FIRE REGIME, VEGETATION DYNAMICS AND LAND COVER CHANGE IN TROPICAL PEATLAND, INDONESIA

Thesis submitted for the degree of

Doctor of Philosophy

at the University of Leicester

by

Agata Hościło

Department of Geography

University of Leicester

(2009)

FIRE REGIME, VEGETATION DYNAMICS AND LAND COVER CHANGE IN TROPICAL PEATLAND, INDONESIA

To my parents

Dla moich Kochanych Rodziców

Fire regime, vegetation dynamics and land cover change in tropical peatland, Indonesia

Agata Hoscilo

ABSTRACT

This thesis seeks to understand and explain the role of fire in land cover change, vegetation and carbon dynamics in the carbon-dense, tropical peat swamp forest ecosystem of Southeast Asia. Following a methodological review, earth observation and ground data are employed to investigate fire regime, post-fire vegetation recovery, and fire-driven carbon losses in 4,500 km^2 of peatland in Central Kalimantan, Indonesian Borneo. Results reveal an increasing trend in deforestation (2.2% yr^{-1} forest loss rate, 1973-1996; 7.5% yr^{-1}, 1997-2005) and identify fire as the principal cause. A step change in fire regime is identified, with increasing fire frequency and reduced return interval following land drainage for the Mega Rice Project (MRP). During the post-MRP period (1997-2005), ~45% of the area was subject to multiple fires; 37% burnt twice and 8% three or more times. Extensive fires in 1997 and 2002 were associated with ENSO droughts, but fires in non-ENSO years (i.e. 2004, 2005) indicate fire incidence has decoupled from ENSO. This study provides a novel approach to quantifying relative magnitude of burn severity using characteristics of the post-fire vegetation regrowth. Combined spectral and ground data are used to demonstrate that enhanced fire frequency and burn severity limit post-fire forest recovery, with fern-dominated communities replacing tree re-growth. The character of post-fire vegetation is an important factor defining burning conditions for a subsequent fire, whilst fire frequency, severity and return interval influence both rate and nature of vegetation regrowth. Methods are proposed for deriving fire-driven carbon losses. Over the period 1973-2005, losses are estimated at 79-113 Mt of carbon (53-83 Mt from peat; 26-30 Mt from vegetation), with the greatest loss occurring during the post-MRP era (65-94 Mt). This work identifies the processes linking fire regime in tropical peatland to changes in vegetation ecology and carbon stocks and assesses the implications for ecosystem rehabilitation.

Acknowledgments

There are many people without whom this thesis would not have been completed at all, or not been half as good. In the first place I would like to thank my main supervisor, Dr. Susan Page, who introduced me to the beauty of the tropical environment and successfully infected me with her scientific enthusiasm. Without her, I would never have learned as quickly or deeply how tropical peatland function as I had done. Many thanks as well, to my second supervisor Dr. Kevin Tansey for helping me with the burning adventure. Thank you both for providing unstinting support, motivation, enthusiasm, constructive criticism, patience, and overall friendly atmosphere.

I want to thank Dr. Suwido Limin and his excellent team from CIMTROP, University of Palangkaraya, for assistance in the field campaign. Without them, there would have been no field data to analyse, or unforgettable experiences to dream of. Thanks to Prof. Chris Brunsdon for statistical advice. I also want to thank the ‘RESTORPEAT family’ for the warm welcome to the project team. Thanks to all the PhD students and staff colleagues in the Department of Geography who I got to know during the last few years for making my life enjoyable and full of interesting experiences.

On a personal note, I would like to thank my Mum, Dad and siblings for endless moral support even from the distance. I should not fail to mention Beata who always believes in me and gives me a strong motivation to work harder. And lastly, I want to thank Artur, whose encouragement and patience, whilst I was undertaking my doctoral labour, has been generous and untiring.

This thesis was funded by the EU-funded RESTORPEAT project (FP6 INCO-DEV project number 510931).

In addition I would like to acknowledge the following:

- DMC International Imaging for providing satellite imagery
- Natural Environment Research Council (NERC) for loaning the spectroradiometer
- Dr. Klaus Wolter for providing the Multivariate ENSO Index dataset

Table of Contents

List of Figures

List of Tables

List of Appendices

Acronyms

Acronym	*Explanation*
AGB	Aboveground Biomass
ANOVA	Analysis of variance
BA	Basal Area
CBI	Composite Burn Index
CIMTROP	Centre for International Cooperation in Sustainable Management of Tropical Peatland, University of Palangkaraya
CO_2	Carbon dioxide
DBH	Diameter at Breast Height
DCP	Digital Canopy Photography
DMC	Disaster Monitoring Constellation
dNBR	Bi-temporal Normalised Burn Ratio
ENSO-MEI	El Niño Southern Oscillation - Multivariate ENSO Index
ETM+	Landsat Enhanced Thematic Mapper
EO	Earth Observation
GHGs	Greenhouse gases
GIS	Geographic Information System
Gt	Gigatone
GV	Fraction of Green Vegetation
HAPs	Human Access Points
IPCC	Intergovernmental Panel on Climate Change
IR	Infrared wavelength
LIDAR	Light detection and ranging
MERIS	MEdium Resolution Imaging Spectrometer
MF	Multiple fires
MF1	Multiple fires (relatively advanced stage of forest regrowth)
MF2	Multiple fires (intermediate stage of forest regrowth)
MF3	Multiple fires (the least advanced stage of woody regrowth)
MF-L	Multiple fires-Low severity fire

MF-M	Multiple fires-Moderate severity fire
MF-H	Multiple fires-High severity fire
Mha	Million hectare
MODIS	Moderate Resolution Imaging Spectroradiometer
MRP	Mega Rice Project
MtC	Megaton of Carbon
MSS	Landsat Multispectral Scanner
Mt	Megaton
NERC	UK's Natural Environment Research Council
NBR	Normalised Burn Ratio
NDVI	Normalised Difference Vegetation Index
NDWI	Normalized Difference Water Index
NOAA/AVHRR	National Oceanic and Atmospheric Administration/Advanced Very High Resolution Radiometer
NPV	Fraction of Non-Photosynthetic Vegetation
PCA	Principal Component Analysis
PSF	Peat Swamp Forest
R	Red wavelength
REDD	Reducing Emissions from Deforestation and Degradation
RMS	Root-Mean-Square
SF	Single fire
SMA	Spectral Mixture Analysis
SPOT	Satellite Pour l'Observation de la Terre
SRTM	Shuttle Radar Topography Mission
SWG	Specific Woody Gravity
SWIR	Short-infrared wavelength
T_{AGB}	Total Aboveground Biomass
TM	Landsat Thematic Mapper
TREES	Tropical Ecosystem Environment Observation by Satellite
UNFCCC	United Nations Framework Convention on Climate Change
UTM	Universal Transverse Mercator

1 INTRODUCTION

1.1 Importance of tropical forests and tropical peatlands

Tropical forests cover only 7-10% of the Earth's surface but they are globally important carbon stores containing 40-50% of all carbon located in terrestrial vegetation (Malhi and Grace, 2000, Nightingale *et al.*, 2004). Through photosynthesis and respiration, tropical forests process approximately six times as much carbon as humans produce from fossil fuel combustion (Lewis, 2006, Malhi and Grace, 2000). Even relatively small perturbations within the tropical forest biome can, therefore, affect significantly the global carbon flux with consequent impacts on climate change.

The tropical forest carbon store encompasses aboveground (i.e., trees and understorey vegetation) and belowground biomass (i.e., living roots), dead organic matter (fallen and standing dead trees, woody debris and leaf litter), and soil organic matter. In general, aboveground biomass represents the largest carbon stock (Lewis, 2006), although, in some tropical ecosystems, such as tropical peat swamp forest, underlying, thick peat deposits represent an additional, substantive carbon stock (up to 20m thick in places in Southeast Asia) (Page *et al.*, unpublished report, Page *et al.*, 2002). Indonesian peatland, for example, may contain on average 2.5 MtC ha^{-1} in the peat and only 0.1-0.15 MtC ha^{-1} in forest biomass (Page *et al.* unpublished report). Tropical peatland has been described as a 'dual ecosystem' comprsed of peatland and rain forest, with both components co-existing throughout thousands of years (Page *et al.*, 2004). The initiation of peat in an acidic, waterlogged environment provided the substrate for swamp forest vegetation which in return, contributed the organic material for continued accumulation of peat.

The largest extent of tropical peatland, with around 70% of the total global area (368,500km^2), occurs in Southeast Asia and more than half of this area (56%) is located in Indonesia (www.carbopeat.org) (Figure 1.1).

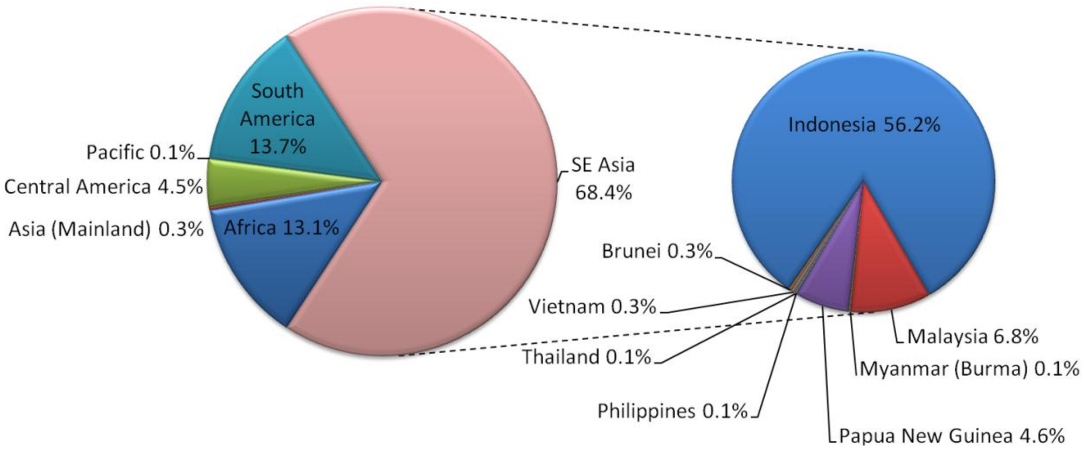


Figure 1.1 Distribution and extent of tropical peatland based on best estimate values (www.carbopeat.org).

The tropical peat deposits of the Southeast Asian region are mostly situated in low-lying coastal and sub-coastal areas. They contain around 61 Gt of carbon (i.e. 95% of the global tropical peat carbon stock) with 52 Gt (80%) stored in the peatlands of Indonesia alone (Figure 1.2). This large carbon pool is responsible for a large proportion of global carbon fluxes between the earth's surface and the atmosphere by sequestration of atmospheric CO_2 during photosynthesis, part of which is stored in plant biomass and ultimately in the peat (Immirzi, 1992, Hooijer *et al.*, 2006).

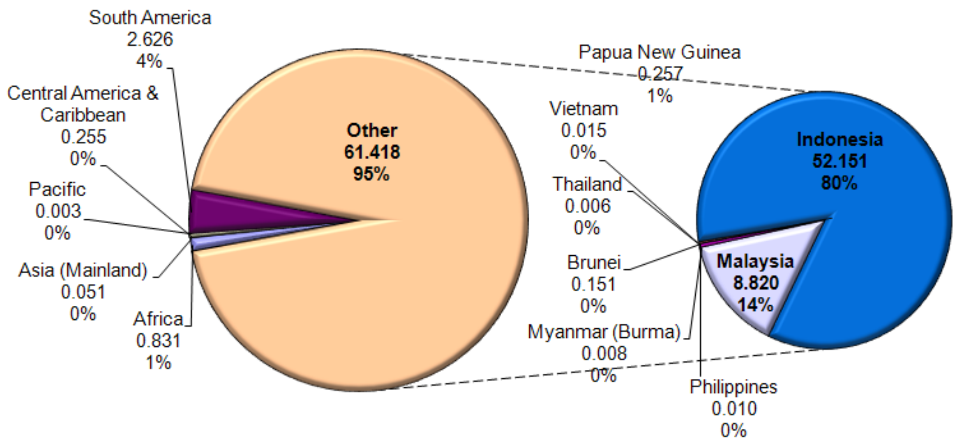


Figure 1.2 Amount of carbon (Gt) stored in tropical peatlands based on the current best estimate (Page et al. unpublished report).

Carbon storage in peat is only possible under waterlogged conditions, which reduce the process of decomposition such that the production of organic matter exceeds its breakdown. Under dry conditions (i.e. when the water table drops below the peat surface) the rate of decomposition increases, enhancing CO_2 emissions back to the atmosphere.

In addition to their role in the global carbon cycle, tropical peatlands play a vital role in the regional water cycle; specific physical and chemical properties of the peat soil and dense forest vegetation regulate the ecosystem water balance, retaining water over the wet season and supplying water to local watercourses during the dry season (Rieley and Page, 2005, Page *et al.*, 1999, Hooijer *et al.*, 2006).

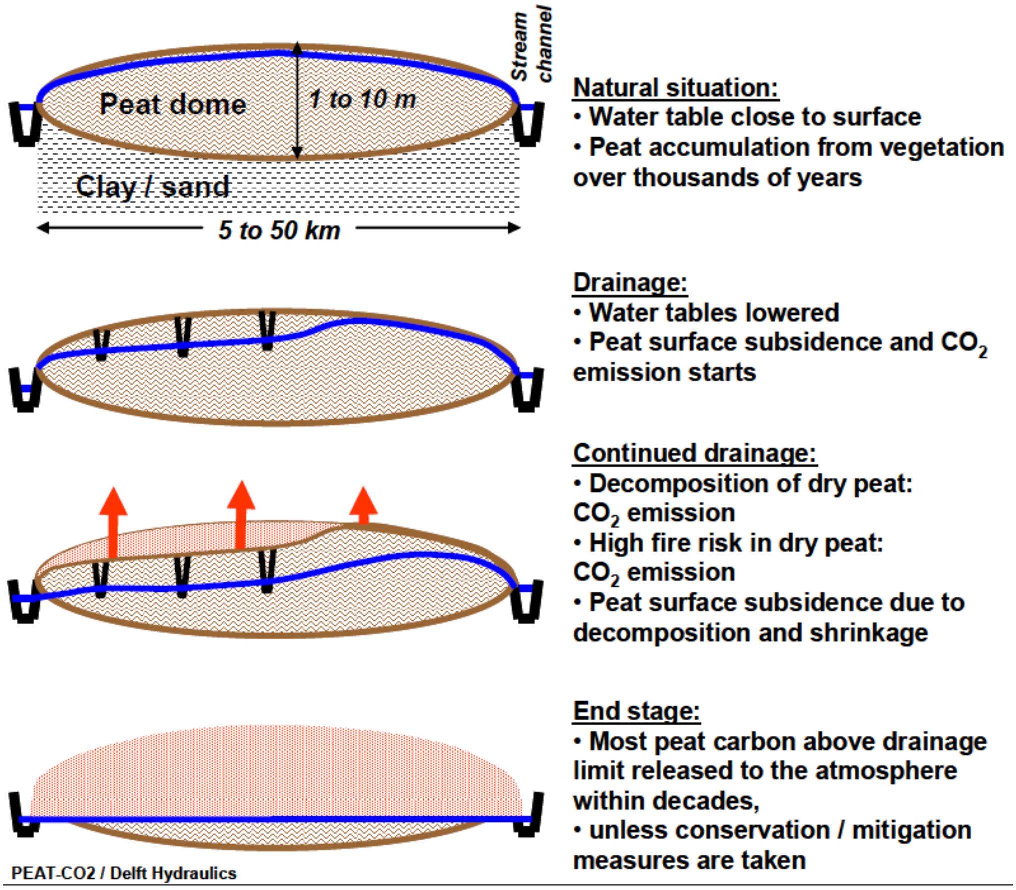
Tropical peat swamp forests (PSF) also support a diverse fauna and flora, with many endangered, protected and unique species, including orangutan, Sumatran tiger, and blackwater fish (Page, 1997, MacKinnon, 1996). These forests may contain up to 240 tree species per hectare (Page *et al.*, 1999, Page, 1997, Anderson, 1976), including a number of highly-valued commercial timber species. Tropical peatlands are also of great importance to local people, since they support livelihoods for indigenous communities through provision of food, medicinal plants, building materials, and, energy resources (Rieley and Page, 2005, MacKinnon, 1996).

1.2 Tropical forests under threat; causes and effects

During the last 50 years, over 74 Mha of Indonesia's forests (equivalent to an area about three times the size of the UK) have been destroyed (i.e. logged, burned or degraded; www.greenpeace.org). Achard *et al.* (2002) showed that between 1990 and 1997, the Southeast Asian region experienced both the world's highest deforestation and degradation rates, at 0.91% (2.5 Mha) yr^{-1} and 0.42% (1.1 Mha) yr^{-1}, respectively. Further intensive forest loss at the rate of 1.9 Mha yr^{-1} took place between 2000 and 2005 (FAO, 2005). Hooijer *et al.* (2006) reported that 25% of current deforestation in the Southeast Asian region was occurring in the peat swamp forests, which have experienced a very high rate of deforestation (1.5% yr^{-1} over the past 20 years). Langner *et al.* (2007) reported an even higher rate of 2.2% yr^{-1} for PSF loss in Borneo.

The large-scale and rapid changes in land cover reported in tropical PSF in Southeast Asia over the last two decades has largely been associated with rapid expansion of plantations, involving both deforestation and drainage, and widespread fires. These unsustainable land and forest management practices have altered the ecosystem hydrological and carbon storage functions, converting a previous carbon sink to a carbon source. Drainage is associated with land conversion for agriculture or commercial plantations and typically requires construction of a canal infrastructure, and also logging, which often involve construction of canals for log extraction. Drainage lowers the water table which in undisturbed forest, stays close to the peat surface (Figure 1.3). This leads to enhanced peat decomposition (biological oxidation) and release of stored carbon to the atmosphere, mostly in the form of carbon dioxide (CO_2) (Wosten *et al.*, 2006, Jauhiainen *et al.*, 2008, Hirano *et al.*, 2008). It has been estimated that decomposition of drained peatlands in Indonesia causes CO_2 emissions of 0.6 Gt yr^{-1} (range 0.4-0.9 Gt yr^{-1}) (Hooijer *et al.*, 2006). The scale of emissions may even increase in coming decades with further peatland developments, driven by rapid expansion of oil palm and commercial timber plantations, both of which requite intensive drainage (Reijnders and Huijbregts, 2008, Danielsen *et al.*, 2009). Figure 1.3 presents the consecutive stages of peatland degradation due to drainage. Peat oxidation (represented by the red arrows in Figure 1.3) results also in progressive surface subsidence (represented by the red shaded area in Figure 1.3) which irreversibly alters the peat properties and increases the risk of flooding during the wet season (Hooijer *et al.*, 2006, Wösten *et al.*, 2008).

Large-scale forest degradation and drainage of tropical peatlands also increase the risk of widespread and intensive fires (Wösten *et al.*, 2008, Dennis *et al.*, 2005). A study by Langner and Siegert (2009) demonstrated the recent increased occurrence of fires within the peatlands of Borneo. They reported that 73% and 75% of the total fire-affected areas in 2002 and 2006 respectively occurred in PSF.



Note: red arrows represent peat lost due to peat decomposition whereas the red shaded area indicates progressive peat surface subsidence.

Figure 1.3 Schematic illustration of consecutive stages of peatland degradation caused by drainage (Hooijer et al., 2006).

Peatland fires consume both the surface vegetation and the underlying peat, thereby releasing large amounts of CO_2 and other GHGs into the atmosphere (Page *et al.*, 2002, van der Werf *et al.*, 2008, Langenfelds *et al.*, 2002). During the El Niño event of 1997-1998, for example, out-of-control fires burned about 8 Mha of peatland in Indonesia, resulting in a loss of 0.8-2.57 GtC, equivalent to 13-40% of the average annual global carbon emissions from fossil fuel (Page *et al.*, 2002). These fires contributed to the peak in global atmospheric CO_2 concentration recorded in 1997-1998 (van der Werf *et al.*, 2006). The subsequent 2002 El Niño-related fires emitted an additional 0.25-0.5 Gt carbon (Bechteler and Siegert, 2004), while a further 0.7-0.8 Gt were released during fires in 2006 (Langner and Siegert, 2007). According to Hooijer *et al.* (2006), currently 0.4 Gt yr^{-1} of carbon (1.4 $GtCO_2$ yr^{-1}) is emitted from peatland fires in Indonesia. These estimates are probably too high since they are all derived from hotspot data and not from burned area estimation (more information on the limitation of active fire products is presented in Section 2.2.1). In addition, they did not consider either the volume of burned peat or fire fuels (i.e. type, quality and amount). Nevertheless, these data do serve to illustrate the scale of emissions resulting from tropical peatland fires.

If emissions from both peat oxidation and combustion are combined then Hooijer *et al.* (2006) calculated that, on average, more than 0.5 Gt yr^{-1} of carbon (equivalent to 2 $GtCO_2$ yr^{-1}) is emitted annually to the atmosphere from the Southeast Asian region alone; this is equivalent to almost 8% of global emissions from fossil fuel burning. In addition to CO_2 emissions, burning biomass emits other harmful chemical compounds such as CO, methane, nitrous oxide, nitric oxide and other GHGs (Crutzen and Andreae M.O., 1990, Crutzen *et al.*, 1979, Levine, 1994, Langenfelds *et al.*, 2002, Edwards *et al.*, 2004, Arellano *et al.*, 2006), which affect human health, plant growth (Davies and Unam, 1999, Nasi *et al.*, 2001) and contribute to regional air pollution episodes. Peatland fires also result in wide-ranging impacts other that those associated with GHGs emissions. They cause irreversible changes in the biological and chemical properties of the peat leading to loss of nutrients, erosion and enhanced decomposition (Rieley and Page, 2005). Fire also causes surface subsidence (Wosten *et al.*, 2006) and alterations in biodiversity and vegetation succession (Curran *et al.*, 2004, Slik *et al.*, 2008, Slik and Eichhorn, 2003), as well as increasing poverty and related socio-economic problems for human populations living in or near peatlands (Aiken, 2004).

In the past, fire incidents in Southeast Asia have been strongly coupled with droughts associated with El Niño Southern Oscillation (ENSO) events (Wang *et al.*, 2004, Wyrtki, 1975, Fuller, 2006, Tacconi, 2006, Goldammer, 1997). The ENSO is defined as an anomalous warming (El Niño) or cooling (La Niña) of surface water in the eastern equatorial Pacific which occurs at irregular intervals between 2 and 7-years, and generally lasts between 12 and 18-months (Glantz, 2001). Variation in the atmospheric CO_2 concentration has been linked to changes in the climatic parameters associated with

El Niño periods. In tropical regions, El Niño periods favour net CO_2 release, whilst La Niña periods favour net uptake (Langenfelds *et al.*, 2002).

In recently decades, degradation of forest ecosystems in the Southeast Asian region has impaired the natural fire suppression function of moist tropical forest and increased the risk of fire during dry seasons, even in non-ENSO years (Gullison *et al.*, 2007). As a consequence, fires have become more regular and extensive (Langner *et al.*, 2007, Hoscilo *et al*, 2008a). Langner and Siegert (2009), for example, demonstrated that around 21% of land in Borneo was subjected to fires over the period 1997-2006, with 6.1% (4.5Mha) of the forest affected more than once.

Undoubtedly emissions from fire-driven deforestation in the tropics are an important driver of climate change (Ramankutty *et al.*, 2007), and, worryingly, several studies have indicated that emissions from forest and peat fires in Southeast Asia are unlikely to be reduced in coming decades (Li *et al.*, 2007, Hooijer *et al.*, 2006). The IPCC Fourth Assessment Report (2007) provides climate change predictions for the Southeast Asian region of warming of 2.5^{o}C by the end of 21st century, and an increase of mean precipitation by about 7%. Even though these predictions vary across the region, there is an explicit pattern of increasing rainfall during the wet season and decreasing during the dry season. Modelling of future climate scenarios for southern Borneo suggests a reduction in rainfall, increased seasonality and enhancement of ENSO intensity (Li *et al.*, 2007). This climatic variability may lead to an enhanced risk of peatland fires and further increases in carbon emissions (Hirano *et al.*, 2007, Suzuki, 1999).

1.3 Remedies for mitigation of climate change processes

Recognition of the regional and global consequences of extensive tropical deforestation and degradation has led to enhanced international concerns and efforts to reduce forest losses and associated emissions of GHGs. This is manifested in global climate policy initiatives, which aim to prevent further deforestation and consequently mitigate climate change processes. The United Nations Framework Convention on Climate Change (UNFCCC) is currently considering a financial mechanism to reduce GHG emissions from deforestation and forest degradation (REDD) in developing countries (UNFCCC, 2007). As a result, there is an ongoing debate on the uncertainties related to estimates of deforestation and degradation rates and the regional and global consequences of forest conversion (De Fries *et al.*, 2007, Miles and Kapos, 2008, Grassi, 2008, Gibbs *et al.*, 2007, Skutsch *et al.*, 2007).

The tropical carbon balance is one of the most poorly understood elements of the global carbon cycle (Coppin *et al.*, 2004, Houghton, 2007). Estimates of the current tropical forest extent and rate of deforestation vary widely (Page *et al.*, 2008, Houghton, 2007, Lewis *et al.*, 2006) as do estimates of above and belowground forest biomass and carbon stocks (Guild *et al.*, 2004, Houghton, 2000, Houghton, 2001, Houghton, 2003, Chave *et al.*, 2005). As a result, estimation of carbon released to the atmosphere from global deforestation and fires in the tropics is still controversial and not well known (De Fries *et al.*, 2002, Guild *et al.*, 2004, Langmann and Heil, 2004, Tansey *et al.*, 2004, Ramankutty *et al.*, 2007). It has been suggested that approximately 50% of the uncertainties surrounding tropical carbon pools and fluxes can be attributed to estimation of biomass, and as much as 80% to estimation of carbon fluxes (Coppin *et al.*, 2004, Houghton, 2003, Houghton, 2005). These wide uncertainty ranges need to be minimised in order to make the REDD initiative successful and to improve understanding of the role played by tropical deforestation in climate change.

Another crucial element of carbon flux estimation is accurate data on the role played by naturally regenerating secondary vegetation in carbon sequestration (Olschewski and Benitez, 2005). Most secondary vegetation regrowth has the capacity to act as a carbon sink and thus has a potential role as a regulator of climate change (Castro *et al.*, 2003, Lucas *et al.*, 2005, Trabucco *et al.*, 2008). In addition, secondary vegetation contributes to restoration of ecological functions that were lost during forest clearance, e.g. soil fertility and protection, vegetation structure, composition and biodiversity (Lu, 2005, Moran *et al.*, 2000, Lucas *et al.*, 2005, Lamb *et al.*, 2005, Hughes *et al.*, 1999, Song and Woodcock, 2002). In tropical peatland areas, regenerating vegetation can also make an important contribution to reducing peat decomposition and the risk of fire (Page *et al.*, 2008).

Given the increased incidence of fire in the Southeast Asian region, there is a particular requirement to understand how the vegetation responds to intensive and often repeated fires. This type of knowledge of the post-fire vegetation response is crucial to: a) understand fire regimes, b) diminish and prevent future fire risk, c) sustainably manage degraded forest land towards a mature stage of regeneration and d) undertake efficient restoration action in locations where natural regeneration is, for whatever reason, halted.

1.4 Research objectives

The aim of this research is to demonstrate the ecological and climatic impacts of recently enhanced fire regimes in tropical peatlands through measures of land cover, vegetation and carbon dynamics. Specifically, this work has the following objectives:

1. **To quantify the land cover and fire regime* dynamics over the period 1973-2005 in the study area (addressed in Chapter 4).**

2. **To examine the effects of fire frequency and burn severity** on the dynamics of post-fire vegetation recovery by studying four restoration indicators: aboveground biomass, vegetation ground cover, canopy cover and diversity of plant species (addressed in Chapter 5).**

3. **To quantify carbon losses from burning aboveground (vegetation) and belowground (peat) biomass (addressed in Chapter 6).**

* The term fire regime used in this study refers to fire extent, frequency, return interval, and fire fuel, where:

Fire frequency is defined as how many times fire affects the same area over a defined period of time.

Fire return interval is defined as the period of time between subsequent fires.

** The term burn severity refers to the long-term (i.e. a period of 4 years or more) effects of fire on vegetation structure.

These research objectives were investigated for a 448,912ha study area forming part of the ex- Mega Rice Project area in Central Kalimantan, Indonesian Borneo.

1.5 Thesis structure

The thesis is divided into seven chapters.

Chapter 1 presents an overview of the importance of tropical forests, particularly tropical PSF, with a focus on the causes and consequences of deforestation and degradation in tropical forest ecosystems. It also highlights the number of uncertainties associated with quantification of tropical carbon pools and fluxes as well as stressing the role played by natural vegetation regeneration.

Chapter 2 synthesises the relevant scientific literature, focusing on the most commonly used EO methods to investigate fire regime and land cover dynamics in tropical ecosystems, with particular attention paid to current knowledge gaps and uncertainties. Clarification of the terminology relating to fire ecology (i.e. terms such as fire intensity, fire severity and burn severity) is provided in Section 2.2. The final Section 2.3 reviews the current state of knowledge and importance of secondary vegetation in the tropics and highlights uncertainties surrounding estimations of carbon pools.

Chapter 3 provides an explanation of the main features of the study area and the development of the Mega Rice Project, which was implemented in the area in the mid-1990s.

Chapter 4 presents methods that have been used to address objective 1; specifically to map and quantify the land cover for the period 1973-2005 and to analyse the fire regime dynamics taking into account fire frequency, extent, interval between subsequent fires and fire fuel. Methods are followed by results, presented in Sections 4.4 and 4.5. The chapter concludes with Section 4.6, which focuses on the analysis of the relationship between fire incident and extent and the ENSO-MEI index.

Chapter 5 contains the methodology used for the quantitative inventory of post-fire vegetation performed during a field campaign. Characteristics of different stages of vegetation regrowth, followed by statistical analyses, are presented in Section 5.2. Finally, Section 5.3 presents the long-term impact of multiple fires on vegetation recovery, quantified by combining *in situ* ground data with remote sensed data.

Chapter 6 uses information derived from Chapters 4 and 5 to determine the changes in aboveground carbon stocks over the period of investigation as well as to estimate the carbon losses arising from combustion of both vegetation and peat, taking into account the occurrence of both single and multiple fires.

Chapter 7 discusses the results obtained from the previous chapters in the context of relevant literature and the thesis research objectives. It focuses on trends in land cover and fire regime dynamics and their causal factors, as well as on the long-lasting effects of fire on vegetation regrowth. Section 7.5 highlights the significant role of fire as the main driver of carbon loss in tropical peatland.

In conclusion, Chapter 8 briefly summarises the results obtained from this study in relation to each of the three research objectives and their contribution to previously identified knowledge gaps. The final section suggests several topics for future research with particular reference to further improvement of quantification of tropical forest carbon pools and fluxes.

2 LITERATURE REVIEW

This chapter discusses the most commonly used Earth Observation (EO) methods to investigate land cover dynamics in tropical ecosystems, with particular attention paid to knowledge gaps and uncertainties. The number of uncertainties in land cover inventory and hence in land-atmosphere interactions identified briefly in Section 1.3 mean that the EO data have an important role to play by providing baseline values for forest cover and for monitoring on-going rates of forest loss, degradation and burning, as well as the types and rates of vegetation regrowth.

This chapter is divided into three major sections covering the research objectives. The first Section concentrates on techniques used to map land cover in tropical ecosystems, the second focuses on the investigation of fire events, since fire is recognised as an important driver of tropical landscape alterations. These two sections are relevant to the first research objective. Sub-section 2.2.3 looks at remote sensing techniques used to quantify burn severity in various non-tropical ecosystems, which refers to objective 1 and 2. The last Section 2.3 reviews the current state of knowledge and importance of secondary vegetation in the tropics. In addition to a review of the relevant literature, each section also contains additional information on the limitations and uncertainties related to monitoring of land cover change in the moist tropical forest biome. A summary of existing uncertainties and knowledge gaps is provided in the final section.

2.1 Analyses of land cover and land cover change in the tropics

Comprehensive information on land cover and land cover change is a crucial step in providing improved estimates of the tropical carbon budget. Several studies have stressed the lack of accurate information on land cover in recent and formerly deforested areas as a limitation in carbon flux estimation (Houghton, 2000, Houghton, 2005, Achard *et al.*, 2004). In addition, Houghton (2007) highlighted uncertainties surrounding rates of tropical deforestation, which relates also to land cover analysis and contributes to the cumulative errors in estimates of the terrestrial carbon balance.

There has been a long period of debate about the most appropriate classification methodologies for mapping land cover in tropical regions. The low/moderate resolution sensors such as the National Oceanic and Atmospheric Administration/Advanced Very High Resolution Radiometer (NOAA/AVHRR), Satellite Pour l'Observation de la Terre (SPOT Vegetation), Moderate Resolution Imaging Spectroradiometer (MODIS) or MEdium Resolution Imaging Spectrometer (MERIS) have been widely used for national and global land cover mapping. The great advantage of these sensors is that their low spatial resolution is compensated by high temporal resolution that significantly increases the probability of obtaining cloud free data. Several studies used these products to obtain information on land cover status and dynamics in the Southeast Asian tropics (Langner *et al.*, 2007, Stibig *et al.*, 2007, Miettinen *et al.*, 2008, Arino *et al.*, 2007, JRC, 2004).

An assessment of the spatial distribution of remote sensing applications to tropical forest monitoring indicates that there is a strong tendency to study the Amazon Basin, whereas the Southeast Asia region seems to be particularly under-investigated (Fuller, 2006). To date, there have been only a few products showing the landscape status of the Southeast Asian region. Some of the most comprehensive sources of land cover information for the entire tropical zone are global sets of data, such as Global Land Cover 2000 (GLC2000) or GLOBCOV2006 derived from low/moderate resolution data. These products generate uniform data sets with consistent legends which are shaped according to the needs of global rather than regional users (Stibig *et al.*, 2007). Therefore a more specific, land cover map for South and Southeast Asia was derived from SPOT Vegetation data (1000m of spatial resolution), based on a 10-day composite of max-NDVI (Stibig *et al.*, 2007). Miettinen *et al.* (2008) subsequently mapped the same region using MODIS (500m) by applying a simplified unsupervised ISODATA method (clustering into 100 classes). All classes were grouped into five classes using visual interpretation, and then further delineation was performed to derive the final twelve land cover classes using the auxiliary data. Langner *et al.* (2007) also applied the same unsupervised method with eight final classes to produce a map of land cover in Borneo. This approach can be used for mapping large areas where there is limited ground-truth reference data (Cihlar, 2000), however it can also lead to misinterpretation with respect to class merging and naming.

Freely available, low resolution data provide a great capacity for monitoring deforestation globally, but the use of these data to map forest loss or forest cover is limited by spatial and spectral resolution (e.g. MODIS has only two bands available at 250m resolution). Consequently these data are not considered suitable for more detailed detection of forest change (i.e. at a regional scale; Trigg *et al.*, 2006). For this purpose, Trigg *et al.* (2006) strongly recommended using high resolution data to derive more precise and detailed information that could be used, for example, to monitor forest cover in protected areas or for land management planning. The review of the literature undertaken for this current study suggests that of the different satellite sensors employed to study tropical forests, Landsat data have been the most commonly used (Baker and Williamson, 2006, Fuller, 2006). This is mainly due to the long data record, extending back to 1972, and the relatively high spatial resolution (30m), that is appropriate for detection of land alteration, canopy condition or forest degradation assessing from burning and logging (Fuller, 2006).

Several methods incorporating spectral information derived from high resolution images have been tested and applied with various results in land cover analysis in tropical ecosystems. For example, Guild *et al.* (2004) tested the capacity of multivariate transformations such as tasseled cap (TC) and principal components analysis (PCA) for land conversion in Brazil and found that TC was more reliable for detecting deforestation fragments than PCA. Dennis and Colfer (2006) also used the TC to derive a series of land cover maps for East Kalimantan, Borneo. Lu *et al.* (2005) showed that image differencing methods or PCA were a more suitable for land cover detection in Amazonian forest than image ratio methods. On the other hand, Hayes and Sader (2001) argued that vegetation index differencing methods were a more accurate way of detecting change in tropical forest compared to PCA. Modification of spectral indices

(Phua *et al.*, 2007) and the use of change vector analysis (Phua *et al.*, 2008) have also been tested to map deforestation at the regional scale in Southeast Asia.

The traditional automatic approaches, reviewed above, deal mainly with a single pixel value, thus are not very effective in assessing heterogeneous surfaces (Mansor *et al.*, 2001, Miettinen and Liew, 2005). Recent research has, therefore, started to focus on analysis of the sub-pixel composition of land cover through spectral mixture analysis (Lu *et al.*, 2004), artificial neural networks (Foody *et al.*, 2001, Ingram *et al.*, 2005, Foody and Cutler, 2003) or object-oriented classification approaches. The object-oriented method has been successfully applied for habitat and agricultural land mapping (Lucas *et al.*, 2007), land cover studies (Whiteside and Ahmad, 2005, Mansor *et al.*, 2001), land cover dynamics (Bontemps *et al.*, 2008) as well as tropical deforestation (Yijun and Ali Hussin, 2003). An advantage of the object-oriented classifier is that the segmentation algorithm does not only rely on the single pixel value but also on the spatial continuity, texture and mutual relations among objects.

Various change detection techniques have been developed and tested, but in many cases an expert-based, manual interpretation still provides the most reliable results for sub-regional scale analysis. This is because landscape heterogeneity in tropical ecosystems is often far greater than that observed in temperate ecosystems and this causes particular problems in automatic classification (Miettinen and Liew, 2005).

An additional concern contributing to uncertainties in determining land cover dynamics is a lack of consensus on terminology and clarification of class definitions. Research by Ramankutty *et al.* (2007) discussed the complexity associated with estimating carbon pools and fluxes, highlighting in the first place the lack of a standardised definition of forest (Lepers *et al.* (2005) reported more than 90 different definitions of forest), discrepancies in the definition of deforestation, a lack of any clear classification of forest degradation, as well as inconsistencies between countries and between assessments. They compared the results of five simulations of global carbon pools and fluxes and suggested that improvements in accuracy could be feasible if studies considered a) the full land cover dynamics prior, during and after the deforestation process, b) the historical land cover dynamics over several decades, and c) future projection of the deforested land and associated carbon losses and gains (e.g. through subsequent regrowth).

2.2 Investigation of fire events

Extensive and repeated fires together with land clearance practices have been reported to be major causes of land cover change in tropical ecosystems, particularly in Southeast Asia. There is therefore, an increasing need to investigate fire regime dynamics. This knowledge is essential to improve our understanding of ecological processes contributing to and caused by combustion and to quantify overall carbon losses and gains from and to the ecosystem.

To describe the spatial and temporal dimensions of fire events, clarification of fire-related terminology is required. Confusion related to the inconsistent use of terminology has been highlighted by several researchers (Smith *et al.*, 2005, Hardy, 2005, Roldan-Zamarron *et al.*, 2006, Keeley, 2009), whilst the paper by Lentile *et al.* (2006) made an attempt to clarify both fire and fire effects terminology. Fire incidents are commonly described using by three terms: fire intensity, fire severity and burn severity. The term *fire intensity* (i.e. referring to an active fire) describes fire behaviour quantified by temperature, and amount of heat released by the fire (Neary *et al.*, 1999, Keeley, 2009). The distinction between *fire severity* and *burn severity* is not clear and these terms are often used interchangeably. Lentile *et al.* (2006) pointed out that temporal gradients separate these two terms. According to these authors the term *fire severity* is usually associated with an active fire thus it is assessed immediately after a fire by measuring how much of the vegetation and soil have been altered, whereas *burn severity* refers to the length of time necessary to return to pre-fire conditions. Key and Benson (2002) defined burn severity as both short- and long-term post-fire effects on the environment, while the National Wildfire Coordinating Group (NWCG, 2008) considered fire severity as the ‘degree to which a site has been altered or disrupted by fire; loosely, a product of fire intensity and residence time’. There is clearly significant variation in definitions of burn severity and fire severity.

2.2.1 *Active fire detection*

Detection of active fire from space has received a great deal of attention over the last decades. Active fire (i.e. fire hotspot) products are available globally from several sources such as the NOAA/AVHRR from 1998 to the present, the Along Track Scanning Radiometer (ATSR) from mid-1996 to the present, Visible and Infrared Spectrometer (VIRS) from 1998 to the present (Giglio *et al.*, 2000, Giglio *et al.*, 2003), the MODIS from early 2000 to the present (Giglio *et al.*, 2009, Justice *et al.*, 2002), and the Meteosat Second Generation1/Spinning Enhanced Visible and Infrared Imagery (MSG1/SEVERI) (Sobrino and Romaguera, 2004, Roberts *et al.*, 2004, Roberts *et al.*, 2005).

There is no doubt that these data are an extremely valuable source of information about fire presence, although, there is a major drawback in the high error of omission associated with cloud presence, which is particularly problematic in the tropics. This occurs because the sensor identifies only those pixels that are associated with active burning when the satellite is passing over. Active fires are detected using the thermal infrared radiation emitted by the fire at around 3.7µm and 12µm, which is the thermal infrared wavelength demonstrated to be useful in measuring the rate of energy released from fires (Fire Radiative Power/Energy) (Smith and Wooster, 2005, Smith *et al.*, 2005, Wooster *et al.*, 2005, Freeborn *et al.*, 2008, Ichoku and Kaufman, 2005, Giglio *et al.*, 2008) and in observations of smoke concentration (Kaufman *et al.*, 2003). In addition, a combination of thermal sensors with other sensors used to detect trace gases (e.g. SCIMACHY, MIPAS, MOPITT CO and GOME) can additionally provide information on the sources and concentration of fire-derived aerosols (Duncan *et al.*, 2003, Arellano *et al.*, 2006).

One of the limitations of active fire products, and at the same time, a great challenge is the quantitative estimation of the area affected by fire from a count of hotspots. Even a small fire covering a small fraction of a pixel (for example no larger than 0.001km^2 of a

1 km^2 area) can saturate the entire pixel. Thus, there is considerable uncertainty about the size of the burn scar associated with one pixel. In fact, there are many factors, such as vegetation type, cloud cover, fire intensity, size and fragmentation of burn scars, and the duration of burning that could potentially influence the relationship between an active fire and burned area. Consequently, hotspot analyses may often lead to overestimation or underestimation of the extent of the fire (Li *et al.*, 2000, Miettinen *et al.*, 2007, Schroeder *et al.*, 2008, Tansey *et al.*, 2008) and authors have suggested a number of different values for the size of the burned area associated with one hotspot, ranging from 15ha for degraded peatlands in Borneo (Tansey *et al.*, 2008) to 75ha for cropland (Smith *et al.*, 2007) and 180ha for boreal forest (Wotawa *et al.*, 2006). By comparison, Langner and Siegert (2009) produced a value for burned area in Borneo by assuming that the area of each hotspot had been completely affected by fire (i.e. 100ha). For that reason, active fire products are commonly accepted as being less suitable for assessing fire extent than actual burned area products (Roy *et al.*, 2008, Langner and Siegert, 2009).

The obstacles and uncertainties associated with burned area assessment can be improved by considering the synergy between hotspot count and burned area derived from high resolution data for specific ecosystems and land cover types.

2.2.2 *Burned area mapping in tropical ecosystems*

Changes in the biophysical properties of the land surface caused by combustion are reflected in alteration of different regions of the electromagnetic spectrum, with the greatest impact on the near-infrared, followed by the short-infrared and red wavelengths (Chuvieco *et al.*, 2006, Pereira and Setzer, 1993). Therefore, the change detection approach is the most commonly used method to derive data on fire-affected area at both regional and global scales. There are many global burned area products available based on the change detection approach, with data representing a single year (Tansey *et al.*, 2004, Simon *et al.*, 2004), as well as multiannual data (Tansey *et al.*, 2008, Roy *et al.*, 2005, Roy *et al.*, 2008).

At the sub-global or regional scale the vast majority of satellite-based methods for burned area mapping utilise an alteration of spectral signals recorded from the affected surface before and after a fire event. Numerous multi-spectral indices have been tested in different regions with various results. Several studies have explored changes in the Normalised Difference Vegetation Index (NDVI) by combining the red (R) (0.68μm) and infrared reflectance (IR) (0.86μm). The NDVI is known to be particularly sensitive to alteration of greenness, which rapidly decreases during combustion. Thus, this index has been widely used to detect burned areas in temporal and boreal forests (Li *et al.*, 2000, Chuvieco *et al.*, 2002). In addition, the NDVI derived from the NOAA/AVHRR sensor in conjunction with active fire or thermal data has been successfully used in detection of burned area in the humid tropics (Fraser *et al.*, 2000, Fuller and Fulk, 2001, Idris *et al.*, 2005).

A number of other indices incorporating R and IR have been tested for burned area mapping, including the Global Environmental-Monitoring Index (GEMI) (Riano *et al.*, 2007), Second Modified Soil-Adjusted Vegetation Index (MSAVI2), and the Burned-Area Index (BAI) (Dempewolf *et al.*, 2007). Nowadays the most popular methods are based on change detection that incorporates short-infrared bands (SWIR) (1.6μm or 2.2μm). This is because the SWIR is particularly sensitive to vegetation moisture but less sensitive to atmospheric effects (Gao, 1996). Combustion of biomass reduces the vegetation moisture, thus there is an immediate decrease in SWIR values after fire due to the rapid absorption by combustion products (Fraser *et al.*, 2000). Hence a combination of SWIR (1.6μm) and IR (0.8μm) can be used to compute the Normalized Difference Water Index (NDWI) and its variations have been successfully applied to examine burned areas in both boreal and temperate forests (George *et al.*, 2006, Dasgupta *et al.*, 2007, Fraser *et al.*, 2000).

Recently, several studies have argued for the superiority of the Normalised Burn Ratio (NBR) over other indices in the context of burned area mapping and burn severity analyses (Epting *et al.*, 2005, Escuin *et al.*, 2008, De Santis and Chuvieco, 2006, Phua *et al.*, 2007). The NBR incorporates the SWIR (2.2μm) and IR (0.8μm); both wavelengths demonstrate sensitivity to moisture content. A recent study by Escuin *et al.* (2008), based on observation of index behaviour in the Mediterranean region, recommended applying a combination of bi-temporal (pre- and post-fire) indices (preferably bi-temporal NBR (dNBR)), with discrimination between unburned and burned pixels, whilst single, post-fire data (preferably NBR) can be used to discriminate between extreme and moderate severity.

In the context of fire detection in tropical forests, the study by Miettinen and Liew (2008) performed on Southeast Asian tropical forests using MODIS data, confirmed again the usefulness of the NBR for burned area detection, whereas the maximum surface temperature method produced surprisingly low separability. The small and fragmented burn scars which dominate in the Southeast Asian region limit the effectiveness of the low/medium resolution data in fire extent analysis (Miettinen *et al.*, 2007, Tansey *et al.*, 2008). A similar problem has been reported in Africa (Silva *et al.*, 2005). Therefore, the analysis of higher resolution data is essential to obtain more accurate and reliable results that can then be incorporated with lower resolution imagery. For example, the research by Phua *et al.* (2007) conducted in a tropical PSF region of Malaysia assessed the utility of a change detection technique based on Landsat ETM; their study confirmed the superiority of dNBR over dNDVI or dNDWI for burned area detection.

