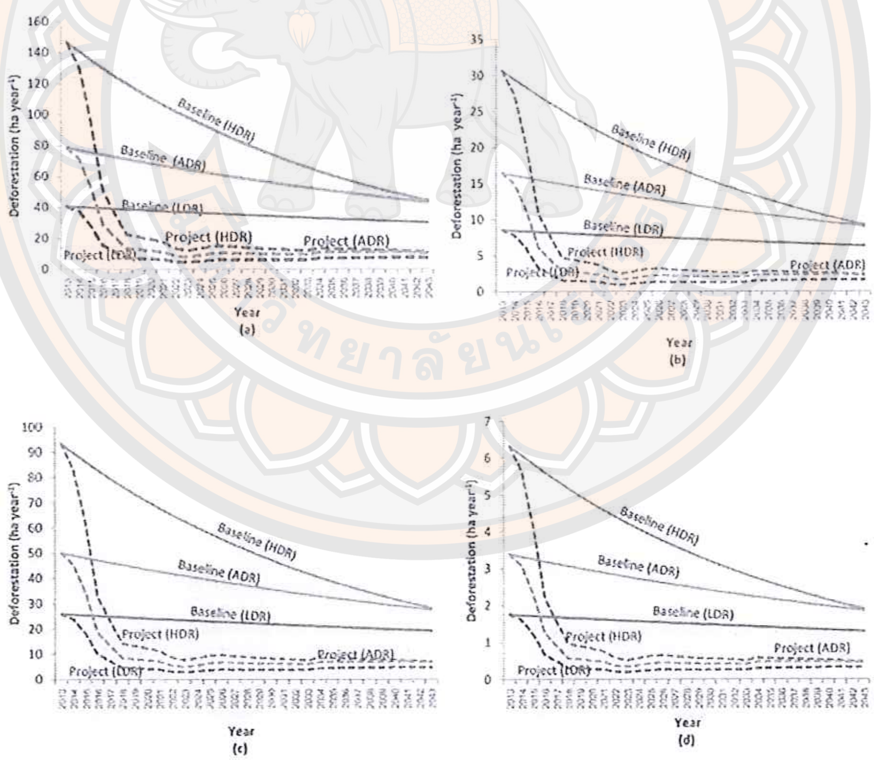

Figure 15 Annual deforestation under the baseline and project scenarios in (a) DMDF, (b) DDF, (c) DF and (d) DHEF

Note: Baseline deforestation occurs in the absence of project While project deforestation occurs when REDD projects is implemented

Baseline and project carbon emission

1. Baseline carbon emission

A baseline or reference emission is the carbon emission in the absence of project activities against which, reduced emissions are compared. It is a benchmark to judge the performance of emission reduction. Baseline carbon emission was estimated by using a baseline deforestation area and carbon stock (CO_2) in each forest.

Although five carbon pools are recommended by IPCC (Intergovernmental Panel on Climate Change, 2006), carbon stored in the aboveground biomass is the most directly impacted by deforestation (Gibbs, et al., 2007). In this study, carbon emission from deforestation considers the following pools: aboveground, belowground, litters and deadwood because soil carbon slowly changes depending on what followed after deforestation.

Due to deforestation, 138744.4 tCO_2 – 331781.6 tCO_2 in the DMDF, 26096.8 tCO_2 – 62405.6 tCO_2 in the DDF, 82562.53 tCO_2 – 203445.7 tCO_2 in the DF and 9127.9 tCO_2 – 21827.6 tCO_2 in the DHEF will be lost between 2013 and 2043 without any conservation measure (Figure 16).

2. Project carbon emissions

During the project's implementation, many activities would be done in order to reduce deforestation. The project will implement activities such as monitoring the project site regarding vehicles, agricultural intensification thinning and removal of invasive species along with making a fire prevention road by clearing some forested area which will emit some amount of carbon. These emission are considered as project emission.

Under the project, total carbon emission over this 30-year project timeframe is estimated at 34720.6 ha - 89790.5 ha in the DMDF, 6530.6 ha – 16888.9 ha in the DDF, 23040.53 ha – 63459.3 ha in the DF, 2284.2 ha – 5907.2 ha in the DHEF for LDR, ADR and HDR, respectively.

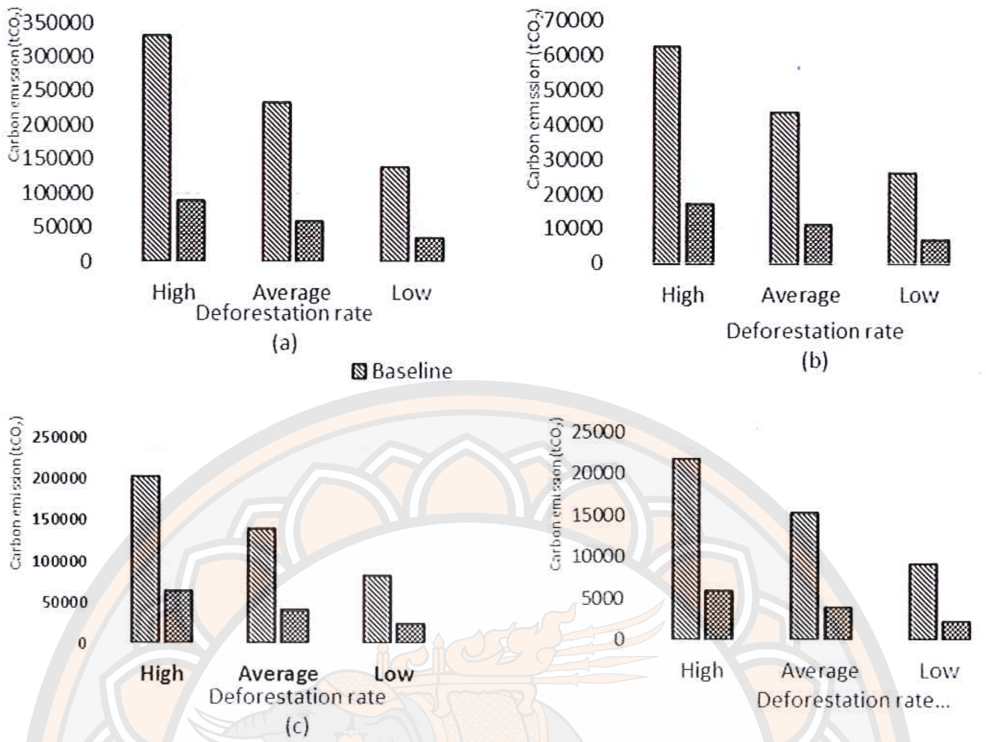

Figure 16 Total emission under baseline and project in (a) DMDF, (b) DDF, (c) DF and (d) DHEF

Carbon emission reduction, Credits and revenues

1. Carbon emission reduction

In order to reduce deforestation or carbon emissions from deforestation, the study attempts to design a project with various activities based on the drivers of deforestation. Various management activitieswere introduced to control deforestation following the Ty's study (Ty, et al., 2011). If the REDD project is implemented to reduce deforestation in the natural forest in the Popa Mountain Park protected area, carbon emissions of about 104023.8 tCO_2, 172564.8 tCO_2 and 104023.8 tCO_2 in the DMDF, 19566.1 tCO_2, 32458.17 tCO_2 and 45516.7 tCO_2 in the DDF, 61285.4 tCO_2, 101666.2 tCO_2 and 142568.5 tCO_2 in the DF and 6843.6 tCO_2, 11352.8 tCO_2 and 15920.3 tCO_2 in the DHEF could be reduced for a 30-year project cycle (Figure 17).

2. Carbon credits

Carbon credits from the REDD project were estimated by measuring carbon emissions from reduced deforestation under the project. However, forestry projects are suspected to be prone to leakage. A project that protects a particular forest land could lead to increased cutting in adjacent lands. Potential for leakage will vary from project to project (Lasco and Pulhin, 2003). In this study, we assumed 30% leakage (lk = 0.30) following (Ty, et al., 2011).

The project leads to accumulated carbon credits of about 69828.2 tCO_2, 115573.1 tCO_2 and 169393.7 tCO_2 in the DMDF, 13696.2 tCO_2, 22720.7 tCO_2 and 31861.7 tCO_2 in the DDF, 42899.8 tCO_2, 71166.3 tCO_2 and 99797.99 tCO_2 in the DF and 4790.54 tCO_2, 7947.1 tCO_2 and 11144.2 tCO_2 in the DHEF under low deforestation rate, average deforestation rate and high deforestation rate, respectively.

3. Carbon Revenues

To facilitate comparison with international literature a reference carbon price US$ 5 per tCO_2 was used, as this is the most common price used in the voluntary market (Busch, et al., 2009; Diaz, et al., 2011; Guyana and Norway, 2009). The project could result in total carbon revenues US$ 349503.3 – US$ 846968.6 in dry mixed deciduous forest, US$ 68481.4 – US$ 159308.5 in dry dipterocarp forest, US$ 214499.1 – US$ 498989.9 in dry forest and US$ 23952.7 – US$ 55721.2 in dry hill or evergreen forest low - high deforestation rates.

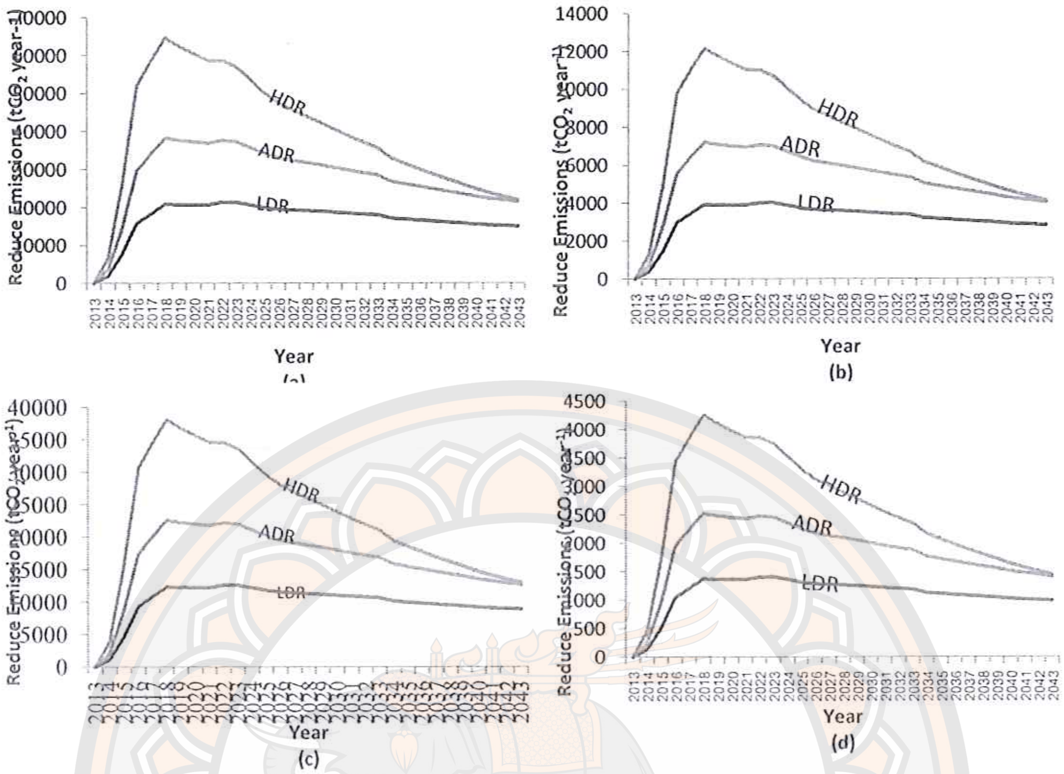

Figure 17 Reduce emissions (avoided emissions) due to REDD project in (a) DMDF, (b) DDF, (c) DF and (d) DHEF

CHAPTER V

DISCUSSION AND CONCLUSION

Discussion

1. Species composition, diversity and stand structure of different forest in PMP

In order to do the vegetative analysis, forest inventory was done by selection the representative sample size. The species-area curves is the best criteria for determining a minimal plot size needed to survey a community adequately. According to the species-area-curve, the number of species increased substantially up to the point of 0.40 ha in DMDF, 0.48 ha in DDF, 0.64 ha in DF and 0.48 ha in DHEF. The increment of species was then less than 10% until the sampling area reached 1.00 ha. According to Cain and and Castro (Cain and Castro, 1959), the minimum representative area will be reached if the number of species increases by less than 10% when expansion of sampling area is 10%. Therefore 1 ha sample plot is represented for each of forest type.