The detection of fire scars in the tropics seems to be much more challenging than in a non-tropical environment, where rather more homogeneous forest structures cover larger areas compared to the heterogeneous and often highly fragmented pattern of forest cover in tropical regions (Arino, 2001, Kauffman *et al.*, 1998, Miettinen *et al.*, 2007).

2.2.3 *Burn severity*

A further remaining challenge surrounding fire events in tropical ecosystems is to assess and quantify the magnitude of degradation caused by combustion. In general, burn severity is not easy to investigate or quantify (van Wagtendonk *et al.*, 2004). It is relatively well-studied in temperate and boreal regions with well-established, published methodologies, owing to the greater homogeneity of landscape structures and the tendency for large burned patches (Arino, 2001). In contrast, there have been very few studies of burn severity in the tropical zone; therefore, this Section focuses predominantly on non-tropical locations.

The magnitude of environmental changes caused by combustion is typically assessed directly after a fire event. A ground-based assessment of the effects of fire on vegetation and soil structure can be performed traditionally by labour-intensive measurements. These are focused on single variables, oriented at alterations in soil characteristics (Neary *et al.,* 1999, Charron and Greene, 2002, Robichaud *et al.,* 2007), canopy consumption (Hudak, 2004), and number of surviving and dead trees (Isaev and Orlick, 2002, van Nieuwstadt and Sheil, 2005). Another less time-consuming method proposed by Key and Banson (2004) is based on visual estimation of fire consequences. This ground-based method, called the Composite Burn Index (CBI), was originally developed for coniferous forests in Montana, USA. The CBI is performed by visual assessment; the method aggregates data for multiple ranging strata which are evaluated individually, including soil, understory, mid-canopy, and overstory vegetation. All values are then summed-up in one cumulative value of burn severity (ranging from 0 to 3) (van Wagtendonk *et al.*, 2004). The CBI approach is relatively quick (30 minutes per plot) but relies mostly on accurate estimation and judgement (Key and Benson, 2006). Thus this method is rather subjective and has given rise to some concerns (Lentile, 2006, Roy *et al.*, 2006, Smith *et al.*, 2007). Field-based measurements or CBI estimations are generally coupled with remote sensed imagery in order to determine the burn severity over a larger area.

In several studies, the capacity of indices combining the R and IR spectrums has been explored from a single (post-fire) and bi-temporal (pre/post-fire) perspective (Diaz-Delgado *et al.*, 2003, Chuvieco *et al.*, 2002). For example, Diaz-Delgado *et al.* (2003) demonstrated a positive correlation between a drop in NDVI values and burn severity. Also a strong correlation between tree mortality and NDVI values was found by Isaev *et al.* (2002).

Recently several works have, however, demonstrated the weakness of indices based on R and IR reflectance in comparison with those that are computed using SWIR. Epting *et al.* (2005) compared thirteen indices including individual bands; ratios and normalized differences for the best combination to discriminate burn severity in Alaska. Overall, the study concluded that the single post-fire NBR had the strongest correlation to the CBI followed by bi-temporal dNBR; the dNBR approach, however, produced a more accurate assessment of burn severity than the single NBR. The reason for this is because the difference between the pre- and post-fire NBR allows obtaining changes in the reflectance (van Wagtendonk *et al.*, 2004).

Several studies performed on high resolution data have confirmed a strong correlation between dNBR and CBI. Thus these methods have been applied widely to determine fire effects in the boreal and temporal mixed-coniferous forests of North America, including Alaska (Key and Benson, 2006, van Wagtendonk *et al.*, 2004, Epting *et al.*, 2005, Miller and Thode, 2007, Allen and Sorbel, 2008, Kokaly *et al.*, 2007, Wimberly and Reilly, 2007, Cocke *et al.*, 2005), in the Mediterranean ecosystem (De Santis and Chuvieco, 2006), and in the eucalyptus forests of southern Australia (Chafer, 2008). Furthermore, the dNBR values derived from low resolution sensors, such as MODIS or MERIS, have been successfully used to obtain burn severity for African savannas (Alleaume *et al.*, 2005), the Mediterranean ecosystem in Spain (Roldan-Zamarron *et al.*, 2006, Gonzalez-Alonso *et al.*, 2007), western Australia (Walz, 2007) and in North America (van Leeuwen, 2008). Recently a few studies have focused on simulation models, called radiative transfer models that can be used to predict how fire will alter the spectral reflectance (Chuvieco *et al.*, 2006, Chuvieco *et al.*, 2007, De Santis and Chuvieco, 2006).

Roy *et al.* (2006) have, however, questioned the utility of the NBR as a superior approach in measuring the effects of fire. They found that NBR was insensitive to fire severity shortly after a fire had occurred. This result contrasts with the work by Escuin *et al.* (2008) who showed that both pre/post-fire dNBR and dNDVI calculated for unaffected pixels were insensitive to spectral changes, but were sensitive to burn severity. Bi-temporal approaches require appropriate sets of images that might be restricted by data availability. Therefore, Roldan-Zamarron *et al.* (2006) highlighted the importance of examining the use of single post-fire imagery, since this approach could reduce both time and cost. They examined three methods in a Mediterranean ecosystem: Matched Filtering, Linear Spectral Mixture Analysis (SMA) and NBR and found that the SMA gave the best classification result for low resolution images (i.e. MODIS or MERIS), whilst for the Landsat imagery the Matched Filtering and NBR generated the most accurate and reliable results. In a tropical context the SMA technique has shown potential for classification of fire impacted forests in the Amazon basin (Gonzalez-Alonso *et al.*, 2007, Cochrane and Souza, 1998, Souza *et al.*, 2003). Epting *et al.* (2005) has, however, pointed out that the use of single imagery (i.e. without pre-fire reference imagery) leads to difficulties in mapping spectrally similar surfaces such as water, wet surfaces, recent burns, or older scars.

It should be stressed that the decision about which is the most appropriate technique is also strongly driven by data availability. Even though NBR has been demonstrated to be superior to NDVI in monitoring of fire extent and effects, sometimes it cannot be computed due to the lack of appropriate spectral bands. For example, the SWIR (2.2µm) wavelength is not available on the SPOT4, SPOT VEGETATION or the Disaster Monitoring Constellation (DMC) satellites. Hence if these satellite data are being used, another approach must be considered such as, for example the NDVI threshold method (Isaev and Orlick, 2002, Hammill and Bradstock, 2006). In addition, it must be emphasised that the dNBR thresholds derived from CBI for the burn severity classes proposed by Key & Benson (2006) are not a universal threshold and are only valid in environments that are similar to those observed in Alaskan forest (Key and Benson, 2006, Lentile, 2006). Therefore, it is important to highlight that these techniques, although appropriate in certain temperate and boreal locations, need to be tested over a range of tropical ecosystems where vegetation structure is far more complicated and where the landscape may be subjected to intensive and repeated fires or other forms of disturbance.

2.3 Secondary vegetation and uncertainties surrounding carbon flux

The regeneration of disturbed tropical forest vegetation is quite a complex and complicated process, the rate and extent of which will depend upon the land use history, the frequency and severity of disturbance, as well as soil conditions and hydrology (Lu, 2005, Hughes *et al.*, 1999). Several studies have been performed to assess and understand the processes of vegetation recovery occurring on abandoned agricultural land (Vieira *et al.*, 2003, Kalacska *et al.*, 2004, Arroyo-Mora *et al.*, 2005, Lucas *et al.*, 2000, Lucas *et al.*, 2002, Griscom *et al.*, 2009) and in selectively logged (Broadbent *et al.*, 2006) or burned (prior to logging) forest (Gerwing, 2002, Asner *et al.*, 2004) in the humid and dry tropical forest regions of Amazonia.

Roberts *et al.* (2005) reported that there is little information on post-disturbance forest recovery in either Southeast Asia or Central Africa. To date there have been a few ecological studies investigating post-fire vegetation regeneration in lowland dipterocarp forest in the Southeast Asian forest on mineral soil following single or repeated fires (Slik *et al.*, 2008, Hiratsuka *et al.*, 2006, Eichhorn, 2006, Kodandapani *et al.*, 2008, van Nieuwstadt, 2002) or selective logging (Slik *et al.*, 2001, Toma *et al.*, 2005, Bischoff *et al.*, 2005) but very few studies of post-fire vegetation regrowth in PSF. Existing limited studies focus on the effects of a single fire on species composition and vegetation structure (Yeager *et al.*, 2003) and biomass (Jaya, 2007). None of these studies have quantified the effect of multiple fires on vegetation structure. Several studies have highlighted slow biomass recovery as a particular feature of areas affected by fires

(Toma *et al.*, 2005, Nykvist, 1996, Chazdon, 2003). The negative effects of fire on forest regeneration are more pronounced in PSFs than in the more fire-resistant lowland forests (Yeager *et al.*, 2003).

Furthermore, to date, accurate forest inventory data for the tropics are rare and not precisely estimated (Houghton, 2007), leading to imprecise estimation of biomass volumes and of carbon emissions from tropical deforestation (Fearnside and Laurance, 2004). Section 5.1.4 provides more information on the difficulties and uncertainties surrounding biomass estimations in the tropics. Nevertheless, estimation of biomass in pristine forest or biomass loss following deforestation seems to be less problematic than the inventory of disturbed or recovering vegetation. This is mainly due to limited knowledge on the ecological processes associated with deforestation/degradation in conjunction with a lack of sufficient understanding of land transformations.

Understanding how vegetation responds to repeated fires on peatland require additional knowledge on the relationship between peat and vegetation; both components affected by fires. One of the consequences of repeated fires on peatland is a land subsidence, resulting in combustion losses of peat volume, which leads to an increased risk of flooding and consequently to the impaired vegetation regeneration (Wosten *et al.*, 2006, van Eijk and Leenman, 2004). The greatest area of uncertainty surrounding carbon flux in degraded peatlands is related to the quantity (volume) of peat available for combustion (Page *et al.*, 2002, Rieley *et al.*, 2008, Spessa *et al.*, 2009). Overall estimation of carbon loss from tropical peatland subjected to repeated fires is not a straight forward procedure and requires many factors to be considered (e.g. those relating to the fire and hydrological regimes).

2.4 Summary – current research gaps and needs

- The literature review has highlighted the following key research gaps, needs and in particular the lack of:Tropical forest inventory, recent land cover/use data and land cover/use change particularly over the recent period of intensive human activity, all of which limit the ability to provide accurate estimates of the rate of deforestation and limits carbon flux estimation.

- Knowledge about the spatial and temporal distribution of above and belowground biomass in tropics leading to inaccurate estimation of how much carbon has been released from burned or cleared biomass (from both above and belowground pools).

- Knowledge on the ecological processes associated with deforestation/degradation in conjunction with a lack of sufficient understanding of land transformations.

- Lack of reliable methods to quantify the degree of land degradation caused by repeated fires together with the short- and long-term effects of fires on vegetation recovery in the tropics.

- Understanding of how single and multiple fires modify the vegetation regeneration processes, and how the path and rate of vegetation succession depends on frequency and severity of disturbance and the fire return interval.

- Information on the capacity of vegetation to recover naturally after severe degradation related to repeated fires.

- Ecological understanding and tested methods on how to estimate the carbon losses from burning peat in the tropical peatlands subjected to different fire regimes.

All of the uncertainties and knowledge gaps listed above strongly influence the carbon fluxes from both above and belowground pools. Research in all of these areas will contribute to reduced uncertainties in estimates of both tropical and global carbon budgets.

3 STUDY AREA

In order to address the aims of this research, a large peatland area that had been subjected to fires over at least the last decade was chosen as an appropriate case study area. The study was carried out in the western part of the ex-Mega Rice Project, located in the southeastern part of Central Kalimantan province, Indonesian Borneo. The study area, known as Block C, covers an area of 448,912ha. It is bordered by the River Sebangau to the West and the River Kahayan to the East, the provincial capital of Palangkaraya to the North and the Java Sea to the South (Figure 3.1).

3.1 History of the Mega Rice Project

The Mega Rice Project (MRP) was established by the Government of Indonesia in order to convert around one million hectares of tropical forest, mostly forested peatlands (61% of the area) into cultivated land, particularly rice fields. This was a very ambitious but ad hoc project which ultimately failed owing to the unique, inimical physical and chemical properties of lowland peatland soils, which are almost completely unsuitable for agriculture (Sulistiyanto, 2004). The project started in 1995 with a phase of intensive infrastructure development, involving construction of a dense system of canals within each of five working Blocks A-E (Figure 3.1). It was intended that the network of more than 4,000km of canals would provide suitable hydrological conditions for rice production. Within a few years it was clear that the project was failing owing to the acidic, infertile peat soils, the malfunctioning irrigation system and the difficulties faced by transmigrant farmers attempting to undertake any form of agricultural development (Muhamad and Rieley, 2002, Sulistiyanto, 2004).

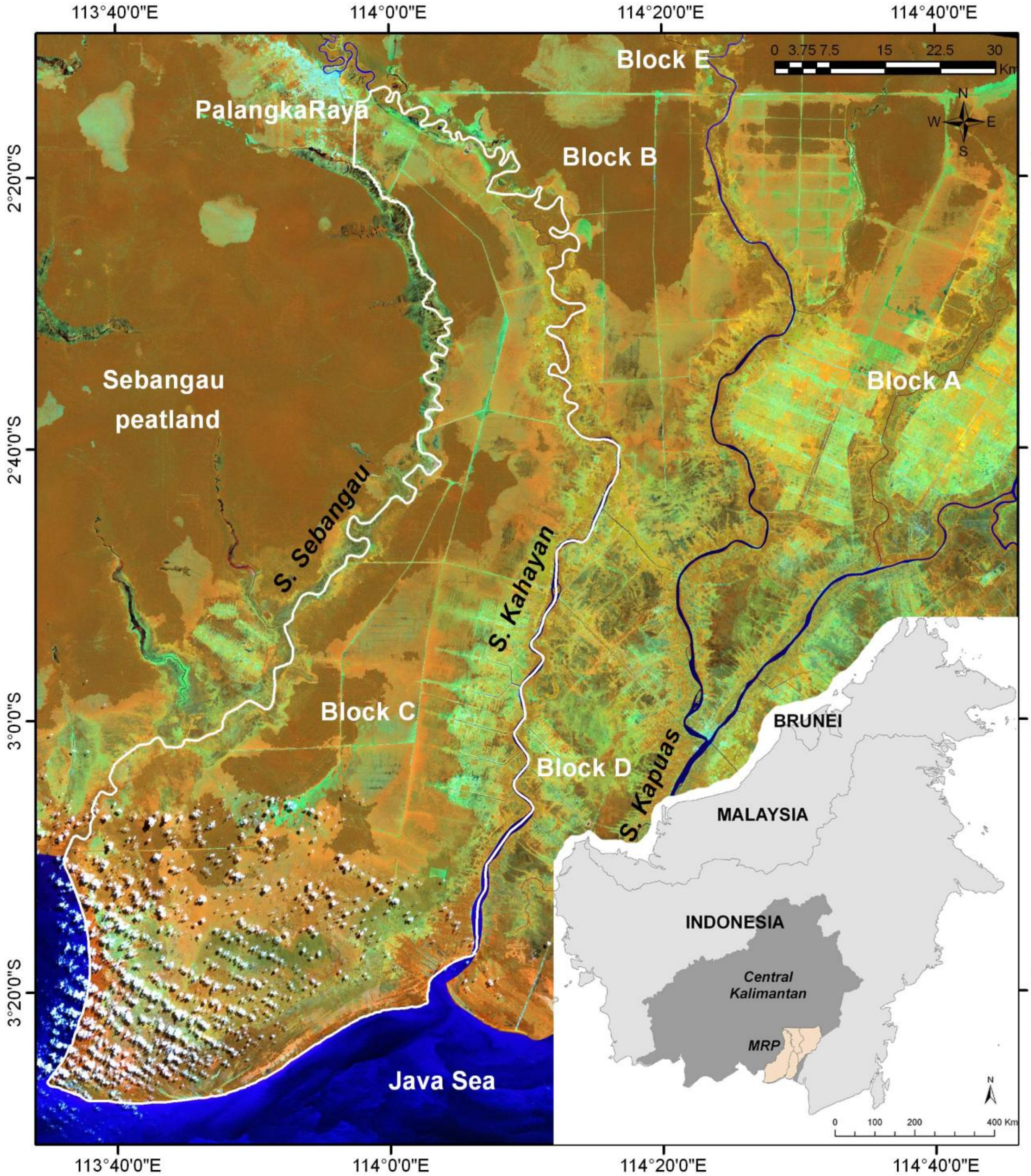


Figure 3.1 Location of the study area: Block C (boundary indicated by a white solid line) in the context of the Mega Rice Project (divided into five Blocks A-E); overlaid on the Landsat ETM+ image (2000-07-16).

The project had, however, transformed the landscape, altering the physical and chemical properties of the peat and a whole range of other ecosystem functions. As a consequence of these changes, devastating fires associated with the prolonged El Niño dry season of 1997 affected around 50% of the MRP area (Page *et al.*, 2002). After this incident, in 1999, the project was officially abandoned without any likelihood of further agricultural use.

In 2007, the Government of Indonesia decided to revitalize the ex-MRP area by undertaking a five-year program for the rehabilitation, conservation and sustainable development of the degraded landscape. The Government of the Netherlands in cooperation with the Governor of Central Kalimantan developed a Master Plan proposal for the former MRP area, the aim of which was to collect sufficient data to provide strategic guidance for long-term land management, considering environmental, socio-economical, cultural and administrative aspects (Euroconsult, 2008). Thus the area may now undergo a phase of restoration and rehabilitation, in which understanding of land cover and vegetation dynamics will be highly relevant.

3.2 Climatic characteristics

The MRP area is characterized by a humid tropical climate with a mean daily temperature fluctuating between 25 and 33^{o}C and humidity in the range 85-90%. Annual rainfall varies between 2000mm yr^{-1} and 3000mm yr^{-1}, increasing gradually from the coastal zone in the south towards the northern part. Annual evapotranspiration in forest is around 1350mm yr^{-1}(Euroconsult, 2008). The analysis of a long-term rainfall data set (1976-2007) undertaken for the Master Plan (Euroconsult, 2008), has revealed a decrease in rainfall over this time period (Figure 3.2), with the most significant decline in the southern part of the MRP. Rainfall is seasonal, with a dry season from May/June until September and a wet season from October until April/May. Rainfall exceeds the rate of evaporation during the rainy season, however, through the dry months evaporation normally exceeds rainfall.

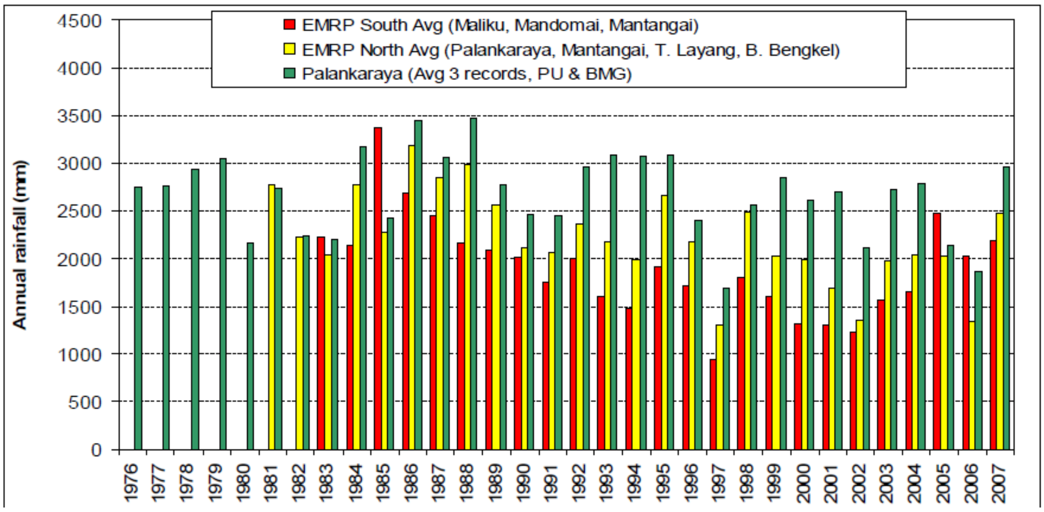


Figure 3.2 Record of annual rainfall for the MRP (Euroconsult, 2008).

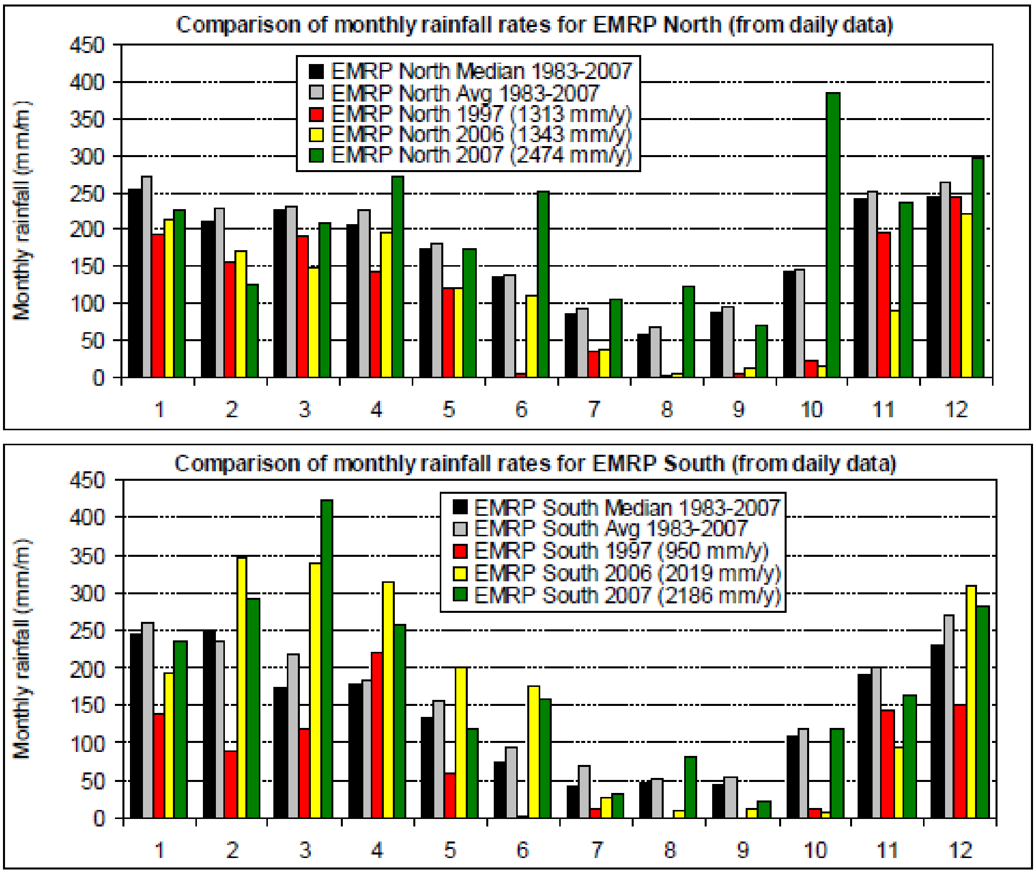


Figure 3.3 Comparison of monthly rainfall rates in the ex-MRP North and South areas, average, median and selected years of dry seasons: dry years 1997, 2006 and wet 2007; evapotranspiration on average equals 112mm per month (Euroconsult, 2008).

During ENSO years (e.g. 1997 and 2006) the dry season may begin earlier and last longer as shown in Figure 3.3. By contrast, in a 'wet' dry season, often associated with the La Niña phase of ENSO, such as the year 2007, the rainfall rate increases significantly. In recent decades there is an increasing tendency for a longer dry season with a reduction in rainfall over the antecedent period from February to May (Euroconsult, 2008). The projection of the future climate by Li *et al.* (2007) forecasted further reduction of rainfall during dry seasons in southern Borneo where the study area is located.

3.3 Topography and soils

The landscape of the study area is almost flat with an average altitude of 9 meters above sea level (measurements based on Shuttle Radar Topography Mission (SRTM) data).

Peat occurs in most of the study area and covers 324,667ha (72%) of Block C, which comprises almost 40% of the total peatland area within the MRP (Rieley and Page, 2005). Peatlands in the study area have a characteristic dome-shaped surface, which is related to the ombrogenous (rain-fed) peat formation (Figure 1.3).

Figure 3.4 shows that in a number of locations in the study area, the peat thickness exceed approximately 5m, with an average of around 4m that is equivalent to a peat volume of 9.4 x 10^9 m^3.

At the riverine margins the peat layer is very shallow (<1.0m) and in some places grades into mineral alluvial soils. In the southern and coastal part of the study area, the soils are sandy with evidence of sandy sediments occurring on a sandstone plateau landform with a well-established network of natural streams and rivers that flow south to the coast. These landscape features have been observed on a high resolution satellite image, but have not been confirmed by ground inspection.

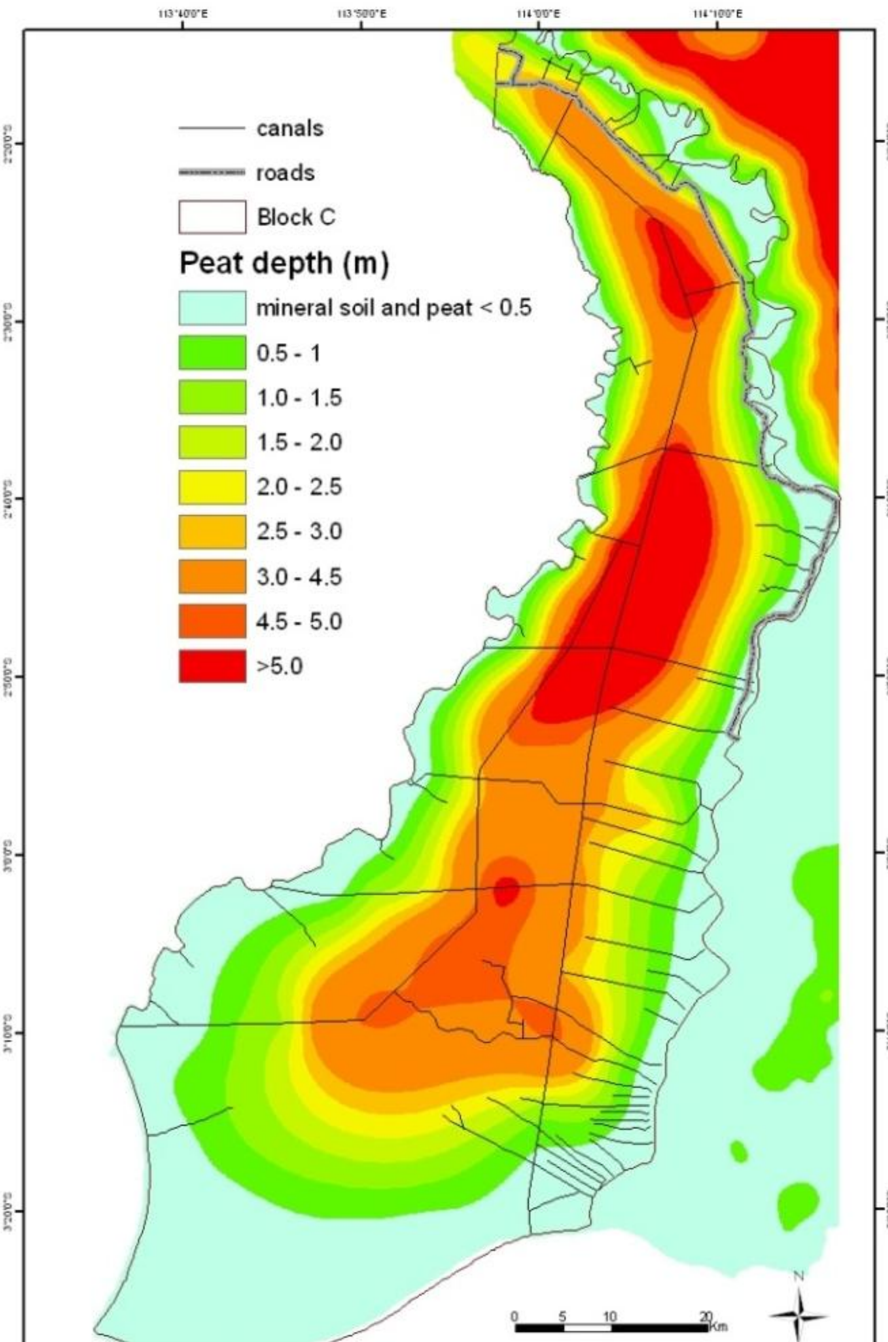


Figure 3.4 Peat depth measured in 2006 for purposes of the Master Plan (Euroconsult, 2008) with overlaid network of canals and roads.

3.4 Vegetation

Rain-fed ombrotrophic peat has a low nutrient concentration and a high level of acidity (Sulistiyanto, 2004), which determines the vegetation variability. The study area is dominated by peat swamp forest (PSF) that varies spatially. On the shallow peat, at the margins of the dome, occurs the most diverse mixed-PSF that supports tall, closed canopy forest. This forest type may contain on average 240 tree species ha^{-1} with several commercially valuable timber trees species such as *Shorea spp.* and *Dipterocarpus spp*., and thus has been heavily exploited. Moving further from the river toward the centre of dome, where the bog is almost flat, the peat becomes deeper and wetter, and simultaneously more nutritionally-limited. Forest in this area, called low pole peat swamp, supports trees with smaller and lower canopies, and is less diverse with only 30-55 tree species ha^{-1} (Rieley and Page, 2005, Page *et al.*, 1999).

On alluvial soils along river valleys, peat swamp forest grades into freshwater swamp forest, whilst marine fringes support mangrove forest. In the southern part of the study area, peat swamp forest is replaced by heath forest, which is typical of heavily leached podzol.

3.5 Population

The area of Block C falls within the administrative boundaries of the district of Pulang Pisau. Pulang Pisau district is one of 14 districts of Central Kalimantan, the third largest province in Indonesia, with Palangka Raya serving as the provincial capital. Within Pulang Pisau district, there are 53 villages divided among 7 sub-districts with a total population in 2005 of 99,201 people (25,036 householders) (Euroconsult, 2008). The settlements are mainly located along the road on shallow peat or mineral soil; there are no permanent settlements on deep peat.

4 FIRE REGIME AND LAND COVER DYNAMICS – METHODS AND RESULTS

This chapter begins with the description of the Earth Observation (EO) data used in the analysis and then moves on to the methods applied to quantify the land cover and fire regime dynamics for the period 1973-2005. Results are presented in Section 4.4 and 4.5 for land cover and fire regime dynamics; the final Section 4.6 focuses on the analysis of relationship between fire incidence and extent and the ENSO-MEI index.

4.1 Satellite data description and pre-processing analysis

This Section contains a description of the series of satellite images used to conduct analysis of the land cover dynamics of the study area and to determine its fire history. Section is divided into two sub-sections which specify a) the type of satellite missions and b) the pre-processing analysis performed on the data.

LANDSAT mission and data availability

The Landsat program, managed by NASA, is one of the longest continuous EO satellite missions providing a 35-year data record beginning in 1972. The satellites are placed in an orbit with an altitude of 705km, with 16 days repeat overpass mode, on a polar, sun-synchronous orbit. This extremely valuable high resolution, multispectral data has been used in a variety of regional and global applications, and has supplied invaluable data for long-term monitoring of land alterations (Rogan and Chen, 2004). The first satellite, the Landsat 1 Multispectral Scanner (MSS) was launched in 1972, followed by Landsat 2 and 3 in 1975 and 1978, respectively, with similar configurations (Table 4.1). The next two satellites, Landsat 4 and 5, were launched in 1984 with a new sensor called the Thematic Mapper (TM). The last satellite, Landsat 7 was equipped with a new Enhanced Thematic Mapper (ETM+) instrument. It was launched in 1999 and is still in operation. However, since May 2003, Landsat 7 has been experiencing an operational problem with the Scan Line Corrector which causes a missing-lines effect visible over a quarter of the scene. Currently data products are available with the missing-lines or optionally filled in using other Landsat7 data selected by the user.

	Landsat 1, 2 & 3	**Landsat 4 and 5**	**Landsat 7**	**DMC**
	Multispectral Scanner (MSS)	Thematic Mapper (TM)	Enhanced Thematic Mapper (ETM+)	
Operation time	1972-1982	Since 1982	Since 1999	Since 2005
Altitude (km)	915	705	705	686
Revisit period (days)	18	16	16	1
Swath width (km)	185	185	185	600
Spatial resolution (m)	79x79	30x30	30x30	32x32
Spectral bands (μm)		0.45-0.52	0.45-0.52	
	0.50-0.60	0.52-0.60	0.52-0.60	0.52-0.60
	0.60-0.70	0.63-0.69	0.63-0.69	0.63-0.69
	0.70-0.80	0.76-0.90	0.76-0.90	0.77-0.90
	0.80-1.10			
		1.55-1.73	1.55-1.73	
		2.08-2.35	2.08-2.35	
	10.4-12.5 (237x237m^2)	10.4-12.5 (120x120 m^2)	10.4-12.5 (80x80 m^2)	
			0.52-0.90 (15x15m^2)	

Table 4.1 Technical specification for the Landsat missions and Disaster Monitoring Constellation (DMC).

A series of thirteen Landsat scenes including MSS, TM and ETM+ (Table 4.2) were used to carry out the analyses of land cover in this study. Fortunately, the entire study area was situated in the centre of a single Landsat scene (orbit parameters: line 118 row 62). Therefore, additional data processing, such as mosaic and contrast adjustment, was not necessary. Moreover, the central position of the study area minimized the missing-line problem appearing on the image since 2003, as only the edges of the study area were affected. The presence of clouds and/or smoke on some of the images meant that not all images could be fully analysed.

To integrate the remote sensed data with other data within the context of a GIS, the pixel-coordinate reference system of the image had to be transformed into an identical coordinate system. All images were therefore stored in the Universal Transverse Mercator (UTM) projection system, zone 50 South. An ortho-rectified image of Landsat ETM+ acquired in July 2000 was used as a reference image to which all other images were co-registered. All images were projected using second polynomial order with total residual Root-Mean-Square (RMS) error of less than a half pixel. A number of around 25 ground control points (GCP) per each scene was used to perform geometric transformation. Every Landsat TM and ETM+ image was atmospherically corrected using the ATCOR model available in PCI Geomatics software.

Satellite mission	Date of acquisition	Additional info
Landsat MSS	1973-10-09	
	1985-06-25	cloudy in southern part
Landsat TM	1991-06-30	
	1993-06-19	
	1994-07-24	patchy clouds in southern part, only 3 bands
	1996-05-10	
	1997-05-29	
	1998-03-29	hazy
Landsat ETM+	2000-02-14	cloudy, only 3 bands
	2000-07-16	
	2001-08-20	partly hazy
	2003-01-14	patchy clouds in southern part
	2005-10-02	partly hazy
DMC	26 images (from 2005-06-18 to 2005-10-26)	some of them cloudy

Table 4.2 Series of satellite images used in this study.

Disaster Monitoring Constellation

The DMC is a collaborative satellite operation system involving Algeria, Nigeria, UK, Turkey and China. It was designed as an alternative to Landsat and in order to carry on the series of valuable observations initiated by that mission. DMC comprises a constellation of five small satellites in a single orbit plane. It is capable of providing unique EO with a daily revisit anywhere in the world. This advanced satellite system delivers data essential for monitoring disasters and earth changes on a daily basis and provides a high frequency product with a high spatial resolution of 32m and a wide swath of 600km (three times wider than Landsat). The frequent revisit time provides greater opportunities for obtaining cloud free observations. The spectral bandwidth and channel characteristics are outlined in Table 4.1 along with the equivalent Landsat ETM+ spectral bands.

The DMC images were provided by *DMC International Imaging* (www.dmcii.com). The registration of data was planned in advance for the dry season of 2005, owing to a high probability of obtaining cloud-free images. In total, twenty-six DMC scenes covering the whole study area were acquired between 18th June and 26th October 2005. Examples of the DMC images used in this study are provided in Appendix 1. All cloud- or partly cloud-free images were geometrically corrected to the reference Landsat ETM+ image, recorded in July 2000, UTM, zone 50 South (using around 30 ground control points per image). The second polynomial order was chosen as a transformation model, with RMS error of less than a half pixel. All images were resampled into a pixel size of 36m x 36m by applying a nearest neighbour algorithm. The DMC data (L1R product) were delivered as scaled radiance (each band scaled individually), so the pixel values had to be rescaled to the radiance using the following equation (4.1):

(4.1)

$$\mathrm{RADIANCE} = \frac{\mathrm{DN}}{\mathrm{RESCALE\ GAIN}} + \mathrm{RESCALE\ BIAS}$$

Where:

DN represents the scaled radiance; the scaling coefficients (rescale gain and bias) are unique for every image and stored in the metadata files that accompany every L1R product. The unit of radiance is Watt per square meter steradian micrometer ($Wm^{-2}sr^{-1}\mu m$) (Crowley, 2008).

As a next analysis stage, the NDVI, based on the red and infrared bands (4.3), was calculated for each image, and finally the 20-day composite image was generated based on a pixel selection of the maximum NDVI value.

4.2 Methodological approach – Land cover mapping

Land cover maps were derived from time series of satellite images obtained from several sensors, including the Landsat MSS, TM, and ETM+. The process of classification consisted of three stages a) field reconnaissance, b) defining an adequate legend and c) mapping the land cover.

4.2.1 *Definition of land cover classes*

Legend establishment is one of the crucial stages in classification. The legend should be consistent over the whole period of investigation and replicable for areas with similar land history, as well as comparable with existing and widely used maps. Therefore the legend was based on the widely used Tropical Ecosystem Environment Observation by Satellite (TREES) classification scheme which had been modified specifically for the Insular Southeast Asia region by Stibig *et al.* (Stibig *et al.*, 2007, Stibig *et al.*, 2002, Stibig *et al.*, 2003). The TREES land cover categories needed to be subsequently amended to provide an appropriate sub-regional context as well as more specific, detailed delineation of vegetation classes that might be important for analysis of post-fire vegetation regrowth (Table 4.3). This was reasonable because originally the TREES classification was based on SPOT Vegetation imagery with a spatial resolution of 1km, whereas the classification in this study was conducted using finer resolution Landsat data. The land cover legend was divided into two main categories, namely forest and non-forest vegetation cover (Table 4.3).

TREES Classification	**Land Cover classification used in this study**	
	Classes	**Sub-classes**
FOREST COVER		
Swamp forest	Peat swamp forest	Mixed-PSF
		Low pole-PSF
		Heavily logged PSF
		Secondary forest
	Freshwater swamp forest	
Mangrove forest	Mangrove forest	Mangrove forest
		Fragmented and/or degraded mangrove forest
Heath forest	Heath forest	
Mosaic of tree cover and other natural vegetation	Mosaic of trees and non-woody vegetation	
NON-FOREST VEGETATION COVER		
Evergreen shrub and re-growth	Non-woody vegetation	
	Sedge swamp (non-woody vegetation)	
Burnt vegetation	Recently burned area (no vegetation)	
Cultivated and managed land	Cultivated land	
	Water	Blackwater lake
		Rivers

Table 4.3 Categories of land cover classes and sub-classes used in this study and their relationship to the TREES classification.

4.2.2 *Ecological description of the classes*

FOREST COVER

Since the entire study area lies below 1000m, the forest was classified as evergreen lowland forest, and subdivided into three forest types: swamp forest, heath forest and mangrove forest. ‘Forest’ vegetation was defined as having a canopy closure of greater than 60%; the threshold was defined based on field measurements of vegetation structure. Below this threshold woody vegetation was defined as heavily logged forest, mosaic of trees and non-woody vegetation or fragmented and/or degraded forest.

Swamp forest

The class ‘swamp forest’ consists of *peat swamp forest* (PSF) and freshwater swamp forest. PSF grows on a thick rain-fed peat layer formed under waterlogged conditions (see Section 3.2.2), whereas *freshwater swamp forest* is periodically flooded or waterlogged by mineral-rich freshwater, and therefore often occurs on mineral soils alongside rivers and lakes.

PSF was further divided into *mixed peat swamp forest* (mixed-PSF) and *low pole peat swamp forest* (low-PSF) according to the forest structure. The most diverse mixed-PSF is often subjected to intensive logging due to the presence of commercially valuable timber species. Intensively logged forest (tree cover almost completely removed) was mapped in this study as a *heavily logged peat swamp forest* class. The low-PSF was less affected by loggers owing to the much lower density of commercially valuable timber species. Low-PSF has greater density of smaller diameter trees and a lower, less dense forest canopy compared to mixed-PSF. Differences in dimensions of the tree canopy crowns, tree density and canopy closure clearly separated the mixed-PSF from the low-PSF class on the satellite image; the mixed-PSF presents a rougher and heterogeneous texture than the more homogenous, smooth low-PSF.

Forest presently undergoing regrowth following disturbance either by intensive logging or fire was classified as *secondary forest*. The term 'secondary forest' in this study refers to the advanced regrowth stage of PSF, with high canopy closure, and uniform texture. In addition, secondary forest reflects more near-infrared energy than primary forest, again providing clear differentiation between these two forest types.

Mangrove forest

Mangrove forest is situated along coastlines and estuaries of rivers. The floristic diversity is much lower than that of other forests classified in this study and is determined by frequency and duration of tidal flooding. Large areas of mangrove have been lost or severely degraded in the last few decades mainly through conversion to shrimp and fish ponds, reclamation for agriculture or logging for charcoal and woodchip, thus most of these fragments were classified as *fragmented and/or degraded mangrove*.

Heath forest

Heath forest (kerangas) occurs on well-drained acidic (pH<4), sandy soils, that are lacking in nutrients (particularly nitrogen). This soil is often called white-sand soil owing to intensive leaching of the upper soil profile (MacKinnon, 1996). In areas where the soils become waterlogged (lose their drainage capabilities), the forest floor becomes swampy, and is known locally as 'kerapah'. A large area of heath forest was identified on a sandstone plateau landform in the southern part of Block C.

Mosaic of trees and non-woody vegetation

This class represents a combination of tree and non-woody vegetation dominated mainly by ferns, grasses, and shrubs. This transition class between secondary forest and non-woody class represents typically the first woody stage of post-fire vegetation regrowth, with progressive succession towards secondary forest. Discrimination between secondary forest and mosaic of trees and non-woody vegetation was based on the proportion of tree canopy cover to non-woody vegetation. Scattered patches of trees were classified as mosaic trees and non-woody vegetation, whereas woody vegetation with a closed canopy was classified as a secondary forest.

NON-FOREST VEGETATION COVER

Non-woody vegetation

The term *non-woody vegetation* is applied indiscriminately to all vegetation types that may be natural and owing their existence to some combination of soil factors or hydrology, or which are a consequence of land use activities. In the study area almost all non-woody vegetation cover was secondary in nature, having arisen from degradation of the original forest cover. Deforestation and regular burning are the principle drivers of this degradation process. The non-woody vegetation class is principally dominated by ferns, grasses, sedges, shrubs and a few trees (Chapter 5).

Sedge swamp

Sedge swamp is a natural non-forest vegetation dominated by sedges (*Cyperus, Eleocharis, Lepironia spp.*, *Pandanus helicopus)* and small woody shrubs located in riverine alluvial plains, where flooding is frequent or prolonged, or the substrata is too unstable to support the establishment of trees (Gupta, 2005). In the study area, however, it is likely that the sedge swamp community that covers the riverine plains is a form of secondary vegetation that occurs following burning of the original riverine forest (this assumption is based on information from indigenous people).

Recently burned area

This class comprises surfaces recently subjected to fire. These have a low or complete absence of vegetation cover, and as a consequence, a large proportion of bare ground. Over time, these areas undergo succession towards the non-woody vegetation class.

Cultivated land

This class comprises both woody and non-woody crops growing mainly on shallow peat or mineral soils. Non-woody cropland consists of fields of maize, cassava, sweet potatoes, rice or vegetables. Tree plantations are relatively small and planted mainly with rubber trees.

Water bodies

This class includes both rivers: the Kahayan River and the Sabangau River that border the study area and blackwater lakes which, prior to drainage and fire damage, were situated on the highest part of the dome in the southern part of Block C.

4.2.3 *Spectral response from different land cover types*

To date there are hardly any ground data available on the spectral properties of pristine and degraded tropical forest, and in particular on post-fire vegetation regrowth. Small scale field spectroscopy was therefore employed in this study in order to a) characterise the spectral response of the various land cover types in the study area, b) understand how fire modifies the spectral reflectance of surfaces subject to burning and to different fire regimes (frequency and severity), and c) characterise the spectral reflectance of post-fire vegetation regrowth.

The first mission to Indonesia in 2005 involved collection of ground reflectance data using a spectroradiometer GER-1500 loaned from the UK's Natural Environment Research Council (NERC). This hand-held instrument records a spectral range from 350nm to 1050nm in 512 channels with a bandwidth of 1.5nm. A series of measurements were taken in different land cover types that had been provisionally identified on the satellite image and which could be subsequently recognised in the field. These cover types comprised: PSF, post-fire vegetation regrowth (i.e. non-woody vegetation and sedge swamp), two types of fern-dominated vegetation, bare peat soil and recently burned areas. All measurements were taken from above the canopy. The reflectance of PSF was measured from the top of the Japanese-founded flux tower located in the northern part of Block C. The variation in the spectral reflectance for different land cover types (Figure 4.1), demonstrates the pronounced effect that fire has on spectral properties of the land surface.

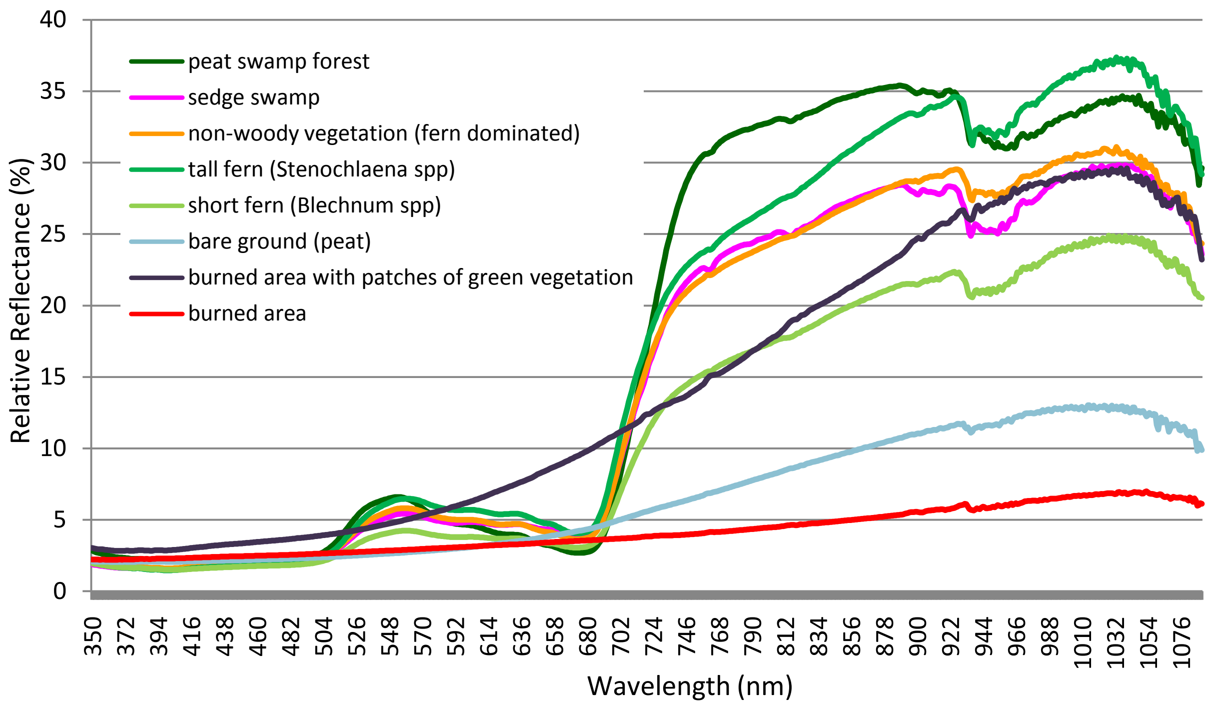


Figure 4.1 Spectral reflectance for different land cover types measured on the ground using spectroradiometer GER-1500.