It was found that dry hill or evergreen forest (DHEF) has the highest mean diameter at breast height (DBH) 24.03 cm and highest volume (610.72 m^3ha^{-1}) while stand density (804 trees ha^{-1}) was the lowest among forest types showing that the DHEF is more mature than other forest types in PMP with less human disturbance in the area. This may be due to the distance from the surrounding settled areas as DHEF was found in high elevation and more remote from the park boundary. The highest stand density (1293 trees ha^{-1}) was found in dry deciduous forest (DDF) with lowest standing volume (126.80 m^3ha^{-1}) signifying that DDF has human disturbance and large trees had been harvested prior to data collection, probably due to nearness of roads and ease of access. DDF was found between 400m - 700m asl and the park circular road was close to the DDF. Forest cover loss was more severe in the areas near roads and therefore the likelihood of encroachment by local communities is high in the accessible forest at low elevations (Htun, et al., 2013).

The diameter distribution of trees was investigated in order to know the population structure of the forest. In all forests, higher species richness and diversity were found in small DBH classes (Table 5). Tree density was also higher in the small DBH classes indicating that small tree were in sufficient numbers to replace mature trees when necessary. Diameter distribution curves show the pattern of population structure. The inverse J shaped curves for the entire investigated stands show that the stands have a growing population structure. However, the mean DBH in all forest is small; 17.50 cm in DMDF, 11.00 cm in DDF, 13.34 cm in DF and 24.03 cm in DHEF. These may be the actual lower thresholds of DBH in the inventory. This study however measured all trees of DBH ≥ 5cm. The highest tree density was found in the DBH classes 5-10cm and 10-15 cm which contribute more than 50 % of total tree density. From the shape of the inversed J shape curve of relative abundance over DBH of small DBH trees, we can conclude that it is a common pattern of stand structure of the forests in logged and deforested areas. Stand structures of the DHEF suggested that this forest is less disturbed because many large trees were still found in this forest compared to the other forests. This may be due to the fact that DHEF is located at high elevations and remote from the park boundaries and human habitation. These findings support Htun (Htun, et al., 2013) whose findings were that forest cover loss was high in areas close to park boundaries and roads and low in remote and less accessible areas.

Patrolling is the main method for forest conservation in the PMP. Limited infrastructure however made conservation activities difficult. In addition, budgets for conservation activities were insufficient. Total government budget for conservation of PMP for 2007-2008 was US$ 115,000, of which government staff salaries consumed 83%, and much of the remainder was used for administration and maintenance rather than for conservation activities (Htun, et al., 2010). Effective conservation of this important park requires more budget as well as infrastructure. With no alternative funding source, it is difficult to maintain and conserve current forest conditions in PMP. Our findings indicate that forests in the PMP are rich in species diversity but stand structures of these forests are similar to that of degraded forests. Therefore, it is possible to conclude that forest conservation in the PMP is still ineffective and there is critical need to control human disturbance. Future study on drivers of deforestation and forest degradation and how to address these drivers would provide the needed information for effective conservation of the PMP.

2. Carbon stock of different forests in PMP

2.1. Selection of equation to estimate tree carbon (AGB and BGB)

Estimation of carbon sequestration in the forest sector should take into consideration changes in carbon stocks in all carbon pools, including aboveground and belowground biomass, litter, deadwood and soil (Takahashi, et al., 2010). The most accurate method for the estimation of biomass is through the cutting of trees and weighing of their parts (T.M. Basuki, et al., 2009). One of the most reliable ways to estimate biomass and carbon sequestration in a forest stand, is to select sample trees. However, obtaining a representative sample of trees for estimating biomass by felling the trees, digging out their root systems, drying and weighing their biomass, etc. is a very difficult and expensive operation. Direct tree harvest data are difficult to acquire in the field, and few published studies are available (J Chave, et al., 2005). Therefore, applying the allometric equations is a preferred approach, using variables such as diameter at breast height (DBH) and tree height (Ht) which can be accurately measured in the field. Estimating of forest stand biomass using allometric equations saves operational costs as well as requiring less time consuming work (Segura and Kanninen, 2005). Allometry enables the total tree biomass of a forest or a stand to be estimated without having to cut any trees, and transport them back to the lab, dry the species in an oven and then weigh all the pieces.

The use of allometric regression models is a crucial step in estimating aboveground carbon stock (Brown, S., et al., 1989). Equation choice is important for AGB estimates. However, these estimates will vary depending on the use of different regression equations. These variations in estimates are caused by environmental, structural, and compositional gradients (Malhi, Y., et al., 2002; Ter Steege, 2000), (Baker, et al., 2004). Previous studies have demonstrated that different equations can give rise to very different AGB estimates when applied to the same forest inventory data (Araújo, Higuchi and Andrade de Carvalho Júnior, 1999). One study (Losi, et al., 2003) reported that in a study of native species carbon sequestration estimation with species growing in tree plantations in Panama and Costa Rica, using the allometric models resulted in an overestimation of 10.2% in the carbon stock values for

D. panamensis plantations. Therefore, comparisons of AGB estimates over large spatial scales need to be based on a consistent regression approach (Baker, et al., 2004). Most of the allometric equations are developed for some range of dbh size, specific forest types and area. Tropical forest possess diverse species, as many as 300 different species (Oliveira and Mori, 1999) and the allometry of tropical trees varies greatly with forest type, and specific allometric equations have been developed separately for wet, moist, and dry forests (Brown, S., et al., 1989; Chave, J., et al., 2005). So, one specific-species equation could not use for another (Brown and Schroeder, 1999). Therefore mixed species tree biomass regression models should be used (Chave, J., et al., 2005).

Many biomass equations have been developed which use various explanatory variables including tree diameter, height, wood density, and tree form factors (Brown, I. F., et al., 1995; Brown, S., et al., 1989). The most important predictive variables are diameter, height, wood specific gravity and forest type (Chave J., et al., 2005). Among them, wood specific gravity is an important predictive variable in all regressions (Chave, J., et al., 2005). The models including wood density exhibit higher R^2 than those without wood density (Chaturvedi and Raghubanshi, 2012). In addition, Baker (Baker, et al., 2004) have shown that ignoring variations in wood density will result in poor overall prediction of the stand. The role of wood density in the allometric equation is more prominent for the mixed species than in the genera (Tyas Mutiara Basuki, 2012) because the diverse age and composition structure of specific stands are the major sources of uncertainty for estimating carbon stock (Intergovernmental Panel on Climate Change, 2003). Thus, the various values of wood density from different species must be considered for estimation of tree carbon. The biomass estimation based on forest volume inventories provides an opportunity to improve the total aboveground biomass estimates because volume data from forest inventories are more abundant and are generally collected from large sample areas using a planned sampling method designed to represent the pollution of interest (Brown, S., et al., 1989).

Therefore, the aboveground carbon (AGC) (tonne C ha^{-1} or tC hereafter) in the study area was estimated using the equation which was based on these factors; stand volume over bark (for each tree), wood density (for each species), biomass expansion factor (for each forest types) and lastly, carbon content default values from IPCC. WD for all tree species were taken from the ICRAF world agroforestry database (Zanne, et al., 2009) and the Global wood density database (ICRAF, 2010). For belowground carbon (root carbon), an allometric equation based on aboveground carbon values was used. This allometric equation was originally developed by Cairns (Cairns, et al., 1997), in which the biomass/ carbon allocation to roots can be estimated based on aboveground density allometries.

2.2 Carbon stock in the investigated stand

Results of the carbon estimation showed that dry evergreen forest had the highest aboveground carbon density among the four forests studied. It contained 200.22 $tCha^{-1}$. The mean DBH was the largest in the dry hill/ evergreen forest (DHEF). Carbon storage was also the highest in this type of forest. However, stand density in the DHEF was lowest among the forests in PMP because of the large basal area of the trees in this forest (47.80 m^2). In contrast, the aboveground carbon storage was lowest in the dry dipterocarp forest (91.28 $tCha^{-1}$) but the stand density was highest in this forest. This may be due to the carbon accumulation being related to tree DBH size. This is probably due to the dry dipterocarp forest having the highest tree density of small trees whereas the DHEF has the largest trees but with the lowest tree density among all the forests in PMP.

Tree species composition of each forest also affected forest carbon stock. Wood density is different in different species. If the species could not identified, this study used wood density at the genus level; the variation of wood density among the genera is higher than within a single genus (Baker, et al., 2004; Tyas Mutiara Basuki, 2012; Jerome Chave, et al., 2006). In this study, tree carbon stock was estimated by using volume over bark and the wood density of the species. Therefore tree species composition and tree size strongly affect the estimation of forest carbon stock.

Various authors have mentioned that the accumulation of biomass and carbon in the forest is influenced by stock density (Brown, S., et al., 1989; Brown, S., 1997; Perez Cordero and Kanninen, 2003). Even though the stand density of the DDF was higher than in the other forests, the aboveground carbon stock was lower than in the other forests in PMP. The mean DBH, 11.00 cm, and basal area, 22.93

m^2ha^{-1}, were both the lowest of the forests in PMP. There are many factors affecting the carbon storage of forest in this study, such as DBH, volume, basal area and wood density and forest disturbances including illegal wood collection and forest fire. In contrast, the highest aboveground carbon was found in the DHEF which has the lowest stand density among the forests in PMP. It is suggested that the mean DBH, volume and basal area were higher in the DHEF because this forest is located above 1000 m asl, remote from the road and the park boundary and was therefore much less accessable to the local communities.

This study also analyzed the **carbon allocation in DBH classes**. The maximum carbon was stored in the 40-60 cm DBH class in DHEF (31.6%) and in the 20-40cm DBH class in each of the other three forest types; DMDF (59.8%), DDF (56.2%) and DF (47.9%). The younger trees (DBH class ≥ 5-20cm), which were the highest density in all forests, stored only 6.2% of total carbon in the DEF, 23.3% in the DMDF, 39.7% in the DDF and 47.9% in the DF. Although the young individuals belonging to the 5-20 cm DBH class dominated in all of the forests in terms of density, the AGB accumulation was greater in the 40-60 cm diameter class. In the DEF, 88.3% of trees were found at the DBH< 40cm and stored 32.3 % of total carbon stock. Eleven point seven percent of the total number of trees were found in DBH > 40cm and stored 67.7% of total carbon. The tree distribution and carbon allocation patterns of the DHEF in this study is similar to that of the dry evergreen forests in Thailand in which the trees are less than 40 cm DBH and have 91.95% of total number of trees, with 22.76% of the total carbon. As well, the DBH > 40 cm trees comprise 9.05 % of the total number of trees, and contain 77.24% of the total carbon. In the DBH of the study area, many trees were found with DBH < 40cm but having high carbon storage at DBH > 40cm. This is because the basal area in large trees is greater than in small trees.

Belowground carbon storage was the highest in the DHEF (36.45 $tCha^{-1)}$ followed by the DMDF (18.50 $tCha^{-1)}$, DF (18.55 $tCha^{-1)}$ and DDF (14.67 $tCha^{-1)}$. These findings are in line with belowground carbon found in Central Amazonian forests which range from 14 – 66 $tCha^{-1}$ (14 – 66 tha^{-1} in biomass) (Edwards and Grubb, 1977). The mean root to shoot ratio in this study (0.16) is close to the mean root to shoot ratio of forests in Malaysia, 0.18 (Niiyama, et al., 2010), 0.15 in

Bangladesh (Yong Shin, Miah and Lee, 2007), and 0.16 in Panama (Kraenzel, et al., 2003). This study findings are also in line with the root to shoot ratio range (0.15 to 0.27) found in mixed species plantations in Myanmar (Oo, 2009). The living tree carbon (aboveground and belowground) is highest in the DEF, followed by DF, DMDF, with DDF the lowest. In PMP, while the DHEF has the lowest stand density it has the highest living tree carbon stock, due to the larger tree stock found in DEF. This suggests that the DHEF is more mature than the other forests and has been subject to less human disturbance than those other forests, which are, in PMP, located near to local communities and are harvested for their livelihood.

There is a wide range of carbon storage in the **litter layer** among the forest types in PMP, from 2.15 to 16.6 $tCha^{-1}$. This is a greater range than that found in Indian forests, which is between 0.16 and 3.26 $tCha^{-1}$ (Mohanraj, 2011). Watershed areas in New Zealand have carbon storage of 0.6±2 $tCha^{-1}$ and the tropical lowland Dipterocarp rainforest in Sabah in Malaysia measures 0.7 $tCha^{-1}$(Saner, et al., 2012; Staley, 2010). The carbon stock in the litter layer is also comparative with the deciduous forest in Alaungdaw Kathapa National Park in Myanmar (3.8 $tCha^{-1}$) and natural secondary forest in China (3 $tCha^{-1}$) (Oo, 2009; Zheng, et al., 2007). As the current study was undertaken with all data collected during the dry season, when the deciduous trees have shed their leaves, the litter mass is at its maximum.

In PMP, the **deadwood** carbon stock in each forest type is DDF (47.46 $tCha^{-1}$), DMDF (28.3 $tCha^{-1}$), DHEF (27.21 $tCha^{-1}$) and DF (19.15 $tCha^{-1}$). It was expected that the dry evergreen forest would have a large amount of deadwood carbon because of the large trees found in that type of forest. This was also explained in Richardson et al., who stated that large, slow-growing tree species support the greatest volumes and biomass of deadwood (Richardson, S. J., et al., 2009). The potential importance of topography to amounts of dead wood was mentioned and discusses in both Gale (Gale, 2000) and Muller (Muller, 2003), who indicated that higher topographic positions have been shown to have higher potential for tree death and therefore higher mass and volume of dead wood. However, the results in PMP are contrary to their findings. The DHEF in PMP was found at the highest elevation, with many large trees and was the more mature forest among all the natural forests in PMP. The deadwood carbon stock in the DHEF was lower than in the DDF and DMDF which are located at lower elevations and have higher stand density with small trees. This study enumerate all standing dead trees, stumps and felled dead trees (above 10 cm diameter and higher than 1.3 m long) in all sample plots. Many dry stumps and coppices are found in the DDF and the DMDF. If dead stump are higher than 1.3 m and larger than 10 cm, this study counted as deadwood. Due to the large numbers of stems are found in the DDF, the mean deadwood carbon storage is the highest in the DDF, 47.46 $tCha^{-1}$. Aboveground carbon stock is lowest in the DDF, however, deadwood carbon stock is highest in the DDF. This findings support the Nascimento's study (Nascimento and Laurance, 2002). He found that there are no significant correlations between large tree biomass and that of any other live or dead biomass components.

Soil organic carbon (SOC) density in 100cm depth was the highest in the DHEF and the next came the DMDF and DDF, DF had the lowest SOC density. To facilitate comparison with international literature a reference depth of 1m is selected, as this is the most common reference depth used in related studies (Han, et al., 2010; Lettens, et al., 2006; Sleutel, et al., 2003). SOC content was statistically higher ($p<0.01$) in the DHEF soils than in other forest soils. It may be due to different forest types, the species composition and forest cover status of the area. Moreover, annual fire, light intensity and microclimate also effect on decomposition of forest floor which may affect accumulation of soil organic carbon. The DHEF is found in high elevation, moist than other forest in PMP and no annual fire. So all of the litter and deadwood decomposed and enter into the soil. Therefore SOC is highest in the DHEF forest in PMP. The SOC decreased with depth in all forests in PMP. The topsoil layer was the most susceptible to land use pattern (Reeder, Schuman and Bowman, 1998). Land use will have a great impact on soil nutrient status (Wang, et al., 2003) as well as soil organic carbon. So, in estimating SOC density, changes in land use, soil types and terrain should be taken into account. SOC density estimation at the forest level could provide baseline data for a large-scale estimation in PMP.

SOC content decreased with an increase of soil depth. The mean was highest but most variable in the topsoil layer. The SOC decreased with an increase soil depth in all land use types but the surface SOC values varied depending on land use (Liu, et al., 2003). Since this study was small, climate could be disregarded. So it was reasonable to assume that land use, soil type and terrain largely accounted for the spatial variability of the SOC content. The SOC content was statistically higher ($P<0.01$) in the DHEF soil than in the other forest soils in PMP especially in the surface horizons. At the same time, land use will have a great impact on soil nutrient status (Wang, et al., 2003). As evaluated by the Ducan Multiple Range Test, SOC content in the DHEF shows significant different among the different forest type for layer of 0-10 cm, 20-40 cm, 40-60 cm and 60-100 cm except the second layer 10 -20 cm.

The total carbon (AGB, BGB, litter, deadwood and soil) stored by the DHEF is the highest followed by the DMDF, DDF and DF. Although total carbon stock are different in different forest, the proportion of carbon storage in carbon pools for all forests are similar. In all forests in PMP, living tree (aboveground and belowground) accumulated 30 % of total carbon. Around 1% and 10% of total carbon are stored in litter and deadwood, respectively. Beside soil organic carbon stored 60 % of total carbon. According to the different stand, the carbon storage capacity are different. But in term of carbon accumulation proportion, there are similar among all forest in PMP. Therefore, forest types especially species composition and density effected on total forest carbon stock. However, it is unclear whether these results can be extrapolated to all forested area in Myanmar, because there are many kinds of forest in Myanmar and wetter than the study area.

3. Estimation of Carbon credits and revenue from reducing deforestation

This study focus only on carbon emission from deforestation. To provide a range of possible deforestation, three rate of deforestation were used to predict future deforestation, namely low, average and high rates. Beside, carbon emission was estimated from aboveground, belowground, litter and deadwood because carbon in these pools are mostly affected by deforestation. Based on the historic deforestation rate, over 30-years period (2013-2043) approximately 20% - 60% of today forest cover will be lost in the investigated natural forest in Popa Mountain Park. This study found that about 256531.6 tCO_2 – 619460.5 tCO_2 will be emitted between 2013 and 2043 without any conservation measures. By managing natural forest (7969 ha) in PMP, carbon emissions could be reduced accounting for 191718.9 tCO_2 – 445996.6 tCO_2 in 30-years project or 6390.6 tCO_2 – 14866.5 tCO_2 annually. This findings highlight that huge amount of carbon would be emitted from deforestation. These emission could be reduced by effective conservation measure like REDD+ project. But the future project implementation activities to reduce deforestation would effect on actual emission reduction. After learned from the experience during the project implementation, these assumption will have to be corrected.

If REDD+ project is implemented in PMP, carbon credits from reducing deforestation were estimated at 4373.8 tCO_2 - 10406.6 tCO_2 or US$ 21881.2 - US$ 52032.9 annually for 30-years of carbon project if carbon priced at US$ 5 tCO_2. This study used the carbon price US$ 5 per tCO2 in order to match with other literature (Arild Angelsen, et al., 2011; Busch, et al., 2009; Mattsson, et al., 2012; Nophea Sasaki, 2011; Nophea Sasaki, et al., 2013). Carbon revenues in this study are affected by the carbon price and therefore future adjustment of the price would be necessary. Nonetheless, Carbon revenues from reducing deforestation could bring funds for forest conservation and livelihood improvement. Moreover, carbon financing be made available to protect the forests in the Popa Mountain Park as well as other parts in Myanmar.

This study address the deforestation and degradation baseline in terms of area with land cover change or rates of land cover change. Ultimately, the quantities of interest are units of carbon emissions avoided by reducing deforestation. Leakage is the extent to which emission controlled in one place simply shift to another place where they are not controlled. Leakage may vary depending on location and nature of the compensation (Andrew and Chomitz, 2009; Murray, B.B.C., McCarl and Lee, 2004; Sohngen and Brown, 2004). Baseline at national level can capture and adjust for within country leakage from local projects, but may not properly account for international leakage unless special provision are put in places (Olander, et al., 2006). This study focus on project level, baseline might not be adjust leakage within one project, thus, leakage was use 30 % following Ty's study (Ty, et al., 2011).

The design of the REDD framework could have a substantial effect on various costs. Costs are directly associated with projects activities to reduce deforestation. The implementation costs include institution and capacity building activities that are necessary to make the REDD+ program happen. Likewise, the expenses associated with the goods, training, research and political, legal and regulatory process, including consultation and government decision-making processes are needed to be accounted for. Benefit sharing is one of the important issues that need to be addressed in REDD+. There are a variety of issues regarding the impact of REDD+ on the rights of indigenous and other communities living in the forest (Larson, 2011). The distribution of tenure rights over carbon and access to forests is critical for benefit sharing, and both are likely to be crucial to the ability of local populations to gain benefits from the increased value of carbon (Agrawal, Nepstad and Chhatre, 2011). Therefore, further research on the cost and benefits sharing of REDD+ inspired projects are recommended.

This study is one important step in estimating the cost of carbon emissions due to deforestation in Myanmar. Most of the decisions of forest conservation are based on biodiversity, sustainable use of forest product and basic need of local people. Where benefits of emissions reduction are consider, a decision based on estimation of carbon revenue from reducing deforestation as done in this study is useful for forest conservation and management. Moreover, this study estimated only carbon credit. The effect of project on biodiversity, livelihood of local people should be investigated as they may significantly affect the result of modelling.

4. Uncertainty of Baseline and Project Emissions Estimates Estimation

Estimation of emission reductions is strongly affected by the assumptions of baseline and effectiveness of project impacts on drivers of deforestation and forest degradation. Trend of forest cover change is important for defining baseline deforestation, against which project deforestation is compared. Two approaches to defining baseline were discussed, namely retrospective and prospective approaches. The former is based on the past trend over a period of time (about 5 - 10 years) to calculate the rate of change in forest cover and make projection linearly or exponentially. In the case of linear projection, two options could be possible, i.e. projection downward or taking average rate of deforestation between at least two points in time and make projection to the future horizontally. Projection downward is not realistic because such projection allows deforestation to continue even if there is no more forest left. Horizontal projection could be acceptable but the rate of change should be revised every 5 to 10 years de- pending on the actual situation in the location or country in question. In fact, discussions on shifting the baseline once every

10 years have been recently discussed (VCS, 2013). The problem is that it would be difficult to estimate an acceptable baseline while a REDD+ project is being implemented and verified. In this study, we used exponential projection for the baseline because this approach takes into account the availability of remaining forests. In practice, as forest cover declines, the rate of deforestation relative to forest cover also declines, and deforestation cannot occur when forests are completely cleared.