For PSF, the spectral reflectance rises sharply between 680nm and 760nm, and remains high in the infrared region between 760nm and 1076nm. If green vegetation is burnt entirely, the level of reflectance declines drastically in both the visual and infrared spectrum (burned area); the curve becomes nearly flat. In comparison, if the burned area contains patches of vegetation, the spectral reflectance curve rises gradually from 400nm to 1054nm, reaching a peak around 1020nm. However, the slope of the curve is steeper than for either burned areas or bare ground. Distinctive differences in the spectral properties of burned and un-burned surfaces clearly separate the fire-affected areas and may also indicate variation with burn severity (e.g. between sites with no or some vegetation remaining).

The spectral curves for regenerating vegetation (i.e. the non-woody vegetation and sedge swamp classes) are very similar in shape, with lower reflectance values in the infrared (750nm - 900nm) compared with PSF. This similarity is a function of the similar species composition in both classes, which comprise a mixture of ferns, sedges, shrubs and a few trees. The non-woody vegetation class is dominated by two species of ferns (*Stenochlacena palustris* and *Blechnum indicum*) and there is a clear distinction between burned areas dominated by 'tall' fern (i.e. *S. palustris*) and those dominated by 'short' fern (*B. indicum)*. The *S. palustris*-dominated areas have quite a high ground cover, even though there are only a few trees present. Thus, the spectral reflectance is substantially higher in the infrared spectrum than in areas dominated by *B. indicum.* The more scattered distribution of *Blechnum spp.* and the larger proportion of bare ground reduce the reflectance between 520nm and 600nm in the infrared spectrum due to an increase in surface moisture content.

4.2.4 *Classification process*

The definitive land cover classification for this study was carried out using a visual-expert interpretation based on knowledge obtained during the field reconnaissance mission. A trial using a digital classification method confirmed (as stressed in Section 3.1) that the heterogeneous character of the land surface, the result of the combination of fire-degraded forest and secondary vegetation (often in different stages of regrowth), made pixel-based classification ineffective. Inconsistent 'salt-and-pepper' results impaired the extraction of objects of interest. In addition, post-fire secondary vegetation, which is a combination of different types and proportions of ferns, sedges, shrubs and trees, showed a similar spectral response to other land cover classes (see Section 4.2.3), which made the pixel-based separation impossible. Thus, visual-expert interpretation was selected as the optimal method to map land cover. A visual-expert interpretation of satellite images involved additional assessment of the context of objects as well as their tone and texture. Context describes the relationship between one object and another, nearby object. The contextual approach requires additional ecological knowledge, which helps to identify the presence of some objects in a certain context. For example, PSF occurs on peat soil whereas freshwater swamp forest does not, and sedge swamp or riverine forest is restricted to the floodplains of rivers. An automatic contextual approach could not be applied because of the lack of relevant software at that time. The visual-expert interpretation was chosen as the optimal method to deliver the best, most consistent and comparable results. Moreover the study area was relatively small (448,912ha), less than half of a Landsat scene, thus the visual-expert classification was not considered too time consuming, whilst, the ecological expertise and knowledge of the landscape obtained from people familiar with the study area provided sufficient relevant information to interpret the images.

Figure 4.2 illustrates the methodology applied to map land cover for the period of investigation (1973-2005) plus extraction of burn scars for the fire that occurred over the period 1973-1997. Land cover mapping and post-classification analyses were performed using ESRI ArcGIS software.

In total, fifteen-land cover classes were distinguished (Table 4.3) and land cover maps were derived for six years: 1973, 1991 (before dry season, i.e. before fires started), 1993, 1997 (before fires started), 2000 and 2005.

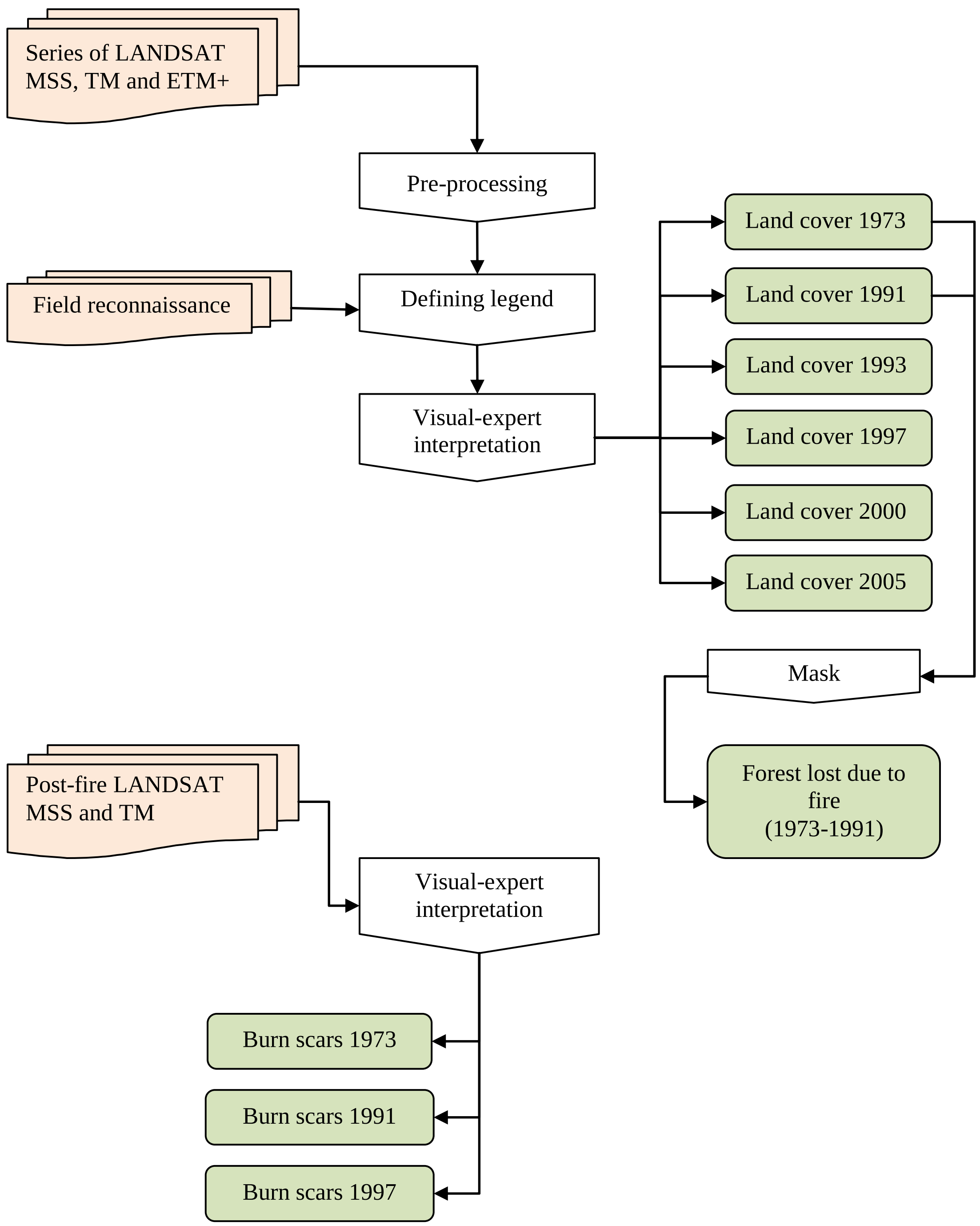


Figure 4.2 Flow chart illustrating the process of land cover mapping for the period 1973-2005 and extraction of burn scars for the fires that occurred over the period 1973-1997.

4.3 Methodological approach – Determining fire regime

This Section presents the methodological approach used to distinguish burned areas and to investigate the fire regime for the period 1973-2005. The methods were chosen based on a) a review of relevant published methodologies (Section 2.2.2) and b) data type and availability.

4.3.1 *Burned area detection*

Detection of burned area requires either both pre- and post-fire images or a single post-fire image taken not long after the fire event. It was important to have information on the length and timing of the burning season in each year under investigation in order to avoid inadequate data purchase and analysis.

The Web Fire Mapper (http://maps.geog.umd.edu) allows for the rapid display and query of current and archive active fire data (from 2001 onwards) for the entire globe. The active fire locations, called hotspots, provided by this source are obtained almost in real time by the spectroradiometer MODIS instrument placed on two satellite platforms, Terra and Aqua. This source was used to ascertain the length and time-range of the burn season for the fires in 2002, 2004 and 2005.

The onset of the fire season was taken as the date on which the hotspots started to cluster spatially and temporally, whereas the end of the fire season was set as the date on which clustering of hotspots no longer occurred (Li *et al.*, 2000).

Due to variation in date of acquisitions and quality (i.e. some of the images were recorded a few years after burning) different techniques had to be applied to derive the fire extent. Burned area detection was undertaken using both the traditional method, based on manual interpretation, and a digital automatic method. The first method was applied only to the archival data obtained before the year 2000 (Figure 4.2); the automatic detection of burned areas on these images was limited by rather poor quality imagery (haze contamination). The burn scars from the 1973, 1991 and the 1997 fires were, therefore, manually interpreted based on single post-fire Landsat MSS and TM scenes. The visual interpretation in conjunction with land cover mapping was used to derive the burn scars resulting from fires during the period 1973-1991.

After the 1997 fire, the pattern of burned areas revealed a more patchy character with a large number of scattered, smaller burn scars that were difficult to delineate manually, thus a more robust, automatic method was applied. The commonly used spectral change detection methods (Li *et al.*, 2000, Chuvieco *et al.*, 2002, Epting *et al.*, 2005, Escuin *et al.*, 2008, Phua *et al.*, 2007; see Section 2.2.2). were chosen to delineate the burn scars resulting from fires in 2002, 2004 and 2005 Figure 4.3 presents the process of burned area extraction for the fire 2002, 2004 and 2005.

A study presented by Phua *et al.* (2007) demonstrated the effectiveness of the bi-temporal Normalised Burn Ratio (dNBR) index for mapping burned areas in the tropical peatlands of Southeast Asia. Several other studies have also recommended this index for use in the discrimination of fire-affected locations (Section 2.2.2). Therefore, the dNBR was chosen as an appropriate method to derive the burned areas for the 2002 fire. The single NBR index was calculated separately for pre- and post-fire images from observations of the surface using IR and SWIR wavelengths (4.2).

Then the dNBR was obtained as follows:

$$NBRpre = IR(4) - SWIR(7) / \; IR(4) + SWIR(7)$$
$$NBRpost = IR(4) - SWIR(7) / \; IR(4) + SWIR(7) \qquad (4.2)$$
$$dNBR = NBRpre - NBRpost$$
$$IR(4) = 0.86\mu m; \; SWIR(7) = 2.2\mu m$$

The pre-fire image was acquired in July 2000 and the post-fire image acquired in February 2003. Both input images were atmospherically corrected using the ATCOR module built into the PCI Geomatics platform.

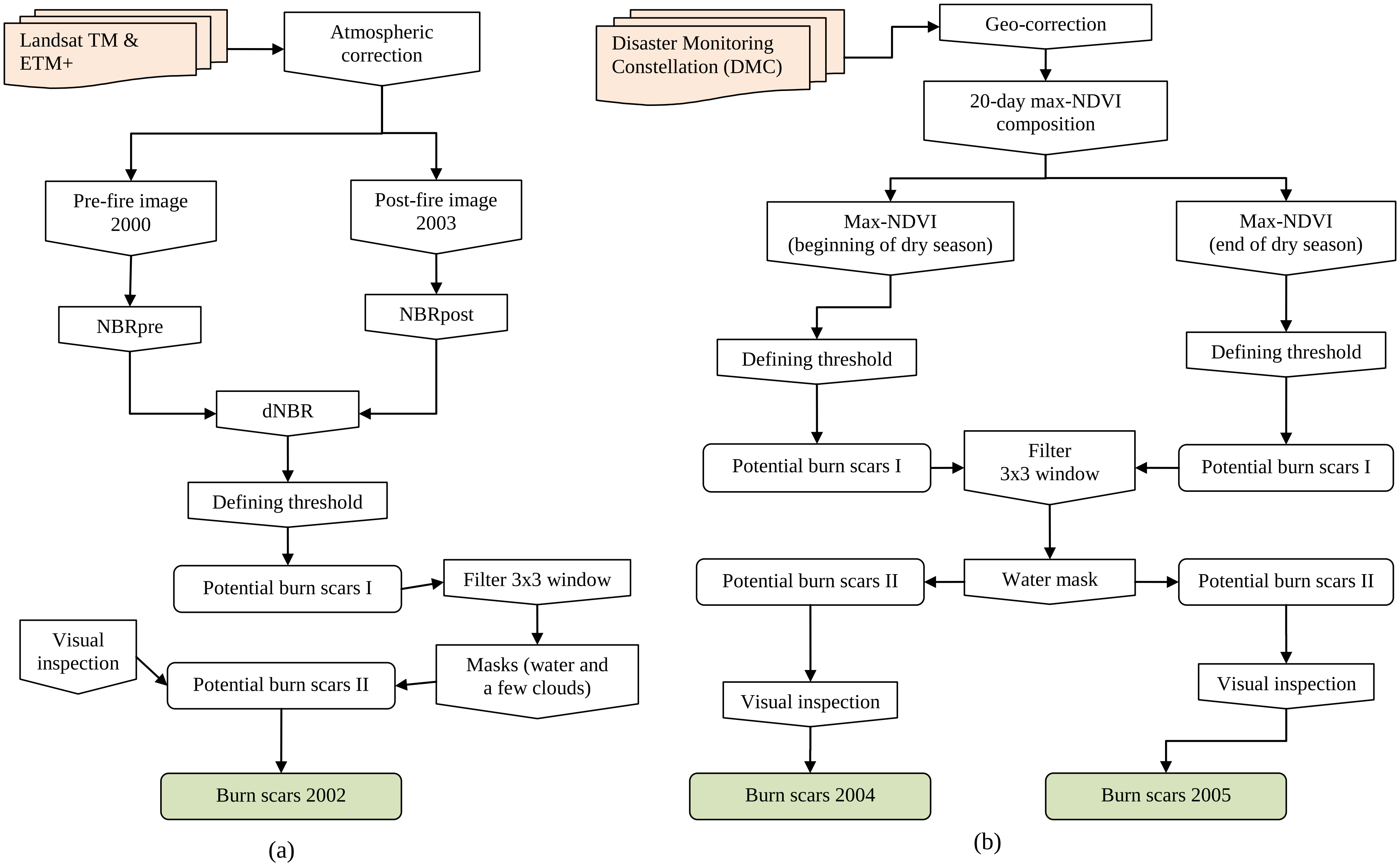


Figure 4.3 Burned area mapping for a) the fires in 2002, using bi-temporal NBR index and b) fires in 2004 and 2005, using NDVI index.

The mask of potential burned area was obtained by thresholding the dNBR values at an empirically-determined level (Loboda *et al.*, 2007). This threshold was determined from the frequency distribution of the dNBR values over sampling areas with known fire presence, based on remote sensing knowledge about spectral properties of burning surface. It was important that selected samples represented a sufficient number of burned and unburned values in the histogram (Figure 4.4). The defined threshold value was checked and adjusted manually based on the post-fire imagery. The threshold of burned pixels was defined as a value equal to or greater than 0.1 (Mean +0.3σ).

The next stage was to apply a 3x3 filter pixel window to potentially burned patches to remove the 'salt-and-pepper' effect, assuming that a potential burned pixel must thus be surrounded by at least four other potential burned pixels within an eight-cell neighbourhood. Finally, masks for water bodies and clouds were used to remove additional noise. These masks were prepared using unsupervised classification. Clouds were mainly present in the southern part of the study area that was less affected by fire.

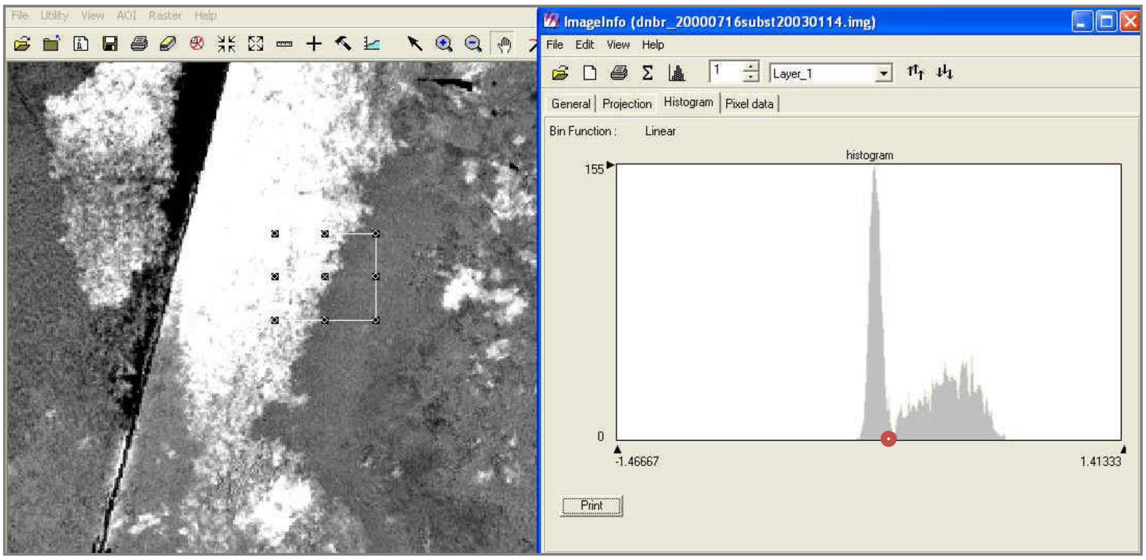


Figure 4.4 Frequency distribution of dNBR values for threshold development test windows; red dot indicates the burned threshold.

The dNBR was used in mapping the burned area from the 2002 fire, but it could not be used to map the fires of 2004 and 2005, because of the limitations caused by Landsat data availability. Hence, a time-series of DMC images were used as a substitute for the Landsat data. The NBR index could not be calculated from the DMC data, however, because of the absence of the SWIR band, which is required to compute the NBR. Thus the masks of burned area for 2004 and 2005 were derived from a time series of 20-day image composites constructed using a max-NDVI value criterion (Figure 4.3). The max-NDVI approach minimizes to a certain extent the impact of haze and clouds. NDVI was calculated based on difference between R and IR spectrum (4.3).

$$NDVI = R(3) - IR(4) / R(3) + IR(4) \tag{4.3}$$

$$R(3) = 0.68\mu m;\ IR(4) = 0.86\mu m$$

The 2004 burn scars were derived from a max-NDVI composition obtained at the beginning of the 2005 dry season (i.e. just before the fire season started; period 2005.07.30–08.21), whereas the burn scars of 2005 were estimated from a post-fire (end of dry season) max-NDVI composite (period 2005.10.06–10.26). The mask of burned area was obtained by thresholding the NDVI values following the same methods used for the 2002 burned area mapping. NDVI threshold values equalled less than 0.15 for the 2004 fire and ranged between 0.35 and 0.59 for 2005. A 3x3 model filter was applied to remove 'salt-and-pepper' effects. The quality of the burn scar map was verified by visual inspection using the Landsat ETM+ 2005 image. This image could not be used to detect the fire scars automatically because of the presence of thin clouds.

4.3.2 *Determining fire frequency and fire-return interval*

Fire frequency indicates how many times fire affects an area over a certain period of time. In other words it is the temporal-spatial distribution of fire recurrence. Fire frequency analyses were undertaken for the entire period of investigation (1973-2005), and for two separate periods, 1973-1996 and 1997-2005. The year 1997 was used as a breakpoint in order to investigate the impact of the implementation of the MRP on fire frequency. The HABITAT platform designed by WL-Delft Hydraulics (http://habitat.wldelft.nl) was used to perform this analysis. This is a spatial analysis software package built around PC-Raster that is specially designed to support ecosystem management. HABITAT allows one to perform ecological analyses in an integrated and flexible manner by generating spatial (maps) and quantitative (tables) results (van der Lee and Haasnoot, 2006).

All burned scars for each year were converted into grid, binary format (.bil) using ESRI ArcGIS software and then imported into HABITAT. The mathematical multiplying operation was used to derive the final fire frequency maps and descriptive statistics, consisting of the area affected by single and multiple fires, for each period.

The fire frequency analysis showed high fire recurrence over the period 1997-2005 (i.e. the post-MRP era); therefore this period was chosen to look at the likelihood of a fire incident in the context of intervals between two subsequent fires. The period 1997-2005 encompassed four fires, i.e. the fires of 1997, 2002, 2004 and 2005, thus six combinations of return intervals were possible (Table 4.4). The total burned area for each individual case of fire-return interval was calculated separately.

Fire return interval (years)	Fire occurrence
1	Burnt in 2004 + 2005
2	Burnt in 2002 + 2004
3	Burnt in 2002 + 2005
5	Burnt in 1997 + 2002
>7	Burnt in 1997 + 2004 Burnt in 1997 + 2005

Table 4.4. Possible combination of fire return interval between two subsequent fires.

Analysis of fire frequency and fire-return interval were necessary to understand and interpret the processes of vegetation regrowth (Chapter 5) and to determine the carbon losses from burning peat and aboveground biomass over the period of investigation (Chapter 6).

4.3.3 *Human access points analysis*

Proximity analysis in this study, referred to human access points (HAPs), was performed using Euclidean distance tools that measure distance from the centre of source cells to the centre of destination cells. HAPs analysis was performed using the HABITAT platform. The term HAPs refers to the network of drainage canals and roads present within the study area, all of which provide opportunities for human access. Each of these features was digitised based on the satellite images, and derived separately for the periods 1973-1996 and 1997-2005. The length of canals and roads was calculated separately for these two periods. A buffer zone of 2500 meters was defined for each canal and road, and then divided into intervals of 250 meters. Within each interval, the total burned area was calculated taking into account fire frequency (i.e. areas burnt once, twice, three, four or more times inside of each interval).

4.4 Results – Land cover analysis 1973-2005

The land cover statistics for the study area for the period of investigation (1973-2005) are presented in Table 4.5, whereas Figure 4.5 illustrates the long-term trend in the land cover dynamics.

In 1973, just after the fires that occurred in that year, forest covered 73% (327,038ha) of the total area of Block C, of which 60% (267,733ha) was defined as PSF. The PSF comprised 57% mixed-PSF and 3% low-PSF (Figure 4.6). Other forest types such as heath, mangrove and freshwater swamp forest occupied 13% of Block C. The remaining 27% of the total land area comprised mainly burned areas (7%), non-woody vegetation (8%), cultivated land (4%), and mosaic of trees and non-woody vegetation (2%).

Eighteen years later, in 1991, just before the fires of that year commenced, forest still dominated a large part of the study area (65%), with 51% occupied by PSF. The rest of the land was covered mainly by non-woody vegetation (13%), cultivated land (11%) and mosaic of trees and non-woody vegetation (4%) (Figure 4.6).

Two years later, in 1993, the forest cover had decreased by 8%, now accounting for 57% of the study area; losses in forest cover were mainly driven by fire and commercial logging. PSF was still the dominant forest type occupying 44% of the land cover, but commercial concession logging had affected around 1,940ha of the mixed-PSF. Non-forest areas were occupied by non-woody vegetation (13%), cultivated land (11%) and recently burned areas (7%). The spatial pattern of land cover type for 1993 is shown in Figure 4.7.

In 1997, just before that year's fire season started, forest still covered more than a half of the study area (56%) of which 43% was PSF and 13% other forest types (Figure 4.7). Non-woody vegetation was the second dominant land cover type accounting for 19% of the entire area, following by cultivated land (12%), mosaic of trees and non-woody vegetation (6%), sedge swamp (2%) and degraded mangrove (2%). An analysis of the land cover for the year 2000 demonstrated major changes in the land cover pattern, showing for the first time the dominance of non-woody vegetation over forest (Figure 4.8). Non-woody vegetation occupied around 45% of the study area at that time, whereas forest covered only 28%, of which 16% was mixed-PSF and only 0.7% low-PSF. The other forest types occupied the remaining 11%, but with a noticeable reduction in freshwater swamp forest to 6%. The residual land was occupied mainly by cultivated land (12%), and mosaic of trees and non-woody vegetation (9%).

This trend in land cover change continued over the subsequent five years. By 2005, 36% of the study area was dominated by non-woody vegetation; less than in 2000 because 10% of the study area occupied by the non-woody vegetation was burnt in the 2005 fire and thus was classified as recently burned area (Figure 4.8). The forest cover was reduced to 22%, including PSF (12%), freshwater swamp forest (6%), heath forest (3%) and mangrove forest (1%). Remaining patches of low-PSF and the area previously occupied by the blackwater lake were now converted to non-woody vegetation. The forest loss was compensated by an increase in secondary re-growth vegetation comprising both mosaic of trees and non-woody vegetation (11%) and secondary forest (4%).

Land cover maps for each analysed year are illustrated in Figure 4.6, 4.7 and 4.8.

LAND COVER TYPE	1973		1991		1993		1997		2000		2005	
	ha	* (%)	ha	* (%)	(ha)	* (%)	(ha)	* (%)	(ha)	* (%)	(ha)	* (%)
Mixed peat swamp forest	253,598	**56.5**	215,841	**48.1**	182,255	**40.6**	177,626	**39.6**	71,027	**16.0**	53,632	**11.9**
Low pole peat swamp forest	14,135	**3.1**	14,135	**3.1**	14,135	**3.1**	14,135	**3.1**	2,934	**0.7**	0	**0.0**
Heath forest	10,929	**2.4**	14,343	**3.2**	14,343	**3.2**	14,343	**3.2**	14,343	**3.2**	13,736	**3.1**
Mangrove forest	9,276	**2.1**	9,276	**2.1**	9,276	**2.1**	9,276	**2.1**	9,276	**2.1**	5,979	**1.3**
Freshwater swamp forest	39,100	**8.7**	37,912	**8.4**	35,314	**7.9**	34,991	**7.8**	26,898	**6.0**	26,640	**5.9**
Fragmented and/or degraded mangrove	9,174	**2.0**	9,174	**2.0**	9,174	**2.0**	9,174	**2.0**	9,174	**2.0**	10,546	**2.3**
Secondary forest	0	**0.0**	2,677	**0.6**	2,677	**0.6**	4,617	**1.0**	1,615	**0.4**	18,262	**4.1**
Non-woody vegetation	36,598	**8.2**	59,243	**13.2**	59,171	**13.2**	85,181	**19.0**	200,792	**44.7**	160,145	**35.6****
Sedge swamp	9,609	**2.1**	9,609	**2.1**	9,609	**2.1**	9,609	**2.1**	9,609	**2.1**	9,059	**2.0**
Mosaic of trees and non-woody vegetation	10,038	**2.2**	18,832	**4.2**	18,905	**4.2**	25,271	**5.6**	40,329	**9.0**	48,843	**10.9**
Heavily logged peat swamp forest	0	**0.0**	0	**0.0**	1,940	**0.4**	0	**0.0**	0	**0.0**	0	**0.0**
Recently burned area	31,109	**6.9**	0	**0.0**	33,198	**7.4**	4,050	**0.9**	0	**0.0**	55,321	**12.3**
Cultivated land	16,617	**3.7**	49,140	**10.9**	50,184	**11.2**	51,942	**11.6**	53,184	**11.8**	43,907	**9.8**
Blackwater lake	4,169	**0.9**	4,169	**0.9**	4,169	**0.9**	4,169	**0.9**	4,169	**0.9**	0	**0.0**
Water body	4,573	**1.0**	4,573	**1.0**	4,573	**1.0**	4,573	**1.0**	4,573	**1.0**	4,573	**1.0**

* indicates percent (%) of Block C =448,912ha

** in 2005 the decrease in the area of non-woody vegetation was caused by the fact that 9.7% of the study area occupied by this class was classified as recently burned area; the area of cultivated land also declined for the same reason

Table 4.5 Land cover statistics for Block C for the period 1973-2005; land cover data for 1991 and 1997 were obtained from images acquired before fires, thus the considerable land cover changes that occurred as a result of the 1991 and 1997 fires are reflected in the figures for 1993 and 2000 respectively.

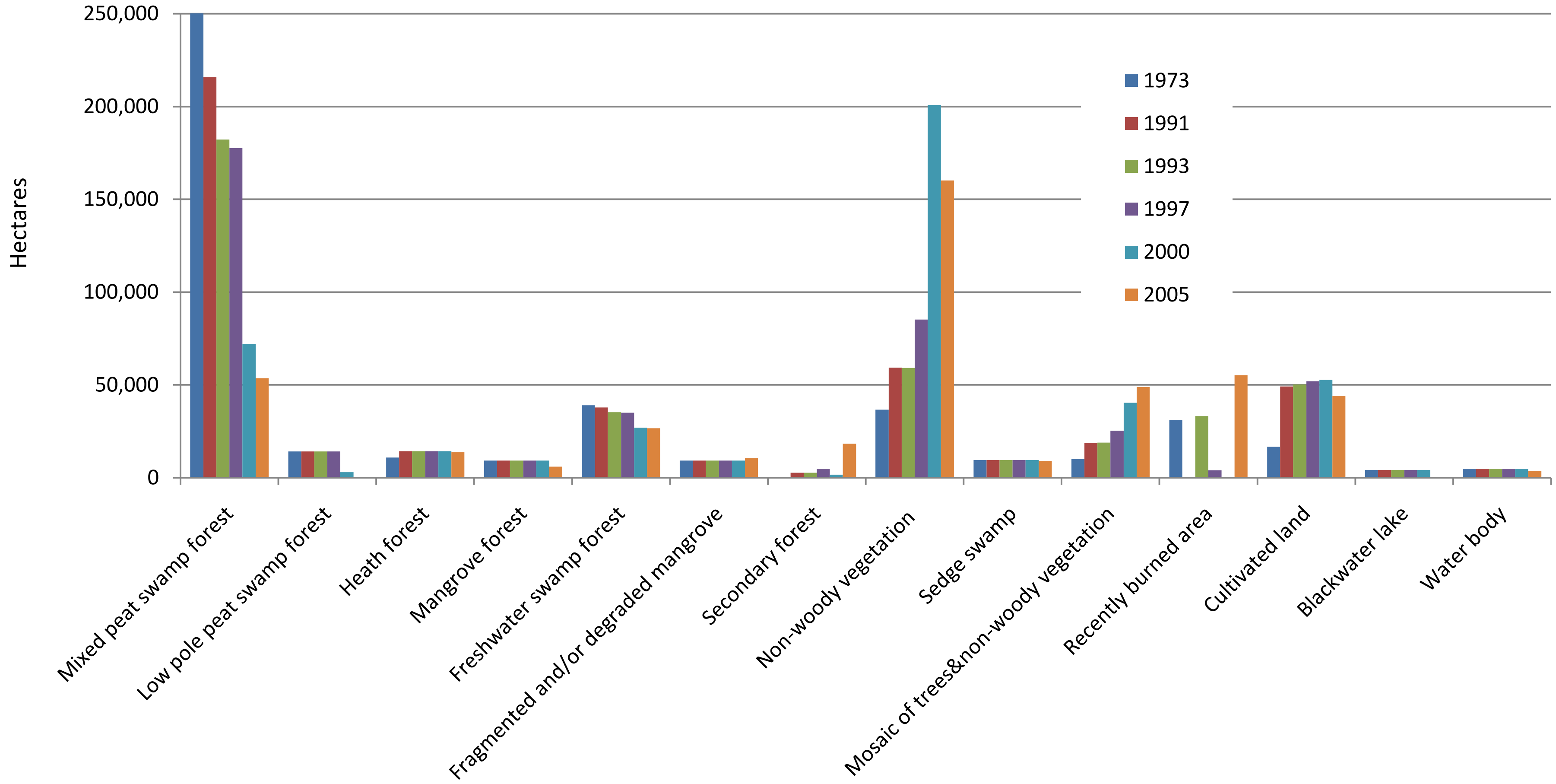


Figure 4.5 Trend in land cover dynamics over the period of investigation (1973-2005); land cover 2005 was obtained from an image acquired after the 2005 fire, thus increase in the area of recently burned class.

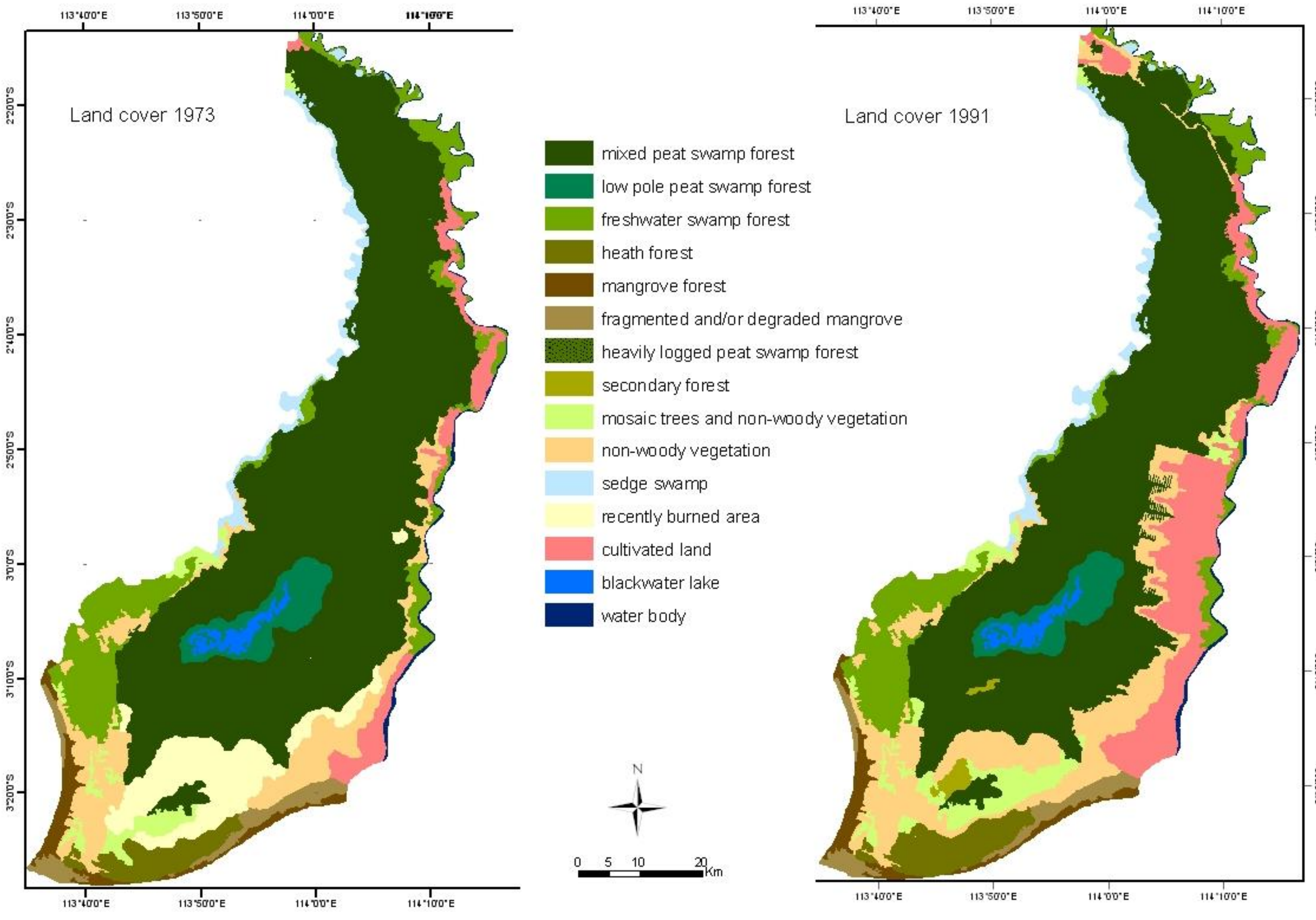


Figure 4.6 Land cover maps for 1973 and 1991.

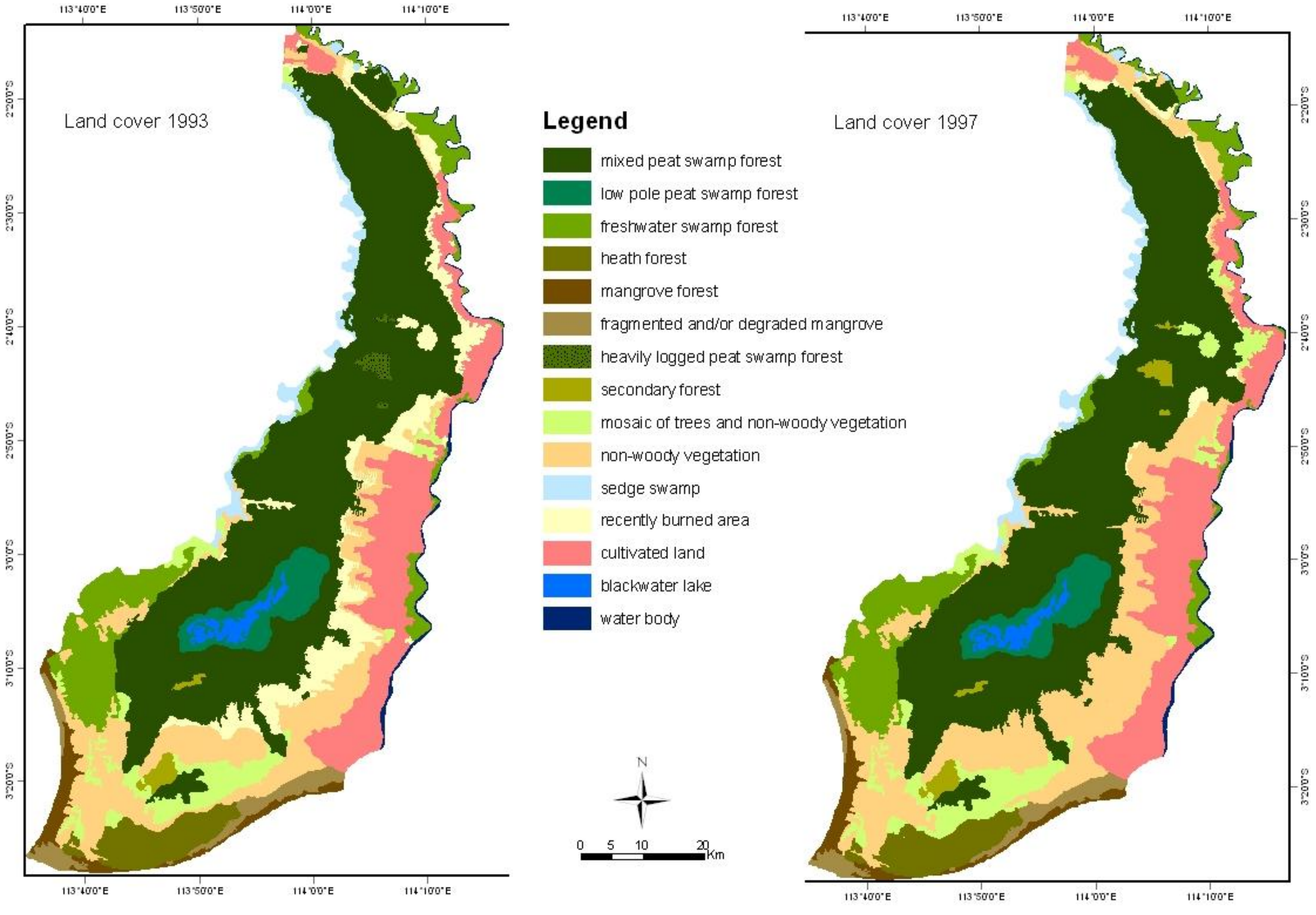


Figure 4.7 Land cover maps for 1993 and 1997.

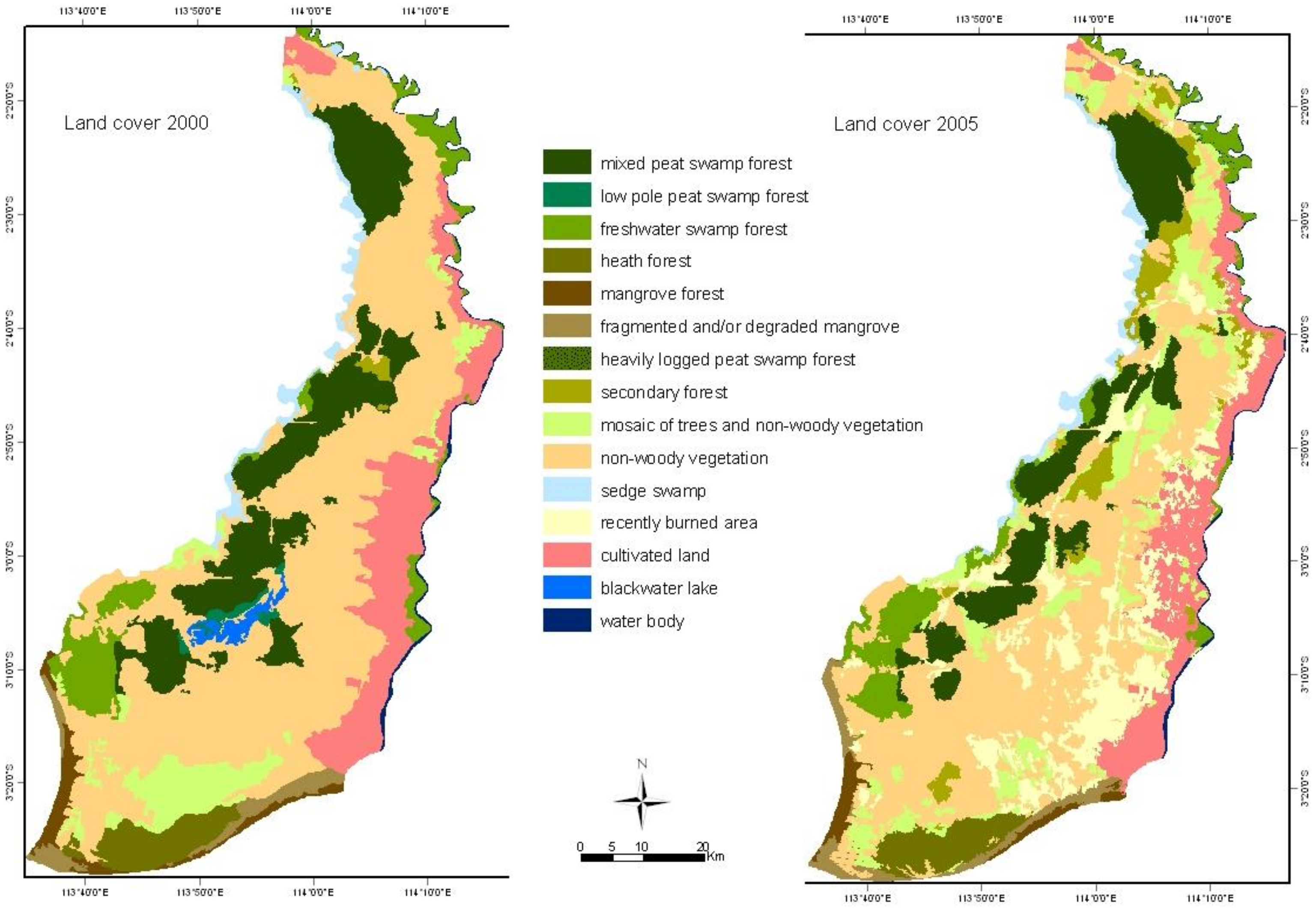


Figure 4.8 Land cover maps for 2000 and 2005.

4.5 Results – Evaluation of the fire regime 1973-2005

This Section presents the results of the evaluation of the fire regime in the study area over the period 1973-2005; emphasis is placed upon changes in fire frequency, fire-return interval and alterations in available fire fuels.

4.5.1 *Fire extent and fire-affected areas by land cover type*

The total burned area and proportion of land cover type affected by each fire over period of investigation is presented in Table 4.6. The fire of 1973 affected approximately 6.9% of Block C (31,109ha) in the southern part of the study area (Figure 4.10). Between 1973 and early 1991 (before that year's dry season started), about 8.6% (38,420ha) of the study area was burnt. Around 98% of the total burned area at that time took place in mixed-PSF, with the remaining 2% in freshwater swamp forest.

Extensive fires leading to forest degradation also occurred during the extended dry season of 1991 (Figure 4.10). These fires affected an additional 7.7% of Block C (34,413ha). The PSF was once again the most affected land cover type, comprising 92% of the total burned area, followed by freshwater swamp forest (7.5%) and mosaic of trees and non-woody vegetation (0.5%).

The 1997 fire was the most severe fire over the entire investigation period in terms of the extent of burned area (Figure 4.10). Fire affected about 34% of Block C (150,486ha); most fires were located in PSF including both mixed- and low-PSF types. Of the total burned area, 79% was in PSF, of which 71.5% was mixed-PSF and 7.5% was low-PSF, and 5.8% was in freshwater swamp forest. The rest of the fire-affected areas were in non-woody vegetation (5.1%), cultivated land (3.2%) and sedge swamp (2.5%), and mosaic of trees and non-woody vegetation (1.6%). For the first time during the three decade period of study, fires affected a large part of the blackwater lake system located on the top of the peat dome in the southern part of Block C.

The subsequent fires of 2002 affected more than 22% of Block C (99,573ha) (Figure 4.11). This time the most fire-affected land cover class was non-woody vegetation (65.1% of total burned area), followed by PSF (16.7%) and mosaic of trees and non-woody vegetation (5%); cultivated land also contributed to the total burned area (6.8%).

During the subsequent dry seasons of 2004 and 2005, fire destroyed 14.3% (64,562ha) and 12.4% (55,349ha) of the study area, respectively, occurring mainly in non-woody vegetation (Figure 4.11 and 4.12). Overall the fire of 2004 affected only 6.1% of the remaining PSF, of which 5.3% was mixed-PSF and 0.8% was low-PSF. Also the over-drained peat formerly occupied by a blackwater lake system burnt for the third time. Of the total area burned during 2004, 72.3% was non-woody vegetation and 11.4% was cultivated land. A similar pattern of fire occurrence took place during the fire of 2005 when overall 78.6% of the total burned area was occupied by non-woody vegetation, 16% by cultivated land, and 4.5% by mosaic of trees and non-woody vegetation, with just 0.2% accounted for by mixed-PSF.

Figure 4.12 illustrates the spatial and temporal distribution of burn scars over the whole period of investigation (1973-2005). In some locations fire occurred more than once, therefore, a fire frequency analysis was performed to identify the pattern of repeated fire scars (see next Section).

Over the first two decades (1973-1996) of this study, fires mainly occurred in areas occupied by PSF as presented in Figure 4.9. Thus, woody forest biomass provided the principal fire fuel. In contrast, during the period following the 1997 fire, regeneration vegetation (i.e. non-woody vegetation, mosaic of trees and non-woody vegetation and secondary forest) became the major source of fire fuel.

LAND COVER TYPE	1973	1974 1990	1991	1997	2002	2004	2005
Total burned area (% of Block C)	6.9	8.6	7.7	33.5	22.2	14.3	12.4
Total burned area (ha)	31,109	38,420	5,421	150,486	99,573	64,562	55,349
Fire-affected areas by land cover type (%)							
Mixed peat swamp forest		98.3	92.0	71.5	14.9	5.3	0.2
Low pole peat swamp forest				7.5	1.8	0.8	
Heath forest					1.1		0.1
Mangrove forest					0.1		0.1
Freshwater swamp forest		1.7	7.5	5.8	1.0	2.5	
Fragmented and/or degraded mangrove					0.3		0.2
Secondary forest					0.3	0.2	0.3
Non-woody vegetation				5.1	65.1	72.3	78.6
Sedge swamp				2.5	1.7	0.7	
Mosaic of trees & non-woody vegetation			0.5	1.6	5.0	5.7	4.5
Cultivated land				3.2	6.8	11.4	16.0
Blackwater lake				2.7	2.1	1.1	
Non-PSF burnt		**1.7**	**8.0**	**21.0**	**83.4**	**93.9**	**99.7**
PSF burnt		**98.3**	**92.0**	**79.0**	**16.7**	**6.1**	**0.2**

Table 4.6 Total burned areas (ha) and proportion (%) of fire-affected areas by land cover type for each fire incident; Block C =448,912ha.

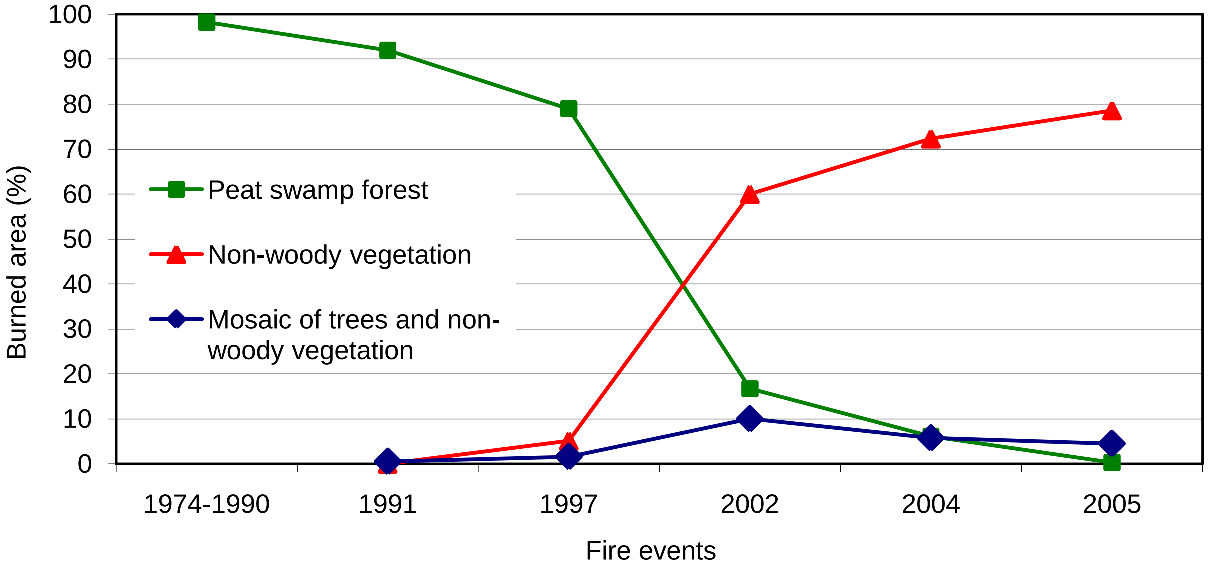


Figure 4.9 Shift in the type of fire fuel from peat swamp forest towards regenerating vegetation, dominated by ferns.

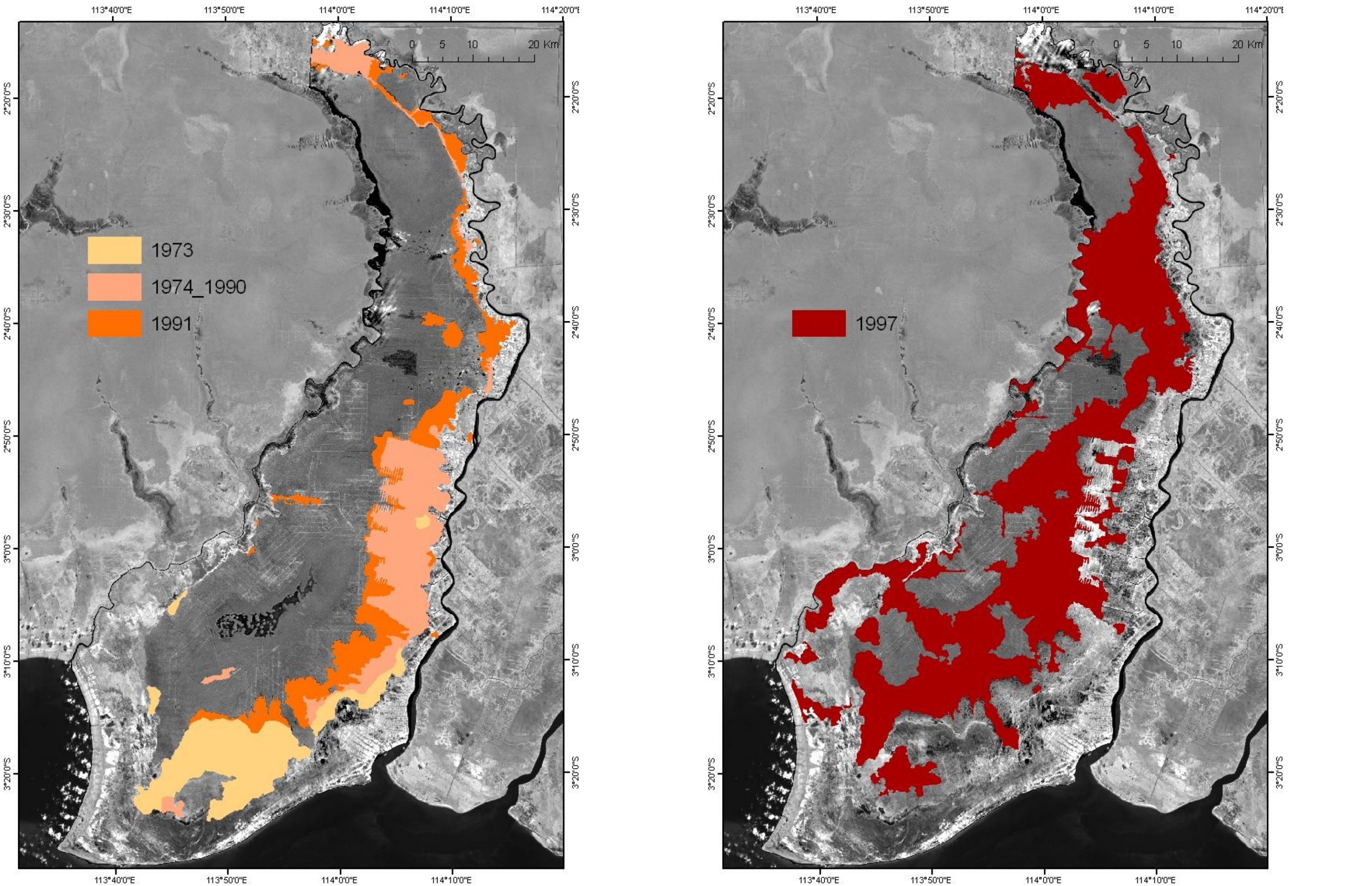


Figure 4.10 Distribution of burn scars in the study area from fires that occurred over the period 1973-1996 (left) and the 1997 fire (right).

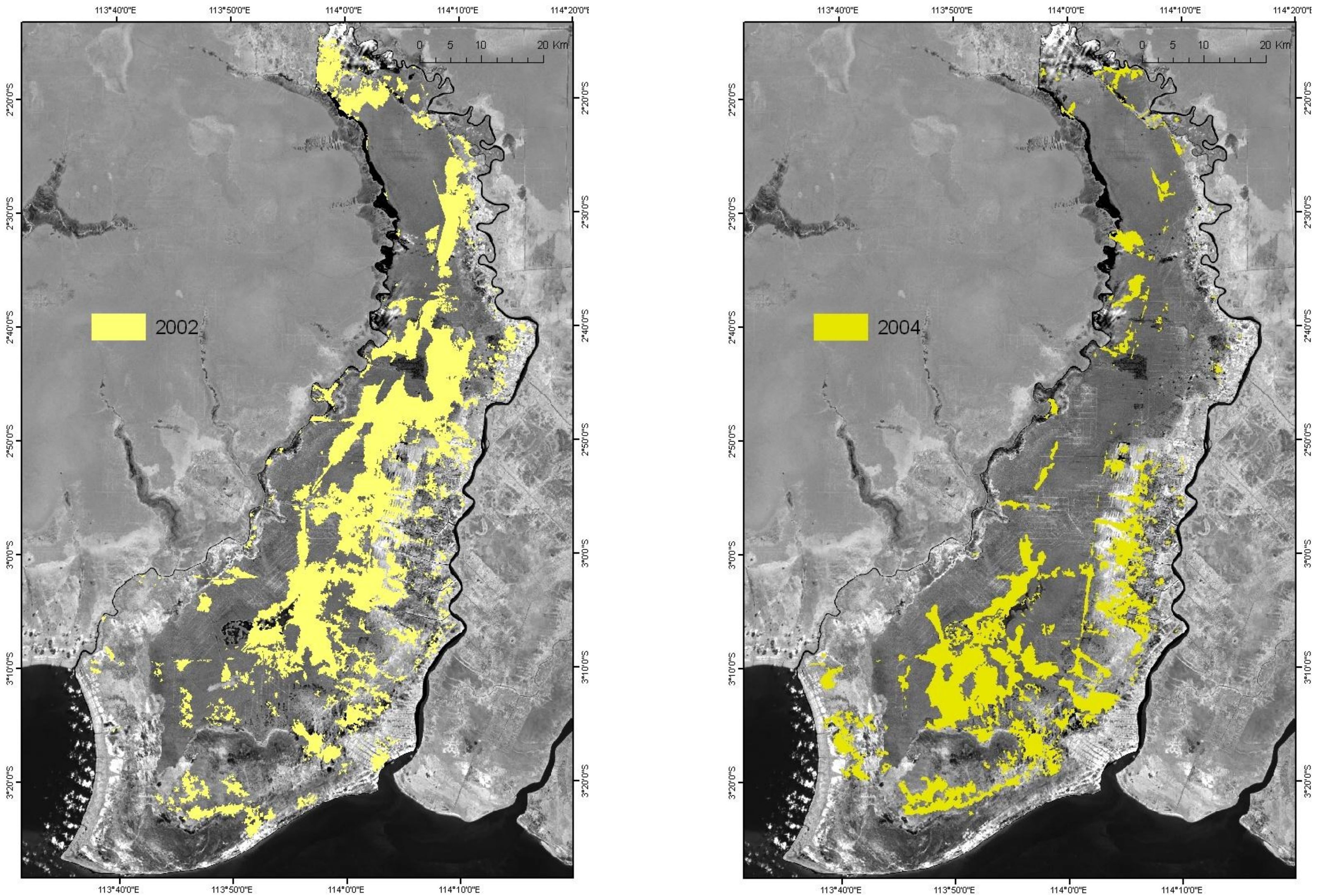


Figure 4.11 Distribution of burn scars in the study area from the 2002 (left) and 2004 fires (right).

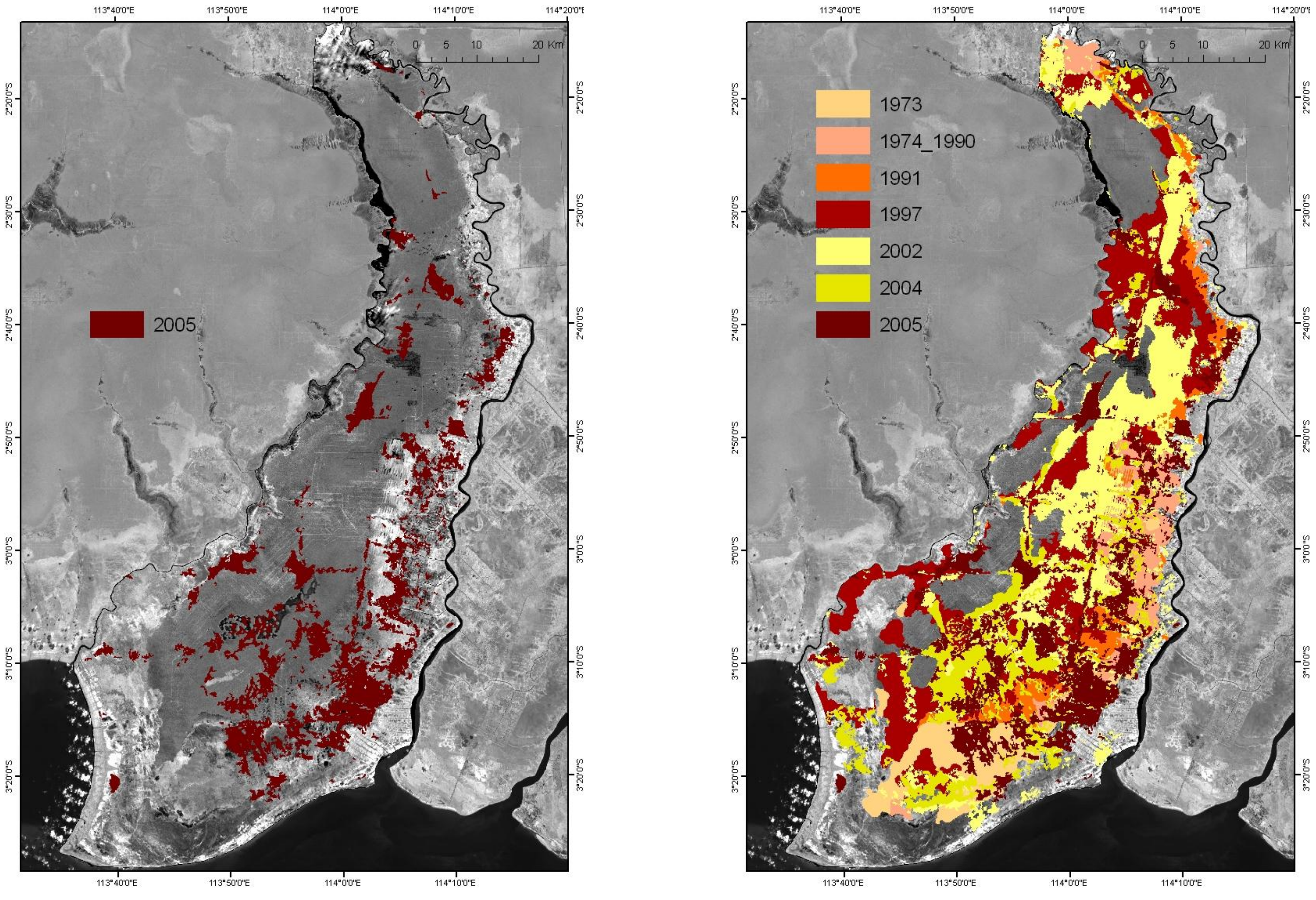


Figure 4.12 Distribution of burn scars from the 2005 fire (left) and accumulate burn scars for the entire period 1973-2005 (right).

4.5.2 *Fire frequency and fire-return interval*

Fire frequency analyses were performed for the three following periods: the entire period of investigation (1973-2005), the pre-MRP era (1973-1996) and the post-MRP era (1997-2005).

Table 4.7 illustrates the proportion of area affected by single and multiple fires (i.e. burnt once, twice, three and more times) over the defined period of investigation. Analysis revealed that around 64% of the total study area burnt at least once during the period 1973-2005. Of the total burned area almost 39% was burnt twice, 10% three times and around 3% more than four times, over a total of seven fire episodes.

Comparison of the pre- and post-MRP periods revealed distinctive variation in the fire frequency pattern with an evident increase in the number of re-burned locations during the period 1997-2005. Over that period more than a half of the total study area was burnt at least once, with 55% of the area affected by a single fire and around 45% affected by multiple fires (36% burnt twice and 8% burnt three or more times). In comparison, analysis of the distribution of burn scars for the period 1973-1996 revealed that each fire occurred in a new location. Thus, there was a very low incidence of repeat fires (Figure 4.14).

Fire frequency	**1973-2005**		**1973-1996**		**1997-2005**	
	*% of burned area	ha	*% of burned area	ha	*% of burned area	ha
1 fire	48.0	137,223	99.5	106,270	55.4	134,735
2 fires	38.8	111,131	0.5	491	36.4	88,613
3 fires	10.3	29,595			6.9	16,889
4 fires	2.6	731			1.2	3,021
5 fires	0.3	794				
Area of Block C affected by fire	63.7	286,174	23.8	106,761	54.2	243,258

* percent of the total burned area for the defined period

Table 4.7 Proportion of the study area affected by single and multiple fires delineated for the three periods.

Figure 4.14 presents a spatial distribution of single and multiple fires for the two periods: pre- and post-MRP eras. It is clear that locations which experience single fire before 1997 and those affected by multiple fires over the period 1997-2005 are mainly clustered in close proximity to HAPs (i.e. canals and roads, overlaid on the fire frequency map) indicating anthropogenic sources of ignition (more on HAPs in the next Section).

Fire return interval analysis

Analysis of the fire-return interval demonstrated that locations with a longer fire-return interval (Table 4.4) were more likely to be re-burnt than those with a short period between subsequent fires. Figure 4.13 demonstrates clearly an increase in the fire extent in areas with a fire-return interval equal to five or more years.

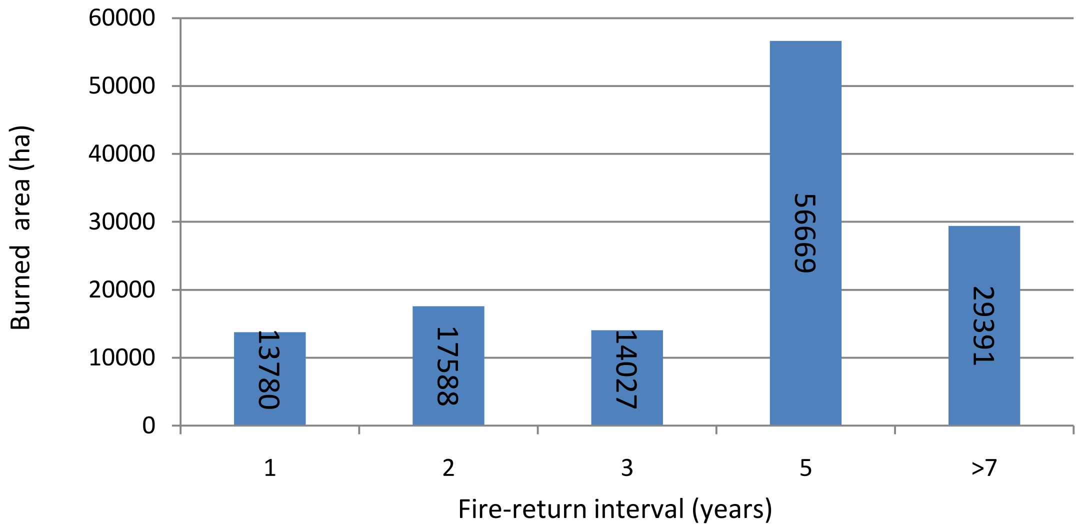


Figure 4.13 Areas affected by repeated fires with different fire-return intervals for the post-MRP period (1997-2005).

Despite the constraints on fire propagation, Figure 4.13 indicates that relatively large areas were affected by fires on an annual or bi-annual basis. This frequent fire rotation is probably more related to sources of ignition (HAPs) (see next Section) rather than the available fuel load.

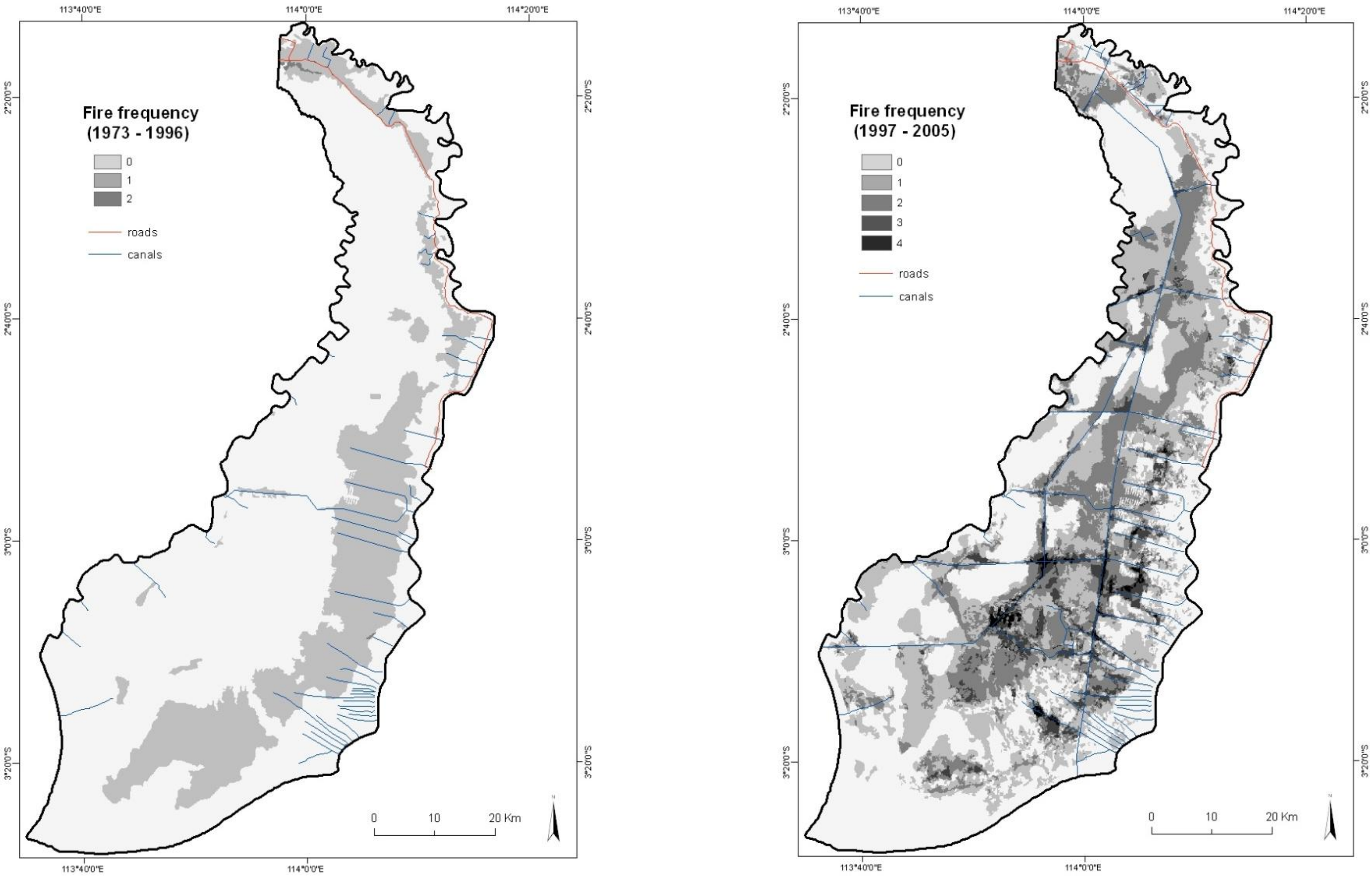
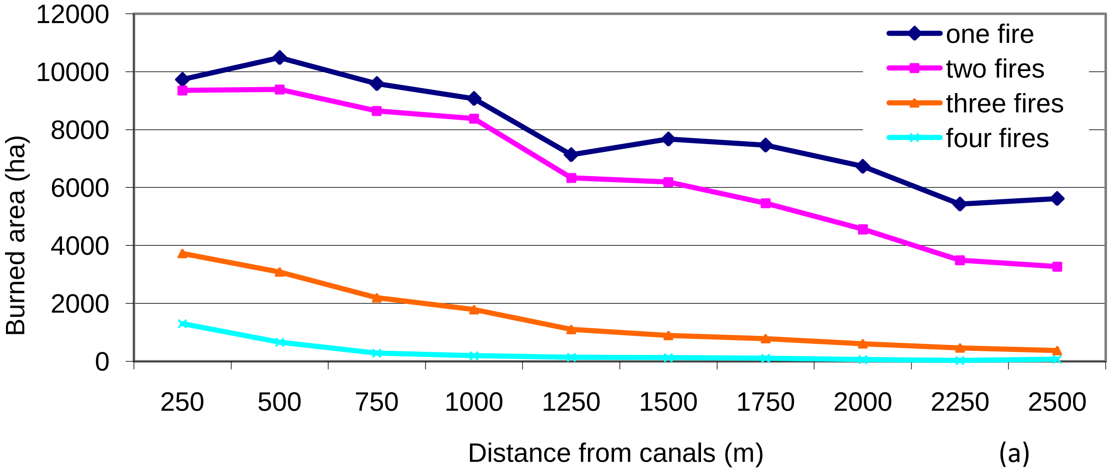


Figure 4.14 Spatial distribution of single and multiple fires for the periods: 1973-1996 (left) and 1997-2005 (right) with overlaid network of canals (in blue) and roads (in red). .

4.5.3 *Accessibility analysis*

The principal human access points (HAPs) in the study area are the canal and road systems (Figure 4.14). The GIS analysis showed that the length of canals in the study area doubled from 342km (before 1997) to 758km (period 1997-2005) as an outcome of the intensive drainage associated with the MRP; by comparison the length of roads is much less (103km) and stayed almost the same throughout the period of investigation.

HAPs analysis showed that areas subjected to three or four fires over the 8-year post-MRP period (1997-2005) were all located within close proximity to the canals and declined with distance away from these features (Figure 4.15a).



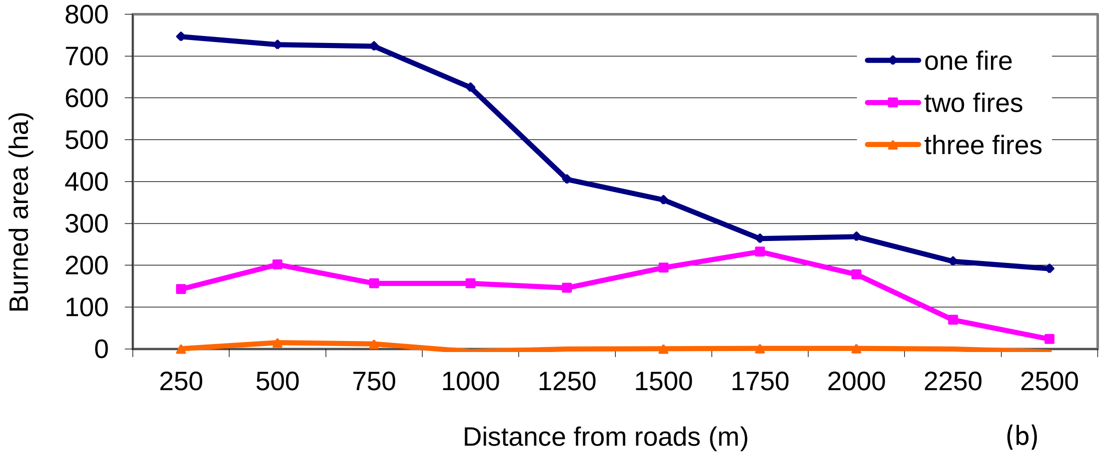


Figure 4.15 Relationships between human access points a) canals and b) roads and fire extent for the period 1997-2005; note: different scale on the axis of burned area.

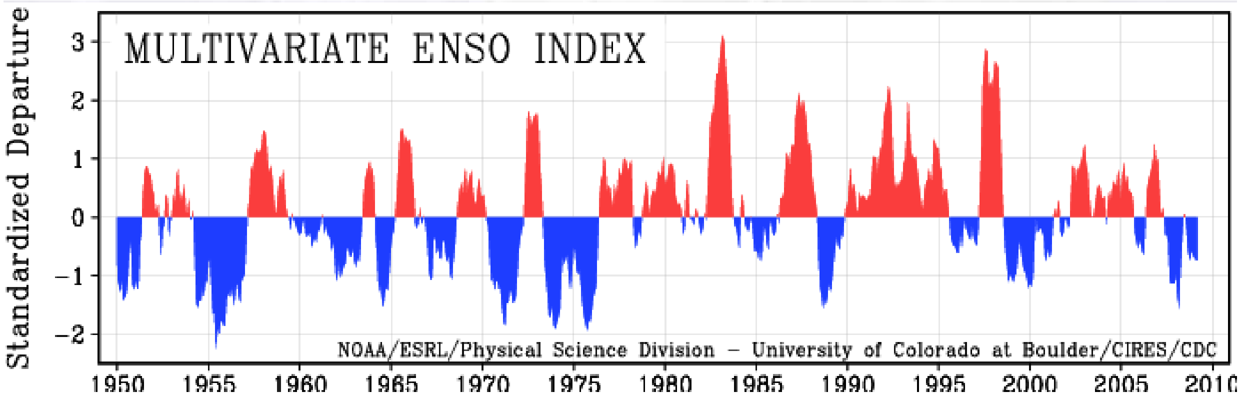
The analysis showed that the dense network of canals had a strong influence on the fire regime in the study area, whilst fires associated with roads, some stretches of which were located in non-peat (i.e. alluvial areas), occurred less frequently and with a lower chance of repeated fires (mainly once or twice burnt and only a very small area affected by three fires) (Figure 4.15b).

Figure 4.15 shows a clear relationship between HAPs and fire frequency, with the probability of fire increasing with proximity to both canals and roads, but with a stronger effect over a greater distance for the canals.

4.6 The relationship between ENSO and fire events

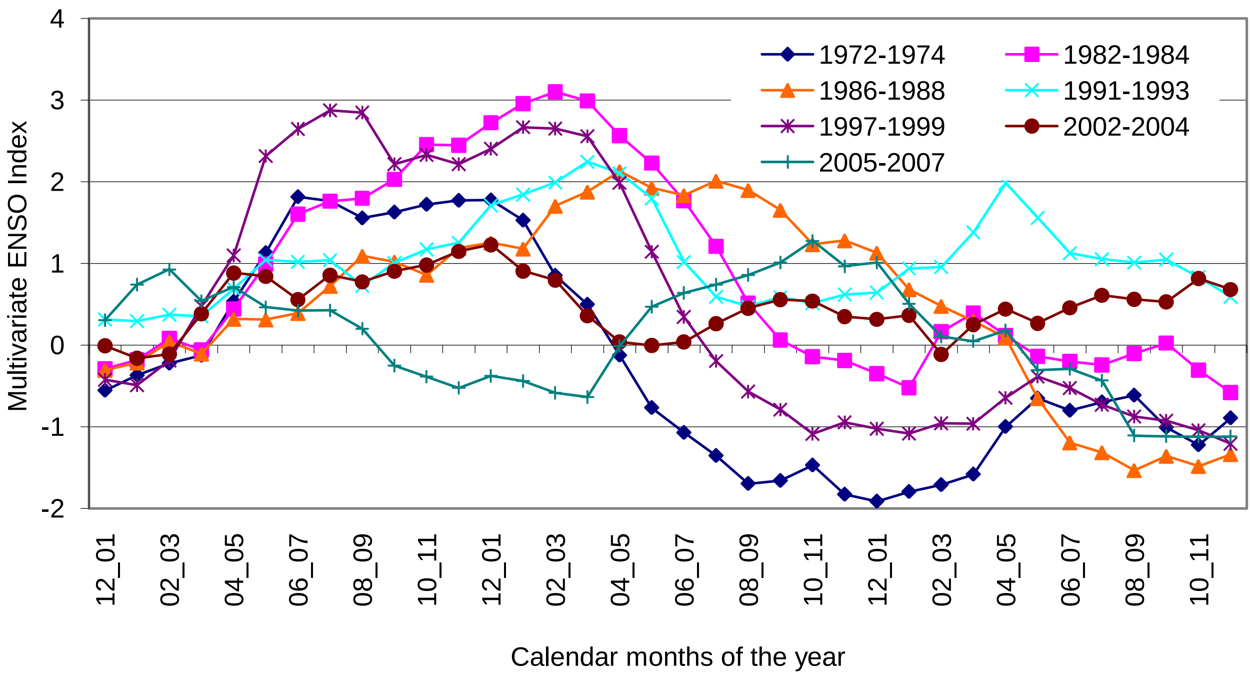
In order to investigate whether fires that occurred in the study area were associated with ENSO phase, the Multivariate ENSO Index (ENSO-MEI) was employed to establish a statistical relationship with fire extent and occurrence for each fire incident. The MEI is based on six main observed variables over the tropical Pacific, namely sea-level pressure, zonal and meridional components of the surface wind, sea surface temperature, surface air temperature, and total cloudiness fraction of the sky (Wolter and Timlin, 1993, Wolter and Timlin, 1998).

The MEI values for the period 1972-2007 were downloaded from the webpage (http://www.cdc.noaa.gov/people/klaus.wolter/MEI/#ref_wt1). The MEI values for each of the fire events reported in the study area were extracted and plotted (Figure 4.17). Negative values of the MEI represent the cold ENSO phase, while positive MEI values represent the warm ENSO phase. Large and prolonged El Niño or La Niña events are indicated by large (MEI>1) or small (MEI< -1) values of the MEI.



Note: negative values (blue bars) represent the cold ENSO phase, while positive values (red bars) represent the warm ENSO phase

Figure 4.16 Multivariate ENSO Index for the period 1950-2008; large and prolonged El Niño/La Niña events are indicated by large/small values of MEI (http://www.cdc.noaa.gov/enso/enso.mei_index.html).



Note: negative values of the MEI represent the cold ENSO phase while positive MEI values indicate the warm ENSO phase

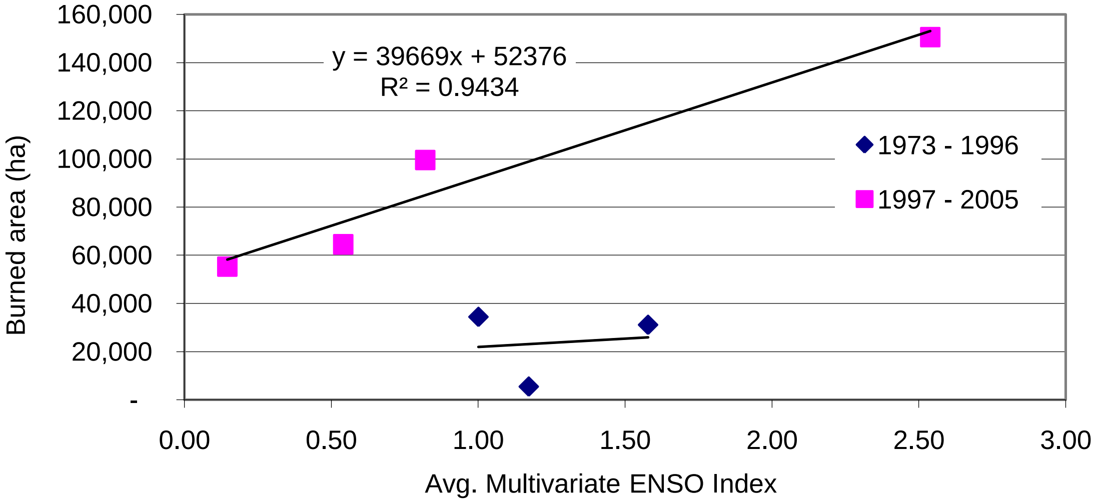
Figure 4.17 Values for the Multivariate ENSO Index for the period 1972-2007.

The El Niño events of 1997/98 and 1982/83 were the longest and severest events recorded over the period of this investigation. In both cases, the period with MEI values greater than 1.0 extended over fourteen months during each event, whilst the MEI value exceeded 2.0 for nine and twelve months during 1982/83 and 1997/98, respectively. The drought of 1997/98 did not exhibit a 'warming up' period, as 1982/83 El Niño did, and the MEI increased steeply from 0.5 in March 1997 to 2.3 in May 1997, reaching a peak value of 2.9 in August 1997. The MEI values remained greater than 2.0 over the next 12 months before rapidly declining to 0.3 in July 1998 initiating a prolonged cold phase (La Niña) (Figure 4.17). The El Niño of 1997/98 followed a similar trajectory to that of 1972/73; however, the associated period of drought was shorter and less intensive in 1972/73. By comparison, the drought associated with the 1991/92 El Niño had a similar pattern to that of the 1982/83 El Niño but the period with increased MEI values was shorter and weaker. In 2002, the MEI increased gradually reaching a value of around 1.0 in June 2002 with a peak of 1.2 in December 2002 (weak El Niño). Over the period 2003-2005, the MEI values fluctuated between 1.0 and -1.0, indicating the neutral phase (non-El Niño years). During 2006, the MEI value rose above zero from May onwards and achieved a peak value of 1.3 in September 2006 and then dropped below -1.0 in August 2007, indicating the start of a La Niña phase.

Fire incidents reported in the study area are strongly associated with ENSO events (i.e. 1973, 1991, 1997 and 2002). However, two widespread fires, in 2004 and 2005, occurred during the neutral phase, when the MEI fluctuated between -1.0 and 1.0.

To investigate whether there was a relationship between fire extent and ENSO, the values of the MEI for the dry season in Central Kalimantan (from May to October) were extracted and then averaged for each fire-year. These average MEI values were then correlated with the burned area for each analysed fire incident derived as a result of this study.

Figure 4.18 illustrates a strong correlation between averaged MEI values and fire extent in the study area with a clear division between the periods 1973-1996 and 1997-2005, i.e. the pre- and post-MRP eras, respectively. During the pre-MRP era, fires affected relatively small areas, even though the value of MEI was quite high (above 1.0) indicating El Niño drought conditions. By contrast, fires claimed much larger areas of land during the post-MRP period; when even during 'normal' dry seasons (MEI<1.0, i.e. indicating non-El Niño conditions) large areas of land were burnt (e.g. the years 2004 and 2005).



Note: the period 1973-1996 referring to the pre-MRP era is explicitly separated from the period 1997-2005 covering the post-MRP era

Figure 4.18 Linear regression between burned area and average value of the MEI calculated for six months of dry season (May-October) in the region.

The regression analysis indicates a linear strong correlation between the MEI index and fire extent over the period 1997-2005 (r^2=0.93), which raises the possibility of using ENSO indices to predict fire extent in degraded tropical peatland. It also shows that the period 1973-1996 is clearly separated from the period 1997-2005.

4.7 Summary

The results presented in this chapter have showed:

- During the period of investigation (1973-2005), land cover dynamics show a decreasing trend in forest cover towards non-woody vegetation. In 1973, forest covered around 73% of the study area, with 60% PSF, whereas in 2005 forest cover was reduced to 22%, with 12% PSF. Most of these changes occurred during the post-MRP period (1997-2005).

- Land cover dynamics are strongly driven by widespread fires that have increased in recent years. During the period 1973-1996 fire affected 23% of the study area occurring mainly in PSF. The fire regime altered following implementation of the MRP project; the 1997 fire affected 34% of the study area, and subsequent fires in 2002, 2004 and 2005 affected 22%, 14% and 12%, respectively.

- Over the first two decades (1973-1997) fires mainly occurred in areas occupied by PSF, thus woody forest biomass provided the principal fire fuel. In contrast, during the period following the 1997 fire, secondary vegetation became the major source of fire fuel.

- Fire frequency increased and fire return interval declined during the period 1997-2005. Over that period around 55% of the study area was affected by single fire and ~45% by multiple fires, of which 36% was burnt twice and 8% burnt three and more times.

- Analysis of fire-return interval demonstrated that locations with a longer fire-return interval were more likely to be re-burnt than those with a short period between subsequent fires.

- There is a clear relationship between human access points and fire extent and frequency, with as increasing probability of fire occurrence with proximity to canals.

- The incidence of fires in the study area was strongly associated with ENSO events (i.e. 1973, 1991, 1997 and 2002). However, recent fires have become decoupled from ENSO, with an increased risk of fire even during ‘normal’ non-ENSO dry seasons (i.e. 2004 and 2005).

5 POST-FIRE VEGETATION DYNAMICS – METHODS AND RESULTS

As demonstrated in the previous chapter, fire is a major driver of land cover change within the study area. Fire frequency, interval between fires and length of time after the last fire strongly control the rate of post-fire vegetation regrowth and the direction of vegetation regeneration. Several previous studies undertaken in the lowland tropical forests of Borneo have demonstrated differences in vegetation structure between sites subject to either single or multiple fires (Slik *et al.,* 2008, Slik and Eichhorn, 2003, Toma *et al.,* 2005, van Nieuwstadt and Sheil, 2005). At multiple fire sites, a short time interval between recurrent fires can potentially increase the magnitude of surface damage and hence reduce the rate of vegetation recovery (Cochrane *et al.*, 1999, Slik *et al.*, 2008, Uhl and Kauffman, 1990). On this basis, it was hypothesised that within the study area there should be differences in vegetation structure between sites recovering from single and multiple fires. Observed variations in vegetation regrowth in areas affected by multiple fires generated a second hypothesis: in locations subject to multiple fires, variation in vegetation regrowth can be related to the burn severity of the last fire. Integration of field measurements with spectral data derived from satellite images was used to address these two hypotheses.

5.1 Quantitative inventory of post-fire vegetation – Field data collection

The main goal of the field data collection was to obtain *in situ* data on post-fire vegetation structure in order to describe the various stages of regrowth. The fire frequency map was used in the first place to select a sufficiently accessible study area in the northern part of Block C (Kalampangan area) that contained representative samples of the different stages of post-fire vegetation regeneration following both single and multiple fires (Figure 5.1). This area had been affected by fires in 1997 and 2002.

Before the fires in 1997, the entire Kalampangan area was covered with mixed-PSF with a few recently excavated canals constructed as part of the MRP running north-south and east-west. The field campaign was divided into two parts: preliminary reconnaissance was undertaken during the dry season of 2005, whilst the principle field data collection was carried out in 2006.

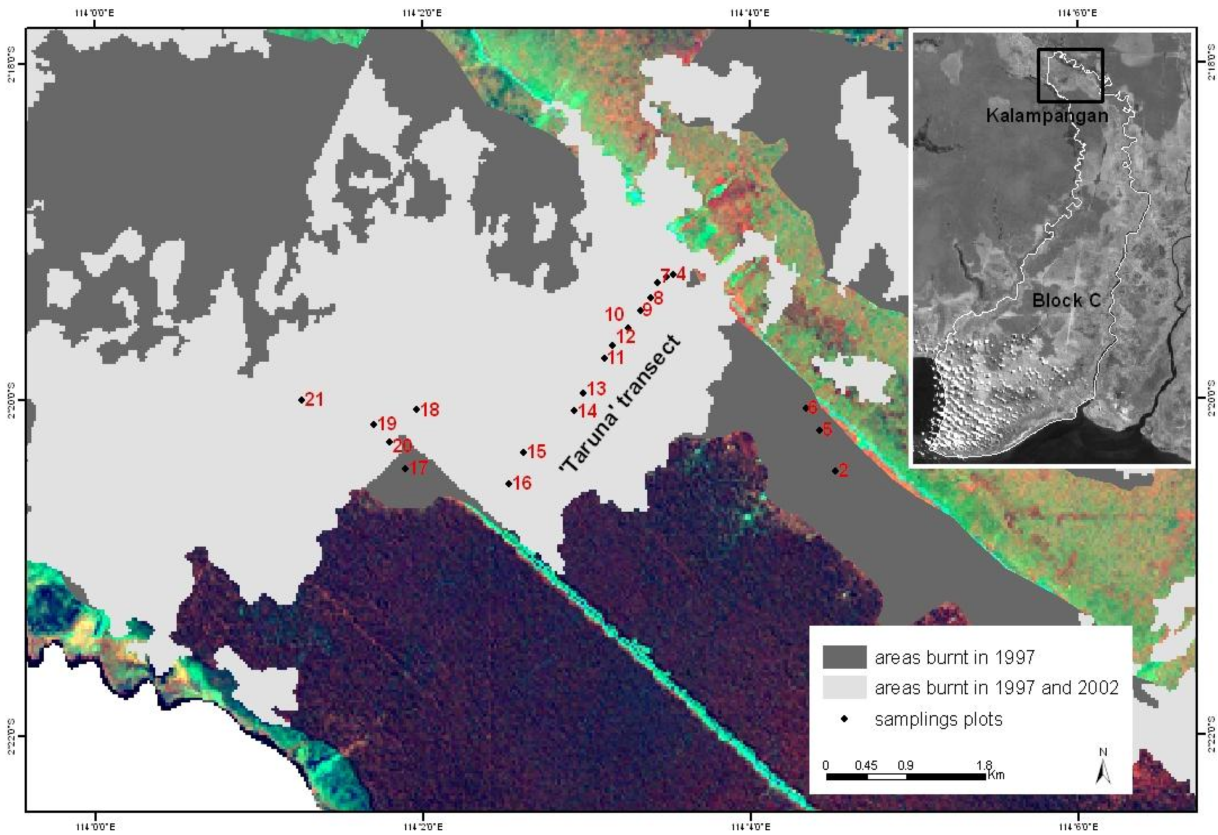


Figure 5.1 Study site: Kalampangan area and fire frequency map, black dots indicate the locations of sampling plots.

Field assistance was provided by a small number of local researchers and workers from the Centre for International Cooperation in Sustainable Management of Tropical Peatland (CIMTROP), University of Palangkaraya; local people also provided valuable experience and knowledge on the fire history of the study area.

5.1.1 *Selection of sampling sites, field logistics and data preparation*

In the next step, locations for the sampling plots representing different classes of regrowth within the study area were chosen. It was important to select sampling plots that were separable both in terms of vegetation stand structure and remote sensing signature (Lu, 2005). The fire frequency analysis (Section 4.5) facilitated the separation of sites subjected to a single fire (SF) from those at which multiple fires (MF) had occurred. SF sites were burnt only once in 1997, while MF sites were subjected to two fires in 1997 and again in 2002 (Figure 5.1).