The latter (prospective baseline) is dependent on national circumstances of individual countries, where rapid economic development was evident after these countries emerged from political instability and isolated economic development. Countries like Myanmar, Cambodia or Laos could adopt this prospective baseline provided that they could provide evidence of past economic growth and governmental reforms. Although we acknowledge that a prospective baseline is important for estimating carbon emissions, estimation of and using this baseline are beyond the scope of this study.

Effectiveness of project activities on drivers of deforestation and forest degradation strongly affects emission reductions and thus carbon revenues. As experience is gained, each driver requires multiple activities while some drivers are not possible to reduce. It is important to understand the scales of drivers before appropriate activities could be introduced. For instance, clearing forest for industrial plantations is global because of the continuous global demands, while illegal logging can occur on a regional scale because of regional (cross-border) demand for timber. Both drivers are difficult to reduce because they require global and regional cooperation. In contrast, such drivers as land speculation and clearing of land by opportunists or land migrants (moving from one province to another) could occur at subnational scale, and they could be somehow reduced through law enforcement. Drivers such as fuelwood consumption, forest fires, clearing forests for small plantations for local needs could occur at local scale, which could be reduced by introducing alternatives for using less wood for fuel (i.e. by introducing efficient cookstoves or providing affordable rural energy, building local infrastructures such as pagoda, school, community clinic, and/or providing environmental education). Worse yet, political conflicts such as war between various factions or countries could also lead to destruction of forests, and yet this kind of drivers cannot be reduced.

Therefore, it is not simple to provide accurate estimates of the relative project impact. Relative project impact should be used with great caution until alternative methods are introduced. For simplicity, some carbon project developers assumed a 20% reduction of deforestation in relation to baseline deforestation until around 80% of deforestation is halted (Poffenberger, DeGryze and Durschinger, 2009). Thereafter, further deforestation could not be reduced unless plantations (enrichment plantings, reforestation or afforestation) are carried out on deforested lands (Chakravarty, et al., 2012).

Conclusions

This study designed to estimate carbon stocks, carbon emission and reductions and related carbon revenues in the event that REDD+ projects are implemented in the Natural forest in Popa Mountain Park (PMP). Forest cover change between 1989 and 2005 was used as past trend of deforestation upon which deforestation rates in PMP were determined. Forest inventory data from 100 sample plots were used to assess tree biodiversity and to estimate carbon stocks in the park. Forest inventory was conducted in four forest types, namely dry mixed deciduous forest (DMDF), dry dipterocarp forest (DDF), dry forest (DF) and dry hill/evergreen forest (DHEF). The distribution of trees in all forest types in PMP displays an inverse J distribution where stem frequencies decrease with the increase in DBH, indicating stable condition of naturally regenerated trees in the study sites. The study also shows that all forests in PMP have a high degree of floristic heterogeneity. In terms of tree species in individual forest, the number of species found in each of the forests does not vary much among DDF, DF and DHEF. However, the total of 74 species in the DMDF is significantly greater than the other forests. The basal area for the DMDF along with the DDF and DF was significantly lower than that of the DHEF. On the contrary, the density of trees was significantly lower in the DHEF than that in other forest types. But the density of trees in the largest size class was higher in the DHEF than that in other forest types. The occurrence of high basal area and high density of trees in the largest size class suggested that the DHEF was less disturbed than other forests in the PMP. The DF and DMDF had a relatively low proportion of trees in larger size classes due to population pressure and timber harvesting. Large trees with DBH greater than

55 cm in DDF were harvested but there were plentiful trees with small DBH, suggesting that natural regeneration capacity for this forest is good.

In order to know the current carbon stock, this study estimated 5 carbon pools (aboveground, belowground, litter, deadwood and soil) in four natural forest types in PMP. Among carbon pools, soil store highest carbon followed by living tree carbon (aboveground and belowground), litter and deadwood. For aboveground carbon stock, the result shows opposite relationship between carbon stock and tree size class. The total carbon storage capacity in different forest in Popa Mountain Park are different according to forest types. The amount of carbon stock ($tCha^{-1}$) among DMDF, DDF and DF are different according to the forest types but not significant different each other. The total carbon stock was significantly higher in the DHEF than other forest in PMP. This may be different soil type, disturbance regime and human impacts. Beside, species composition and wood density affected the carbon storage capacity of forests. This carbon stock was used as current carbon in the study area and baseline deforestation and carbon emission was estimated by using historical deforestation rate 1989-2005. Baseline scenario calculations show that 20% - 60 % of today forest cover will be loss between 2013 and 2043, resulting in a release of additional 0.25 million tCO_2 – 0.61 million tCO_2. Under the REDD scenario, project is likely to reduce carbon emissions 0.20 million tCO_2 – 0.44 million tCO_2 over a period of 30 years. Using carbon price of US \$5 per tCO_2, a 30-year project would create carbon revenues of about US \$656436.5 to US \$ 1560988.2. Our results highlight the contribution of REDD+ project to reducing deforestation and forest degradation, carbon emissions, and generating carbon revenues.

Our study findings suggested that PMP was rich in terms of tree species but many large trees in accessible forests were harvested except in dry hill or evergreen forest, where large trees still remained in the forests. This was due to less population pressure and limited access to this forest. Considering the structural diversity and the substantially lower density observed in large DBH classes, the PMP has not been effectively protected although it is a designated protected area by the government. Lack of effective mechanisms and funding may have contributed to this failure. Management interventions that take into account the need of local people, tree diversity, and transition of forest stand structures would protect this PMP. Law enforcement mechanism is also important to ensure that government regulation and policies regarding protected area are not violated or punished otherwise. In addition, it is necessary to clarify the land use policy and to cooperate with local communities to maintain the integrity of designated protected areas. It is also fairly necessary to provide incentives to forest dependent communities for their activities to protect this protected area. Forest-dependent communities should be allowed to participate in all decision making processes for sound management of protected areas. Implementing activities on the ground such as restoring degraded forests and protecting the park require participation from both government agency and local communities.

Effective forest conservation and restoration strategies need to know the condition of the degree of forest degradation and deforestation, tree species composition and stand structures. Information provided in our study is useful for introducing future policy interventions, conservation measures, and forest restoration in Popa Mountain Park. However, to achieve this adequate budgets are essential. REDD+ financial incentive could materialize these conservation activities in addition to providing employment to local people. This findings highlight the contribution of REDD+ project to reducing deforestation, carbon emissions reduction, and generating carbon revenues. The expected income from carbon payments for REDD could increase the capacities for forest conservation and help to enforce the existing laws and regulations for the forest management and protection of forest reserves. They could also support activities for sustainable forest management (SFM) and development in communities next to the reserves and help to reduce the risk of encroachment and illegal logging. Therefore, reducing deforestation could result in a huge emission reductions and carbon-based revenues, while improving livelihood of forest dependent communities. Actual emission reductions depend on assumptions of the future implementation of project actions in order to reduce the drivers of the deforestation. These assumption will have to be revised when experience from the project is obtained.

The carbon emission and reduction emissions estimated in this study are for the Popa Mountain Park as a whole; one must have caution in applying them to any specific region or specific forest. Local estimation carbon emission based on deforestation rate and driver of deforestation from the area of interest will generally be more reliable than our estimation which are intended for a more overall area in dry region. Carbon revenues in this study is affected by the carbon price and therefore future adjustment is needed when carbon price is known for some degree of uncertainty. As forest cover change, carbon stocks and drivers of deforestation affect the estimation of emission reductions, further studies on these variables by district are important reducing study biases or uncertainties. This study provides an important step for estimating carbon emissions and reductions due to deforestation and forest degradation in Myanmar. Where benefits of emission reductions are considered, a decision making based on better-informed information on emission reductions and carbon revenues as done in our study would be useful for the adoption of sound forest conservation and management that could achieve multiple benefit.

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APPENDIX

Appendix A Species with the highest importance value in Dry Mixed Deciduous Forest in PMP

Table 17 Species with the highest importance value in Dry Mixed Deciduous Forest in PMP

Scientific Name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Shorea obtusa Wall.	103	3.788	29.486	9.71	2.25	13.76	8.58
Croton roxburghianus N.P.Balakr	100	1.912	11.955	9.43	5.63	6.95	7.34
Pittosporum napaulensis (DG) Rehder Wilson	76	0.932	5.15	7.16	3.94	3.39	4.83
Bixa orellana L.	64	1.087	6.843	6.03	3.10	3.95	4.36
Terminalia crenulata (Heyne) Roth	41	1.833	14.074	3.86	2.25	6.66	4.26
Flacourtia cataphracta Roxb.	45	0.972	5.713	4.24	3.10	3.53	3.62
Litsaea glutinosa (Lour) C. B. Cl.	43	0.945	6.146	4.05	3.38	3.43	3.62
Strychnos potatorum L.f	40	0.52	2.659	3.77	3.38	1.89	3.01
Dipterocarpus tuberculatus Roxb.	17	1.327	16.14	1.60	1.69	4.82	2.70
Diospyros spp.	29	0.47	3.188	2.73	3.38	1.71	2.61
Shorea siamensis (Kurz) Miq.	26	0.862	7.063	2.45	1.97	3.13	2.52
Eriobotrya bengalensis (Roxb.) Hook. f.	30	0.676	4.961	2.83	2.25	2.46	2.51
Dalbergia cultrata Grah.	24	0.593	3.921	2.26	3.10	2.15	2.51
Uncaria pilosa Roxb.	28	0.249	1.15	2.64	3.94	0.90	2.50
Protium serratum Engl.	20	0.611	4.154	1.89	3.10	2.22	2.40
Vitex limnifolia Wall.	23	0.391	2.686	2.17	3.38	1.42	2.32
Albizzia chinensis (Osbeck) Merr.	20	0.455	2.965	1.89	2.54	1.65	2.02
Schleichera oleosa (Lour) Oken.	20	0.269	1.52	1.89	2.82	0.98	1.89
Syzygium cumini (L.) Skeels.	18	0.534	3.782	1.70	1.69	1.94	1.78
Dalbergia oliveri Gamble	15	0.31	2.303	1.41	2.54	1.13	1.69
Careya arborea Roxb.	12	0.444	3.016	1.13	1.97	1.61	1.57
Senna siamea (Lam.) Irwin & Barneby	12	0.708	8.348	1.13	0.85	2.57	1.52
Berrya mollis Wall.ex Kurz	11	0.431	3.147	1.04	1.69	1.57	1.43
Anogeissus acuminata Wall.	9	0.632	9.71	0.85	1.13	2.30	1.42
Millettia pulchra Benth.	13	0.504	3.582	1.23	1.13	1.83	1.39
Pajanelia longifolia (Willdr.) K. Sehum	13	0.228	1.489	1.23	1.97	0.83	1.34
Gmelina arborea Roxb.	5	0.718	6.549	0.47	0.85	2.61	1.31
Bridelia ovata Decne.	10	0.114	0.704	0.94	2.54	0.41	1.30
Mangifera indica L.	15	0.288	1.572	1.41	1.41	1.05	1.29
Prunus cerasoides (Osbeck) Merr.	10	0.308	3.875	0.94	1.69	1.12	1.25
Bauhinia malabarica Roxb.	11	0.285	3.393	1.04	1.41	1.04	1.16
Heterophragma adenophylla (Wall.) Seem. ex Benth. & Hook.	12	0.184	1.027	1.13	1.41	0.67	1.07

Table 17 (cont.)