Field reconnaissance revealed a great variability in vegetation structure inside the MF areas and additionally provided an opportunity to experience the constraints on undertaking fieldwork in the study area, which was invaluable in developing logistics for the main field campaign. The major concerns were site accessibility and health and safety issues. It was difficult to gain access to remote areas and walking long distances was exhausting. In the end, a combination of motorbike followed by walking was the most reliable method. For example, recently burned locations were extremely hazardous owing to deep water-filled holes hidden beneath dense ferns and fallen trees. The dense growth of ferns meant that tracks had to be cut, whilst the lack of trees meant that there was almost no shelter from the sun and heat. All these factors needed to be carefully considered and were factored into the location of the study sites. Due to accessibility, the majority of sites were set up along or close to the ‘Taruna’ transect (indicated in Figure 5.1), a walking route cut relatively clear of overgrowth and running west-east from the main road towards the Kalampangan canal in the West. This transect was cleared by researchers from CIMTROP in order to undertake hydrological monitoring. This route provided the necessary access to almost all MF study plots.

Before going to the field, all available satellite images and the fire frequency map were converted into appropriate formats and uploaded into a hand-held PDA with integrated

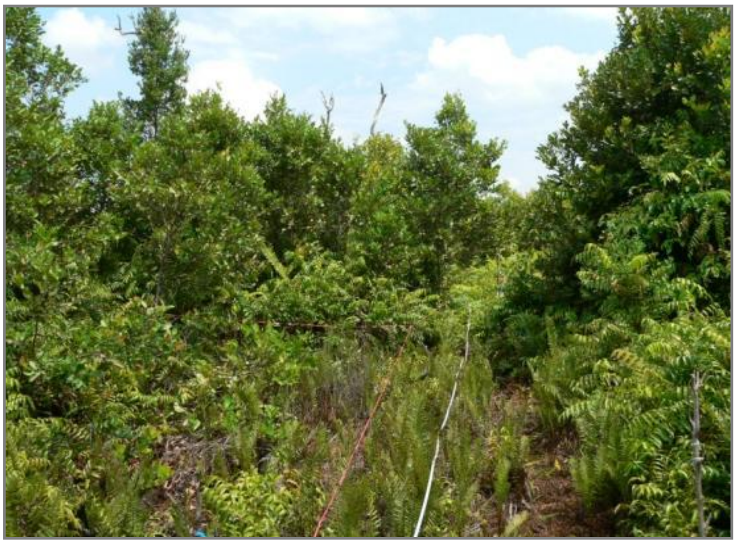
GPS. Laminated hard copy maps were also used in the field. The PDA fed with geo-located satellite data was found to be of particular value in such a heterogeneous environment. Firstly there was no point of reference in the flat, fern-covered landscape, making it difficult to locate sites and, second, it was even harder to get back to exactly the same position by relying only on hard copies of satellite images.

All four SF plots were located at least 500m away from the edge of any remaining forest fragments in order to diminish the edge effect. MF areas were occupied by far more heterogeneous vegetation which varied from relatively dense woody vegetation to non-woody vegetation with exposed ground (i.e. bare peat). From the initial reconnaissance, three classes of regrowth (MF1, MF2 and MF3) were identified visually inside the MF locations taking into account vegetation composition, presence of a woody canopy and the proportion of bare ground (Figure 5.2). A total of 16 sampling plots representing the MF categories were established (i.e. MF1, MF2 and MF3 were represented by five, five and six plots respectively).

Single fire (SF) plot
(burnt in 1997)

9-year old advanced woody regrowth

Figure 5.2 Examples of the four classes of regrowth, as defined in the field based on vegetation formation, canopy presence and proportion of bare ground.



Multiple fires plot
(fires in 1997 and 2002)

MF1 – relatively advanced stage of forest regrowth

heterogeneous structure
dominated by trees and saplings,
relatively dense woody vegetation,
canopy cover: more than 50%

Multiple fires plot
(fires in 1997 and 2002)

MF2 – intermediate stage of forest regrowth

heterogeneous structure
dominated by trees, saplings and dense fern,
evidence of bare ground,
canopy cover: less than 50% but more than 10%

Multiple fires plot
(fires in 1997 and 2002)

MF3 – the least advanced stage of woody regrowth

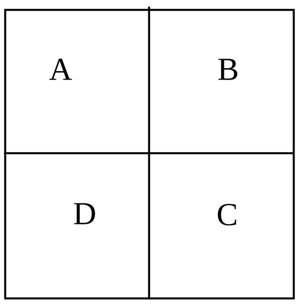
homogenous structure dominated by ferns with a few trees and saplings
more evidence of bare ground,
canopy cover less than 10%

Figure 5.2 (cont.) Examples of the four classes of regrowth, as defined in the field based on vegetation formation, canopy presence and proportion of bare ground.

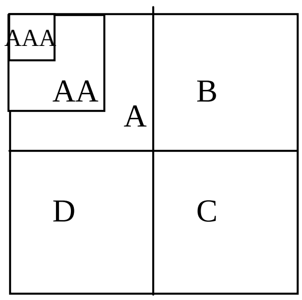
5.1.2 *Plot design*

The preliminary field reconnaissance revealed that the post-fire vegetation structure was quite heterogeneous and mainly dominated by small, often multi-stemmed trees with a DBH of less than 10cm. Thus, a traditional inventory method that considered all trees greater than 10cm DBH (as is usually applied in studies of mature tropical forest) was inappropriate and the inventory method for small trees proposed by Nascimento & Laurence (2002) was considered more suitable. Using this method, trees were classified as having a woody stem greater then 5cm DBH, saplings had stems in the range 1-4.9 cm DBH and seedlings had stems less than 1cm DBH. Each sampling SF plot, size 20 x 20m, was divided into four 10 x 10m subplots, within which all trees were measured (Figure 5.3). The sapling inventory was performed within one 5 x 5m subplot nested in one corner of the plot, while seedlings were counted individually within a 2.5 x 2.5m subplot nested in the same corner as the sapling subplot.

Further modification of this sampling strategy was made for MF plots with fewer woody plants. In these sites, instead of counting saplings within a nested subplot, all saplings and seedlings were counted within each of the four 10 x 10m subplots (Figure 5.3).



Multiple fires (MF)



Single fire (SF)

Figure 5.3 Plot design for the multiple fires and single fire plots 20 x 20m; A, B, C and D indicate the subplots 10 x 10m, AA–nested sapling subplot 5 x 5m, and AAA–seedlings subplot 2.5 x 2.5m.

5.1.3 *In situ measurements*

In order to comprehensively explain the vegetation dynamics resulting from different fire regimes, various vegetation variables were measured in the field (Table 5.1).

In order to facilitate fieldwork logistics, an inventory form was prepared before going into the field (Appendix 2) and training was arranged for field assistants.

Stand structure – woody vegetation		**Field measurements**
Trees	greater than or equal to 5cm DBH	height DBH canopy crown diameter percentage of canopy cover number of single- and multi-stemmed trees species composition
Saplings	DBH ranging from 1- 4.9cm	
Seedlings	DBH less than 1cm but height greater then 50cm above the ground surface	DBH number of single- and multi-stemmed trees species composition
Stand structure – non woody vegetation		**Field measurements**
Ferns	non-woody vegetation dominated by two different species of fern	species composition average height ground cover (for each subplot) photographs taken vertically and perpendicular to the ground
Additional data		
Photographs	taken at each corner and in the plot centre, clockwise at every 45 degrees (provide a record of the vegetation cover, not only inside the plot but also in the surrounding area)	
GPS position	recorded at each corner and in the plot centre	
Vegetation profile	a vertical profile of the vegetation structure was drawn in each sampling plot	
Ground cover	estimation of proportion of bare ground in each plot	

Table 5.1 Vegetation variables measured or collected in the field; DBH–diameter at breast height.

The following parameters were measured in each plot:

- Single-stemmed vs. multi-stemmed canopies

Field inspection preceding the main data collection revealed the presence of a large number of multi-stemmed trees/saplings, which were particularly characteristic of the early stages of re-growth. A tree/sapling was registered as multi-stemmed if the tree/sapling split below the breast height measure into two or more stems. The multi-stemmed trees/saplings quite frequently had a shrubby shape and therefore could have been misclassified as shrubs without detailed field investigation. In the field, trees/saplings were recorded as either single- or multi-stemmed and the data were expressed as the total numbers of individual trees/saplings per plot.

- Diameter at breast height (DBH)

The DBH of trees or saplings is an easy, relatively accurate, common measurement for forest inventory. DBH was measured at a standard height of 130cm above the ground surface for each individual single-stemmed tree/sapling and for each stem of a multi-stemmed tree/sapling.

- Height

The height of all trees/saplings in the MF sites was measured using a hand-held clinometer; measurements were made to the top of the highest foliage of each canopy. In the case of multi-stemmed trees/saplings the height of the tallest stem was measured, whilst heights of the remaining lower stems were obtained as a function of DBH and the height of the tallest stem. In the SF plots, the trees generally had a similar height, therefore representative trees were measured using the clinometer and then the height of the remaining trees nearby was estimated. This was done to reduce sampling time and obstacles related to canopy density.

- Woody species composition

Every tree, sapling and seedling was identified and given a local name in the field; subsequently, Latin names were assigned using the knowledge of locally trained staff and the herbarium in CIMTROP. Woody species that could not be identified were recorded as ‘unknown’ (a negligible proportion of all trees). The total number of species, their frequency and the name of the dominant species were obtained separately for trees, saplings and seedlings in each plot. Appendix 3 contains the list of tree species recorded in the study area.

- Canopy cover

Two techniques were applied to derive a value for canopy cover. The first method was based on measuring the diameter of the canopy crown in the maximum and minimum direction; this was used for all individual single- or multi-stemmed trees and saplings within the MF classes. This technique was feasible due to the sparse canopy structure that allowed precise measurement of canopy cover at these plots. At the SF plots, with relatively dense structure of the woody regrowth, canopy cover was measured using Digital Canopy Photography (DCP). In each plot, canopy photographs were taken in 16 locations, 4 in each subplot. Photographs were taken using a camera placed on a tripod and held at a height of 130cm above the ground, corresponding to the height at which DBH measurements were made. All photographs were taken using the same digital camera (Panasonic Lumix DMC-FZ20) with a fixed parameter: lens= 34mm, resolution= 2MG pixels. To minimise the effect of brightness, all photos were taken on overcast days or when the sun was obscured by clouds.

The following attributes of the non-woody vegetation were measured in each plot:

- Species composition

In every plot all non-woody species (mostly fern species) were identified and given a local name in the field and, subsequently, a Latin name.

- Fern and bare ground cover

The proportion of the ground covered by ferns and the proportion of bare ground in each of the MF plots were assessed by visual estimation in each subplot. In addition, a sketch map was prepared to show the spatial distribution of ferns as well as of bare ground inside each plot.

- Fern biomass collection

There is a lack of published methodology on the measurement of fern biomass; therefore a method previously used in temperate grasslands (Dabrowska-Zielinska *et al.*, 2009) was adopted for this purpose. A metal frame square of size 0.5 x 0.5m was used to sample the fern biomass. All of the aboveground portions of each fern species present within the square were cut down to the ground and placed in sealed plastic bags to prevent moisture loss. Each bag was labelled with the plot number, the species name and average height of the vegetation. In addition, each sampling location was described in a field notebook and recorded by taking photographs, which gave an overview of fern density, fern formation and ground cover. A total of 16 samples of fern were randomly selected within the MF plots (four sampling sites for each MF class).

5.1.4 *Retrieval of vegetation variables from the in situ measurements*

- Total aboveground biomass (T_{AGB})

T_{AGB} is defined as the total amount of aboveground biomass of living organic matter expressed as dry tonnes per unit area (FAO, 2005). For the purposes of this research, the term T_{AGB} refers to the sum of both the woody-AGB and the non-woody-AGB (5.1):

$$T_{AGB} = \sum woody\text{-}AGB + \sum non\text{-}woody\text{-}AGB \tag{5.1}$$

Where,

$$\sum woody\text{-}AGB = \sum trees\text{-}AGB + \sum saplings\text{-}AGB + \sum seedlings\text{-}AGB$$

$$\sum non\text{-}woody\text{-}AGB = \sum dry\ biomass\ of\ ferns$$

Estimation of total woody-AGB

The most accurate estimation of woody-AGB requires destructive methods, which are time consuming and have negative effects on the sampling locations. On the other hand, destructive methods provide allometric regression equations that can be applied in situations where biomass is being assessed without tree harvesting. The use of an allometric approach is, however, essential in situations where non-destructive AGB estimates are required, although its use can propagate many uncertainties and errors during the different steps in the estimation process and, therefore, standardisation of models is essential (Chave *et al.*, 2005).

To improve the estimation of woody-AGB in tropical forests, Chave *et al*. (2005) integrated inventory data from previous studies performed in 27 different tropical regions into more standardized models. In addition, the authors provided a critical reassessment of the quality and robustness of the proposed new models and demonstrated that these models were reliable in predicting AGB across a broad range of tropical forest types. Previously, Chave *et al.* (2004) had indicated that around 20% of the error in biomass estimates could be due to the choice of allometric equation, although this could be reduced to 13% if values for specific wood gravity (SWG) (wood density; i.e. dry weight per unit volume of wood) were included (Chave *et al.* 2005). Other authors have also recognised that SWG is an important parameter that should be considered when estimating woody biomass (Kettering *et al.*, 2001, Suzuki, 1999, Chave *et al.*, 2005, Baker *et al.*, 2004). The SWG is correlated with wood strength, gap formation and the growth rate of trees with, in general, lighter woods containing more air (Suzuki, 1999, Osunkoya *et al.*, 2007, Baker *et al.*, 2004). The use of different SWG values can change the results of biomass calculations quite significantly, particularly these that are generic. Therefore they should be chosen carefully. Suzuki (1999) suggested that on average the SWG for tree species in 2-6 year old secondary forest is about 0.31 g cm^{-3}, which is nearly half of that for primary forest. Unfortunately, the pioneer tree species (*Combretocarpus rotundatus*), which dominated within the study area, was not on the list of species measured by Suzuki (1999). Therefore, the medium value of SWG for mature trees of *Combretocarpus rotundatus* was selected from the

Wood Density Database[1]. A half of that value (0.38 g cm^{-3}) was used in the calculation of biomass for this study, following Suzuki's assumption that the SWG of young regrowth is reduced by half when compared to the SWG of the same species growing in primary forest.

Woody-AGB values for single- and multi-stemmed trees were calculated based on the allometric equation proposed by Chave *et al.* (2005) for tropical moist forest. The same equation was applied in the case of saplings and seedlings, following the suggestion made by Chave *et al.* (2004) that, if present, all small trees, saplings and/or lianas should be included in biomass estimations.

The following allometric equations were used to obtain values for woody-AGB (5.2):

$$<AGB>_{est} = exp\ (-2.977 + ln\ (pD^2H)) \equiv 0.0509 * pD^2H \tag{5.2}$$

Where,

D- DBH in cm, H- height in cm, p – SWG equal to 0.38 g cm^{-3}

The biomass of seedlings in the SF plots was derived by applying destructive methods, (i.e. by cutting down the seedlings present in the 2.5 x 2.5m nested plots). Values were obtained for fresh weight and, following oven-drying, dry weight (see next section).

Finally, values for woody-AGB were calculated separately for trees, saplings and seedlings within each plot and then summed to provide total woody-AGB expressed in tons per hectare (t ha^{-1}).

Estimation of total non-woody-AGB

Estimation of fern biomass was performed following a standard method used for grass biomass. The fern samples collected in the field were weighed in the laboratory (facilities provided by University of Palangkaraya) to obtain a wet biomass value, then oven-dried at 80°C, until the samples maintained a constant weight, indicating they

[1] (http://www.worldagroforestry.org/sea/Products/AFDbases/WD/)

were completely dried out. Each sample weight was recorded and totalled for each plot, taking into account the proportion of the ground covered by fern. Fern biomass values were expressed in t ha^{-1}.

- Canopy cover analysis

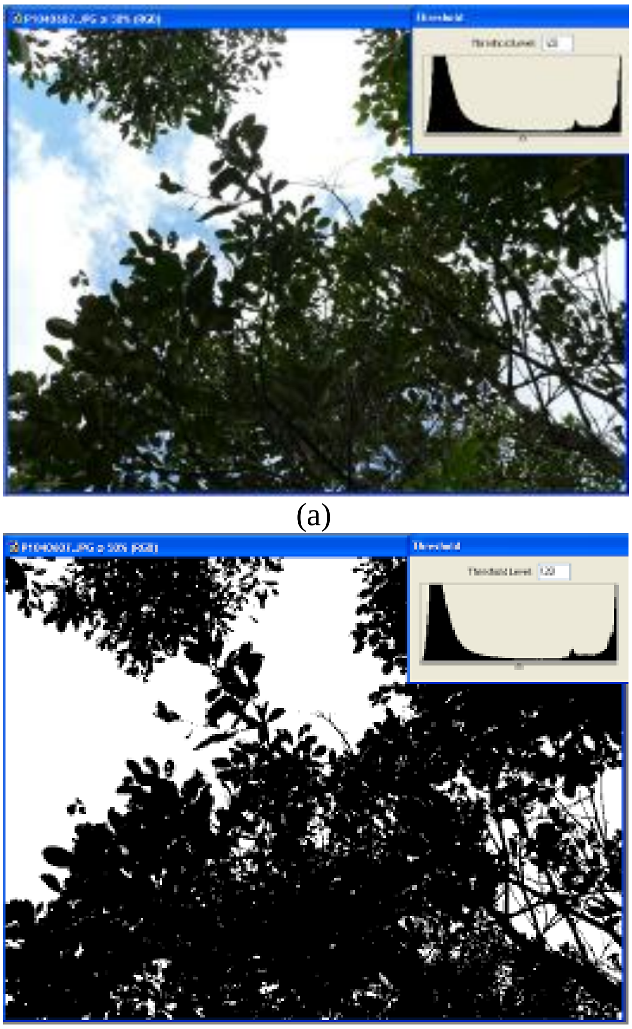
The DCP method, used in this study to measure canopy cover in the SF plots, was similar to that prescribed by Feeley (2005) following Engelbrecht and Herz (2001). All digital photographs were transformed from colour into black and white, where pure black pixels represented 'leaves' whilst pure white pixels corresponded to 'sky' (Figure 5.4), using the 'histogram' option available in Adobe Photoshop software. In the next step, the percentage of black pixels, indicating canopy closure, on each image was determined (Engelbrecht and Herz, 2001). Feeley (2005) pointed out that even though this method does not correct for the area of a hemisphere, the results are comparable with those obtained through hemispherical photography or LAI-2000 Plant Canopy Analyzers (Engelbrecht and Herz, 2001, Guevara-Escobar *et al.*, 2005). Average values for canopy closure were calculated for each plot and then averaged for the entire SF class.

- Basal area

Basal area (BA) is the cross section area of the trunk of a tree estimated at breast height; in other words it is a sum of the trunk areas of trees at DBH height for all species per unit ground area. The following equation (5.3) was applied to derive a BA for trees/saplings for each plot:

$$BA = \sum \pi/4 \; x \; (DBH)^2 \tag{5.3}$$

Values for BA were expressed as m^2 ha^{-1}.



(a)

(b)

Figure 5.4 A digital photograph transformed from a) colour to b) pure black (leaves) and pure white (sky), using the Adobe Photoshop histogram option.

5.2 Characteristics of different stages of vegetation regrowth – Analyses and results

One of the main goals of the field data collection was to collect sufficient information to produce a comprehensive description of the post-fire vegetation associated with different fire regimes and at various stages of regrowth. Statistical analyses were used to find out whether the vegetation classes delineated in the field could be statistically separated based on a range of ecological vegetation variables.

5.2.1 *Statistical analyses*

Correlation analysis

The statistical measure of the strength and direction of the relationship between two random variables is normally expressed by correlation coefficients. The correlation coefficient ranges from +1 to -1, where a correlation +1 indicates a perfect positive relationship between the variables. The Kolmogorov-Smirnov normality test and the skewness scores showed that the relation between both the vegetation and spectral variables had a non-linear form, thus Spearman's rank correlation was considered an appropriate analysis technique (Figure 5.5). Spearman's rank correlation with two–tailed tests of significance was applied to examine the relationship between selected vegetation variables and spectral data (Section 5.3).

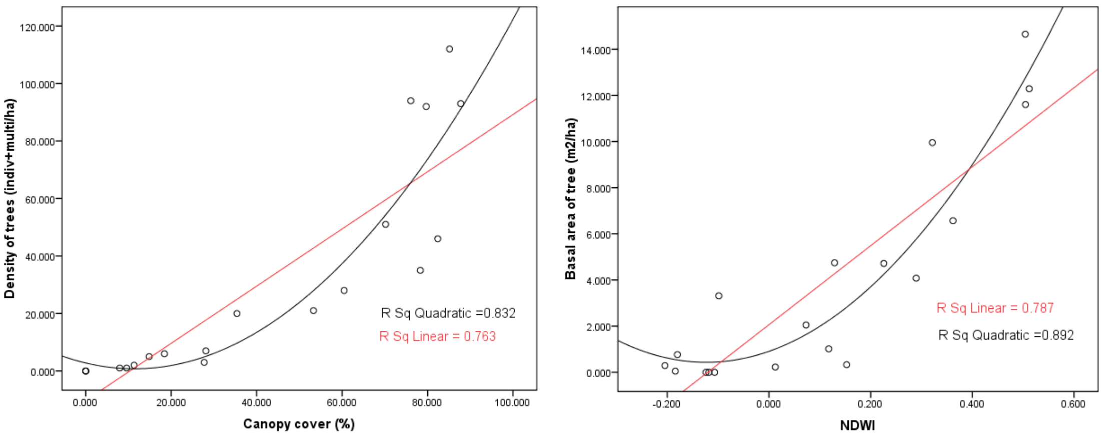


Figure 5.5 Examples of linear and non-linear regression based on vegetation and spectral variables; a non-linear function best characterises the form of the relationship.

Analysis of variance

In order to determine whether or not one regrowth classes statistically differed from each other, the analysis of variance technique (ANOVA) was applied. The ANOVA procedure is used to assess statistical separability across all the plots or, in statistical terms, whether the samples come from a population with similar distribution. The

ANOVA analysis compares the ratio between group variance and within group variance. If there are significant differences between classes, then the variability between groups would be expected to be much higher than the variability within the group, giving a higher F-test value. The F-test statistic is the ratio of between and within group variances. All statistics were tested at a significance level of 0.05. The Tukey multiple comparisons of means test was used to make all possible pairwise comparisons between means, in other words, which pair of means were significantly different from each other. In the first place, ANOVA was applied to test whether there was a statistically significant separation between classes of vegetation regrowth that were established on the basis of the vegetation structure assessed in the field. This was done to identify those variables (e.g. basal area, density of trees, canopy cover, biomass of trees, total biomass) that could be used as potential indicators of different stages of vegetation regrowth (Section 5.2.2). ANOVA analysis was also used to test whether stages of regeneration could be statistically explained by difference in the magnitude of burn severity (Section 5.3).

Descriptive analysis

Once the separation between classes was confirmed, the descriptive statistics for each stage of regrowth were used to summarize and assess the general characteristics of the dataset. The mean, standard deviation, minimum and maximum were examined to explain the vegetation characteristic of each class. In descriptive statistics, a box-plot is a convenient way of graphically representing numerical data through statistical summaries (the smallest, lower quartile, median and upper quartile and the largest observation). The box-plot may also indicate which observations, if any, might be considered outliers, numerically distanced from the rest of the data.

All statistical analyses were performed using SPSS Statistical Analysis Software version 16.

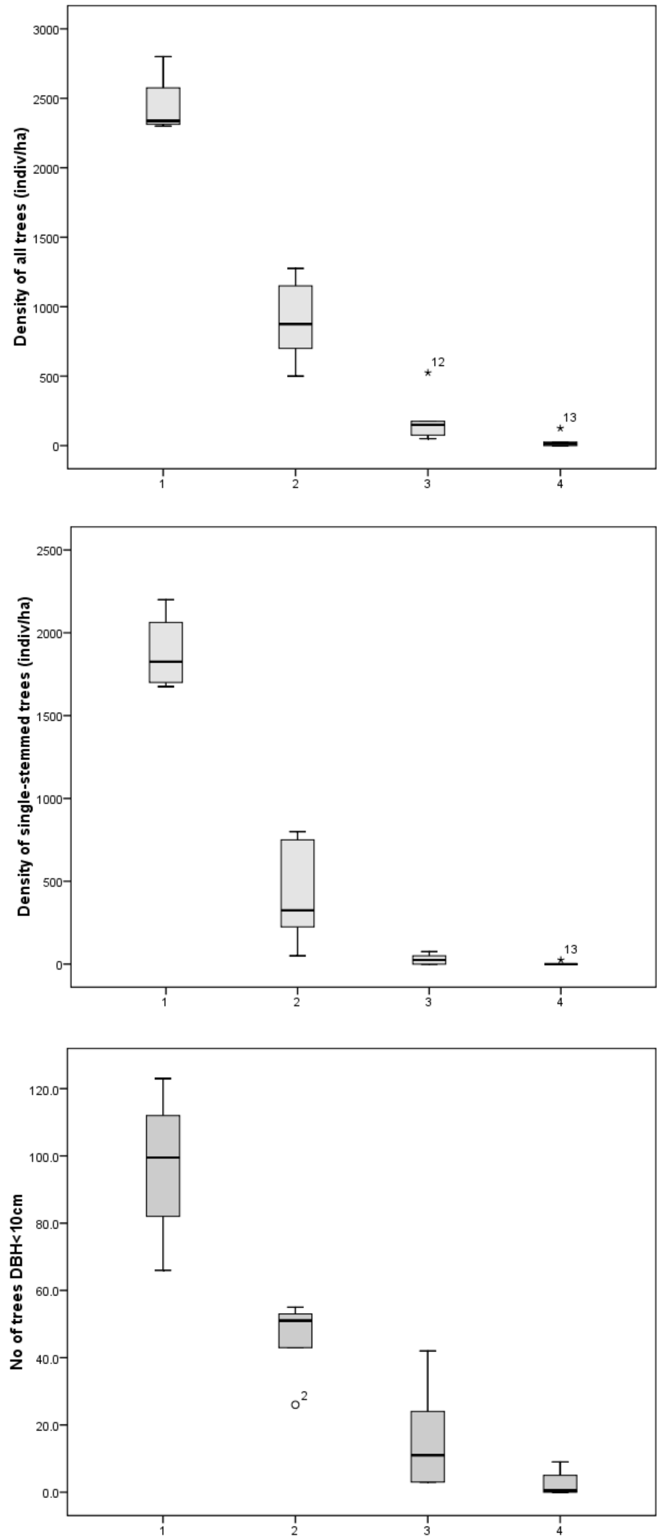
5.2.2 *Statistical variation between different classes of regrowth*

The ANOVA analysis showed that qualitative grouping of post-fire regrowth into four classes adequately captured the trend of vegetation regeneration. Table 5.2 clearly showed that some vegetation variables performed better than others in the separation of the four classes of regrowth such as the density of all trees, BA of trees, total BA, canopy cover and AGB of trees. Table 5.2 presents the list of vegetation variables that showed the best delineation between the four classes of vegetation regrowth ($p<0.002$); the rest of the analysed variables are included in Appendix 4.

The density of all trees (single- and multi-stemmed), followed by the density of single-stemmed trees, both showed a strong significant difference between various classes of regrowth ($F=118$ and 89, $p<0.0005$) (Table 5.2), indicating that these variables can be used as important indicators of vegetation regeneration. Density of trees clearly delineated the SF from the MF classes, due to a much higher number of trees in the SF class, more than double that of MF1 (Figure 5.6). Also classes MF1, MF2 and MF3 were clearly separated from each other; the MF1 class showed the greatest number of single-stemmed trees compared to an almost absence of this growth form in MF2 and MF3. The difference between classes was less significant for density of multi-stemmed trees ($F=16$) and density of multi-stemmed saplings ($F=8$). The number of trees smaller than 10cm DBH also clearly separated the SF and MF classes, and less strongly delineated the MF classes (Figure 5.6). In contrast, the number of trees with DBH from 10cm to 15cm provided a less significant separation of the different regrowth classes ($F=7$).

ANOVA		Sum of Squares	df	Mean Square	F	Sig.
Density of single- and multi-stemmed trees	Between Groups	1.624E7	3	5412118.056	118.178	.0005
	Within Groups	732739.583	16	45796.224		
	Total	1.697E7	19			
Density of single-stemmed trees	Between Groups	1.023E7	3	3411461.806	89.190	.0005
	Within Groups	611989.583	16	38249.349		
	Total	1.085E7	19			
Basal area of trees	Between Groups	563.446	3	187.815	46.101	.0005
	Within Groups	65.184	16	4.074		
	Total	628.629	19			
No of trees with DBH<10cm	Between Groups	23958.300	3	7986.100	37.716	.0005
	Within Groups	3387.900	16	211.744		
	Total	27346.200	19			
Basal area – total	Between Groups	727.240	3	242.413	37.078	.0005
	Within Groups	104.608	16	6.538		
	Total	831.848	19			
Canopy cover	Between Groups	18884.959	3	6294.986	32.381	.0005
	Within Groups	3110.454	16	194.403		
	Total	21995.413	19			
Biomass of trees	Between Groups	2070.816	3	690.272	17.574	.0005
	Within Groups	628.464	16	39.279		
	Total	2699.279	19			
Total woody biomass	Between Groups	2410.529	3	803.510	16.444	.0005
	Within Groups	781.832	16	48.864		
	Total	3192.361	19			
Density of multi-stemmed trees	Between Groups	955968.750	3	318656.250	15.902	.0005
	Within Groups	320625.000	16	20039.062		
	Total	1276593.750	19			
Biomass of fern	Between Groups	225.325	3	75.108	14.060	.0005
	Within Groups	85.473	16	5.342		
	Total	310.798	19			
Ratio of basal area of trees to total basal area	Between Groups	14338.301	3	4779.434	8.348	.001
	Within Groups	9160.228	16	572.514		
	Total	23498.530	19			
Total biomass (woody and fern)	Between Groups	1190.375	3	396.792	8.129	.002
	Within Groups	780.951	16	48.809		
	Total	1971.326	19			
Ratio of basal area of saplings to total basal area	Between Groups	93.570	3	31.190	8.019	.002
	Within Groups	62.236	16	3.890		
	Total	155.805	19			
DBH of tree	Between Groups	82.445	3	27.482	7.855	.002
	Within Groups	55.981	16	3.499		
	Total	138.426	19			
Density of multi-stemmed saplings	Between Groups	228604.167	3	76201.389	7.585	.002
	Within Groups	160739.583	16	10046.224		
	Total	389343.750	19			
No of trees with DBH from 10 to 15cm	Between Groups	699.250	3	233.083	7.465	.002
	Within Groups	499.550	16	31.222		
	Total	1198.800	19			

Table 5.2 Analysis of variance (one-way ANOVA) for selected vegetation variables sorted by values of F-test; df –degrees of freedom, F–F-test of significance, Sig.–level of significance; N=20.



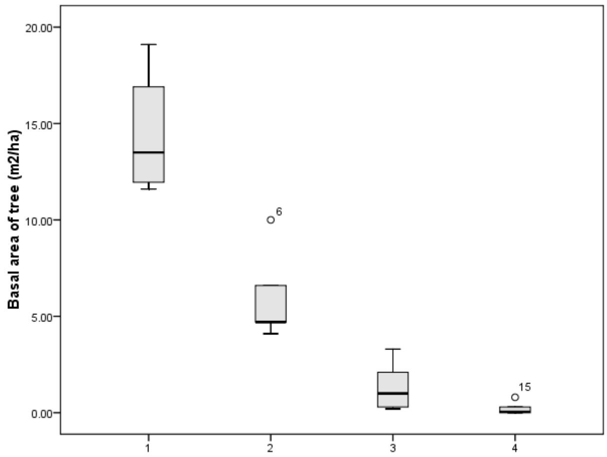
a) density of single- and multi-stemmed trees

b) density of single-stemmed trees

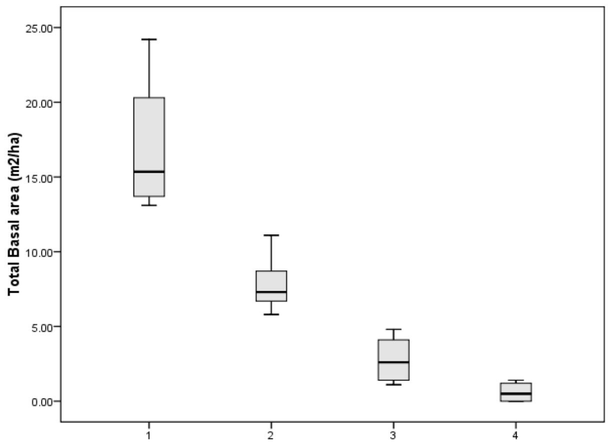
c) number of trees with DBH<10cm

Figure 5.6 Trends in a) density of single- and multi-stemmed trees, b) density of single-stemmed trees, and c) number of trees with DBH<10cm across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

BA of trees and total BA were recognised as significant variables separating all classes of regrowth (Figure 5.7). The BA of trees separated the four classes of regrowth slightly better than the total BA (F=46 and 37 respectively). This was due to the contribution of saplings to the total BA (Figure 5.8). Since BA is related to both canopy density and diameter, the same decreasing trend from the SF through to MF3 class was also observed. In contrast, the ratios of BA of trees to the total BA and the ratio of saplings to the total BA were less statistically significant ($p<0.001$ and $p<0.002$, respectively) (Table 5.2).



a) basal area of trees (single- and multi-stemmed)



b) total basal area (trees and saplings)

Figure 5.7 Trends in a) basal area of trees (single- and multi-stemmed) and b) total basal area (trees and saplings) across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

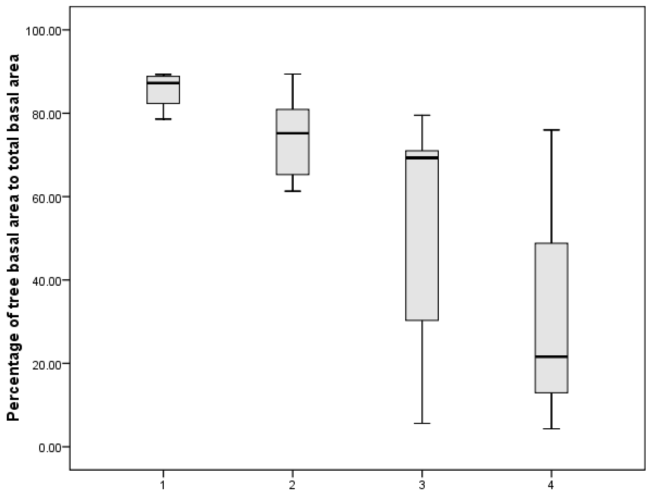
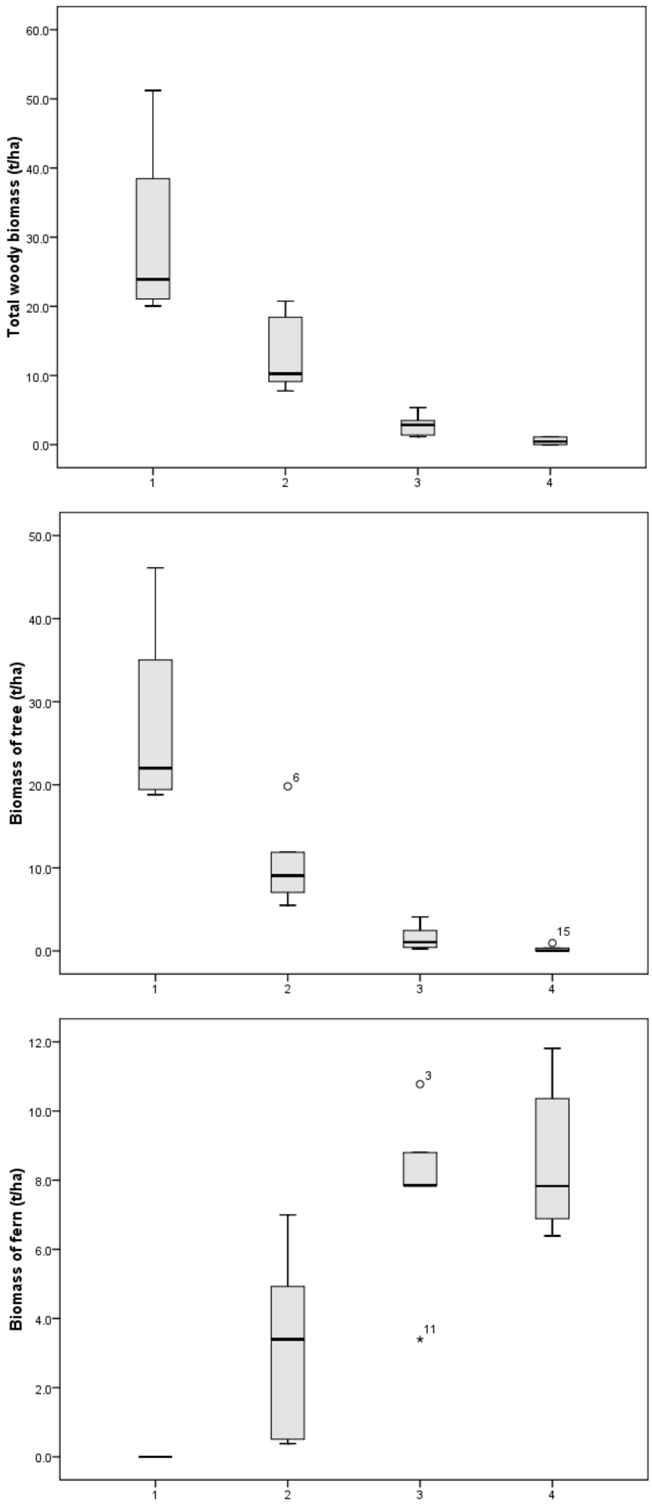


Figure 5.8 Trends in the percentage of trees basal area to total basal area across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

The biomass of trees and the total woody-AGB clearly delineated all four classes of regrowth (F=18 and 16, respectively) (Table 5.2 and Figure 5.9 a, b). Total woody-AGB is a function of DBH and height; it combines together the AGB values for trees, saplings and seedlings and therefore reflects the fact that the more advanced stages of vegetation regrowth generally accumulate greater amounts of AGB. The SF class showed much higher values of woody-AGB and biomass of trees with larger ranges of variability than in the MF classes. The amount of woody-AGB decreased gradually from the SF class, through MF1 to MF3.

In contrast, fern biomass, increased from MF1 through MF2 to MF3, but was less significant (F=14) in separating the different classes (Figure 5.9c). Consequently the total biomass, containing both woody and non-woody components, demonstrated a lower level of significance (F=8, $p<0.002$) (Table 5.2).



a) total woody-AGB

b) tree biomass

c) non-woody-AGB (fern)

Figure 5.9 Trends in a) total woody-AGB b) tree biomass, and c) non-woody-AGB (fern) across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

Canopy cover showed a relatively good separation of the MF classes with some overlap between the smallest and the largest quartiles (F=32) (Figure 5.10). The overlap between the SF and MF1 class may be associated with the different techniques used to derive canopy cover in the SF and MF plots. There was less variation in this variable for the SF class compared to the MF classes.

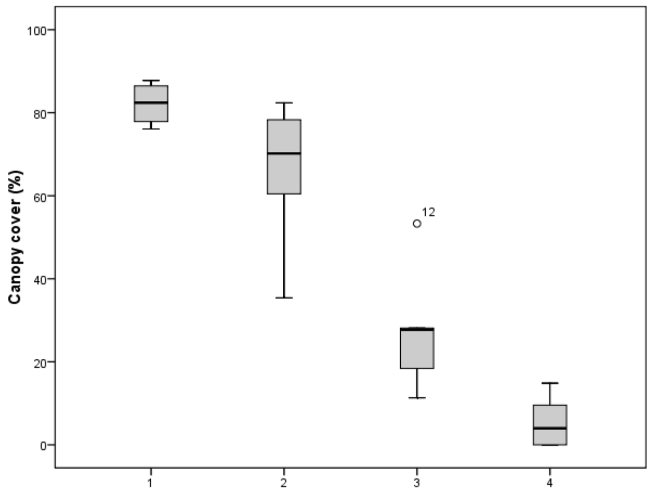


Figure 5.10 Trends in canopy cover across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

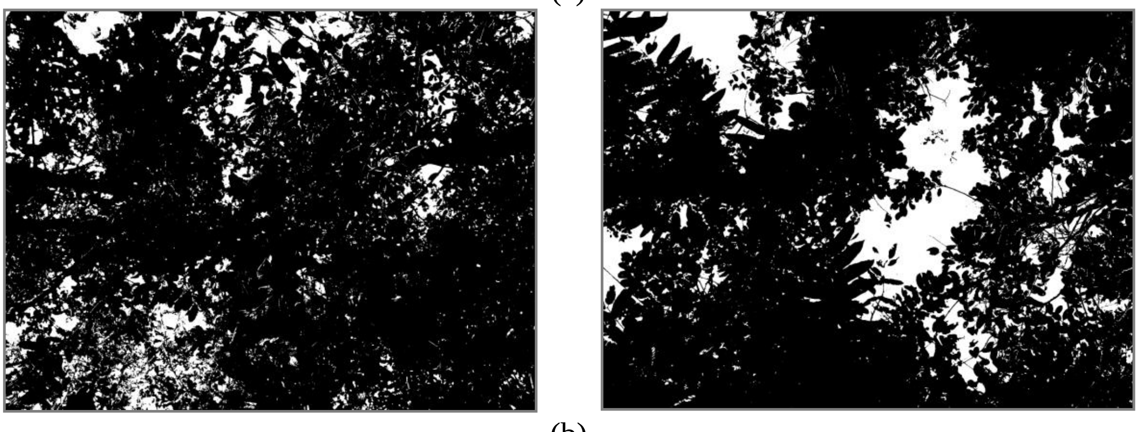
5.2.3 *Ecological description of the four classes of regrowth*

The comparative statistical analyses of the different classes of vegetation regrowth described above confirmed the existence of four different stages of post-fire vegetation regrowth. This Section provides a more detailed description of the vegetation characteristics of each of these classes.

Single fire (SF) class

The SF class containing 9-year old vegetation regrowth following a single fire in 1997 showed a relatively advanced stage in regeneration back to closed-canopy forest (Figure 5.11). Of the four SF study plots, one (no 17) was established in closer proximity (around 500m) to unburned forest compared to the other plot. This plot had higher recorded values for AGB, number of tree species, and density of trees (Table 5.3).

(a)



(b)

Figure 5.11 Advanced stage of regrowth (9-year old) following a single fire in 1997; a) vertical profile of vegetation and b) canopy closure.

VEGETATION VARIABLES (SF)	Mean	SD	Min	Max
Number of tree species (no/plot)	**16.3**	9.0	10.0	29.0
Density of single-stemmed trees (individual/ha)	**1881.0**	238.0	1675.0	2200.0
Density of multi-stemmed trees (individual/ha)	**562.5**	116.4	400.0	675.0
Density of single- and multi-stemmed trees (individual/ha)	**2444.0**	238.0	2300.0	2800.0
Ration of multi-stemmed trees to total no of trees	**22.9**	4.6	17.2	38.0
Density of single-stemmed saplings (individual/ha)	**325.0**	257.4	125.0	700.0
Density of multi-stemmed saplings (individual/ha)	**0.0**	0.0	0.0	0.0
Density of single- and multi-stemmed saplings (individual/ha)	**5900.0**	3598.0	3200.0	11200.0
Density of single- and multi-stemmed seedlings (individual/ha)	**8000.0**	1306.0	6400.0	9600.0
Ratio saplings/trees	**2.5**	1.6	1.4	5.0
DBH of trees (cm)	**7.7**	0.9	7.2	9.1
Basal area of trees (m^2/ha)	**14.4**	3.4	11.6	19.1
Basal area of saplings (m^2/ha)	**2.6**	1.7	1.5	5.2
Total basal area (m^2/ha)	**17.0**	5.0	13.1	24.2
Ration of basal area of trees to total basal area	**85.6**	4.9	78.6	89.3
Trees DBH <10cm (% of total trees no)	**84.8**	14.5	63.0	92.0
Trees DBH (10cm-15cm) (% of total trees no)	**13.8**	11.5	0.0	31.0
Trees DBH >15cm (% of total trees no)	**1.5**	3.0	0.0	6.0
No of trees DBH<10cm	**97.0**	23.5	66.0	123.0
No of trees DBH (10cm-15cm)	**15.3**	11.9	8.0	33.0
No of trees DBH >15cm	**1.5**	3.0	0.0	6.0
Canopy cover (%/plot)	**82.4**	4.7	77.5	87.5
Fern cover (% /plot)	**0.0**	0.0	0.0	0.0
Bare ground (% /plot)	**0.0**	0.0	0.0	0.0
Biomass of trees (t/ha)	**27.2**	12.8	18.8	46.1
Biomass of saplings (t/ha)	**2.5**	1.8	1.1	5.1
Total woody biomass (t/ha)	**29.8**	14.5	20.1	51.2
Fern biomass (t/ha)	**0.0**	0.0	0.0	0.0
Total biomass (t/ha)	**29.8**	14.5	20.1	51.2
Plot number and Long/Lat coordinates				
No 2: 114.0754E -2.3404S No 5: 114.0738E -2.3364S No 6: 114.0727E -2.3337S No 17: 114.0309E -2.3396S				

Table 5.3 Characteristics of vegetation regenerating after a single fire (SF); N=4.

Multiple fires (MF)

A total of sixteen plots were selected to assess vegetation regrowth following multiple fires (1997 and 2002). Three classes of regrowth (i.e. MF1, MF2, and MF3) were represented by five, five and six plots, respectively.

1) Multiple fires: MF1

The MF1 class revealed the most advanced vegetation regrowth within the MF category. A significantly lower number of tree species, lower tree density and consequently woody-AGB and BA separated this class from the SF class. The woody-AGB (13t ha^{-1}) was made up of a large number of saplings and trees. The tree stratum of this class was dominated by multi-stemmed trees whilst single-stemmed trees were dominant in the SF class (Table 5.4 and Figure 5.12).

Figure 5.12 The most advanced vegetation regrowth, class MF1, following multiple fires.