Scientific Name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Buchanania lanzan Spreng.	10	0.216	1.328	0.94	1.41	0.78	1.05
Bauhinia racemosa Lam.	8	0.228	1.427	0.75	1.13	0.83	0.90
Terminalia bellerica Roxb.	6	0.311	2.629	0.57	0.85	1.13	0.85
Xylia xylocarpa (Roxb.) Toub.	5	0.374	3.549	0.47	0.56	1.36	0.80
Ficus hispida L.f.	4	0.227	1.381	0.38	1.13	0.82	0.78
Trema tomentosa (Roxb.)Hara	14	0.062	0.22	1.32	0.56	0.23	0.70
Grewia tiliifolia Vahal.	6	0.07	0.384	0.57	1.13	0.25	0.65
Schrebera swietenioides Roxb.	4	0.272	2.534	0.38	0.56	0.99	0.64
Premna pyramidata Wall.	6	0.059	0.253	0.57	1.13	0.21	0.64
Tectona grandis L.f	7	0.154	0.613	0.66	0.56	0.56	0.59
Eucalyptus Camaldulensis Dehnh.	7	0.226	2.405	0.66	0.28	0.82	0.59
Emblica officinalis Gaertn.	6	0.039	0.15	0.57	0.85	0.14	0.52
Engelhardtia spicata Blume	4	0.089	0.601	0.38	0.85	0.32	0.52
Cassia glauca Lam.	5	0.034	0.142	0.47	0.85	0.12	0.48
Chionanthus ramiflora Roxb.	4	0.048	0.323	0.38	0.85	0.17	0.47
Vitex pubescens Vahl	4	0.022	0.055	0.38	0.85	0.08	0.43
Ficus mysoriensis Heyne	1	0.229	2.094	0.09	0.28	0.83	0.40
Sapium baccatum Roxb.	1	0.229	4.397	0.09	0.28	0.83	0.40
Ficus altissima Blume	1	0.204	1.569	0.09	0.28	0.74	0.37
Diospyros burmanica Kurz	3	0.132	1.096	0.28	0.28	0.48	0.35
Wendlandia grandis (Hook. f.) Cowan	4	0.018	0.056	0.38	0.56	0.07	0.34
Cissus discolor Blume	3	0.017	0.045	0.28	0.56	0.06	0.30
Ficus hirta Vahl.	2	0.031	0.285	0.19	0.56	0.11	0.29
Lophopetalum wallichii Kurz	4	0.054	0.317	0.38	0.28	0.20	0.28
Terminalia chebula Retz.	2	0.105	1.027	0.19	0.28	0.38	0.28
Bombax insigne Wall.	2	0.021	0.081	0.19	0.56	0.08	0.28
Zizyphus rugosa Lam.	2	0.021	0.126	0.19	0.56	0.08	0.28
Stereospermum colais (Buch.-Ham.ex Dillwyn) Mabb.	2	0.089	0.889	0.19	0.28	0.32	0.26
Rhus paniculata Wall.	2	0.011	0.035	0.19	0.56	0.04	0.26
Ficus obtusifolia Roxb.	1	0.096	0.44	0.09	0.28	0.35	0.24
Lannea coromandelica (Houtt). Merr.	2	0.053	0.367	0.19	0.28	0.19	0.22
Chukrasia velutina Roem.	1	0.071	1.034	0.09	0.28	0.26	0.21
Ficus obscura Blume	2	0.039	0.208	0.19	0.28	0.14	0.20
Cassia fistula L.	2	0.021	0.089	0.19	0.28	0.08	0.18
Hiptage candicans Hook.f.	2	0.01	0.022	0.19	0.28	0.04	0.17
Bridelia retusa (L.) A.Juss	1	0.02	0.085	0.09	0.28	0.07	0.15
Wendlandia tinctoria DC.	1	0.013	0.073	0.09	0.28	0.05	0.14

Table 17 (cont.)

Scientific Name	SD (trees ha-1)	BA (trees ha-1)	Vol (trees ha-1)	RD (%)	RF (%)	RBA (%)	IVI (%)
Vitex peduncularis Wall	1	0.008	0.029	0.09	0.28	0.03	0.14
Toona ciliate M. Roemer	1	0.006	0.023	0.09	0.28	0.02	0.13
Cordyline fruticosa (L.) A. Chev.	1	0.004	0.014	0.09	0.28	0.01	0.13
Diospyros montana Roxb.	1	0.004	0.018	0.09	0.28	0.01	0.13
Cassia renigera Wall.	1	0.003	0.008	0.09	0.28	0.01	0.13
	1061	**27.52**	**214.69**	**100**	**100**	**100**	**100**

SD = stand Density, BA = Basal Area, Vol = Volume, RD = Relative Density, RF = Relative Freequency, RBA = Relative Basal Area, IVI = Important value index.

Appendix B Species with the highest importance value in Dry Dipterocarp Forest in PMP

Table 18 Species with the highest importance value in Dry Dipterocarp Forest in PMP

Scientific Name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Shorea obtusa Wall.	251	6.63	34.82	19.41	6.19	28.91	18.17
Dipterocarpus tuberculatus Roxb.	172	5.19	34.21	13.30	5.93	22.65	13.96
Shorea siamensis (Kurz) Miq.	121	2.16	12.38	9.36	5.67	9.43	8.15
Terminalia crenulata (Heyne) Roth	93	1.42	6.86	7.19	6.19	6.20	6.53
Dalbergia oliveri Gamble	80	0.97	5.93	6.19	5.67	4.23	5.36
Buchanania lanzan Spreng.	49	0.99	6.03	3.79	5.93	4.33	4.68
Premna pyramidata Wall.	47	1.05	6.33	3.63	4.64	4.58	4.29
Chionanthus ramiflora Roxb.	54	0.41	1.52	4.18	4.12	1.79	3.36
Diospyros burmanica Kurz	49	0.59	2.93	3.79	3.61	2.56	3.32
Wendlandia tinctoria DC.	44	0.34	1.15	3.40	3.87	1.47	2.91
Rhus paniculata Wall.	34	0.17	0.52	2.63	3.61	0.76	2.33
Careya arborea Roxb.	38	0.23	0.91	2.94	2.58	1.00	2.17
Xylia xylocarpa (Roxb.) Toub.	21	0.29	1.87	1.62	2.84	1.26	1.91
Vitex peduncularis Wall.	25	0.15	0.55	1.93	3.09	0.66	1.90
Pittosporum napaulensis (DG) Rehder Wilson"	20	0.24	1.19	1.55	2.32	1.06	1.64
Strychnos nuxblanda A.W. Hill	17	0.34	1.55	1.31	2.06	1.50	1.63
Strychnos potatorum L.f	13	0.15	0.73	1.01	2.58	0.66	1.42
Flacourtia cataphracta Roxb.	13	0.15	0.74	1.01	2.32	0.67	1.33
Tectona grandis L.f	12	0.09	0.31	0.93	2.32	0.37	1.21
Vitex limonifolia Wall.	14	0.14	0.75	1.08	1.80	0.63	1.17
Lannea coromandelica (Houtt).Merr.	11	0.13	0.52	0.85	2.06	0.55	1.15
Morinda tinctoria Roxb.	11	0.11	0.50	0.85	2.06	0.47	1.13
Zizyphus rugosa Lam.	14	0.07	0.24	1.08	1.80	0.32	1.07
Bridelia ovata Decne.	10	0.05	0.21	0.77	2.06	0.23	1.02
Hiptage candicans Hook.f.	12	0.09	0.36	0.93	1.55	0.40	0.96
Dalbergia cultrata Grah.	9	0.10	0.47	0.70	1.03	0.44	0.72

Table 18 (cont.)

Scientific Name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Protium serratum Engl.	6	0.07	0.38	0.46	1.29	0.30	0.68
Lophopetalum wallichii Kurz	4	0.09	0.47	0.31	1.03	0.41	0.58
Uncaria pilosa Roxb.	7	0.03	0.13	0.54	1.03	0.15	0.57
Terminalia chebula Retz.	4	0.03	0.12	0.31	0.77	0.11	0.40
Heterophragma adenophylla (Wall) Seem. ex Benth&Hook.	4	0.04	0.18	0.31	0.52	0.19	0.34
Tectona hamiltoniana Wall.	3	0.12	0.68	0.23	0.26	0.50	0.33
Emblica officinalis Gaertn.	3	0.01	0.04	0.23	0.52	0.06	0.27
Pajanelia longifolia (Willdr.) K. Sehum	2	0.03	0.16	0.15	0.52	0.13	0.27
Schleichera oleosa (Lour) Oken.	2	0.02	0.11	0.15	0.52	0.10	0.26
Croton roxburghianus N.P.Balakr	2	0.02	0.12	0.15	0.52	0.10	0.26
Acacia laevis Parker	2	0.02	0.06	0.15	0.52	0.09	0.25
Gardenia sessiliflora Wall.	2	0.02	0.07	0.15	0.52	0.07	0.25
Bauhinia racemosa Lamk.	2	0.01	0.03	0.15	0.52	0.05	0.24
Engelhardtia spicata Blume	2	0.04	0.23	0.15	0.26	0.18	0.20
Bixa orellana L.	1	0.03	0.14	0.08	0.26	0.12	0.15
Diospyros spp.	2	0.01	0.03	0.15	0.26	0.03	0.15
Terminalia oliveri Brandis	1	0.01	0.10	0.08	0.26	0.06	0.13
Phyllantus spp.	1	0.01	0.04	0.08	0.26	0.04	0.13
Bombax insigne Wall.	1	0.01	0.07	0.08	0.26	0.04	0.12
Berrya mollis Wall.ex Kurz	1	0.01	0.02	0.08	0.26	0.03	0.12
Cassia fistula L.	1	0.01	0.01	0.08	0.26	0.03	0.12
Melanorrhoea usitata Wall.	1	0.01	0.02	0.08	0.26	0.03	0.12
Carissa carandas L.	1	0.01	0.02	0.08	0.26	0.02	0.12
Acacia pennata (L.) Willd.	1	0.00	0.02	0.08	0.26	0.02	0.12
Chukrasia velutina Roem.	1	0.00	0.01	0.08	0.26	0.02	0.12
Walsura villosa Wall.ex.W&A	1	0.00	0.01	0.08	0.26	0.02	0.12
Antidesma diandrm Roth.	1	0.00	0.00	0.08	0.26	0.01	0.12
	1293	**22.93**	**126.80**	**100**	**100**	**100**	**100**

SD = stand Density, BA = Basal Area, Vol = Volume, RD = Relative Density, RF = Relative Freequency, RBA = Relative Basal Area, IVI = Important value index.

Appendix C Species with the highest importance value in Dry Forest (PMP)

Table 19 Species with the highest importance value in Dry Forest (PMP)

SF name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Tectona hamiltoniana Wall.	610	20.32	160.97	70.03	17.12	75.59	54.25
Terminalia oliveri Brandis	75	3.48	39.71	8.61	13.01	12.94	11.52
Tectona grandis L.f	42	0.50	2.51	4.82	6.16	1.88	4.29
Lannea coromandelica (Houtt). Merr.	18	0.28	1.74	2.07	8.22	1.03	3.77
Dalbergia oliveri Gamble	18	0.56	5.05	2.07	6.16	2.10	3.44
Diospyros burmanica Kurz	10	0.12	0.52	1.15	2.74	0.45	1.44
Morinda tinctoria Roxb.	8	0.07	0.21	0.92	2.74	0.24	1.30
Acacia catechu Willd.	6	0.12	0.74	0.69	2.74	0.43	1.29
Dalbergia cultrata Grah.	4	0.09	0.52	0.46	2.74	0.32	1.17
Albizzia chinensis (Osbeck) Merr.	3	0.25	2.80	0.34	2.05	0.94	1.11
Schleichera oleosa (Lour) Oken.	6	0.06	0.18	0.69	2.05	0.22	0.99
Terminalia crenulata (Heyne) Roth	5	0.03	0.12	0.57	2.05	0.11	0.91
Boscia variabilis Collett & Hemsl.	5	0.17	0.80	0.57	1.37	0.63	0.86
Bombax insigne Wall.	3	0.03	0.07	0.34	2.05	0.10	0.83
Premna pyramidata Wall.	3	0.02	0.12	0.34	2.05	0.09	0.83
Lagertroemia villosa Wall. ex. Kurz	6	0.05	0.27	0.69	1.37	0.17	0.74
Capparis glauca Wall.	3	0.13	0.42	0.34	1.37	0.47	0.73
Xylia xylocarpa (Roxb.) Toub.	5	0.04	0.13	0.57	1.37	0.14	0.70
Anogeissus acuminata Wall.	3	0.01	0.05	0.34	1.37	0.04	0.59
Flacourtia cataphracta Roxb.	2	0.03	0.21	0.23	1.37	0.12	0.57
Shorea siamensis (Kurz) Miq.	2	0.03	0.18	0.23	1.37	0.12	0.57
Acacia pennata Willd.	2	0.01	0.06	0.23	1.37	0.05	0.55
Bridelia ovata Decne.	2	0.01	0.05	0.23	1.37	0.05	0.55
Vitex limonifolia Wall.	2	0.01	0.02	0.23	1.37	0.03	0.54
Grewia tiliifolia Vahal.	4	0.04	0.34	0.46	0.68	0.15	0.43
Protium serratum Engl.	2	0.08	0.48	0.23	0.68	0.29	0.40
Millettia pendula Benth.	3	0.03	0.07	0.34	0.68	0.11	0.38
Miliusa velutina Hook.f. & Thoms	1	0.08	0.88	0.11	0.68	0.30	0.37
Pterocarpus macrocarpus Kurz	1	0.06	0.37	0.11	0.68	0.21	0.34
Cassia racemosa Lamk.	2	0.01	0.03	0.23	0.68	0.03	0.32

Table 19 (cont.)

SF name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Buchanania lanzan Spreng.	1	0.04	0.21	0.11	0.68	0.14	0.31
Shorea obtusa Wall.	1	0.03	0.05	0.11	0.68	0.10	0.30
Azadirachta indica A. Juss.	1	0.02	0.17	0.11	0.68	0.09	0.30
Chukrasia velutina Roem.	1	0.02	0.09	0.11	0.68	0.07	0.29
Croton roxburghianus N.P.Balakr	1	0.01	0.04	0.11	0.68	0.04	0.28
Lagerstroemia speciosa (L.) Pers.	1	0.01	0.05	0.11	0.68	0.04	0.28
Bauhinia velutina Wall.	1	0.01	0.01	0.11	0.68	0.03	0.28
Aegle marmelos (L.) Coorea.	1	0.01	0.04	0.11	0.68	0.03	0.28
Erythrina stricta Roxb.	1	0.01	0.02	0.11	0.68	0.02	0.27
Osyris wightiana Wall.	1	0.01	0.01	0.11	0.68	0.02	0.27
Albizia lebbek (L.) Benth.	1	0.00	0.01	0.11	0.68	0.01	0.27
Diospyros montana Roxb.	1	0.00	0.01	0.11	0.68	0.01	0.27
Millettia pulchra Benth.	1	0.00	0.01	0.11	0.68	0.01	0.27
Uncaria pilosa Roxb.	1	0.00	0.01	0.11	0.68	0.01	0.27
Zizyphus rugosa Lam.	1	0.00	0.01	0.11	0.68	0.01	0.27
	871	**26.876**	**220.329**	**100**	**100**	**100**	**100**

SD = stand Density, BA = Basal Area, Vol = Volume, RD = Relative Density, RF = Relative Freequency, RBA = Relative Basal Area, IVI = Important value index.

Appendix D Species with the highest importance value in Dry Hill/ Evergreen Forest (PMP)

Table 20 Species with the highest importance value in Dry Hill/ Evergreen Forest (PMP)

Scientific Name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Vitex canescens Kurz	263	20.072	284.518	32.71	12.72	41.96	29.13
Rapanea af. Neriifolia (Seib & Zucc) Mez.	160	5.467	50.423	19.90	8.09	11.43	13.14
Bixa orellana L.	63	1.954	15.201	7.84	9.25	4.09	7.06
Eriobotrya bengalensis (Roxb.) Hook. f.	32	3.17	57.186	3.98	5.78	6.63	5.46
Syzygium cumini (L.) Skeels.	22	2.226	24.490	2.74	8.09	4.65	5.16
Wendlandia tinctoria DC.	43	1.314	13.114	5.35	6.36	2.75	4.82
Croton roxburghianus N.P.Balakr	44	1.483	11.175	5.47	4.62	3.10	4.40
Litsaea glutino (Lour)C.B.Cl.	30	2.623	25.793	3.73	3.47	5.48	4.23
Cinnamomum obtusifolium (Roxb.) Nees	27	0.618	5.119	3.36	4.62	1.29	3.09
Cissus discolor Blume	20	0.191	0.897	2.49	5.20	0.40	2.70
Sapium baccatum Roxb.	4	1.741	24.220	0.50	2.31	3.64	2.15
Berrya mollis Wall.ex Kurz	5	0.605	8.812	0.62	2.89	1.26	1.59
Flacourtia cataphracta Roxb.	5	1.144	20.356	0.62	1.73	2.39	1.58
Toona ciliata M. Roemer	8	1.005	16.767	1.00	1.16	2.10	1.42
Holarrhena pubescens Wall. ex G. Don	3	0.792	14.879	0.37	1.73	1.66	1.25
Albizzia chinensis (Osbeck) Merr.	5	0.544	5.730	0.62	1.73	1.14	1.16
Dalbergia oliveri Gamble	8	0.359	3.392	1.00	1.73	0.75	1.16
Pittosporum nepaulensis (DC.) Rehoto & Wilson	10	0.464	5.155	1.24	1.16	0.97	1.12
Michelia champaca L.	5	0.455	6.980	0.62	0.58	0.95	0.72
Mangifera indica L.	6	0.1	0.620	0.75	1.16	0.21	0.70
Diospyros spp.	4	0.092	0.574	0.50	1.16	0.19	0.62
Schrebera swietenioides Roxb.	2	0.165	3.689	0.25	1.16	0.34	0.58
Protium serratum Engl.	6	0.195	1.724	0.75	0.58	0.41	0.58
Cordyline fruticosa (L.) A. Chev.	3	0.074	0.935	0.37	1.16	0.15	0.56
Artocarpus heterophyllus	3	0.019	0.102	0.37	1.16	0.04	0.52
Engelhardtia spicata Blume	2	0.349	4.117	0.25	0.58	0.73	0.52
Uncaria pilosa Roxb.	2	0.033	0.118	0.25	1.16	0.07	0.49
Zizyphus rugosa Lam.	2	0.015	0.039	0.25	1.16	0.03	0.48
Acacia laevis Parker	2	0.01	0.071	0.25	1.16	0.02	0.48

Table 20 (cont.)

Scientific Name	SD (trees ha^{-1})	BA (trees ha^{-1})	Vol (trees ha^{-1})	RD (%)	RF (%)	RBA (%)	IVI (%)
Pinus insularis Engl.	2	0.225	2.472	0.25	0.58	0.47	0.43
Chionanthus ramiflora Roxb.	2	0.079	0.395	0.25	0.58	0.17	0.33
Xylia xylocarpa (Roxb.) Taub.	2	0.068	0.502	0.25	0.58	0.14	0.32
Dalbergia cultrata Grah.	1	0.075	0.670	0.12	0.58	0.16	0.29
Cassia fistula L.	2	0.01	0.038	0.25	0.58	0.02	0.28
Toddalia aculeata Pers.	1	0.025	0.113	0.12	0.58	0.05	0.25
Careya arborea Roxb.	1	0.023	0.028	0.12	0.58	0.05	0.25
Diospyros burmanica Kurz	1	0.019	0.127	0.12	0.58	0.04	0.25
Grewia tiliifolia Vahal.	1	0.015	0.123	0.12	0.58	0.03	0.24
Senna siamea (Lam.) Irwin & Barneby	1	0.009	0.042	0.12	0.58	0.02	0.24
Carissa carandas L.	1	0.004	0.010	0.12	0.58	0.01	0.24
	804	**47.831**	**610.716**	**100**	**100**	**100**	**100**

SD = stand Density, BA = Basal Area, Vol = Volume, RD = Relative Density, RF = Relative Freequency, RBA = Relative Basal Area, IVI = Important value index.

Appendix E Species Carbon storage in the Dry Mixed Deciduous Forest

Table 21 Species Carbon storage in the Dry Mixed Deciduous Forest

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Shorea obtusa Wall.	20.11	103	3.79	29.49	8.58	20.21
Dipterocarpus tuberculatus Roxb.	29.41	17	1.33	16.14	2.70	8.52
Terminalia crenulata (Heyne) Roth	20.85	41	1.83	14.07	4.26	8.43
Croton roxburghianus N.P.Balakr	14.31	100	1.91	11.96	7.34	6.14
Anogeissus acuminata Wall.	24.33	9	0.63	9.71	1.42	6.05
Shorea siamensis (Kurz) Miq.	18.17	26	0.86	7.06	2.52	4.64
Senna siamea (Lam.) Irwin & Barneby	26.67	12	0.71	8.35	1.52	4.16
Flacourtia cataphracta Roxb.	14.71	45	0.97	5.71	3.62	3.77
Pittosporum napaulensis (DG) Rehder Wilson	11.68	76	0.93	5.15	4.83	2.68
Litsaea glutinosa (Lour) C. B. Cl.	15.67	43	0.95	6.15	3.62	2.66
Eriobotrya bengalensis (Roxb.) Hook. f.	14.25	30	0.68	4.96	2.51	2.66
Gmelina arborea Roxb.	38.60	5	0.72	6.55	1.31	2.31
Dalbergia cultrata Grah.	16.71	24	0.59	3.92	2.51	2.27
Syzygium cumini (L.) Skeels.	18.67	18	0.53	3.78	1.78	2.22
Xylia xylocarpa (Roxb.) Toub.	29.80	5	0.37	3.55	0.80	2.21
Bixa orellana L.	13.31	64	1.09	6.84	4.36	2.01
Berrya mollis Wall.ex Kurz	20.64	11	0.43	3.15	1.43	2.00
Protium serratum Engl.	16.70	20	0.61	4.15	2.40	1.98
Careya arborea Roxb.	19.25	12	0.44	3.02	1.57	1.96
Bauhinia malabarica Roxb.	14.45	11	0.29	3.39	1.16	1.83
Prunus cerasoides (Osbeck) Merr.	16.00	10	0.31	3.88	1.25	1.79
Dalbergia oliveri Gamble	15.00	15	0.31	2.30	1.69	1.72