VEGETATION VARIABLES (MF1)	Mean	SD	Min	Max
Number of tree species (no/plot)	**8.8**	4.1	5.0	15.0
Density of single-stemmed trees (individual/ha)	**430.0**	330.4	50.0	800.0
Density of multi-stemmed trees (individual/ha)	**470.0**	198.0	175.0	650.0
Density of single- and multi-stemmed trees (individual/ha)	**900.0**	317.7	500.0	1275.0
Ratio of multi-stemmed trees to total no of trees	**54.8**	27.1	34.8	92.9
Density of single-stemmed saplings (individual/ha)	**580.0**	170.8	425.0	800.0
Density of multi-stemmed saplings (individual/ha)	**270.0**	153.5	150.0	450.0
Density of single- and multi-stemmed saplings (individual/ha)	**850.0**	236.5	600.0	1175.0
Density of single- and multi-stemmed seedlings (individual/ha)	**115.0**	94.5	0.0	225.0
Ratio saplings/trees	**1.0**	0.2	1.0	1.0
DBH of trees (cm)	**7.3**	0.9	6.3	8.2
Basal area of trees (m^2/ha)	**6.0**	2.4	4.1	10.0
Basal area of saplings (m^2/ha)	**1.9**	0.7	1.1	2.6
Total basal area (m^2/ha)	**7.9**	2.1	5.8	11.1
Ratio of basal area of trees to total basal area	**74.4**	11.4	61.3	89.4
Trees DBH <10cm (% of total tree no)	**86.8**	9.7	79.0	100.0
Trees DBH (10cm-15cm) (% of total tree no)	**12.5**	9.1	0.0	21.0
Trees DBH >15cm (% of total tree no)	**0.8**	1.5	0.0	3.0
No of trees DBH<10cm	**45.6**	11.9	26.0	55.0
No of trees DBH (10cm-15cm)	**6.2**	4.3	0.0	12.0
No of trees DBH >15cm	**0.9**	0.4	0.0	2.0
Canopy cover (%/plot)	**67.7**	21.4	35.4	92.4
Fern cover (% /plot)	**29.0**	21.6	5.0	57.0
Bare ground (% /plot)	**0.0**	0.0	0.0	0.0
Biomass of trees (t/ha)	**10.7**	5.6	5.5	19.8
Biomass of saplings (t/ha)	**1.5**	0.5	0.9	2.1
Total woody biomass (t/ha)	**13.3**	5.9	7.8	20.8
Fern biomass (t/ha)	**3.2**	2.9	0.4	7.0
Total biomass (t/ha)	**16.5**	3.8	12.5	21.3
Plot number and Long/Lat coordinates				
No 3: 114.0582E -2.3211S No 7: 114.0573E -2.3218S No 8: 114.0566E -2.3232S No 9: 114.0556E -2.3245S No 10: 114.0543E -2.3263S				

Table 5.4 Characteristics of vegetation regenerating after multiple fires: MF1-respectively advanced regrowth; N=5.

2) Multiple fires: MF2

The MF2 class illustrated an intermediate stage of vegetation regrowth. Reduction in the density of trees was compensated for by an increase in the number of saplings and seedlings compared to the MF1 class; even so, on average 77% of the plot area was covered by fern with some patches of bare ground (3%). Non-woody-AGB contributed the greatest amount to the total AGB, with 7.7t ha^{-1} compared to the woody-AGB of 2.8t ha^{-1}. The number of multi-stemmed trees exceeded the number of single-stemmed trees (Figure 5.13 and Table 5.5).

Figure 5.13 The intermediate stage of vegetation regrowth, class MF2, following multiple fires.

VEGETATION VARIABLES (MF2)	Mean	SD	Min	Max
Number of tree species (no/plot)	**4.0**	2.0	2.0	7.0
Density of single-stemmed trees (individual/ha)	**30.0**	32.6	0.0	75.0
Density of multi-stemmed trees (individual/ha)	**165.0**	170.1	25.0	450.0
Density of single- and multi-stemmed trees (individual/ha)	**195.0**	191.5	50.0	525.0
Ratio of multi-stemmed trees to total no of trees	**80.5**	27.5	33.3	100.0
Density of single-stemmed saplings (individual/ha)	**1140.0**	2136.2	0.0	4950.0
Density of multi-stemmed saplings (individual/ha)	**155.0**	123.0	50.0	300.0
Density of single- and multi-stemmed saplings (individual/ha)	**1295.0**	2085.5	100.0	5000.0
Density of single- and multi-stemmed seedlings (individual/ha)	**890.0**	1306.4	75.0	3200.0
Ratio saplings/trees	**16.9**	28.5	1.0	67.0
DBH of trees (cm)	**6.6**	0.5	6.2	7.5
Basal area of trees (m^2/ha)	**1.4**	1.3	0.2	3.3
Basal area of saplings (m^2/ha)	**1.4**	1.4	0.4	3.8
Total basal area (m^2/ha)	**2.8**	1.6	1.1	4.8
Ratio of basal area of trees to total basal area	**51.1**	31.8	5.6	79.5
Trees DBH <10cm (% of total tree no)	**98.3**	4.1	90.0	100.0
Trees DBH (10cm-15cm) (% of total tree no)	**1.7**	4.1	0.0	10.0
Trees DBH >15cm (% of total tree no)	**0.0**	0.0	0.0	0.0
No of trees DBH<10cm	**16.6**	16.6	3.0	42.0
No of trees DBH (10cm-15cm)	**0.0**	0.0	0.0	0.0
No of trees DBH >15cm	**0.0**	0.0	0.0	0.0
Canopy cover (%/plot)	**27.8**	15.9	11.3	53.3
Fern cover (% /plot)	**77.4**	18.0	47.0	92.0
Bare ground (% /plot)	**3.0**	3.0	0.0	8.0
Biomass of trees (t/ha)	**1.7**	1.6	0.3	4.1
Biomass of saplings (t/ha)	**1.0**	1.2	0.0	3.1
Total woody biomass (t/ha)	**2.8**	1.7	1.1	5.4
Fern biomass (t/ha)	**7.7**	2.8	3.4	10.8
Total biomass (t/ha)	**10.6**	3.8	4.5	14.3

Plot number and Long/Lat coordinates

No 4:	114.0589E	-2.3209S
No 11:	114.0519E	-2.3292S
No 12:	114.0527E	-2.3279S
No 13:	114.0497E	-2.3326S
No 18:	114.0327E	-2.3343S

Table 5.5 Characteristics of vegetation regenerating after multiple fires: MF2-intermediate stage of regrowth; N=5.

3) Multiple fires: MF3

The MF3 class representing the least advanced stage of woody regrowth had the lowest value for woody-AGB with only 0.5t ha^{-1}, and fern biomass (8.5t ha^{-1}) made up the largest portion of the total AGB value. This class was characterised by an almost complete absence of trees and only a few saplings; the ground surface had an average 97% cover by ferns (Figure 5.14 and Table 5.6).

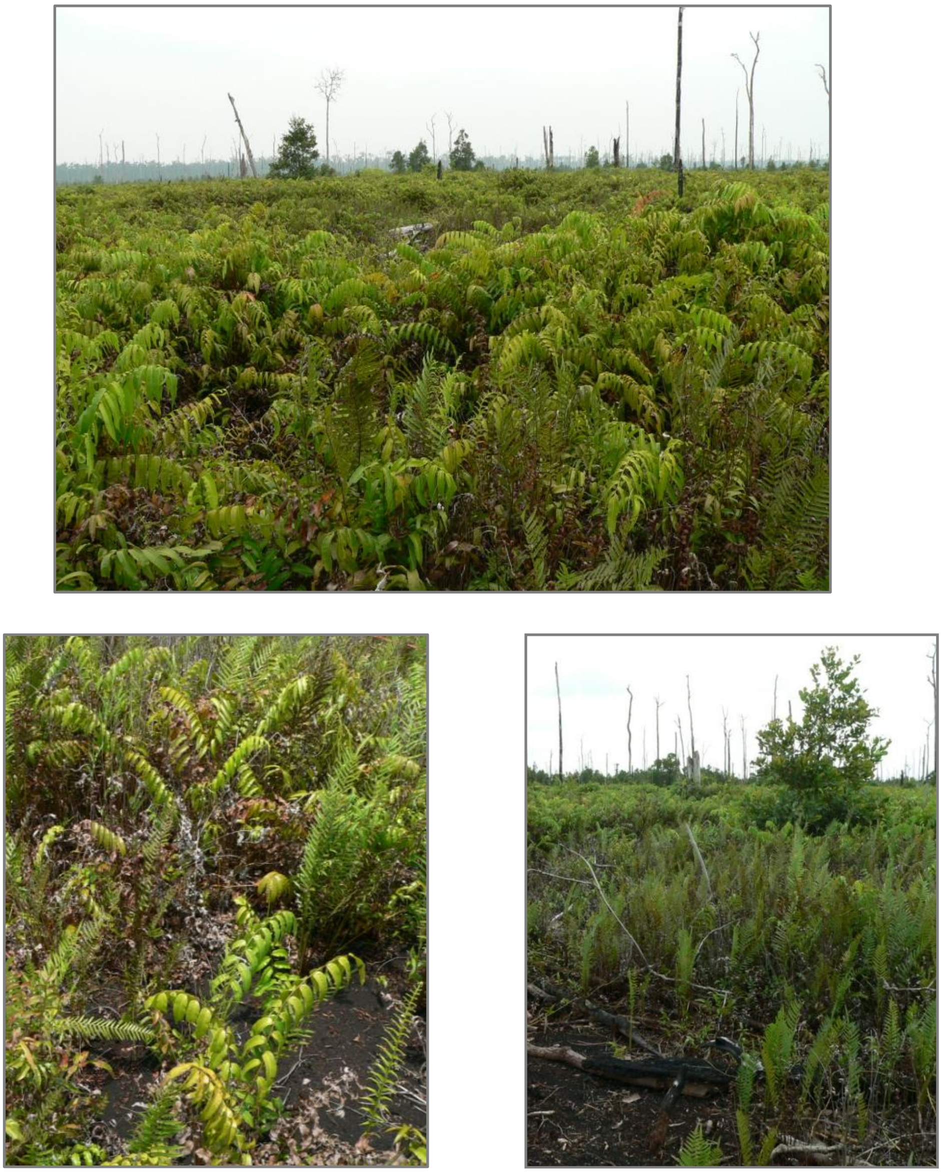


Figure 5.14 The least advanced stage of vegetation regrowth, class MF3, following multiple fires.

VEGETATION VARIABLES (MF3)	Mean	SD	Min	Max
Number of tree species (no/plot)	**2.7**	3.5	0.0	8.0
Density of single-stemmed trees (individual/ha)	**4.2**	10.2	0.0	25.0
Density of multi-stemmed trees (individual/ha)	**25.0**	38.7	0.0	100.0
Density of single- and multi-stemmed trees (individual/ha)	**29.2**	48.5	0.0	125.0
Ratio of multi-stemmed trees to total no of trees	**93.3**	11.5	80.0	100.0
Density of single-stemmed saplings (individual/ha)	**616.7**	932.4	0.0	1975.0
Density of multi-stemmed saplings (individual/ha)	**20.8**	33.2	0.0	75.0
Density of single- and multi-stemmed saplings (individual/ha)	**637.5**	963.7	0.0	2025.0
Density of single- and multi-stemmed seedlings (individual/ha)	**312.5**	450.5	0.0	1150.0
Ratio saplings/trees	**14.7**	14.7	0.0	69.0
DBH of trees (cm)	**5.7**	0.7	5.1	6.5
Basal area of trees (m^2/ha)	**0.2**	0.3	0.0	0.8
Basal area of saplings (m^2/ha)	**0.4**	0.1	0.0	1.1
Total basal area (m^2/ha)	**0.6**	0.7	0.0	5.2
Ratio of basal area of trees to total basal area	**34.0**	37.4	4.3	76.0
Trees DBH <10cm (% of total tree no)	**100.0**	0.0	100.0	100.0
Trees DBH (10cm-15cm) (% of total tree no)	**0.0**	0.0	0.0	0.0
Trees DBH >15cm (% of total tree no)	**0.0**	0.0	0.0	0.0
No of trees DBH<10cm	**2.5**	3.7	0.0	9.0
No of trees DBH (10cm-15cm)	**0.0**	0.0	0.0	0.0
No of trees DBH >15cm	**0.0**	0.0	0.0	0.0
Canopy cover (%/plot)	**5.4**	6.3	0.0	14.8
Fern cover (% /plot)	**96.7**	4.2	91.0	100.0
Bare ground (% /plot)	**3.2**	2.4	1.0	7.0
Biomass of trees (t/ha)	**0.2**	0.4	0.0	0.9
Biomass of saplings (t/ha)	**0.3**	0.3	0.0	0.7
Total woody biomass (t/ha)	**0.5**	0.6	0.0	1.1
Fern biomass (t/ha)	**8.5**	2.2	6.4	11.8
Total biomass (t/ha)	**9.0**	2.7	6.4	12.7
Plot number and Long/Lat coordinates				
No 14: 114.0488E -2.3344S No 15: 114.0436E -2.3385S No 16: 114.0421E -2.3416S No 19: 114.0283E -2.3357S No 20: 114.0327E -2.3375S No 21: 114.0211E -2.3333S				

Table 5.6 Characteristics of vegetation regenerating after multiple fires: MF3- the least advanced stage of wood regrowth; N=6.

5.2.4 *Plant species diversity*

With increasing intensity and frequency of fire disturbance, the number of tree species reduced greatly from an average of 16.3 species in the SF class to 8.8 - 2.7 species in the MF classes, with only two dominant species of woody recolonizer, namely *Combretocarpus rotundatus* and *Cratoxylon glaucum* (Table 5.7 and Figure 5.15). The SF class showed the greatest variation in tree species composition. The SF plot located closest to the edge of the remaining forest contained a mixture of primary forest species, with *Cratoxylon spp.*, *Shorea spp.* and *Litsea spp.* representing 19%, 13% and 9% of the total number of trees, respectively.

Classes of regrowth	**SF**	**MF1**	**MF2**	**MF3**
Average no of tree species	16.3	8.8	4.0	2.7
Dominant species of tree (% dominance amongst trees in plot)	*Combretocarpus rotundatus* (1.1-88.2%) *Cratoxylon glaucum* *Shorea spp.* *Litsea spp.* Various	*C. rotundatus* (74.5-100%)	*C. rotundatus* (95.2-100%)	*C. rotundatus* (57.7-100%)
Dominant species of sapling	Various	*C. rotundatus* (41.2-77%)	*C. glaucum* (18.7-87.5%) *C. rotundatus*	*C. glaucum.* (73.9-93.8%) *C. rotundatus*
Dominant species of seedling	Various	Various	*C. glaucum & C. rotundatus* (45.5-100%)	*C. glaucum & C. rotundatus* (72.0-100%)
Fern *Stenochlaena palustris* (% cover)	0	25.8	55.2	54.2
Fern *Blechnum indicum* (% cover)	0	1.2	18.8	38.2
Bare ground (% cover)	0	0	3.0	3.2

Table 5.7 Dominant species of trees, saplings and seedlings, and percentage cover of ferns and bare ground for each class of regrowth; (plot size 20 x 20m).

By contrast, in the SF plots further away from the forest edge, the primary forest species were replaced by pioneer species, dominated by *Combretocarpus spp.*

At the highest level of degradation (MF1 and MF2 plots) the number of trees species dropped on average to 2.7-4.0 species per class, with woody vegetation largely replaced by less structured plant communities dominated by ferns (*Stenochlaena palustris* and *Blechnum indicum*). In these two classes, plots with *Blechnum indicum* had a higher proportion of bare ground because the vertical structure of the fern fronds revealed a larger area of the underlying peat surface (Figure 5.16). In plots dominated by *Stenochlaena palustris*, which has a more sprawling habit, there was greater ground cover and hence less exposed peat surface (Table 5.7).

The list of tree species reported within each of four classes of regrowth is presented in Appendix 5.

Figure 5.15 Recolonizing tree species: Combretocarpus rotundatus (top view) and Cratoxylon glaucum (bottom); the fern, Stenochlaena palustris, is growing in the foreground.

Figure 5.15 Recolonizing tree species: Combretocarpus rotundatus (top view) and Cratoxylon glaucum (bottom); the fern, Stenochlaena palustris, is growing in the foreground.

Avg. height: 120-150 cm
Avg. biomass: 12.7 t ha^{-1}

Family: *Blechnaceae*
Genus: *Stenochlaena*
Species: *Stenochlaena palustris*
Local name: *Kalakai*

Avg. height: 60-80 cm
Avg. biomass: 7.2 t ha^{-1}

Family: *Blechnaceae*
Genus: *Blechnum*
Species: *Blechnum indicum*
Local name: *Lampasas*

Figure 5.16 Vertical and nadir view of the two dominant fern species: Stenochlaena palustris and Blechnum indicum.

5.3 Examination of burn severity

As demonstrated in the previous section, the post-fire vegetation was separated into four classes of vegetation regrowth. Differences between the SF class and the MF classes were clearly caused by a more advanced phase of vegetation regrowth in the former associated with a longer post-fire recovery period. It was more difficult, however, to explain the variation in vegetation structure and composition in areas subjected to multiple fires, where three classes of regrowth could be distinguished. It was hypothesised that the variation in the vegetation structure of the MF plots could be related to the burn severity of the most recent fire affecting the study area. In this Section, the integration of *in situ* data with spectral data derived from satellite images was tested to address this hypothesis.

There is a lack of tested methods to support analysis of burn severity in the tropics. Therefore, numerous spectral approaches, some of them successfully used in non-tropical situations (Section 2.2.3), were examined in order to determine the most reliable method which could be applied to quantify burn severity in a tropical context. Several single and bi-temporal indices (i.e. NDVI, NBR and NDWI) and spectral fractional endmembers derived from images obtained by the Landsat ETM+ sensor were tested against post-fire vegetation variables to find out:

A. If/how spectral data were correlated with vegetation variables derived from *in situ* data collected four years after a fire event. This would explain whether the characteristics of the regenerating vegetation and the separation of the vegetation classes could be explained by differences in the magnitude of burn severity which, in addition to fire frequency, was hypothesised to be a dominant control on vegetation regrowth.
B. Which spectral data (indices or mixed fractions of vegetation), if any, could be used to distinguish and define the magnitude of burn severity in the context of tropical peatland vegetation.

C. Whether vegetation response to fire can be used to quantify the magnitude of burn severity.

Several single and bi-temporal indices, namely Normalised Burn Ratio, Normalised Difference Vegetation Index and Normalised Difference Water Index, were calculated from either a single post-fire image or a combination of pre- and post-fire images. The pre-fire image was acquired in June 2000 (three years after the first fire), while the post-fire image was obtained in January 2003 (a few months after the 2002 fire).

Definitions of spectral indices derived from Landsat ETM+ used in this study:

NDVI is defined as:

$$NDVI = R(3) - IR(4) \ / \ R(3) + IR(4)$$
$$dNDVI = NDVIpre - NDVIpost \tag{5.4}$$

R(3) = 0.68µm; IR(4) = 0.86µm

NDWI is defined as:

$$NDWI = IR(4) - SWIR(6) \ / \ IR(4) + SWIR(6)$$
$$dNDWI = NDWIpre - NDWIpost \tag{5.5}$$

IR(4) = 0.86µm ; SWIR(6) = 1.24µm

NBR is defined as:

$$NBR = IR(4) - SWIR(7) / \ IR(4) + SWIR(7)$$
$$dNBR = NBRpre - NBRpost \tag{5.6}$$

IR(4) = 0.86µm ; SWIR(7) = 2.2µm

Spectral mixture analysis (SMA), which had been used in a previous study to quantify the level of forest degradation caused by burning (Section 2.2.3), was also tested. The SMA approach assumes that the reflectance of each pixel can be decomposed into three endmember fractions: green vegetation (GV), non-photosynthetic vegetation (NPV) and Shade. The pixel-purity index, available in ENVI4.4, was used to identify the candidates for image endmembers. The endmembers were selected from post-fire Landsat image based on spectral shape and image contents; the GV endmember was extracted from green vegetation (grass or fern), NPV from heavily senesced vegetation (along canal banks) and Shade from deep, dark water (rivers). The SMACC Endmember Extraction model available in ENVI4.4 was applied to estimate the proportion of each of the endmembers within the Landsat pixel. In the next step, the three types of endmembers were extracted from the post-fire reflectance image and incorporated with the vegetation variables.

Figure 5.17 illustrates the stages involved in integrating the vegetation variables and spectral data in order to derive the magnitude of burn severity (burn severity classes) for the 2002 fire.

5.3.1 *Relationship between spectral data and vegetation variables*

Spearman's rank correlation was used to compare the strength of the association between the spectral data and those vegetation variables previously selected as the best separators of the classes of vegetation regrowth (Section 5.2). This was done in order to determine which of the spectral data could be used to determine burn severity. The SF plot (no 17) that was located in closer proximity to unburned forest lies outside of the pattern of a distribution, in the distant from the rest of the plots, thus was excluded from analyses.

dNBR showed the strongest correlation with a number of vegetation variables amongst both the single and bi-temporal spectral variables (Table 5.8). Of the single indices, NBR showed the strongest correlation followed by NDWI, GV and NPV; by contrast, NDVI and dNDVI showed the weakest correlations. NDVI did, however, reveal a strong correlation with fraction of GV, perhaps because both GV and NDVI are particularly sensitive to the quantity of greenness. Those vegetation variables that are highly correlated with dNBR can be considered as potential indicators of long-term fire effects (i.e. total woody-AGB, BA of trees, or density of trees). Figure 5.18 and 5.19 show the logarithmic character of the correlation between vegetation variables and dNBR and NBR.

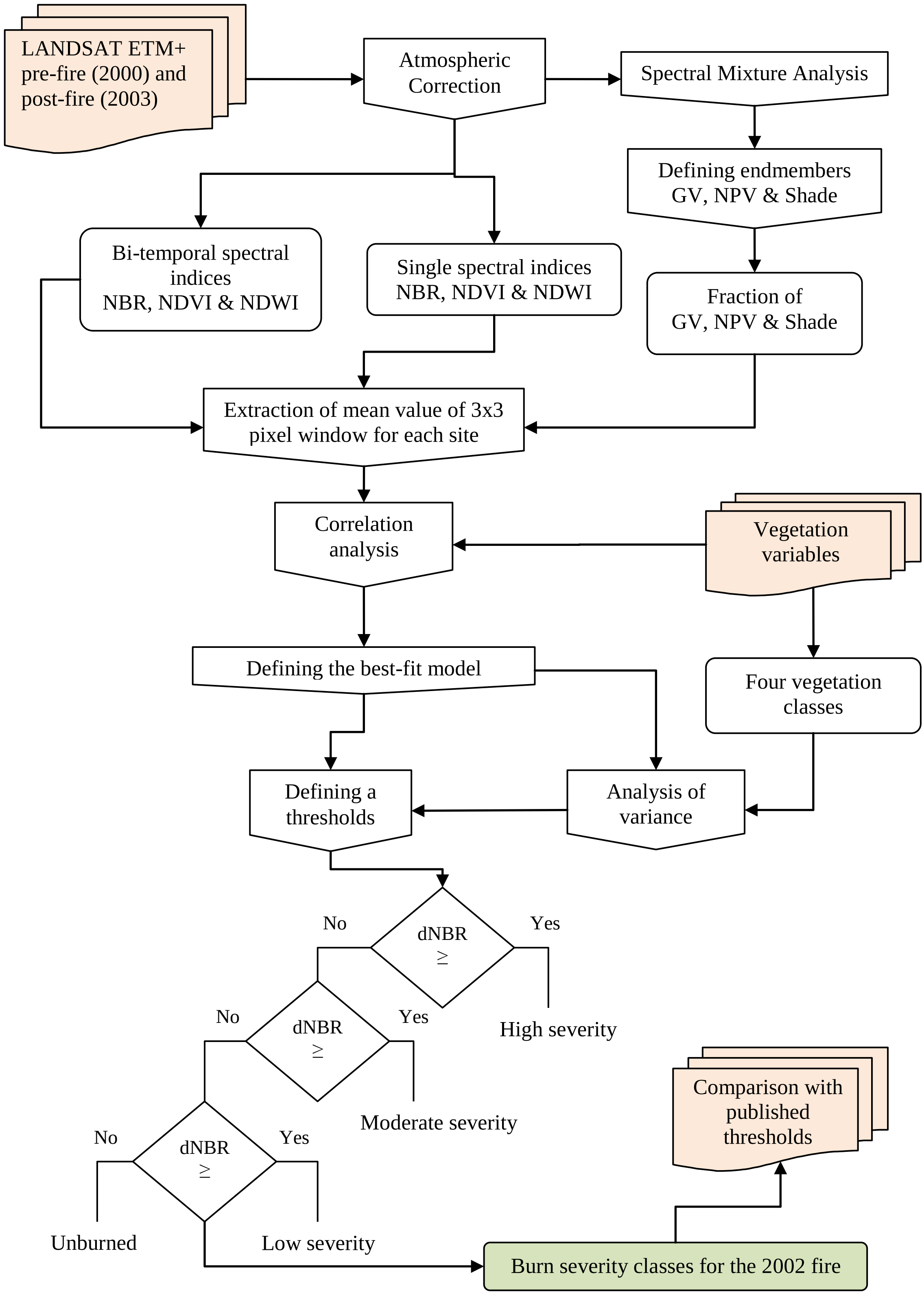


Figure 5.17 Integration of vegetation variables and spectral data to determine burn severity classes for the 2002 fire.

VARIABLES		NBR	**dNBR**	NDVI	**dNDVI**	NDWI	**dNDWI**	GV	Shade	NPV
Total woody-AGB	Corr.	.889**	**-.891****	.812**	**-.793****	.886**	**-.868****	.842**	.632**	-.826**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.004	.000
Basal area of trees	Corr.	.866**	**-.872****	.786**	**-.786****	.863**	**-.870****	.817**	.545*	-.873**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.016	.000
AGB of trees	Corr.	.865**	**-.870****	.782**	**-.786****	.861**	**-.866****	.814**	.570*	-.873**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.011	.000
Density of all trees	Corr.	.854**	**-.864****	.782**	**-.777****	.847**	**-.861****	.804**	.543*	-.820**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.016	.000
Total basal area	Corr.	.851**	**-.859****	.794**	**-.772****	.847**	**-.840****	.817**	.585**	-.786**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.009	.000
Number of trees with DBH<10cm	Corr.	.840**	**-.851****	.758**	**-.762****	.837**	**-.853****	.790**	.546*	-.852**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.016	.000
Canopy cover	Corr.	.838**	**-.831****	.765**	**-.747****	.831**	**-.819****	.800**	.527*	-.779**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.021	.000
Density of single-stemmed trees	Corr.	.811**	**-.826****	.748**	**-.743****	.808**	**-.807****	.774**	.650**	-.703**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.003	.001
AGB of fern	Corr.	-.828**	**.803****	-.828**	**.830****	-.831**	**.818****	-.836**	-.019	.790**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.937	.000
Total AGB (woody +fern)	Corr.	.677**	**-.686****	.623**	**-.595****	.672**	**-.651****	.642**	.720**	-.607**
	Sig.	.001	**.001**	.004	**.007**	.002	**.003**	.003	.001	.006
Number of species	Corr.	.633**	**-.635****	.635**	**-.571***	.622**	**-.590****	.626**	.499*	-.566*
	Sig.	.004	**.004**	.004	**.011**	.004	**.008**	.004	.030	.012
GV	Corr.	.977**	**-.965****	.991**	**-.975****	.982**	**-.947****	1.000	.233	-.888**
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	NULL	.336	.000
Shade	Corr.	.364	**-.387**	.202	**-.190**	.343	**-.352**	.233	1.000	-.310
	Sig.	.125	**.102**	.407	**.435**	.150	**.140**	.336	NULL	.197
NPV	Corr.	-.912**	**.912****	-.882**	**.904****	-.918**	**.930****	-.888**	-.310	1.000
	Sig.	.000	**.000**	.000	**.000**	.000	**.000**	.000	.197	NULL
*. Correlation is significant at the 0.05 level (2-tailed);**. Correlation is significant at the 0.01 level (2-tailed)										

Note: one outlier: SF plot (no 17) located in closer proximity to unburned forest

Table 5.8 Correlation of co-efficient values (r of Spearman's rho) between selected vegetation variables and post-fire single and bi-temporal spectral indices (NBR, NDVI and NDWI) and the fractions of green vegetation (GV), non-photosynthetic vegetation (NPV), and Shade; in situ data were collected four years after the 2002 fire; N=19 (excluding one outlier).

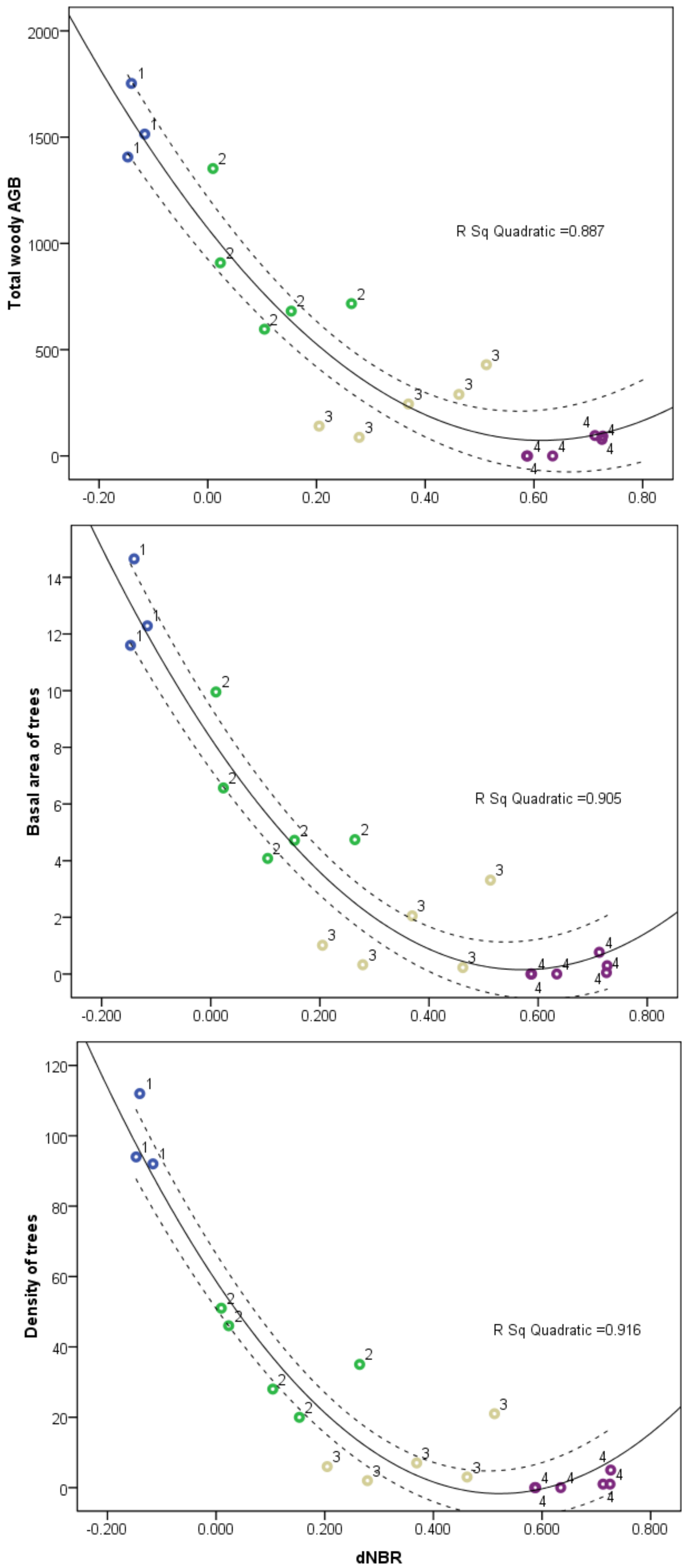


Figure 5.18 Dependence of the vegetation variables: total woody-AGB, tree basal area and tree density against variation of the dNBR values; 1–SF, 2–MF1, 3–MF2 and 4–MF3; N=19.

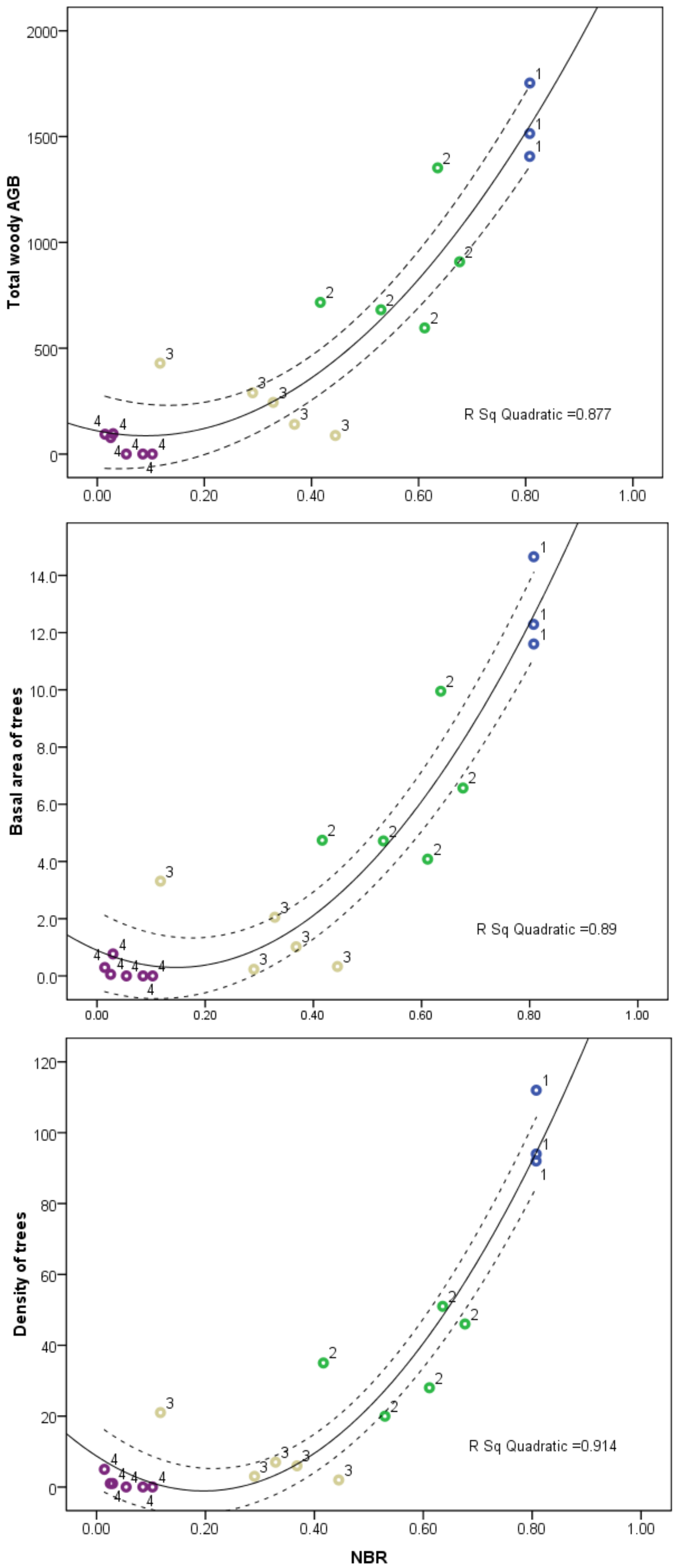


Figure 5.19 Dependence of the vegetation variables: total woody-AGB, tree basal area and tree density against variation of the NBR values; 1–SF, 2–MF1, 3–MF2 and 4–MF3, N=19.

5.3.2 *Magnitude of burn severity*

The strong correlation between vegetation variables and spectral data (in particular dNBR, NDWI and NBR) demonstrates the potential for delineating burn severity in tropical peatland ecosystems using the post-fire vegetation response approach. Nevertheless, the correlation coefficients do not confirm whether the selected statistically separated classes of vegetation regrowth can also be differentiated spectrally (i.e. by taking into account the effect of fire on the spectral properties of the land surface). Therefore, the same set of spectral data (indices and endmembers) was tested for separability between and within the different classes of vegetation regrowth (Table 5.9).

An ANOVA test indicated that there was a significant difference existing among all of the classes considering all spectral indicators ($p<0.0005$), except the Shade endmember (Table 5.9). Out of the whole group of tested variables, both NBR and NDWI provided the best differentiation of the four classes of regrowth, ($F=73$ and $F=68$, respectively), followed by GV and dNBR ($F=63$ and $F=62$) (Figure 5.20 and 5.21). The Tukey multiple comparisons of mean test were carried out to determine specifically where the significance difference lay. This test showed that NBR, dNBR, and NDWI can separate each class at a 95% confidence level ($p<0.02$) (Table 5.10). The results of the Tukey multiple comparisons of mean test are presented in Appendix 6.

Figure 5.20 shows graphically that the NBR and dNBR differentiated the four classes better than any of the other single indices; however, there was a slight overlap between the largest and the smallest quartiles of the MF stages. Both the NDWI and the dNDWI showed a similar trend with relatively good separation between classes. The fraction of GV also separated the four classes relatively clearly (Figure 5.21). The poorest distinction of regrowth classes was obtained using NDVI, dNDVI and NPV fraction (Table 5.10).

ANOVA		Sum of Squares	df	Mean Square	F	Sig.
NBR	Between Groups	1.529	3	.510	73.108	.0005
	Within Groups	.112	16	.007		
	Total	1.641	19			
dNBR	Between Groups	1.647	3	.549	62.526	.0005
	Within Groups	.140	16	.009		
	Total	1.787	19			
NDVI	Between Groups	.394	3	.131	26.178	.0005
	Within Groups	.080	16	.005		
	Total	.474	19			
dNDVI	Between Groups	.404	3	.135	20.066	.0005
	Within Groups	.107	16	.007		
	Total	.511	19			
NDWI	Between Groups	1.106	3	.369	67.828	.0005
	Within Groups	.087	16	.005		
	Total	1.193	19			
dNDWI	Between Groups	1.241	3	.414	39.967	.0005
	Within Groups	.166	16	.010		
	Total	1.406	19			
GV	Between Groups	2.263	3	.754	63.104	.0005
	Within Groups	.191	16	.012		
	Total	2.455	19			
Shade	Between Groups	.024	3	.008	2.784	.075
	Within Groups	.045	16	.003		
	Total	.069	19			
NPV	Between Groups	1.726	3	.575	47.273	.0005
	Within Groups	.195	16	.012		
	Total	1.921	19			

Table 5.9 Analysis of variance (One-way ANOVA) for spectral data for four classes of regrowth; df –degree of freedom, F–F-test of significance, Sig.–level of significance.

Spectral indicators	Classes undistinguishable at level p<0.02
NBR	All separated
dNBR	All separated
NDVI	[1,2] [2,3] [3,4]
dNDVI	[1,2] [2,3] [3,4]
NDWI	All separated
dNDWI	[1,2] [2,3]
GV	[3,4]
Shade	[1,2] [2,3] [3,4]
NPV	[2,3] [3,4]

Table 5.10 Confusion between classes (p<0.02) for each spectral indicator based on the Tukey multiple comparisons test; 1–SF, 2–MF1, 3–MF2 and 4–MF3.

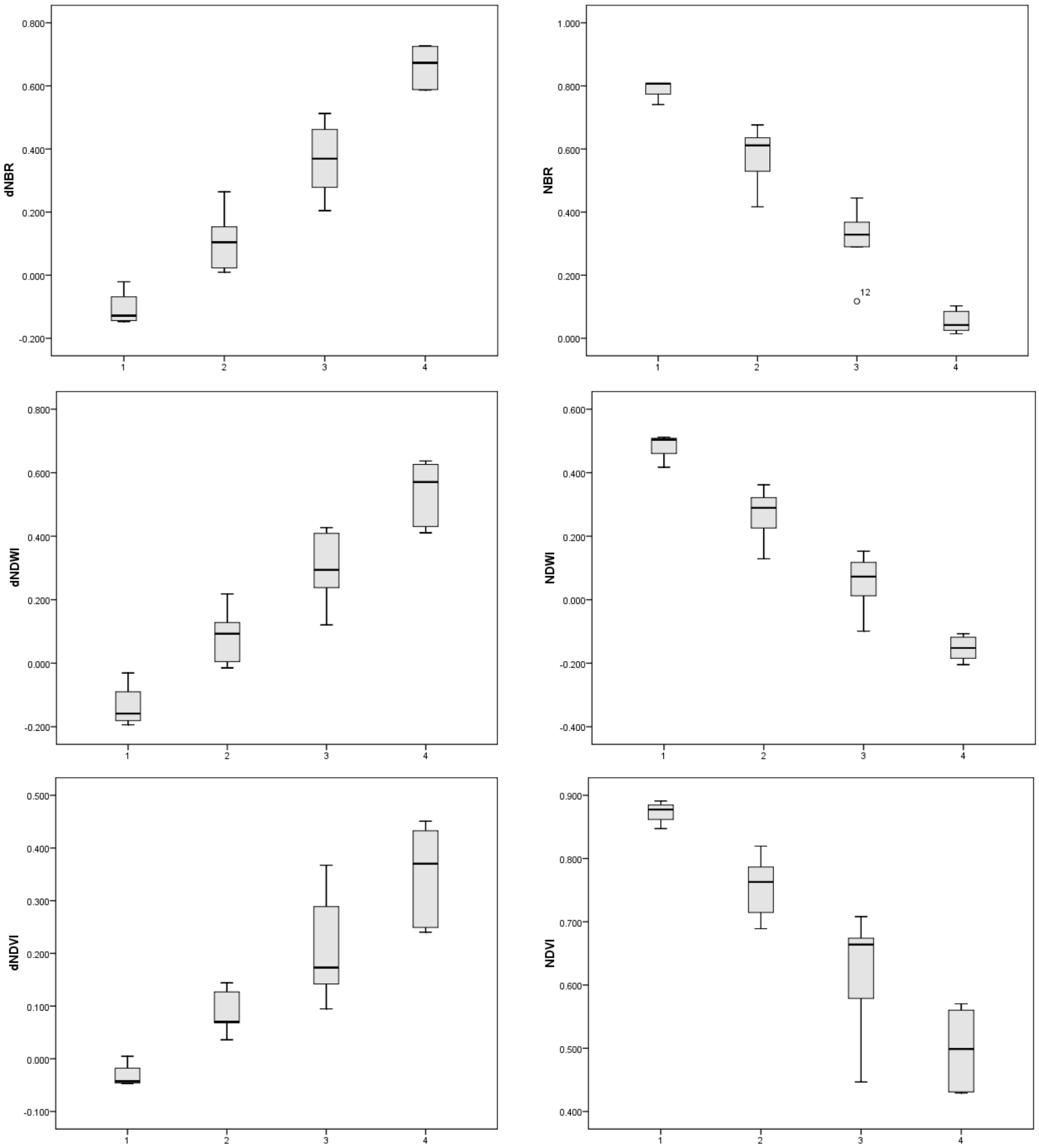


Figure 5.20 Trends in NBR, NDWI and NDVI, single indices (right column) and bi-temporal spectral indices (left column) across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

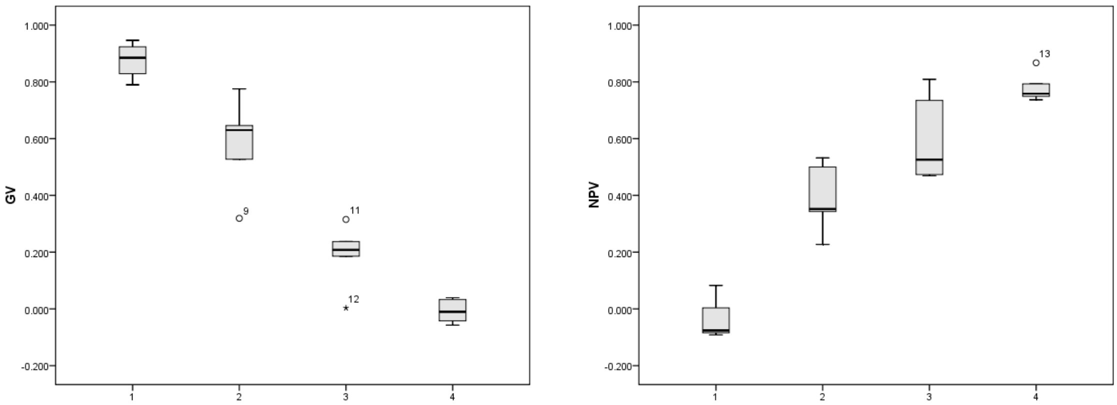


Figure 5.21 Trends in the green fraction (GV) and non-photosynthetic fraction (NPV) across the four classes of regrowth: 1–SF, 2–MF1, 3–MF2 and 4–MF3.

Separation of the SF from MF class can be confidently performed using NPV and GV fractions as well as NDWI and NBR, whilst three classes of regrowth following multiple fires can be distinguished using dNBR, NBR and NDWI (Table 5.11).

The dNBR, NBR, and NDWI could be recommended for separation of both the SF from MF class and all three classes within the areas affected by multiple fires.

Separation between classes	**Rank of the best spectral indicators**	**Level of significance**
SF – MF1	NPV	0.0005
	NDWI	0.002
	GV	0.005
	NBR	0.007
	dNBR	0.016
MF1 – MF2	GV	0.0005
	NBR	0.001
	NDWI	0.002
	dNBR	0.003
MF2 – MF3	dNBR	0.0005
	NBR	0.001
	NDWI	0.002
	dNDWI	0.006

Table 5.11 Separation between pairs of classes; spectral indicators are sorted according to the level of significance ($p<0.02$); based on the Tukey multiple comparisons test.

5.3.3 *Classification of burn severity, determining threshold values*

On the basis of the results obtained in the correlation and separability analyses, the spectral data with the greatest capacity for assessing burn severity (i.e. dNBR and NBR) were selected and a classification of burn severity was performed. Figure 5.22 shows the location of the sampling plots overlaid on the dNBR image calculated for burn scars of the 2002 fire, presented in continues scale. As a result, the four classes of vegetation regrowth could be confirmed as being associated with the following classes of burn severity for the 2002 fire:

SF = unburned (in 2002)
MF1 = low severity fire (MF-L)
MF2 = moderate severity fire (MF-M)
MF3 = high severity fire (MF-H)

Figure 5.23 illustrates the distribution of the highly correlated spectral indices: dNBR and NBR and selected vegetation variables for each sampling plot, sorted by the dNBR values. There is a good agreement between the vegetation variables and the dNBR values except for plot 10. High values for woody-AGB, tree density, and tree BA led to this plot being placed within the MF1 regrowth class. On the basis of spectral features alone, however, the relatively high dNBR value would have placed this plot within the MF2 class. This apparent misclassification can be perhaps related to differences in the burn severity of the previous fire that occurred in 1997 (see the following Section).