Table 21 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Diospyros spp.	13.07	29	0.47	3.19	2.61	1.61
Eucalyptus Camaldulensis Dehnh.	19.57	7	0.23	2.41	0.59	1.61
Millettia pulchra Benth.	21.38	13	0.50	3.58	1.39	1.55
Diospyros burmanica Kurz	21.00	3	0.13	1.10	0.35	1.53
Terminalia bellerica Roxb.	24.50	6	0.31	2.63	0.85	1.50
Vitex limnifolia Wall.	12.54	23	0.39	2.69	2.32	1.46
Strychnos potatorum L.f	11.75	40	0.52	2.66	3.01	1.42
Schrebera swietenioides Roxb.	28.88	4	0.27	2.53	0.64	1.39
Sapium baccatum Roxb.	54.00	1	0.23	4.40	0.40	1.16
Schleichera oleosa (Lour) Oken.	11.88	20	0.27	1.52	1.89	1.13
Albizzia chinensis (Osbeck) Merr.	15.90	20	0.46	2.97	2.02	1.02
Mangifera indica L.	15.07	15	0.29	1.57	1.29	0.77
Ficus mysoriensis Heyne	54.00	1	0.23	2.09	0.40	0.77
Uncaria pilosa Roxb.	10.00	28	0.25	1.15	2.50	0.72
Terminalia chebula Retz.	25.50	2	0.11	1.03	0.28	0.66
Bauhinia racemosa Lam.	18.38	8	0.23	1.43	0.90	0.66
Chukrasia velutina Roem.	30.00	1	0.07	1.03	0.21	0.54
Heterophragma adenophylla (Wall.) Seem. ex Benth. & Hook.	13.00	12	0.18	1.03	1.07	0.46
Ficus hispida L.f.	25.00	4	0.23	1.38	0.78	0.45
Stereospermum colais (Buch.-Ham.ex Dillwyn) Mabb.	22.00	2	0.09	0.89	0.26	0.42
Bridelia ovata Decne.	11.50	10	0.11	0.70	1.30	0.39
Buchanania lanzan Spreng.	15.13	10	0.22	1.33	1.05	0.38
Pajanelia longifolia (Willdr.) K. Sehum	13.62	13	0.23	1.49	1.34	0.36

Table 21 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Tectona grandis L.f	15.29	7	0.15	0.61	0.59	0.32
Ficus altissima Blume	51.00	1	0.20	1.57	0.37	0.26
Engelhardtia spicata Blume	15.25	4	0.09	0.60	0.52	0.21
Grewia tiliifolia Vahal.	12.05	6	0.07	0.38	0.65	0.21
Ficus obtusifolia Roxb.	35.00	1	0.10	0.44	0.24	0.16
Chionanthus ramiflora Roxb.	11.50	4	0.05	0.32	0.47	0.16
Premna pyramidata Wall.	10.83	6	0.06	0.25	0.64	0.13
Lannea coromandelica (Houtt). Merr.	17.00	2	0.05	0.37	0.22	0.12
Ficus hirta Vahl.	12.50	2	0.03	0.29	0.29	0.11
Zizyphus rugosa Lam.	11.25	2	0.02	0.13	0.28	0.10
Lophopetalum wallichii Kurz	12.25	4	0.05	0.32	0.28	0.10
Emblica officinalis Gaertn.	8.67	6	0.04	0.15	0.52	0.09
Cassia glauca Lam.	9.20	5	0.03	0.14	0.48	0.08
Ficus obscura Blume	15.50	2	0.04	0.21	0.20	0.08
Cassia fistula L.	11.50	2	0.02	0.09	0.18	0.06
Trema tomentosa (Roxb.)Hara	7.29	14	0.06	0.22	0.70	0.06
Bridelia retusa (L.) A.Juss	16.00	1	0.02	0.09	0.15	0.04
Wendlandia tinctoria DC.	13.00	1	0.01	0.07	0.14	0.04
Wendlandia grandis (Hook. f.) Cowan	7.50	4	0.02	0.06	0.34	0.03
Vitex pubescens Vahl	8.25	4	0.02	0.06	0.43	0.03
Bombax insigne Wall.	11.50	2	0.02	0.08	0.28	0.02
Cissus discolor Blume	8.33	3	0.02	0.05	0.30	0.02
Vitex peduncularis Wall	10.00	1	0.01	0.03	0.14	0.02
Rhus paniculata Wall.	8.00	2	0.01	0.04	0.26	0.01
Diospyros montana Roxb.	7.00	1	0.00	0.02	0.13	0.01

Table 21 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Hiptage candicans Hook.f.	8.00	2	0.01	0.02	0.17	0.01
Toona ciliate M. Roemer	9.00	1	0.01	0.02	0.13	0.01
Cordyline fruticosa (L.) A. Chev.	7.00	1	0.00	0.01	0.13	0.01
Cassia renigera Wall.	6.00	1	0.00	0.01	0.13	0.00
		1061	**27.52**	**214.69**	**100.00**	**116.65**

*DBH = Diameter at Breast High, SD = Stand Density, BA = Basal Area, Vol = Volume, IVI = Important Value Index, AG-C = Aboveground Carbon.

Appendix F Species Carbon storage in the Dry Dipterocarp Forest

Table 22 Species Carbon storage in the Dry Dipterocarp Forest

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Shorea obtusa Wall.	16.00	251	6.629	34.817	0.25	28.30
Dipterocarpus tuberculatus Roxb.	17.50	172	5.193	34.205	0.12	21.30
Shorea siamensis (Kurz) Miq.	13.90	121	2.163	12.381	0.12	9.64
Terminalia crenulata (Heyne) Roth	12.50	93	1.422	6.857	0.24	4.92
Dalbergia oliveri Gamble	11.50	80	0.971	5.929	0.12	5.23
Diospyros burmanica Kurz	11.70	49	0.587	2.927	0.15	4.84
Premna pyramidata Wall.	15.70	47	1.051	6.328	0.12	3.89
Buchanania lanzan Spreng.	14.60	49	0.992	6.032	1.02	2.04
Xylia xylocarpa (Roxb.) Toub.	12.40	21	0.290	1.870	4.68	1.38
Strychnos nux-blanda A.W. Hill	13.90	17	0.344	1.550	2.17	1.29
Chionanthus ramiflora Roxb.	9.50	54	0.410	1.517	0.12	0.88
Wendlandia tinctoria DC.	9.30	44	0.336	1.152	0.12	0.75
Pittosporum napaulensis (DG) Rehder Wilson"	11.40	20	0.242	1.189	3.36	0.73
Careya arborea Roxb.	8.40	38	0.229	0.911	0.12	0.70
Flacourtia cataphracta Roxb.	11.50	13	0.154	0.741	0.26	0.58
Vitex limonifolia Wall.	11.10	14	0.144	0.750	0.72	0.48
Tectona hamiltoniana Wall.	21.30	3	0.115	0.679	5.36	0.48
Strychnos potatorum L.f	11.30	13	0.152	0.725	3.32	0.46
Vitex peduncularis Wall.	8.60	25	0.152	0.550	0.15	0.35
Dalbergia cultrata Grah.	10.90	9	0.101	0.471	13.96	0.32
Morinda tinctoria Roxb.	10.70	11	0.107	0.504	0.27	0.26
Rhus paniculata Wall.	7.90	34	0.174	0.522	0.20	0.23
Lannea coromandelica (Houtt).Merr.	11.80	11	0.125	0.517	1.33	0.19
Zizyphus rugosa Lam.	8.00	14	0.073	0.241	0.25	0.23
Protium serratum Engl.	11.10	6	0.068	0.382	0.34	0.22

Table 22 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Lophopetalum wallichii Kurz	14.70	4	0.093	0.474	0.96	0.18
Hiptage candicans Hook.f.	9.50	12	0.091	0.357	1.15	0.18
Tectona grandis L.f	9.20	12	0.085	0.305	0.58	0.19
Bridelia ovata Decne.	7.90	10	0.052	0.206	0.12	0.13
Engelhardtia spicata Blume	15.50	2	0.041	0.233	1.13	0.10
Heterophragma adenophylla (Wall) Seem. ex Benth&Hook.	10.60	4	0.043	0.177	0.27	0.09
Uncaria pilosa Roxb.	7.60	7	0.034	0.134	0.13	0.10
Schleichera oleosa (Lour) Oken.	12.00	2	0.023	0.107	1.64	0.09
Terminalia chebula Retz.	8.30	4	0.025	0.120	4.29	0.09
Croton roxburghianus N.P.Balakr	11.50	2	0.022	0.116	0.68	0.07
Terminalia oliveri Brandis	13.50	1	0.014	0.097	2.33	0.06
Bixa orellana L.	19.00	1	0.028	0.140	0.26	0.05
Pajanelia longifolia (Willdr.) K. Sehum	12.50	2	0.029	0.162	18.17	0.05
Gardenia sessiliflora Wall.	9.70	2	0.015	0.072	8.15	0.05
Acacia laevis Parker	10.50	2	0.020	0.056	1.63	0.03
Emblica officinalis Gaertn.	7.70	3	0.014	0.039	1.42	0.03
Phyllantus spp.	11.50	1	0.010	0.038	1.21	0.03
Bombax insigne Wall.	11.00	1	0.009	0.066	0.33	0.02
Bauhinia racemosa Lamk.	8.20	2	0.012	0.030	0.40	0.02
Berrya mollis Wall.ex Kurz	9.40	1	0.007	0.019	6.53	0.01
Diospyros spp.	6.80	2	0.007	0.025	0.13	0.02
Melanorrhoea usitata Wall.	9.00	1	0.006	0.017	0.57	0.01
Carissa carandas L.	8.00	1	0.005	0.017	1.17	0.01
Cassia fistula L.	9.00	1	0.006	0.012	1.90	0.01
Chukrasia velutina Roem.	7.00	1	0.004	0.014	0.12	0.01

Table 22 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Acacia pennata (L.) Willd.	7.50	1	0.004	0.015	2.91	0.01
Walsura villosa Wall.ex.W&A	7.00	1	0.004	0.007	1.91	0.01
Tatal		1293	22.930	126.806	100.00	91.28

*DBH = Diameter at Breast High, SD = Stand Density, BA = Basal Area, Vol = Volume, IVI = Important Value Index, AG-C = Aboveground Carbon.

Appendix G Species Carbon storage in the Dry Forest

Table 23 Species Carbon storage in the Dry Forest

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Tectona hamiltoniana Wall.	18.00	610	0.01	0.05	0.27	93.68
Terminalia oliveri Brandis	19.90	75	0.02	0.19	0.27	21.70
Dalbergia oliveri Gamble	17.30	18	0.01	0.06	0.70	3.65
Albizzia chinensis (Osbeck) Merr.	32.20	3	0.13	1.57	3.44	1.63
Tectona grandis L.f	11.10	42	0.01	0.01	0.28	1.25
Diospyros burmanica Kurz	11.70	10	0.02	0.11	0.59	0.70
Lannea coromandelica (Houtt). Merr.	12.90	18	0.01	0.03	0.40	0.53
Acacia catechu Willd.	14.70	6	0.01	0.03	54.25	0.50
Boscia variabilis Collett & Hemsl.	18.50	5	0.06	0.35	1.11	0.47
Miliusa velutina Hook.f. & Thoms	32.00	1	0.08	0.88	0.38	0.39
Dalbergia cultrata Grah.	16.30	4	0.02	0.09	0.73	0.29
Capparis glauca Wall.	22.70	3	0.04	0.10	0.86	0.27
Protium serratum Engl.	22.00	2	0.05	0.29	0.30	0.22
Pterocarpus macrocarpus Kurz	27.00	1	0.06	0.37	0.29	0.18
Grewia tiliifolia Vahal.	11.00	4	0.02	0.18	0.55	0.18
Lagertroemia villosa Wall. ex. Kurz	9.60	6	0.02	0.14	0.43	0.15
Schleichera oleosa (Lour) Oken.	10.50	6	0.00	0.01	0.28	0.13
Flacourtia cataphracta Roxb.	13.80	2	0.03	0.19	0.55	0.12

Table 23 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Shorea siamensis (Kurz) Miq.	12.80	2	0.03	0.17	0.28	0.11
Azadirachta indica A. Juss.	17.00	1	0.02	0.17	1.30	0.10
Morinda tinctoria Roxb.	9.90	8	0.02	0.05	0.32	0.09
Terminalia crenulata (Heyne) Roth	8.60	5	0.01	0.04	0.27	0.07
Premna pyramidata Wall.	9.70	3	0.01	0.03	0.30	0.06
Buchanania lanzan Spreng.	22.00	1	0.04	0.21	0.91	0.06
Chukrasia velutina Roem.	16.00	1	0.02	0.09	0.83	0.05
Shorea obtusa Wall.	19.00	1	0.03	0.05	0.28	0.04
Anogeissus acuminata Wall.	7.00	3	0.01	0.02	1.45	0.03
Aegle marmelos (L.) Coorea.	10.00	1	0.01	0.04	4.29	0.03
Millettia pendula Benth.	10.80	3	0.01	0.02	0.37	0.03
Bridelia ovata Decne.	9.00	2	0.01	0.04	0.99	0.03
Acacia pennata Willd.	9.00	2	0.01	0.01	11.52	0.02
Lagerstroemia speciosa (L.) Pers.	12.00	1	0.01	0.05	0.54	0.02
Croton roxburghianus N.P.Balakr	12.00	1	0.01	0.04	0.74	0.02
Bombax insigne Wall.	9.80	3	0.01	0.03	1.17	0.02
Cassia racemosa Lamk.	7.50	2	0.00	0.01	0.83	0.02
Osyris wightiana Wall.	9.00	1	0.01	0.01	0.31	0.01
Bauhinia velutina Wall.	10.50	1	0.01	0.01	1.29	0.01
Diospyros montana Roxb.	7.00	1	0.00	0.01	0.57	0.01

Table 23 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Xylia xylocarpa (Roxb.) Toub.	9.60	5	0.00	0.01	0.27	0.01
Zizyphus rugosa Lam.	6.00	1	0.00	0.01	0.27	0.01
Vitex limonifolia Wall.	6.50	2	0.00	0.01	0.27	0.01
Albizia lebbek (L.) Benth.	7.00	1	0.00	0.01	3.77	0.00
Erythrina stricta Roxb.	9.00	1	0.01	0.02	0.57	0.00
Millettia pulchra Benth.	6.00	1	0.00	0.01	0.34	0.00
Uncaria pilosa Roxb.	6.50	1	0.00	0.01	0.27	0.00
Total		**871**	**0.86**	**5.83**	**100.00**	**126.87**

*DBH = Diameter at Breast High, SD = Stand Density, BA = Basal Area, Vol = Volume, IVI = Important Value Index, AG-C = Aboveground Carbon.

Appendix H Species Carbon storage in the Dry Hill or Evergreen Forest

Table 24 Species Carbon storage in the Dry Hill or Evergreen Forest

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Vitex canescens Kurz	24.93	263	20.07	284.52	29.13	82.21
Eriobotrya bengalensis (Roxb.) Hook. f.	27.84	32	3.17	57.19	5.46	22.76
Rapanea af. Neriifolia (Seib & Zucc) Mez.	18.63	160	5.47	50.42	13.14	19.52
Syzygium cumini (L.) Skeels.	32.36	22	2.23	24.49	5.16	10.68
Flacourtia cataphracta Roxb.	52.80	5	1.14	20.36	1.58	9.99
Litsaea glutino (Lour)C.B.Cl.	30.25	30	2.62	25.79	4.23	8.30
Holarrhena pubescens Wall. ex G. Don	55.00	3	0.79	14.88	1.25	5.76
Wendlandia tinctoria DC.	16.74	43	1.31	13.11	4.82	5.36
Sapium baccatum Roxb.	74.00	4	1.74	24.22	2.15	4.75
Croton roxburghianus N.P.Balakr	19.75	44	1.48	11.18	4.40	4.27
Berrya mollis Wall.ex Kurz	38.90	5	0.61	8.81	1.59	4.18
Toona ciliata M. Roemer	36.00	8	1.01	16.77	1.42	3.93
Bixa orellana L.	17.91	63	1.95	15.20	7.06	3.32
Michelia champaca L.	31.10	5	0.46	6.98	0.72	2.13
Pittosporum nepaulensis (DC.) Rehoto & Wilson	22.85	10	0.46	5.16	1.12	2.00
Dalbergia oliveri Gamble	22.75	8	0.36	3.39	1.16	1.89
Cinnamomum obtusifolium (Roxb.) Nees	15.39	27	0.62	5.12	3.09	1.67

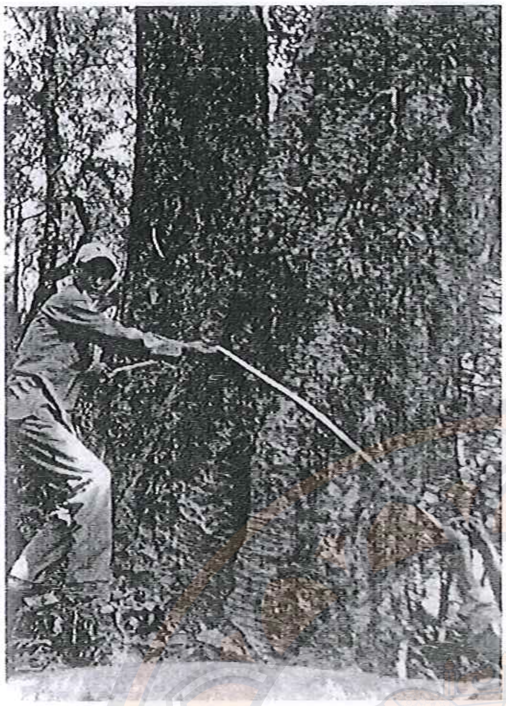
Table 24 (cont.)

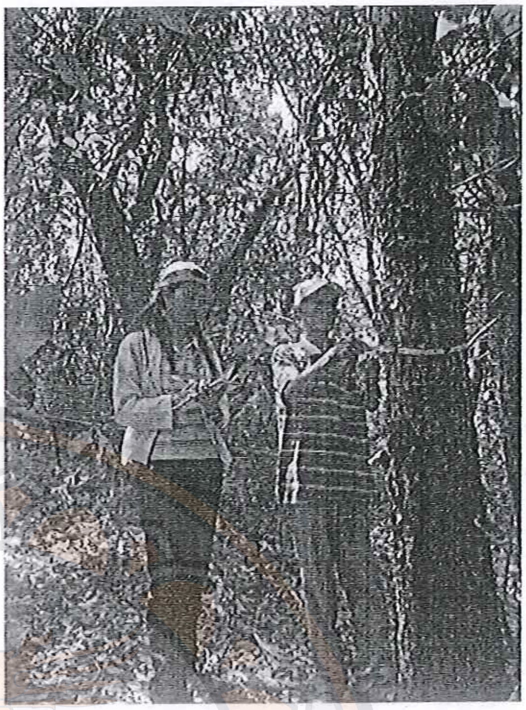
Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Schrebera swietenioides Roxb.	26.75	2	0.17	3.69	0.58	1.51
Albizzia chinensis (Osbeck) Merr.	36.80	5	0.54	5.73	1.16	1.47
Engelhardtia spicata Blume	44.25	2	0.35	4.12	0.52	1.06
Pinus insularis Engl.	37.25	2	0.23	2.47	0.43	0.76
Protium serratum Engl.	17.17	6	0.20	1.72	0.58	0.61
Cordyline fruticosa (L.) A. Chev.	15.50	3	0.07	0.94	0.56	0.30
Cissus discolor Blume	10.40	20	0.19	0.90	2.70	0.29
Dalbergia cultrata Grah.	31.00	1	0.08	0.67	0.29	0.29
Xylia xylocarpa (Roxb.) Taub.	20.50	2	0.07	0.50	0.32	0.23
Mangifera indica L.	14.08	6	0.10	0.62	0.70	0.23
Diospyros spp.	16.75	4	0.09	0.57	0.62	0.22
Chionanthus ramiflora Roxb.	22.00	2	0.08	0.40	0.33	0.14
Diospyros burmanica Kurz	15.50	1	0.02	0.13	0.25	0.13
Uncaria pilosa Roxb.	14.50	2	0.03	0.12	0.49	0.06
Grewia tiliifolia Vahal.	14.00	1	0.02	0.12	0.25	0.05
Toddalia aculeata Pers.	18.00	1	0.03	0.11	0.25	0.04
Artocarpus heterophyllus	9.00	3	0.02	0.10	0.52	0.03
Acacia laevis Parker	8.00	2	0.01	0.07	0.48	0.03
Zizyphus rugosa Lam.	9.50	2	0.02	0.04	0.48	0.02
Cassia fistula L.	8.00	2	0.01	0.04	0.28	0.02
Senna siamea (Lam.) Irwin & Barneby	11.00	1	0.01	0.04	0.24	0.02

Table 24 (cont.)

Scientific name	Mean DBH (cm)	SD (tree ha^{-1})	BA (m^2 ha^{-1})	Vol (m^3 ha^{-1})	IVI (%)	AG-C (tC ha^{-1})
Careya arborea Roxb.	17.00	1	0.02	0.03	0.25	0.01
Carissa carandas L.	7.00	1	0.00	0.01	0.24	0.00
		804	**47.83**	**610.72**	**100.00**	**200.22**

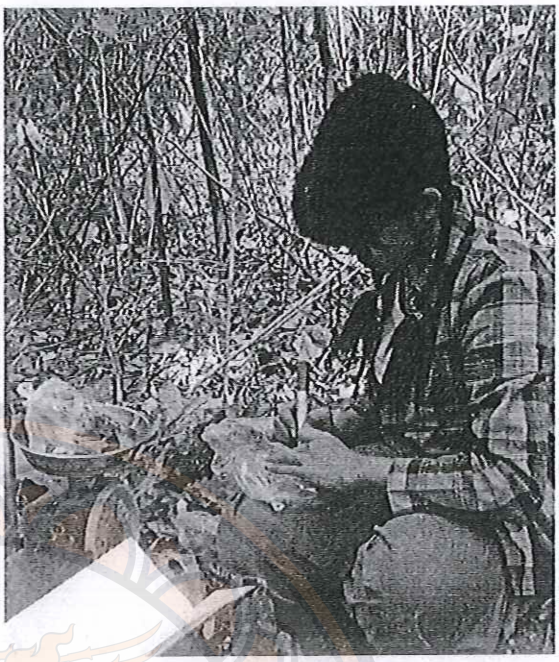
*DBH = Diameter at Breast High, SD = Stand Density, BA = Basal Area, Vol = Volume, IVI = Important Value Index, AG-C = Aboveground Carbon.





Measuring the trees in the sample plot

Process of collecting samples from litters



Process of collecting soil samples

Deadwood in the sample plots

GLOSSARY

GLOSSARY

BAU	Business-as-usual
CDM	Clean Development Mechanism
COP	Conference of the Parties, United Nations Climate Change Conference
FAO	Food and Agriculture Organization of the United Nations
FRA	Forest Resource Assessment
GHGs	Greenhouse Gas
GIS	Geographic Information System
IPCC	Intergovernmental Panel on Climate Change
ITTO	The International Tropical Timber Organization
IUCN	International Union for Conservation of Nature
LULUCF	Land Use, Land Use Change and Forestry
MAI	Mean Annual Increasement
MRV	Measurement, Reporting and Verification
NMFA	National Forest Monitoring and Assessment
NGO	Non-governmental Organization
PAs	Protected Areas
PES	Payment for Environmental Services
Ppm	Parts per million
RED	Reducing Emissions from Deforestation
REDD	Reducing Emissions from Deforestation and forest Degradation
REDD+	Reducing Emissions from Deforestation and forest Degradation in developing countries; and the role of Conservation, Sustainable management of forests and Enhancement of forest carbon stocks in developing countries
RS	Remote Sensing
SFM	Sustainable Forest Management
SPSS	Statistical Package for Social Science
UN	United Nations
UNDP	United Nations Development Programme

GLOSSARY (CONT.)

UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
UNFF	United Nations Forum on Forests
UN-REDD	The United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries
UN-REDD PT	The United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries Programme Team