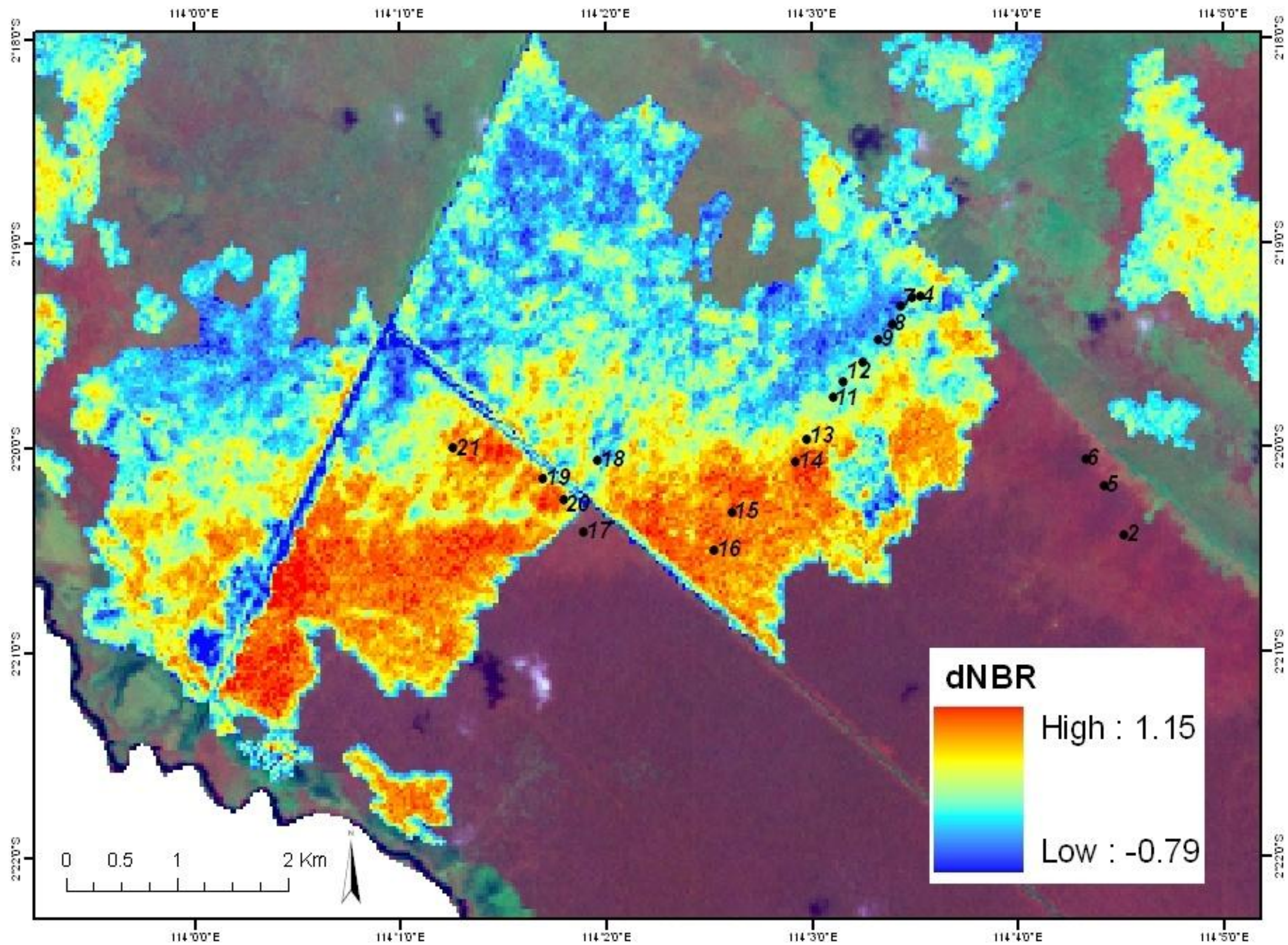


Figure 5.22 Bi-temporal NBR values defined for the 2002 fire; black dots show the location of sampling sites.

Finally, a threshold procedure was performed on the dNBR and NBR values to derive the categories of burn severity based on the classes of vegetation regeneration.

Table 5.12 shows the threshold values derived for dNBR and NBR based on four classes of vegetation regrowth, excluding plot no 10.

Classes of burn severity based on dNBR	**N**	**Mean**	**Std. Deviation**	**Std. Error**	**95% Confidence Interval for Mean**		**Min**	**Max**
					Lower Bound	Upper Bound		
SF=unburned in 2002	4	-.10622	.058320	.029160	-.19902	-.01342	-.147	-.021
MF1=MF_L low severity	4	.07257	.068207	.034103	-.03596	.18110	.009	.153
MF2=MF_M medium severity	5	.36534	.126842	.056725	.20785	.52284	.205	.512
MF3=MF_H high severity	6	.66213	.067165	.027420	.59165	.73262	.587	.726

(a)

Classes of burn severity based on NBR	**N**	**Mean**	**Std. Deviation**	**Std. Error**	**95% Confidence Interval for Mean**		**Min**	**Max**
					Lower Bound	Upper Bound		
SF=unburned in 2002	4	.79069	.033425	.016713	.73751	.84388	.741	.808
MF1=MF_L low severity	4	.61323	.061947	.030974	.51465	.71180	.530	.677
MF2=MF_M medium severity	5	.30973	.121875	.054504	.15840	.46106	.117	.445
MF3=MF_H high severity	6	.05188	.035546	.014512	.01457	.08918	.014	.103

(b)

Table 5.12 Threshold values of a) dNBR and b) single NBR for each class of severity related to the classes of vegetation regrowth; N=19, excluding plot 10.

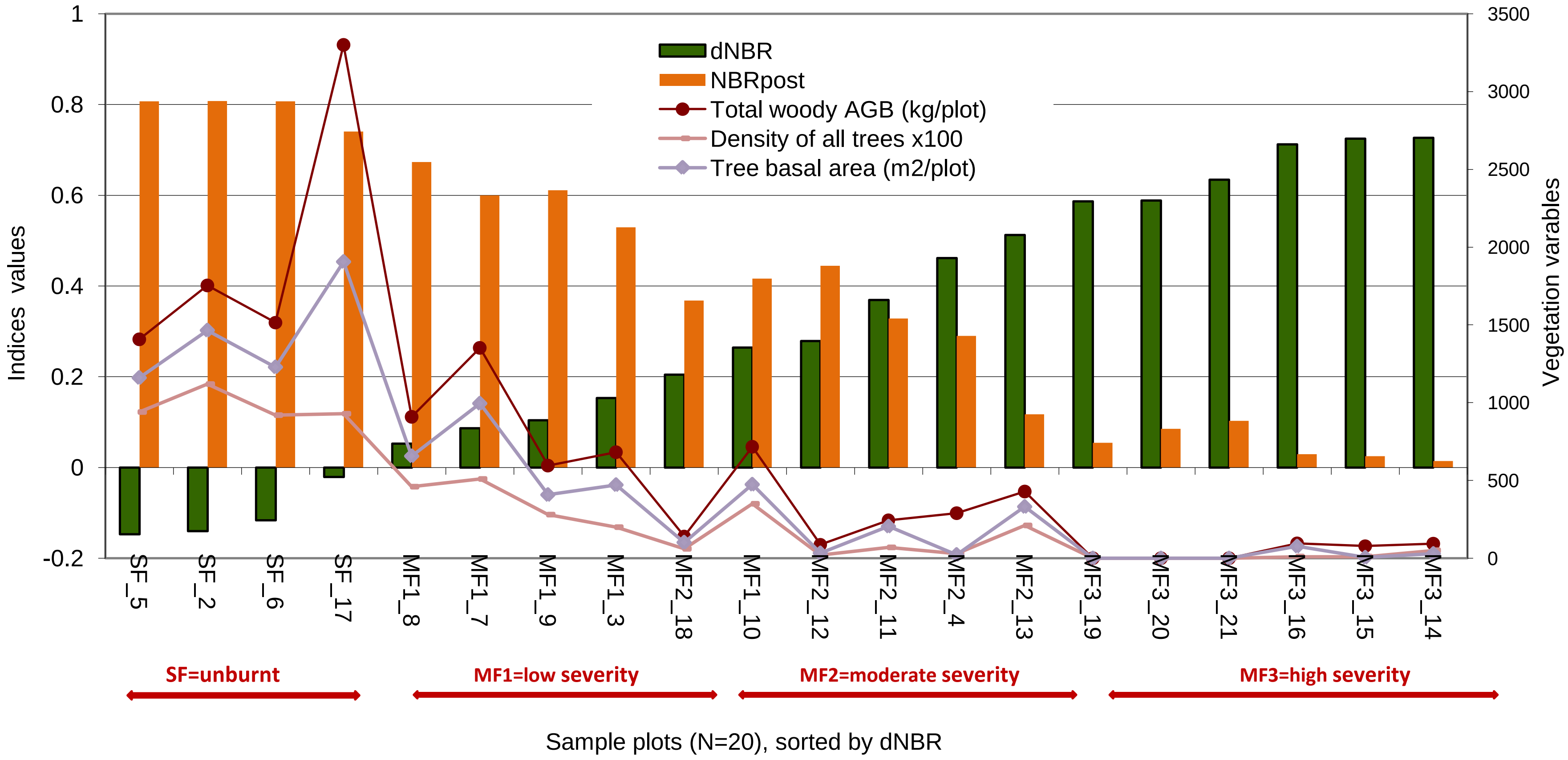


Figure 5.23 Variation in the NBR and dNBR values for each sampling plot, sorted by dNBR; SF–unburnt in 2002, but burnt in 1997, MF–burnt twice: in 1997 and 2002.

5.3.4 *Assessment of burn severity as a result of the 1997 fire*

The previous results indicated that pre-fire vegetation type and condition were strong determinants of post-fire vegetation structure. Consequently, if areas have been affected by multiple fires then the effect of repeated burning should, where possible, be considered. Within the study area, two fires occurred over a five year time interval, in 1997 and 2002; however assessment of the severity of the first fire (1997) was difficult due to the lack of appropriate EO data. Unfortunately, the only available post-fire image was for the year 2000 (i.e. three years after the fire) by which time vegetation regrowth had already covered the burned surfaces. As, however, the dNBR index had proved efficient in determining the burn severity of the 2002 fire, the same technique was applied to look at alteration in the dNBR values caused by the 1997 fire. Thus dNBR values were derived from pre- and post-fire images acquired in 1997 (pre-fire) and 2000, respectively.

Figure 5.24 presents the variation of these values for both fires (i.e. 1997 and 2002), associated with the sampling plots. In general, a low value of dNBR indicates a less severe fire, thus it is obvious that some plots (e.g. plots 14, 15 and 16) were lightly affected by the first fire but heavily burned by the subsequent fire, perhaps owing to the larger fuel load (standing and fallen dead timber) remaining after the first fire. By contrast, plots 19, 20 and 21 were burnt twice at high severity which explains the almost complete absence of a tree canopy and dominance by ferns.

Figure 5.25 shows a strong linear correlation between the dNBR index values for the 2002 and 1997 fires for the MF3 plots (r^2=0.87).

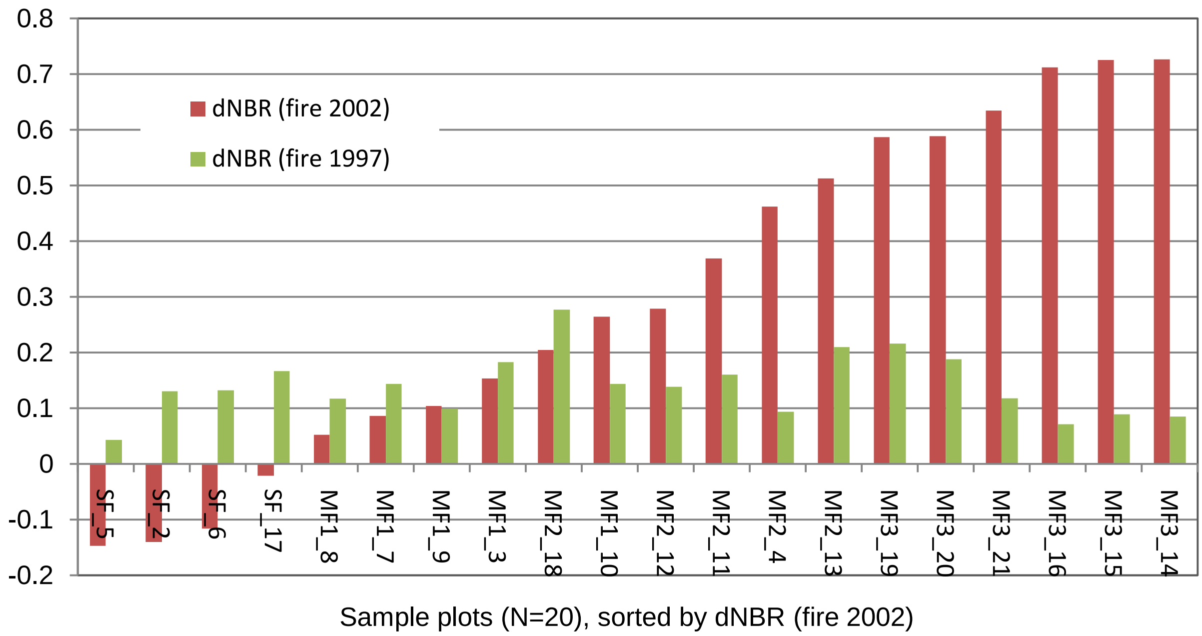


Figure 5.24 Variation in the dNBR calculated for the 2002 fire and 1997 fire; plots are labelled by numbers and sorted by dNBR of the 2002 fire; lower value of dNBR indicates less severe fire.

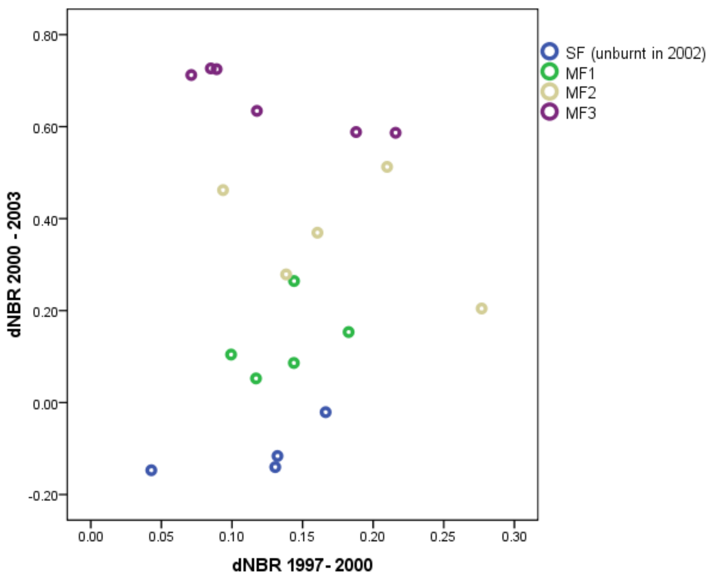


Figure 5.25 Scatter plot between the dNBR of the 2002 fire and the dNBR of the 1997 fire for the unburned in 2002 (SF), low (MF1), moderate (MF2) and high (MF3) severity fire of 2002 (linear correlation coefficient for each of classes: r^2=0.51, 0.16, 0.20 and 0.87 respectively).

5.4 Summary

- The ANOVA analysis showed that qualitative grouping of post-fire regrowth into four classes adequately captured the trend of vegetation regeneration.

- Plots established in the areas subjected to a single fire revealed the most advanced stage of regeneration back to closed-canopy forest.

- Locations affected by multiple fires (MF) presented less advanced stages of regeneration and were further separated into three sub-classes representing to most advanced vegetation regrowth (MF1), intermediate (MF2), and the least advanced stage (MF3).

- Woody-AGB showed a decreasing trend from the SF plots (20-51t ha^{-1}) through to the MF3 plots (0-1t ha^{-1}). The process of recovery of biomass and vegetation structure can be stimulated by proximity of burned areas to remnant patches of primary forest.

- The density of trees clearly delineated from the SF to MF classes, due to a much higher number of trees in the SF class, more than double that of MF1, and an almost total absence of trees in MF3.

- Proportion of ground covered by fern decreased and proportion of bare ground increased from MF1 to MF3; whereas these two features were not present in the SF class.

- The number of tree species declined from an average of 16 species in the SF class to 9-3 species in the MF classes, with only two dominant species of woody recolonizer in the MF plots: *Combretocarpus rotundatus* and *Cratoxylon glaucum*.

- Second fires had more profound effects on the ecosystem, with very slow or no recovery of woody-biomass and invasion by non-woody plant communities dominated by two species of fern, *Stenochlaena palustris* and *Blechnum indicum.*

- Non-woody (i.e. fern) biomass on average reached 7.7t ha^{-1} and 8.5t ha^{-1} in areas subjected to second fires. The proportion of fern cover increased to 77% and 98% in areas subject to moderate and high severity burns, where canopy cover was only 28% and 5%, respectively.

- A strong correlation was established between spectral indices (i.e. dNBR and NBR, NDWI) derived from pre/post satellite imagery and sets of vegetation parameters collected four years after the last fire.

- The ANOVA test indicated that both NBR and dNBR followed by NDWI and GV fraction provide the best differentiation among any of the regrowth classes. The Tukey multiple comparisons test confirmed that NBR, dNBR and NDWI can delineated between all classes at a 95% confidence level ($p<0.02$).

- The variation in the structure and species composition of post-fire vegetation regrowth observed in the field (i.e. MF1, MF2 and MF3 plots) could be explained by differences in the magnitude of burn severity for the 2002 fire. Second fires of moderate or high severity had more profound effects on the ecosystem, with very slow or no recovery of woody-biomass and invasion by non-woody plant communities dominated by two species of fern.

6 CARBON DYNAMICS – STOCKS AND LOSSES

As was reported in Chapters 4 and 5, repeated fires modify the tropical PSF ecosystem leaving behind a heavily degraded landscape in which secondary vegetation succession back to forest is impeded. Tropical peatlands store much greater amounts of carbon below ground (in the peat) than above ground (in the forest biomass). Peatland fires, which involve combustion of both AGB and peat, generate large atmospheric emissions of carbon compounds. This result in emissions of GHGs, particularly CO_2 and CO, and particulate matter to the atmosphere and thus make a contribution to global climate change and regional pollution episodes.

This chapter explores dynamic trends in carbon stocks in the study area over the 32-year period of investigation (1973-2005) and emphasises carbon losses resulting from combustion of both AGB and peat. Estimation of carbon stocks and losses in the study area was performed using GIS analyses using the series of spatial-temporal data such as land cover, fire frequency, type of land cover affected by each fire, interval between fires and above and belowground biomass data obtained for the period of investigation 1973-2005 (Chapters 4 and 5). The method used to quantify the carbon stocks and losses is presented in Figure 6.1.

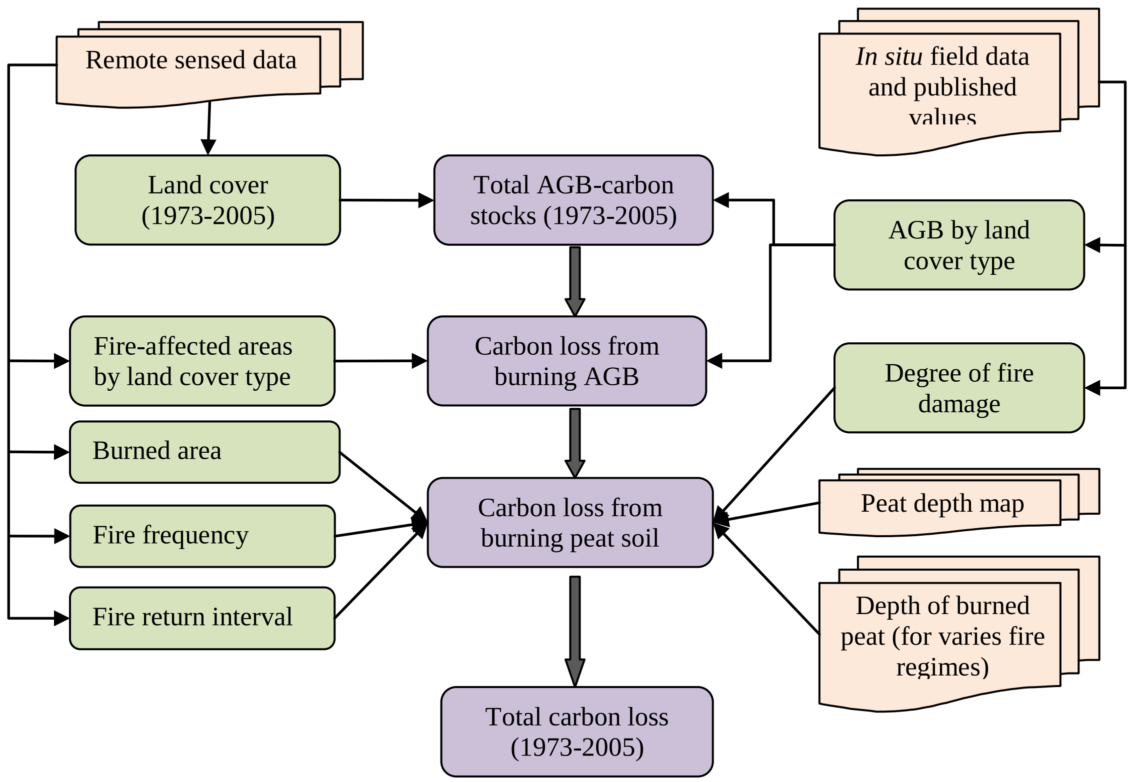


Figure 6.1 Methodology for estimating carbon stocks and losses from burning AGB and peat for the period of investigation (1973-2005).

6.1 Estimating aboveground biomass carbon stocks and losses

6.1.1 *Aboveground biomass carbon stocks*

The carbon stocks in AGB in the study area were estimated using the land cover data derived for the period 1973-2005 (Chapter 4). The biomass of different land cover types may vary regionally according to different growth conditions, thus where possible (Table 6.1) the biomass values used in the carbon stock and loss calculations were measured either as part of this study (Chapter 5) or at other sites in close proximity to the study area and/or elsewhere within the same province (Central Kalimantan).

Land cover type	Aboveground biomass (t ha^{-1})	Biomass used for calculations (t ha^{-1})	Data sources
Mixed peat swamp forest, Central Kalimantan	312	313	(Waldes and Page, 2001)
	314		(Sulistiyanto, 2004)
Low pole peat swamp forest, Central Kalimantan	249	251	(Waldes and Page, 2001)
	252		(Sulistiyanto, 2004)
Freshwater swamp forest, Malaysia, Indonesia	220	220	(WWF, 2008)
Heath forest, Central Kalimantan	225	225	(Miyamoto *et al.*, 2007)
Mangrove forest, East Kalimantan	216	216	(Komiyama *et al.*, 1988)
Secondary forest, Central Kalimantan	30	30	This study (SF)
Mosaic of trees and non-woody vegetation, Central Kalimantan	17	17	This study (MF1)
Sedge swamp, Asia	37	37	(WWF, 2008)
Non-woody vegetation, Central Kalimantan	9	9	This study (MF3)
Cultivated land, Indonesia	56	56	(WWF, 2008)
Recently burned area, Central Kalimantan	9	9	Assumption* (equal MF3)

* based on observation of vegetation structure

Table 6.1 Aboveground biomass values used for calculation of carbon stocks and losses for each land cover type; values obtained from WWF (2008) indicate the average biomass values for specific land cover type.

The carbon content of the AGB was set at 50%, following the IPCC GPG guidelines (IPCC, 2006). The difference in biomass of secondary vegetation between years was not considered and set up as a constant (Table 6.1).

The total AGB for each analysed year was calculated as:

$$T_{AGB} = \sum A_n * AGB_n \tag{6.1}$$

Where:

A_n = area of n land cover type (ha)

AGB_n = total AGB for n land cover type (Mt ha^{-1})

Then, the total amount of AGB-carbon stored in Block C for the period 1973-2005 was calculated as:

$$C_t = \sum_{i}^{i+j} T_{AGB} * C_c \tag{6.2}$$

Where:

C_t = total AGB-carbon stock for the period 1973-2005 (Mt)

T_{AGB} = total AGB (Mt)

C_c = amount of carbon content of AGB (dry weight), set at 50%

i = initial year 1973

j = next year, including years 1991, 1993, 1997, 2000 and 2005

Table 6.2 shows the total AGB-carbon stock in the study area for the years of investigation with additional separation of the AGB-carbon stocks in PSF and post-fire, vegetation regrowth. Vegetation regrowth carbon stocks comprised secondary forest, mosaic of trees and non-woody vegetation, and non-woody vegetation (Table 6.3).

Years	**Block C**		**Peat swamp forest**		
	AGB-carbon stock (Mt)	% of carbon loss, 1973 initial year	AGB-carbon stock (Mt)	% of AGB-carbon stock in Block C	% of carbon loss, 1973 initial year
1973	49.61		41.46	83.6	
1991	44.81	9.7	35.55	79.4	14.3
1993	39.41	20.6	30.30	76.9	26.9
1997*	38.68	22.0	29.57	76.5	28.7
2000	20.27	59.1	11.64	57.4	71.9
2005	16.86	66.0	8.39	49.8	79.8

*pre-fire image

Table 6.2 AGB-carbon stocks (Mt) and percentage AGB-carbon losses compared to the initial year of 1973, calculated for the entire study area (Block C=448,912ha) and peat swamp forest for each year of investigation.

Years	Vegetation regrowth	
	AGB-carbon stock (Mt)	% of AGB-carbon stock in Block C
1991	0.30	0.7
1993	0.31	0.8
1997	0.65	1.7
2000	2.66	13.2
2005	0.91	7.1

Table 6.3 AGB-carbon stock in secondary vegetation for each year of investigation.

Initially in 1973, AGB in the study area stored around 50 MtC, of which 41 Mt (84% of AGB-carbon stock in Block C) was stored in PSF alone. During the following decades (1973-1991), the AGB-carbon stock was reduced by 10% across the entire study area and by 14% in PSF, with stocks declining to 45 Mt and 36 Mt respectively (Table 6.2). Further reductions to 39 MtC (22% reduction since 1973) within the study area occurred between 1991 and 1997 (pre-fire). The greatest carbon stock reduction was observed following the 1997 fire, with a reduction of 18 Mt, when carbon stock declined to 20 Mt in the study area (59% of the 1973-stock) and to 12 Mt (72% of the 1973-stock) in PSF. By the end of the 2005 dry season, the AGB-carbon stock had declined to 17 Mt across the entire study area and 8 Mt in PSF; thus 66% of the AGB-carbon stock in the study area was lost over the period 1973-2005 and 80% of the PSF AGB-carbon stock (Figure 6.2).

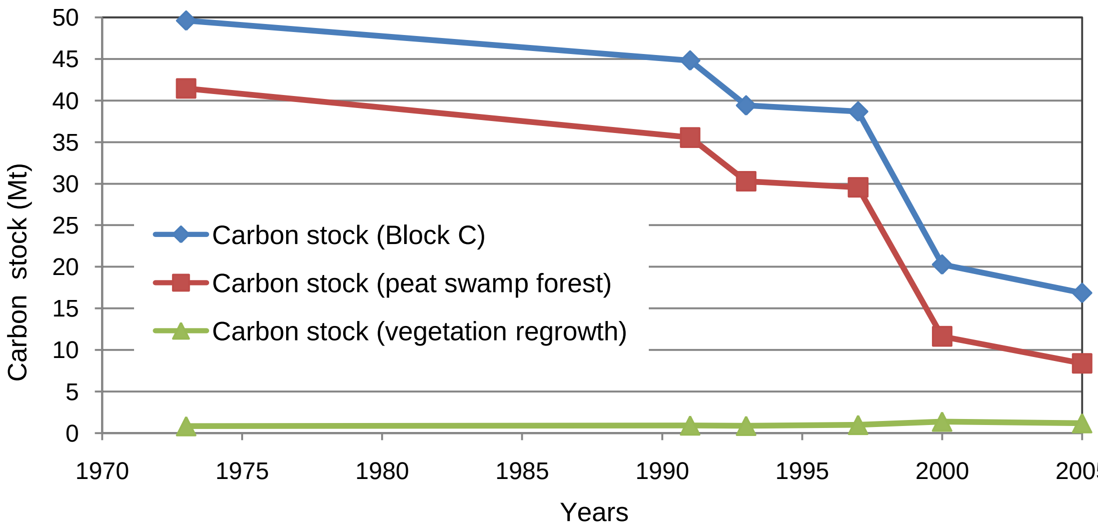


Figure 6.2 Aboveground biomass carbon stock for: Block C, peat swamp forest and vegetation regrowth for the period 1973-2005.

Between 1973 and 1997 (pre-fire), the AGB-carbon stored in regenerating vegetation fluctuated around 0.3-0.6 Mt (less than 1% of the entire AGB-carbon stock) (Table 6.3 and Figure 6.2). By 2000 the carbon stock in vegetation regrowth had increased to 2.7 Mt, (13% at the AGB-carbon stored in Block C), a consequence of the transformation of fire-affected forested areas into secondary vegetation. In 2005, the AGB-carbon stored in vegetation regrowth accounted for 0.9 Mt, which was more than 7% of the AGB-carbon in the study area (17 MtC).

6.1.2 *Carbon losses due to combustion of aboveground biomass*

Determining carbon losses from AGB combustion is a complex task because there are virtually no data available on fuel loads, fire affected areas, burn intensities, degree of fire damage and many other important parameters (WWF, 2008). In this study the AGB-carbon loss was estimated considering the land cover types affected by each fire (Chapter 4), type and amount of fuel loads derived from Table 6.1, and degree of fire demage. Van Nieuwstadt and Sheil (2005) studied fire mortality in tropical lowland forest in East Kalimantan, and showed that the mortality of trees <10cm DBH reached 74% and 80% for trees >10cm DBH during the 21 months after a fire. They also pointed out that tree mortality was lower shortly after the fire and increased with time. It was decided, therefore, to use the mortality rate measured by van Nieuwstadt and Sheil (2005) as a conservative and reliable value for estimating post-fire mortality in the study area, in the absence of data specific to PSF.

The total volume of carbon lost from burning AGB was derived separately for each fire and then summed-up to obtain the total carbon loss for the study area.

Total amount of burned AGB for each fire was derived as:

$$TB_{AGB} = \sum BA_n * AGB_n * \alpha_{ij} \tag{6.3}$$

Where:

BA_n = area of n land cover type affected by fire (ha)

$AGBn$ = total AGB for n land cover type (Mt ha^{-1}) (Table 6.1)

α_{ij} = degree of fire damage, *ij:* 74%-80%, respectively for trees >10cm DBH and <10cm DBH (van Nieuwstadt and Sheil, 2005)

Total carbon loss due to AGB combustion was calculated as:

$$CB_t = \sum_{k}^{k+l} TB_{AGB} * C_c \tag{6.4}$$

Where:

CB_t = total carbon loss due to AGB burning for the period 1973-2005 (Mt)

TB_{AGB} = total amount of burned AGB (Mt)

C_c = amount of AGB-carbon content was set at 50%

k = first fire

$k+l$ = next fire

Table 6.4 illustrates the residual and summed values of carbon losses from burning AGB in Block C; data are presented for both the total loss of carbon from biomass burning AGB and the loss from PSF over the period of investigation.

Fire events	**Total carbon loss (CB_t) Block C (Mt)**	**Carbon loss peat swamp forest (Mt)**
1974-1990	4.47 - 4.83	4.37 - 4.73
1991	3.88 - 4.19	3.66 - 3.96
1997	14.51 - 16.85	13.78 - 14.89
2002	2.09 - 2.76	1.92 - 2.07
2004	0.62 - 0.80	0.04 - 0.04
2005	0.46 - 0.49	0.01 - 0.02
SUM	**25.61 – 29.47**	**24.20 – 25.18**

Table 6.4 Carbon losses from burning AGB for the study area and PSF over period of investigation (1973-2005).

During the period 1973-2005, between 26-29 MtC was lost in the study area as a result of AGB combustion, half of which was lost as a result of the 1997 fire alone.

Figure 6.3 illustrates the proportion of carbon lost from PSF and non-PSF land cover types. Over the period of study, PSF contributed more significantly to AGB-carbon losses than non-PSF land cover type.

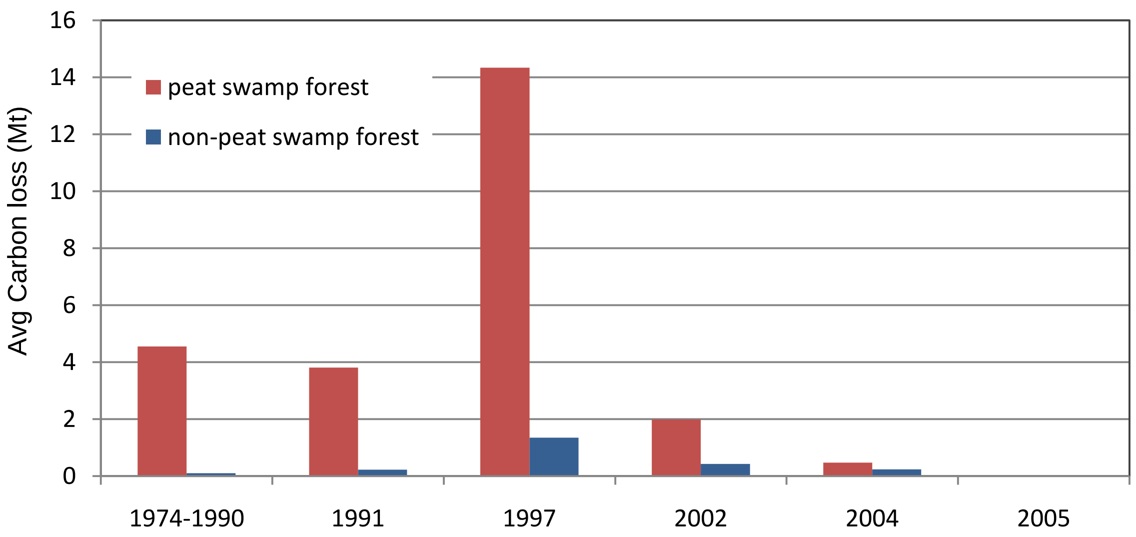


Figure 6.3 Carbon losses from burning AGB in PSF and non-PSF, based on the land cover types affected by fires.

For example in 2002, of the total burned area, only 16% was PSF and more than 70% regenerating vegetation (Section 4.5.1). However the amount of carbon released from burning PSF in 2002 was about five times greater than that from other land cover types (Figure 6.3).

6.2 Estimation of carbon loss from burning peat soil

Estimation of carbon loss from peat soil combustion is not a straight forward procedure, mainly due to the difficulties in obtaining accurate measures of the average depth of burned peat over a large area subjected to varying fire regimes.

This study focuses exclusively on the peat volume loss and subsequent peat carbon loss as a result of burning. This was achieved by looking at the dynamics of the fire regime in the study area, taking into account fire extent, frequency and interval between subsequent fires. Due to limited data on the depth of burn scars for single and subsequent fires, some conservative assumptions had to be made based on existing measurements, field observations and ecological knowledge (Table 6.5).

The following assumptions were made, which are also summarised in Table 6.5:

1) a depth of 0.51 ±0.05m was used for the average depth to which peat burned in the 1997 fire (the most severe fire over the period of investigation) (Page *et al.*, 2002);
2) a burn depth of 0.30 ±0.12m was used if fire occurred for the first time in either 2002, 2004 or 2005; this value was based on LIDAR measurements of burn scars conducted in the study area and adjacent peatlands after the 2006 fire (Siegert, unpublished data);
3) a burn depth of 0.22 ±0.12m was applied to locations that burnt twice (Limin, unpublished data);
4) in situations where a 3rd fire occurred within 1- or 2-years of a previous fire, a shallower burn depth of 0.10 ±0.05m was applied, on the basis of the reduced fuel load;
5) in situations where fire returned after more than a 7-year interval, the burn depth was assumed to be 0.30 ±0.12m, due to substantial re-accumulation of AGB fuel load (Siegert (unpublished data) and field observations);
6) in locations where fire occurred before the year 1997 (i.e. prior to MRP development), a burn depth of 0.20 ±0.05m was applied. This assumption was based on ecological knowledge of the properties of undrained peat (i.e. it is high moisture content, which would prevent a deep burn even during periods of drought).

GIS techniques were used to extract peat areas from a peat depth map for the study area (Euroconsult, 2008) and to intersect it with the fire frequency, fire extent and interval maps. The different values of peat depth burned away (Table 6.5) were assigned to the individual burn scars taking into account variation in fire regimes (fire frequency and intervals).

Figure 6.4 illustrates spatial pattern of average depth by which peat burnt down during the period 1997-2005 in Block C.

		First fire, period 1997-2005						
	Year	**1997**	**2002**	**2004**	**2005**	**1997**	**1997**	**2002**
Following fires	**1997**	0.51 ±0.05	0	0	0	0.51 ±0.05	0.51 ±0.05	0
	2002	0.22 ±0.12	0.30 ±0.12	0	0	0	0	0.30 ±0.12
	2004	0.10 ±0.05	0.22 ±0.12	0.30 ±0.12	0.0	0.30 ±0.12	0	0
	2005	0.10 ±0.05	0.10 ±0.05	0.22 ±0.12	0.30 ±0.12	0.22 ±0.12	0.30 ±0.12	0.22 ±0.12
	Fires 1973 - 1996: 0.20 ±0.05							

Table 6.5 Estimates of the average depth (in meters) by which peat burnt down in each fire; for burn scars affected by multiple fires the values of peat burnt for each individual fire were summed.

Finally, the peat volume loss was converted into a value for carbon loss by multiplying by a value for peat bulk density and carbon content; values of 0.09 g cm^{-3} and 56% were applied, respectively, based on the most recent assessment of these parameters (www.carbopeat.org, 2008).

Data for carbon losses arising from combustion of AGB and peat over the period of investigation are provided in Table 6.6. The data are further separated into two periods in order to compare the scale of fire-related losses during the pre-MRP era (1973-1996) and the post-MRP era (1997-2005).

These results show that between 53 and 83 Mt of peat carbon was lost over the period 1973-2005. From 1997 onwards, fire frequency and extent increased, affecting around 66% of the peatland in the study area; in contrast only 23% was affected by fire during the first two decades (1973-1996). The estimate of around 213,000ha of peat burnt during the period 1997-2005 provides a carbon loss value of 48-74 Mt, whilst during the period 1973-1996 losses were much smaller, at between 6-9 Mt, with a total burned area of 74,000ha. Figure 6.5 shows the extent of carbon loss from burning peat over the period 1997-2005 in Block C.

The total peat-derived carbon loss of 53-83 Mt represents a reduction of between 11 and 18% of the peatland carbon stored in Block C.

Study area	**1973-1996**	**1997-2005**	**1973-2005**
Peatland area (ha)	324,667		
Peat volume (Mt)	9,404		
Carbon store (Mt)	473.99		
Area of fire damaged peatlands (ha)	74,010	213,022	287,032
Percentage of peatland area damaged	22.8%	65.6%	88.4%
Peat volume loss (Mt)	111.0 – 185.0	939.7 – 1436.2	1050.7 – 1621.2
Peat carbon loss (Mt)	5.60 – 9.33	47.93 – 73.91	53.52 – 83.24
Percentage of carbon loss from storage	1.2 – 2.0%	10.1 – 15.6%	11.3 – 17.6%
AGB-carbon loss (Mt)	8.35 – 9.02	17.27 – 20.45	25.61 – 29.47
Total carbon loss (Mt)	**13.95 – 18.35**	**65.20 – 94.36**	**79.13 – 112.71**

Table 6.6 Effect of multiple fires on peat carbon loss and AGB-carbon loss for the period 1973-2005, with two sub-periods, pre-MRP (1973-1996) and post-MRP (1997-2005).

The combined amount of carbon released from burning both peat and surface vegetation in the study area was estimated at 14-18 MtC for the period 1973-1996, but at four times greater than this (65-94 Mt) for the period 1997-2005.

The total amount of carbon lost from both peat and AGB burning over the entire period of investigation (1973-2005) was estimated to be in the range 79-113 Mt.

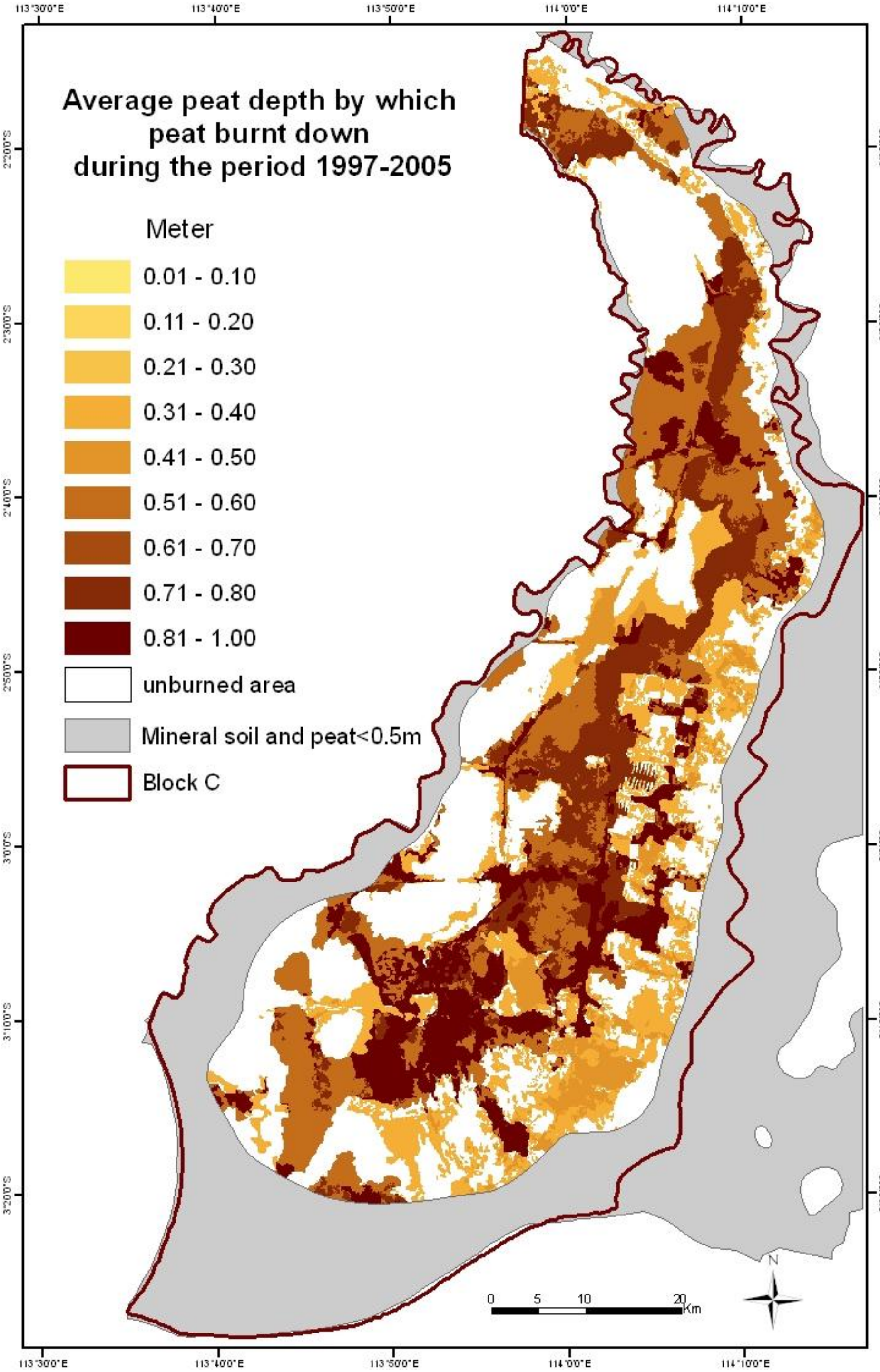


Figure 6.4 Average depth (in meter) by which peat burnt down during the period 1997-2005, it was assigned taking into account variation in fire regime (fire frequency, fire extent and interval between subsequent fires).

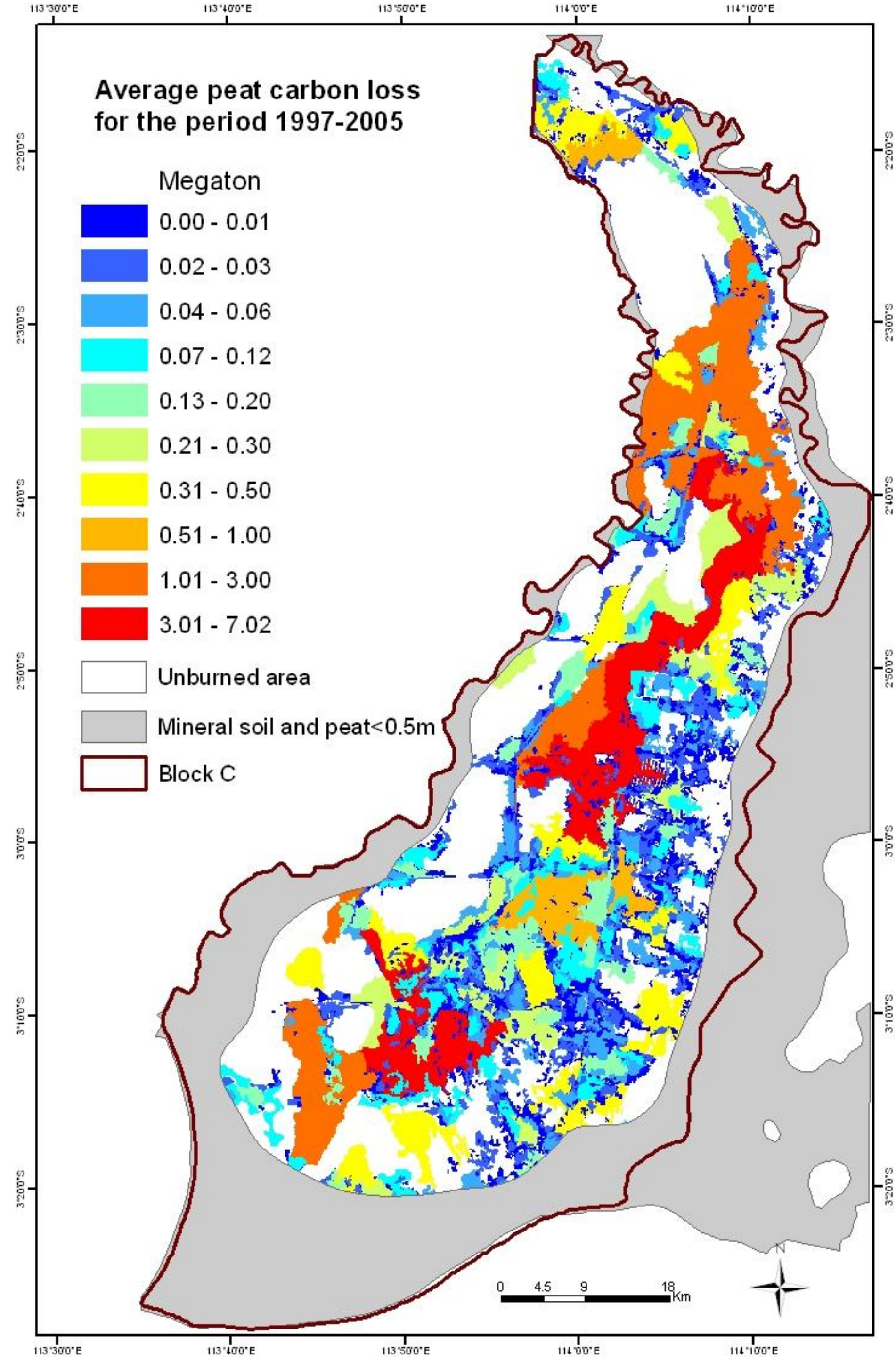


Figure 6.5 Extent of carbon loss from burning peat over the period 1997-2005 in Block C.

6.3 Summary

Investigation of carbon stocks and lossess have shown that:

- Due to a lack of appropriate methodology, this study proposed a conservative approach to estimating the carbon losses from burning peat, taking into account a number of fire variables (i.e. fire extent, fire frequency, return interval, amount of available fuel, and fire intensity).

- The AGB-carbon stock in the study area declined by 66% from 50 Mt to 17 Mt over the period 1973-2005; most of this change was in the PSF carbon stock, which declined by 80% from 41 Mt to 8 Mt, largely as a result of recurrent and extensive fires.

- By contrast, post-fire secondary vegetation, which has a much lower carbon density, made a small contribution to fire-related AGB-carbon losses. Between 1973 and 2000 the AGB-carbon stored in regenerating vegetation increased from 0.3 to 2.7 Mt, and decreased to 0.9 Mt in 2005, as a consequencis of the 2005 fire.

- During the last three decades (1973-2005), between 79 and 113 Mt of carbon were lost from burning AGB and peat. Most of these losses occurred during the period following development of the MRP. Carbon losses from burning peat were around two and a half times greater (53-83 Mt) than the carbon losses from burning AGB (26-29 Mt).

- The loss of peat carbon represents a reduction of between 11-18% of the total peat carbon store in the study area. The largest proportion of these losses took place between 1997 and 2005, with an estimated 65-94 Mt (~83% of total) of carbon lost over this period.

7 DISCUSSION

The aim of this research was to demonstrate the ecological and climatic impacts of recently enhanced fire regimes in tropical peatlands through measures of land cover, vegetation and carbon dynamics. This study has clearly demonstrated the importance of fire as a main driver of land cover and vegetation dynamics within the study region. This chapter discusses the results presented in Chapters 4, 5 and 6 in the context of relevant literature and the thesis research objectives. The final Section 7.6 provides recommendations for further research.

The research objectives were as follow:

1. **To quantify the land cover and fire regime dynamics over the period 1973-2005 in the study area (discussed in section 7.1, 7.2 and 7.3).**

2. **To examine the effects of fire frequency and burn severity on the dynamics of post-fire vegetation recovery by studying four restoration indicators: aboveground biomass, vegetation ground cover, canopy cover and diversity of plant species (discussed in section 7.4).**

3. **To quantify carbon losses from burning aboveground (vegetation) and belowground (peat) biomass (discussed in section 7.5).**

7.1 Trends in land cover dynamics

In Southeast Asia, tropical PSFs have been put under tremendous pressure from unsustainable land and forest management (i.e. logging, land conversion, drainage and settlements). During the entire period of investigation (1973-2005), the land cover changes within the study area have revealed a reduction in forest cover towards non-woody vegetation. Every year, around 2.2% of the forest, equivalent to around 7,000ha yr^{-1}, was converted to some other land cover (Table 7.1). A similar deforestation rate was reported by Hooijer *et al*. (2006) for lowland peatland forest in Central Kalimantan over the period 1985-2000. Of the forest types in the study area, PSF was subjected to the highest level of deforestation with a reduction of 2.5% yr^{-1} (6,687ha yr^{-1}). Over the 32-year study, 80% of the PSF was lost; the deforestation rate increased steeply after 1997, when the PSF extent dropped from 43% to 17% (Table 7.2). Disturbed and drained PSF is a particularly fire-prone type of forest, compared to other lowland forests, due to combustion of the peat substrate (Yeager *et al.,* 2003, Nishimua *et al.,* 2007, Page *et al.,* 2009). Fires in PSF may kill trees directly, but many are felled by loss of peat and hence stability around shallow root systems (Nishimua *et al.*, 2007). The remaining non-PSF forest types (i.e. freshwater swamp, heath, and mangrove forests) were also reduced by 22% compared to the initial year (1973).

LAND COVER TYPE	**1973-1996**		**1997-2005**		**1973-2005**	
	%/yr	**ha/yr**	**%/yr**	**ha/yr**	**%/yr**	**ha/yr**
Mixed peat swamp forest	-1.2	-3,165	-8.7	-15,484	-2.5	-6,245
Low pole peat swamp forest	0.0	0	-12.5	-1,767	-3.1	-442
Heath forest	1.3	142	-0.5	-70	0.8	89
Mangrove forest	0.0	0	-4.4	-406	-1.1	-101
Freshwater swamp forest	-0.4	-171	-3.0	-1,042	-1.0	-389
Fragmented and/or degraded mangrove	0.0	0	2.0	187	0.5	47
Secondary forest	0.0	192	37.3	1,724	0.0	575
Non-woody vegetation	5.5	2,024	17.4	14,803	14.3	5,219
Sedge swamp	0.0	0	-0.7	-69	-0.2	-17
Mosaic of trees and non-woody vegetation	6.3	635	12.9	3,257	12.9	1,290
Recently burned area	-3.6	-1,127	-12.5	-506	-3.1	-972
Cultivated land	8.9	1,472	0.2	104	6.8	1,130
Blackwater lake	0.0	0	-12.5	-521	-3.1	-130
Peat swamp forest (mixed- and low-PSF)	**-1.2**	-3,165	**-9.0**	-17,251	**-2.5**	-6,687
Non-peat swamp forest	**-0.1**	-29	**-2.6**	-1,518	**-0.7**	-401
Forested area	**-1.0**	-3,194	**-7.5**	-18,768	**-2.2**	-7,088

Table 7.1 Land cover change over a 32-year period 1973-2005; negative values indicate loss; 1973-1996 refers to the pre-MRP period, whereas 1997-2005 values are for the post-MRP period.

The trend in land cover dynamics over the period 1973-2005 has a non-linear character; two explicit periods can be delineated with the year 1997 marking the trend break point. Over the period 1973–1996, there were relatively minor forest cover alterations (Table 7.2). By 1991, almost half of the deforested land had been converted into cultivated land owing to the development of transmigration schemes. The residual deforested land (i.e. land not converted to agriculture) was not under any form of economic land use, as indicated by the increase in the area of non-woody vegetation, at a rate of 5.5% yr^{-1} (2,024ha yr^{-1}), and mosaic of trees and non-woody vegetation, at a rate of 6.3% yr^{-1} (635ha yr^{-1}).

FOREST TYPE	PRIMARY FOREST COVER (%) IN BLOCK C							
	1973	**1991**	**1993**	**1997**	**2000**	**2002**	**2004**	**2005**
Mixed peat swamp forest	56.5	48.1	40.6	39.6	16.0	12.7	12.0	11.9
Low pole peat swamp forest	3.1	3.1	3.1	3.1	0.7	0.3	0.1	0.0
Total peat swamp forest	**59.6**	**51.2**	**43.7**	**42.7**	**16.7**	**13.0**	**12.0**	**11.9**
Total peat swamp forest (ha)	**267733**	**229976**	**196389**	**191761**	**74962**	**58424**	**53665**	**53632**
PSF loss between periods		**8.4**	**7.5**	**1.0**	**26.0**	**3.2**	**1.0**	**0.01**
PSF loss in comparison to initial year 1973		**14.1**	**26.6**	**28.4**	**72.6**	**78.2**	**79.96**	**80.0**
Heath forest	2.4	3.2	3.2	3.2	3.2	N/A	N/A	3.1
Freshwater swamp forest	8.7	8.4	7.9	7.8	6.0	N/A	N/A	5.9
Mangrove forest	2.1	2.1	2.1	2.1	2.1	N/A	N/A	1.3
Total non-PSF	**13.2**	**13.7**	**13.2**	**13.1**	**11.3**	N/A	N/A	**10.3**
Total forest cover	**72.9**	**65.0**	**56.9**	**55.8**	**28.0**	N/A	N/A	**22.3**

N/A lack of data due to cloud cover

Table 7.2 Changes in primary forest cover (%) in Block C during the period of investigation; the 1991 and 1997 data were obtained from images dating to before fires occurred in these years, thus fire-related forest losses are reflected in the forest cover figures for 1993 and 2000, respectively.

A rapid acceleration of land cover dynamics, with a significant increase in the rate of deforestation, occurred during 1997-2005. Over this period, forest cover declined at a rate of 7.5% yr^{-1}, with PSF experiencing the highest deforestation rate of 9% yr^{-1} (~17,000ha yr^{-1}). For the first time, frequent fires also affected low-PSF and the blackwater lake located on the top of the peat dome in the southern part of the study area, resulting in the complete loss of these two land cover classes. In a pre-disturbance condition, low-PSF and the lake would have occupied the wettest parts of the peatland dome, thus the occurrence of fire in these locations is indicative of the extent of drainage caused by the MRP canal system. The remaining non-PSF forest types also declined at an overall rate of 2.6% yr^{-1} over this period (mainly mangrove and freshwater swamp forest). It is important to note that most of the deforested land during 1997-2005 was converted to either non-woody vegetation or a mosaic of trees and non-woody vegetation, rather than cultivated land. A as a result, the area of both these classes doubled at a rate of 17% yr^{-1} (14,800ha yr^{-1}) and 13% yr^{-1} (3,257ha yr^{-1}) respectively. Furthermore, the area of secondary forest also underwent a rapid increase, at a rate of 37% yr^{-1}, accounting for 1,724ha yr^{-1}; this was mostly as a result of transformation of the mosaic of trees and non-woody vegetation class into more advanced stages of woody regrowth.

7.2 Fire regime dynamics and causal factors

In recent years, fires in tropical forests have become more frequent and widespread resulting in an increased need to evaluate fire impacts at a landscape scale. As was shown in Section 4.5.1, within the first two decades of this study (i.e. the period 1973-1996) fires affected a relatively small area; in total 23% of the study area. This situation changed markedly during the last decade (1997-2005), largely as a result of various human activities, discussed later in this chapter. The widespread, intensive fires of 1997 affected about 34% of the land within Block C (i.e. 10% more than the total burned area for the period 1973-1996). Five years later, in 2002, extensive fires returned, affecting 22% of the study area; this was again equal to the total area burnt during the period

1973-1996. In 2004 and 2005 further fires affected 14% and 12% of the study area respectively.

These widespread and repeated fires took place in a humid tropical peatland environment, where the water table is naturally close to or above the peat surface for much of the year, and where fire propagation is controlled (i.e. limited) by moisture (Siegert and Ruecker, 2000, Wösten *et al.*, 2008). The high humidity of intact PSF acts as a fire suppression barrier, protecting the forest vegetation and the peat surface from burning. This was confirmed by this study (Section 4.5), which demonstrated that prior to the MRP development most burn scars were located along forest edges (i.e. in disturbed forest) and usually in close proximity to human settlements which provided a source of ignition. Intact forests remained unaffected by fire, even during the extended droughts associated with the El Niño events of 1973, 1982 and 1991. It has been suggested that tropical PSF may be less tolerant to drought than other forest types due to the shallow nature of the tree rooting-systems, which are adapted to absorb nutrients from the relatively nutrient-enriched top layer of peat soil (Nishimua and Suzuki, 2001). However, other studies have shown that undisturbed PSF is resilient to fire, even during extreme drought conditions (Page *et al*., 2002). This study supports this latter finding.

7.2.1 *Anthropogenic pressure on the ecosystem*

Several studies have highlighted that the majority of fire events in tropical peatland forests are directly or indirectly associated with human activities (Langner *et al.*, 2007, Langner and Siegert, 2009, Siegert *et al.*, 2001). Historically, indigenous people have used fire as the way to clear and maintain their land, keeping it free of woody growth (Rieley and Page, 2005, MacKinnon, 1996). In the study area, this practice was used to triple the area of cultivated land between 1973 and 1991. This impressive 'achievement' was only possible owing to settlement and small-holder land clearance under the Transmigration Programme, supported by the Indonesian government. The aim of this initiative was to improve the lives of millions of poor and landless people from over-crowded Indonesian islands by offering land and jobs in less populated provinces

(Fearnside, 1997). Transmigrants settled in the peatland areas of Central Kalimantan in the early 1980s experienced various obstacles, including nutrient-poor, acidic peat and potential acid-sulphate soils, plant diseases, weed invasion, as well as a lack of knowledge of crop cultivation on peat (Levang, 1984). Traditionally, the indigenous Dayak people of Borneo cultivated only shallow peats not more than 0.5m thick underlain with nutrient-enriched clay; thicker peat was considered unsustainable for agriculture (Rieley and Page, 2005, MacKinnon, 1996, Sulistiyanto, 2004). In spite of the long experience of local communities, however, the Indonesian government agreed to establish transmigrant settlements on deeper peat. Analysis of GIS data layers in this study showed that only 5% of forest clearance for new settlements in the study area occurred on peat less than 0.5m deep, whereas 52% was in locations where peat thickness ranged from 0.5 to 2m and 43% was on peat thicker than 2m.

A number of natural obstacles forced the transmigrants to acquire more farmland or to look for alternative sources of income such as labour-intensive illegal logging, although this was less widespread prior to the fall of the Suharto government in 1998. During the period 1973-1991, many legal logging concessions were established in the PSF in the study area. By 1991, a network of log extraction tracks covered almost the entire area of mixed-PSF, but excluding the less commercially valuable low-PSF. Over-logging, combined with some illegal activities particularly in the vicinity of settlements, degraded the forest condition making logged forests more vulnerable to future fires. Several other studies have indicated a susceptibility of previously logged forest to fires (Woods, 1989, Cochrane *et al.*, 1999, Asner *et al.*, 2004, Siegert *et al.*, 2001). As a result, between 1991 and 1993, half of the heavily logged PSF was burnt.

The most extensive forest loss in the study area occurred following the implementation of the MRP (1996-1998). The intensive drainage infrastructure, consisting of numerous canals, had an impact on peat flammability through drainage, particularly during the dry season when water tables fell to 1m or more (Wösten *et al.*, 2008). This greatly impaired the peatland hydrological system contributing to the desiccation of the peat, thus increasing the risk of combustion. It did not take long before the negative impacts of over-drainage of the peat became apparent, with widespread fires during 1997 claiming one-third of the study area (~150,000ha), mainly affecting previously disturbed (logged) and drained PSF. Drainage lowers the peat water table, exposing a greater volume of dry peat to combustion; this effect is greatest closest to the drainage feature. In addition, the canals allowed easier access for people into previously remote areas of peatland and their activities provided ignition sources (cigarette ends, cooking fires etc.). This study showed that areas subjected to multiple fires (i.e. three or four fires) over the 8-year post-MRP period were all located within close proximity to canals and the extent of burn scars declined with distance away from these features (Section 4.5.3.).

7.2.2 *El Niño phase as a natural driver of fire*

Fires are not a totally new phenomenon in tropical forests and neither are they entirely dependent upon human activity. Lightning strikes have been noted as a natural source of ignition for tropical forest fires, although these events are usually associated with heavy rainfall and thus rarely lead to forest fires (Tutin *et al.*, 1996). The remote location of the extensive burn scars (24,700ha) recorded on the 1973 image in the southern part of the study area may suggest a natural source of ignition since the fire occurred some considerable distance from habitation. Ignition would be more likely to occur if fire had either been preceded by a severe period of drought or if lightning struck a fire prone surface. In the case of this particular fire, both factors could have been important. Firstly, 1973 was an El Niño year, and the fire location was possibly covered by fire prone vegetation. This hypothesis was based on the structure of the regenerating vegetation that, even three decades after burning, showed no progression to forest, suggesting that the area had been burnt more than once prior to the 1973 fire. Lightning-ignited fires have been noted in the past in PSF (Brünig, 1973). In addition, a sharp and explicit border excluding the burn scar from the adjacent PSF confirmed the resilience of the undisturbed forest to fire (i.e. the 1973 fire) was confined to an area of secondary vegetation. Other fires within the region in the post-1973 period appear to have been strongly linked to human access, thus mostly occurred in close distance from the settlements or human access points.

In Section 4.6, the relationship between fire incidents and ENSO-related droughts was clearly illustrated since the most extensive fires that have occurred in the study area coincided with ENSO events (i.e. 1973, 1991, 1997 and 2002). Several previous studies have indicated a strong relationship between ENSO events and fire occurrence in the Southeast Asian region (Fuller and Murphy, 2006), including the 1972/73 fires (Wyrtki, 1975, Aiken, 2004), the 1982/83 fires, called the Great Fire of Borneo (Fuller and Murphy, 2006, Aiken, 2004, Malingreau *et al.*, 1985, Goldammer, 1997), the 1991/92 fire (Kasperson *et al.*, 1995), the 1997/98 fire, which was one of the severest fires in the 20th century (Tacconi, 2006, Siegert *et al.*, 2001, Fuller and Murphy, 2006) and the 2002 and 2006 fires (Langner *et al.*, 2007, Langner and Siegert, 2009). There is also evidence to suggest that the incidence of fire reported by this study has become uncoupled from ENSO drought events and, that in this highly degraded landscape, the risk of fire incidence is increasing even in non-El Niño years (i.e. 2004 and 2005). The phenomenon of fire incidents decoupled from the El Niño phase was also reported by Gullison *et al.* (2007) who studied fire incidents in the Amazon Basin and Southeast Asia over the last 5 years. The increase in fire incident in the study area, even in the absence of ENSO-related drought, can be explained by the high fire-risk posed by degraded peatland but may also be linked to the decreasing trend in the rainfall pattern observed within the region over the period 1976-2007 (Euroconsult, 2008). In addition, the projection of future climate by Li *et al.* (2007) forecasted further reduction of rainfall during dry seasons in southern Borneo where most peatlands are located, which may lead to further enhancement of ENSO intensity.

Analysis of the relationship between fire extent and magnitude of ENSO-droughts with respect to variation in the MEI index (Section 4.6) showed that during the period 1973-1996 (pre-MRP era), fires affected relatively small areas, even though the value of the MEI indicated El Niño drought conditions. By contrast, fires claimed much larger areas of land during the post-MRP period (1997-2005) when even in the ‘normal’ dry season (non-El Niño phase) large areas of land were burnt (e.g. the years 2004 and 2005). The regression analysis presented in Figure 4.15 indicates a strong linear correlation between the ENSO-MEI index and fire extent over the period 1997-2005 (r^2=0.93), which raises the possible use of ENSO-MEI indices to predict fire extent in degraded tropical peatland. Successful predictive capacity would be helpful in implementing fire hazard mitigation strategies.

7.2.3 *Fire frequency and fuel load*

During the period 1997-2005 more than a half of the study area (~243,000ha) was burnt at least once over four intensive fire events. Of this total burned area, about 45% (108,500ha) was subjected to multiple fires, with 37% burnt twice and 8% burnt three or more times (Table 4.7). The increasing trend in fire occurrence has also been observed on a larger scale by Langner *et al.* (2009) who demonstrated (using hotspot data) that 6.1% (4.5 Mha) of forest in Borneo had been affected by fire more than once during the period 1997-2006. These results may indicate the existence of a positive feedback between fire, future fire susceptibility, fuel loading, fire intensity and burn severity as pointed out by Cochrane (1999) from studies in Amazonia.

The frequency of fires observed in the study area indicates the availability of sufficient fuels to maintain repetitive fires. The term fire fuel refers to many factors influencing both fuel type and quality of available fuel (i.e. fuel composition, moisture, continuity, distribution, spatial arrangement) whilst the amount of flammable material (dry biomass) ready to burn is referred to fuel load (Cochrane, 2009). All of these factors determine fire regimes, types, and spreading capacity.

In principle, pre-mature secondary forest or non-woody vegetation both catch fire at lower temperatures than large diameter trees (i.e. pristine forest) because less energy will be consumed to raise any particular pieces to the ignition temperature. Moreover, smaller trees or non-woody plants easily catch fire and burn quicker than larger-woody pieces due to the proportion of the mass that is exposed to oxygen. Hence, if there is a sufficient amount of flammable fuel available, the fire will burn more intensively and spread faster than with a small amount. In other words, the greater the availability of high quality fuel, the more intensive the fire will be. Figure 7.1 illustrates the fuel load alteration according to the land cover type, which determines fire intensity and influences peat combustion.

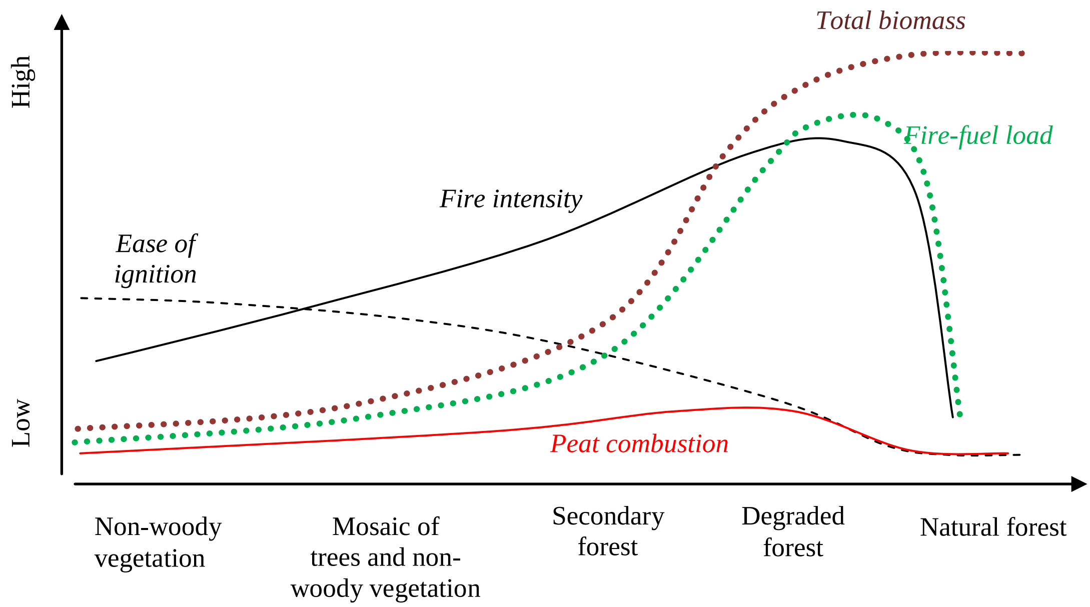


Figure 7.1 Schematic relationship between total biomass, fire-fuel load, fire intensity ignition and peat combustion with respect to land cover type.

Over-drained PSF, with around 313t ha^{-1} of dry woody-AGB, was a major supplier of fire fuel for the extensive 1997 fires, which explains why this fire event has been recognised by this study and confirmed by other authors (Tacconi, 2006, Siegert *et al.*, 2001, Fuller and Murphy, 2006) as the most intensive and extensive fire in the region. Furthermore, once-burned forest can become more fire prone owing to the high amount of dry and flammable materials left over from the previous fire and the lower humidity (Uhl and Kauffman, 1990, Siegert *et al.*, 2001, Cochrane and Schulze, 1999). Fire modifies the forest microclimate, causing greater sunlight penetration due to canopy openness that accelerates drying out of the forest floor and results in an increase of the potential fuel stock (Nepstad *et al.*, 2004, Nepstad *et al.*, 1999). A study by van Nieuwstadt (2005) in East Kalimantan demonstrated the long-lasting effect of fire on tree mortality that reached 74% for trees >10cm DBH and 80% for trees <10cm DBH, 21 months after widespread fires in 1997/98.

After a second fire, most of the aboveground fuel is eliminated and more time is required before the vegetation can re-build sufficient amount of biomass and hence fuel load to sustain a new intensive fire. Analysis of the fire-return interval (Section 4.5.2), demonstrated that locations with a longer fire-return interval (more than 5-years) were more likely to be re-burnt than those with a shorter period (less than 5-years) between subsequent fires. This is perhaps indicative of the time interval required for sufficient AGB fuel to accumulate to enable fire to occur and propagate. As was demonstrated in Section 5.2, two severe fires occurring within a 5-year interval reduce drastically the process of tree regeneration, and as a result the vegetation becomes dominated exclusively by ferns. The fires that occurred during the dry seasons of 2004 and 2005, which followed the extensive fires of 1997 and 2002, consumed mainly non-woody vegetation, which comprised 72% and 79% of the total burned areas in each year, respectively. Non-woody, highly flammable fern can easily catch fire (Figure 7.1); however fire propagation is slow and restricted by both low fuel quality and load. As a consequence, the four- or five-times burned areas (dominated by ferns) were mainly clustered close to sources of ignition, at distances of 1000-1500m from HAPs (mainly canals) (Figure 4.15). This indicates that the probability of burning declined during the third fire owing to low fuel availability. This negative fire feedback with an annual burning regime was also observed by Balch *et al.* (2008) in a transitional forest in Southeastern Amazonia.

The shift in the fuel type after 1997, described in Section 4.5.1 (Figure 4.9), from woody-AGB (mainly PSF) towards a largely non-woody-AGB, dominated by regenerating vegetation determined the subsequent fire regime. The high calorific woody-AGB of the PSF was replaced by a lower calorific fuel dominated by fern, with greatly reduced dry non-woody-AGB values (6 to 10t ha^{-1}). This alteration influences fire intensity and consequently the type of combustion.

7.2.4 *Peat combustion and loss*

Discussion of fire fuels in tropical peatland ecosystems would not be complete without a consideration of the important role played by the available sub-surface fuel. The sub-surface fuel includes the peat plus vegetation root mats, duff and partly decomposed woody debris. Saturated peat does not burn thus the water depth determines how much peat is available for burning. Hooijer (in Euroconsult, 2008) compared fire extent data (derived from this study) with long-term modelled water depth data in order to determine the critical water-depth threshold for fire establishment. He demonstrated a strong correlation between fire occurrence and a water table depth of 0.8m; the critical level for fire risk occurred when the water table dropped to or below 1m (Figure 7.2).

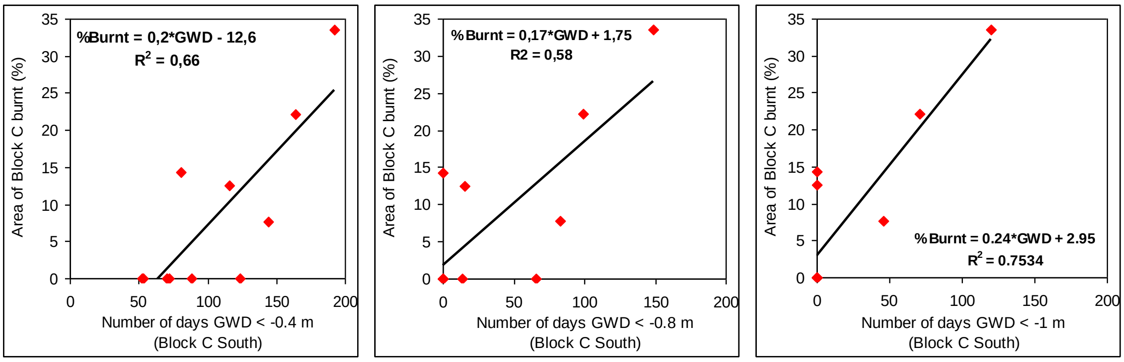


Figure 7.2 Water depth threshold exceedance in relation to annual burn extent in Block C (using fire scar data from this study) (Euroconsult, 2008).

The strong relationship between fire risk and drainage was also confirmed by Wosten *et al*. (2008). The network of man-made canals in the MRP area has accelerated the dryness and increased the flammability of both AGB and peat (Hooijer *et al.*, 2006).

The flammability of peat is also driven by changes in vegetation cover. Closed-canopy PSF acts as a moisture ‘reservoir’; the high humidity under the canopy reduces evaporative water losses and keeps the peat surface moist. Once the water balance is altered by forest disturbance, both the vegetation and the peat surface dry out, adding to the total amount of fuel available to burn. Furthermore, the more high quality surface fuel that is available, the more peat can be burnt away enhancing fire intensity (Figure 7.1). Less intensive fires will support only surface fire propagation, whereas more intensive fires will penetrate deeper into the peat layer supporting ground combustion (Figure 7.3).

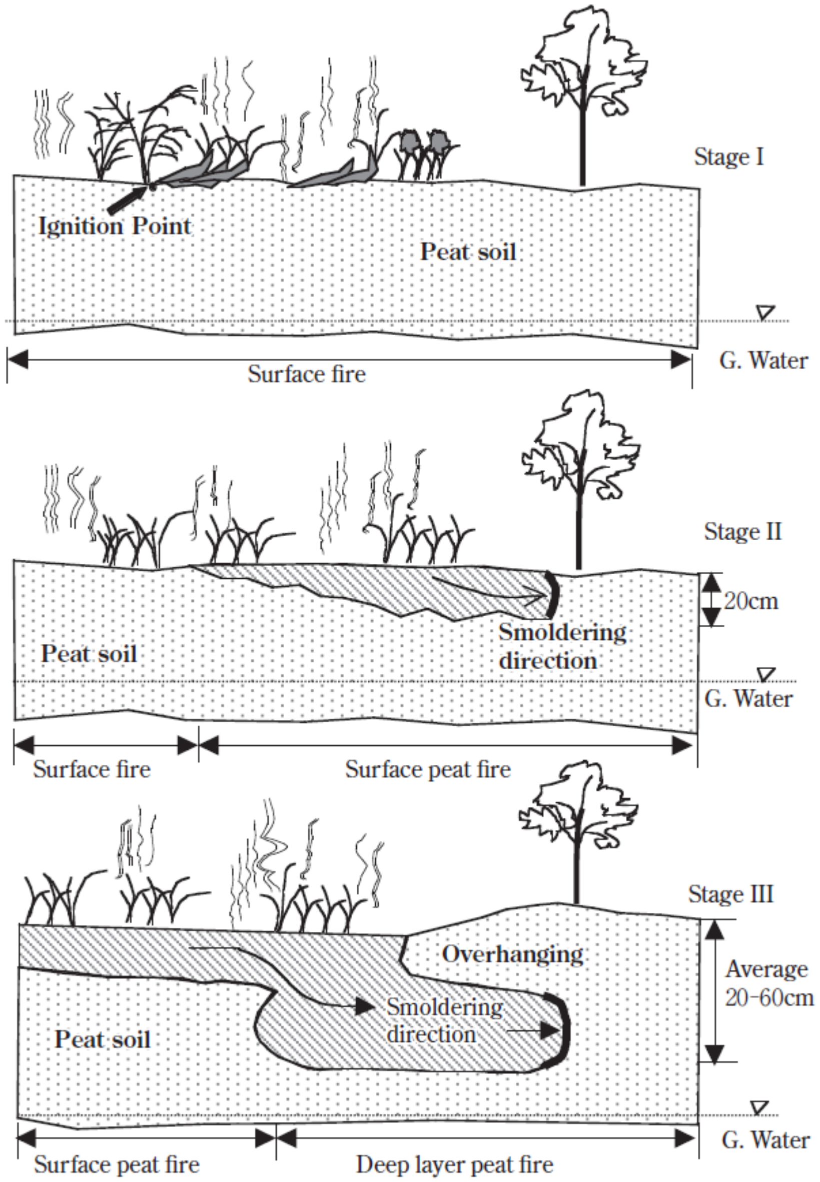


Figure 7.3 Fire development in tropical peatland fires, stage I: surface fire, stage II: surface peat fire burns at <20cm depth; stage III: deep peat fire burns at >20cm depth (Usup et al., 2004).

Ground fires penetrating deep into the peat are extremely hazardous, difficult to extinguish, and result in heavy mortality of the vegetation and a large amount of smouldering emissions (Usup *et al.*, 2004).

The quantity of sub-surface fuel can greatly exceed the surface fuel in the peat swamp ecosystem (Section 6.2). However, estimating the amount of sub-surface fuel combusted during each fire is much more difficult than quantification of vegetation fuel (Page *et al.*, 2009). Thus, there is a considerable uncertainty surrounding quantification of peat volume available for combustion (Page *et al.*, 2002, Rieley *et al.*, 2008, Spessa *et al.*, 2009).

7.3 Assessment of burn severity in tropical peatland

This study has presented a novel approach to quantifying relative magnitude of burn severity through characterisation of the post-fire condition of vegetation. Section 3.2.3 presented an overview of several studies undertaken in the extra-tropics that have used various remote sensing techniques to map the magnitude of burn severity; however, to date these studies have not been extended to the tropics, where in recent decades, some of the largest fires have occurred (Cochrane, 2003, Cochrane, 2009). Roy *et al.* (2006) pointed out difficulties in designing one universal index that could perform accurately across ecosystems in various climatic zones. Therefore, different approaches need to be tested over a range of tropical ecosystems where vegetation structure is far more complicated than in the temperate zone and where the landscape is subject to intensive and repeated fires. Investigation of burn severity in tropical forests is crucial for several reasons: a) to improve understanding of the uncertainties associated with fire-related fluxes, b) to understand the process of post-fire vegetation recovery, and c) to assess the type, quality and amount of fuel available for future fires (fire risk).

Burn severity is typically assessed immediately after a fire and thus refers to short-term effects of fire on the environment (Kasischke and Hoy, 2007). This is normally done by synergy of spectral data (particularly dNBR) and the ground-based Composite Burn

Index. The CBI was designed to make a quick assessment of the magnitude of burn severity in coniferous forests after fire (Section 3.2.3). In the tropics, however, assessment of fire effects shortly after burning can be difficult, if not impossible, owing to the weather conditions, since the end of burning season always coincides with the start of the wet season when the acquisition of optical data is confounded by cloud cover. In addition, in the time frame of this study, the assessment of fire effects shortly after burning was not feasible; therefore the main focus was to investigate whether the recovering vegetation could be used to assess burn severity of the last fire.

As was presented in Section 5.3.1, this study has established a strong correlation between spectral data derived from pre- or/and post-fire images and sets of vegetation variables collected four years after the last fire. The high correlation demonstrates the long-lasting effect of multiple fires on vegetation structure and recovery, which supports the hypothesis that the variation in the characteristics of post-fire vegetation regrowth observed in the field could be explained by differences in the magnitude of burn severity. Secondly, it also confirms that some of the vegetation variables derived from *in situ* measurements collected four years after a fire event can be proposed as indicators characterising the magnitude of burn severity in tropical peatlands subjected to multiple fires. Several other studies have also demonstrated the relationship between burn/fire severity and ecosystem response such as recovery of belowground systems in pine forest in USA (Neary *et al.* 1999), forest biomass regeneration in Canadian boreal forests (Lecomte *et al.,* 2006) and species richness in Mediterranean shrublands (Keeley *et al,* 2005). The recent study by Keeley *et al.* (2008) showed that fire severity (expressed by dNBR) was not correlated with vegetation recovery in the chaparral shrublands. By contrast, this study undertaken in tropical forest, demonstrated a strong correlation between dNBR and recovering vegetation.

Amongst the range of spectral single/bi-temporal indices and spectral endmembers tested in this study, dNBR showed the strongest correlation with vegetation variables, in particularly with total woody-AGB and tree BA, tree biomass and density of all trees. Several studies conducted in non-tropical countries had previously recommended the use of dNBR for the assessment of burn severity (using CBI) (Key and Benson, 2006, De Santis and Chuvieco, 2006, Escuin *et al.*, 2008, Epting *et al.*, 2005). This study has also shown that NDWI and dNDWI correlate well with the vegetation variables. The NDWI is known as being particularly sensitive to water content (De Alwis *et al.*, 2007, Xiao *et al.*, 2002), and it performed very well in areas subjected to two intensive fires, where there was a large reduction in canopy cover. Vegetation and surface moisture content is an important factor determining the spectral characteristic of the surface. To date, the NDWI has not been considered in analysis of burn severity. However, as this study has demonstrated, it can be used as an appropriate index for delineating burn severity, at least in tropical peatlands subject to repeated fires.

The ANOVA analysis presented in Section 5.3.2 demonstrated that both NBR and dNBR differentiate classes of regrowth better than any of the other spectral indices. The Tukey multiple comparisons of mean test showed that NBR, dNBR, and NDWI can delineate each of the four classes of vegetation at a confidence level $p<0.02$ (Table 5.10). The poorest distinction between classes was obtained using NDVI, dNDVI and NPV fraction. Separation of the SF from the MF classes can be confidently performed using NPV and GV fractions as well as NDWI, NBR and dNBR, whilst three classes of regrowth following multiple fires can be distinguished from each other using NBR, dNBR, and NDWI. These indices that incorporate IR and SWIR wavelengths (i.e. NBR and NDWI) performed better then the NDVI (based on R and IR) due to particular sensitivity of SWIR spectrum to vegetation and ground moisture. Regenerated vegetation after two subsequent fires is mainly characterised by combination of bare ground with mosaic of trees and non-woody vegetation, thus the NDVI, that is particularly sensitive to greenness, did not perform very well.

The threshold values of dNBR derived from this study for three classes of burn severity plus the unburned class showed to be comparable with the threshold values obtained in non-tropical ecosystems (Table 7.3).

Ecosystems	Study	Range of dNBR threshold for three burn severity levels			
		Unchanged	Low	Moderate	High
Mediterranean	(Miller and Thode, 2007)	<41	41-176	177-366	>=367
Alaska (forest)	(Epting *et al.*, 2005)	<90	90-275	275-680	>=680
Boreal forest (Montana)	(Key and Benson, 2006)	<99	100-269	270-659	>=660
Tropical peatland (Central Kalimantan)	This study	<10	11-179	180-550	>=550

Note: values are scaled by 10^3

Table 7.3 Range of severity thresholds derived from dNBR values in various ecosystems; including this study.

The dNBR thresholds for the burn severity classes proposed for the non-tropical ecosystems were derived using the CBI approach proposed by Key & Benson (2006), whereas the threshold obtained for tropical peatlands was based on field measurements. The threshold obtained for tropical peatland, subjected to multiple fires, is closer to the threshold derived for the Mediterranean ecosystem, which may be due to similarities in the vegetation pattern. The smallest value of the low severity class for tropical peatland is clearly lower than the other values for non-tropical ecosystems, which may be related to the period between the last low severity fire and ground-data collection. Those sites that have been affected less severely have a greater potential to recover quicker than those affected by fires of greater severity, thus this class could be difficult to separate from non-burned areas four years after a fire.

Discussion of burn severity in areas affected by multiple fires would not be complete without consideration of the effects of the first fire, in particular where there is a short fire return interval between the first and second fire. Section 5.3.4 provided an assessment of dNBRs derived for two fires occurring over a 5-year period. However, it needs to be stressed that the results are not entirely comparable because of limited data availability. In fact, the lack of essential remote sensed data to derive either fire extent or burn severity is a main constraint in tropical regions owing to high cloud cover. Nevertheless, Figure 5.22 showed that some plots that experienced a less severe first fire were heavily burnt by the subsequent fire; perhaps owing to the large fuel load remaining after the first fire. In contrast, some other plots experienced two high severity fires, which explain the almost complete absence of trees and invasion by ferns. Post-fire vegetation dynamics are discussed in more detail in the next Section.

To conclude, this study addressed a different approach to estimate burn severity by looking into the pattern of vegetation regrowth. The post-fire vegetation recovery model can be used to predict burn severity in tropical peatlands.

7.4 Long-lasting effects of fire on vegetation dynamics

Within the study area, both vegetation structure and species composition explicitly reflect the importance of the fire regime as a driver of land cover dynamics. In addition, the character of the post-fire vegetation is an important factor defining the burning conditions for the next fire event (discussed above) whilst the frequency, severity and return interval of fires influence the rate and nature of vegetation regrowth (Section 5.2) (Hoscilo *et al.* 2008b). The duration and trajectory of vegetation recovery depend also on pre-disturbance landscape conditions (Page *et al.*, 2008, Cochrane and Schulze, 1999, Goldammer, 1999), and ecosystem vulnerability (Yeager *et al.*, 2003).

In tropical peatlands, sites that have been subjected to a single fire recover quickly and undergo a progressive vegetation succession to secondary forest (Figure 7.4). With increased frequency of fire disturbance, the potential for natural recovery declines significantly and secondary succession back to forest becomes almost impossible since progressive succession is replaced by retrogressive change to low growing, less structured plant communities dominated by ferns. If the process of land degradation under a 'business-as-usual' scenario (i.e. no change to the current fire regime) continues, the tropical forest will be permanently transformed into a savanna phase where the landscape is largely composed of non-woody plant communities. This type of transition has been already discussed by other authors (Goldammer, 1999, Cochrane and Schulze, 1999).

A conceptual model of vegetation recovery following single and multiple fires (Figure 7.4), shows a simplified temporal relationship between fire frequency and the phases of vegetation succession. As was demonstrated in this study, however, the processes of vegetation regrowth are driven not only by fire frequency but also by several other factors (i.e. burn severity, interval between subsequent fires and proximity to remaining forest patches). The impact of these factors on post-fire vegetation regeneration processes are discussed in the following sub-sections.

7.4.1 *Differences in vegetation structure and biomass between once and twice burned forest*

Field data for 9-year old vegetation re-growth following a single fire (SF) in 1997 showed a relatively advanced stage of succession to forest with a significant number of trees (2444 trees ha^{-1}) greater than 5cm DBH, although on average 85% of all trees were represented by individuals with stems of less than 10cm DBH, while only 15% was made up of trees with a DBH>10cm. The density of trees greater than 10cm DBH (367 trees ha^{-1}) was comparable with the figure of 358 trees ha^{-1} reported by Yeager *et al.* (2003) for once-burned PSF in Central Kalimantan. The large proportion of small trees increases the fire risk owing to the fact that trees of small diameter are more fire prone than larger trees (Slik and Eichhorn, 2003, van Nieuwstadt and Sheil, 2005, Nishimua *et al.*, 2007). Therefore, regenerating forest is more vulnerable to fire than undisturbed forest.

The woody-AGB of secondary regrowth after the 1997 fire reached an average 30t ha^{-1} (ranging from 20 to 51t ha^{-1}) within a 9-year period, which is around 10-15% of the primary biomass in PSF (314t ha^{-1} reported by Sulistiyanto (2004)). This is a slower rate of biomass recovery than noted in non-peatland forests. Nykvist (1996), for example, showed that in lowland dipterocarp forest in southwest Sabah, Borneo, 24% of the pre-fire biomass was restored over an 8-year period. This indicates the fragility of tropical PSF and the long-lasting negative effects of fire on forest regeneration compared with more fire-resistant lowland forests (Yeager *et al.*, 2003).   .

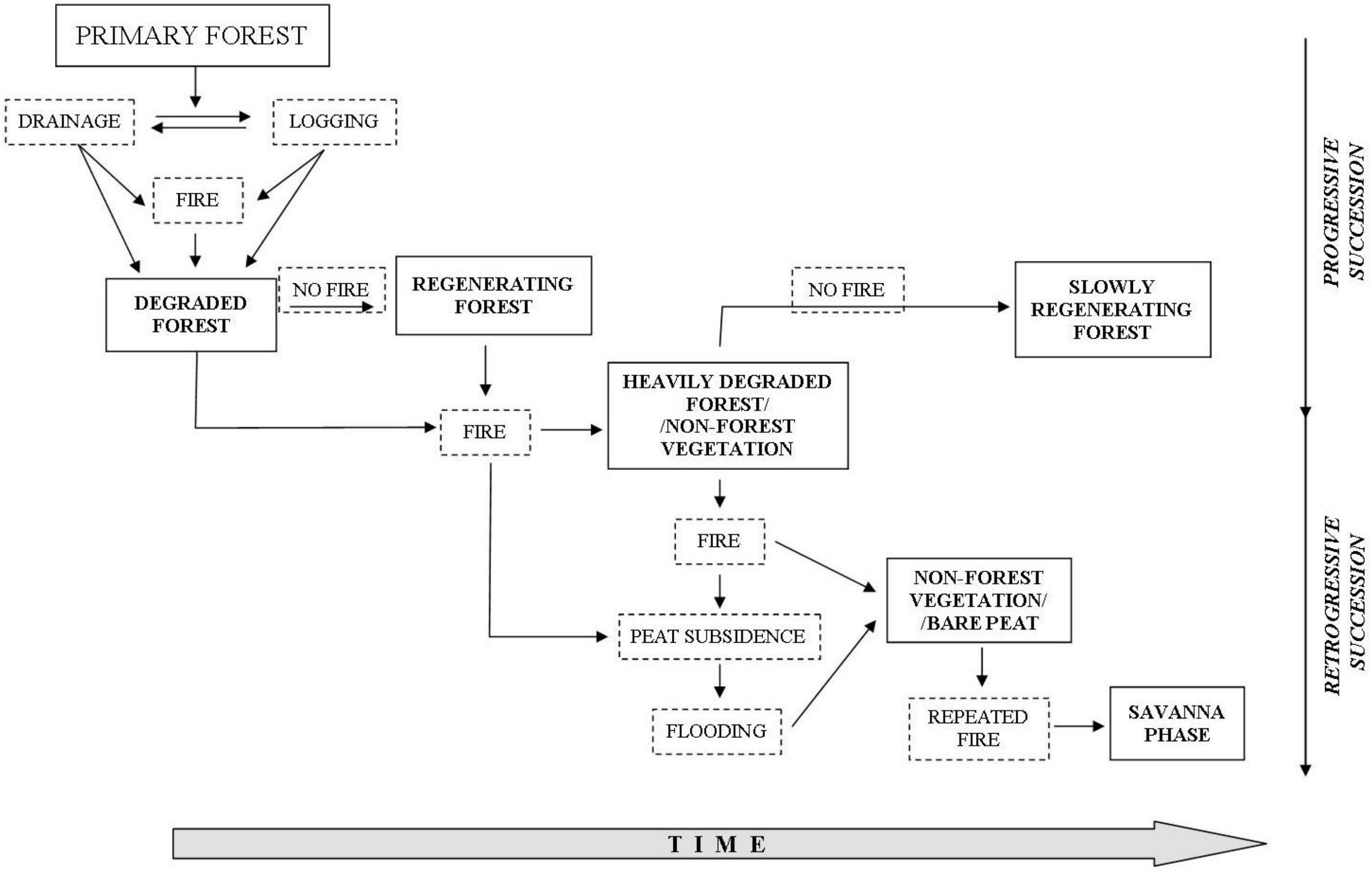


Figure 7.4 Conceptual model showing relationship between single and multiple fires and phases of vegetation succession placed on a time scale (modified from Page et al. (2008)).

In a review paper, Chazdon (2003) reported that in burned dipterocarp forest in Borneo a period of around 60 years was required for biomass to recover to the level of an unburned primary forest. Extrapolating the results on biomass recovery presented in this thesis indicates that PSF requires also at least 60 years to recover to a pre-disturbance level, assuming a proportional logarithmic incremental. Secondary forests accumulate biomass at a rate up to 3.3t ha yr^{-1} in the early stage of the regrowth (based on an average AGB value of 30t ha^{-1} for 9-year forest regrowth). The process of recovery of AGB and vegetation structure can be stimulated by proximity to remnant patches of forest since undisturbed forest provides a source of seeds and seed dispersers (Chazdon, 2003, Gould *et al.*, 2002). The sampling plot located in close proximity to remaining forest showed a higher post-fire biomass increment, up to 51t ha^{-1}, and tree density (2800 tree ha^{-1}) than plots established further away from the forest edge.

Repeated fires reduce the ability of disturbed forest to regenerate naturally (Section 5.2). Cochrane *et al.* (1999) highlighted that variation in fire intensity and burn severity are crucial aspects influencing the pattern of vegetation regrowth. This was clearly observed in this study (Section 5.2), where an increase in the magnitude of burn severity reduced the capacity of the vegetation to recover naturally. Unfortunately, some of the forested areas that burnt in 1997 had only 5-years to regrow before the fire of 2002 destroyed the secondary regrowth. In general the woody-AGB of secondary vegetation in the multiple fire plots (MF) was much lower than that of the single fire (SF) plots, although the woody-AGB of MF plots subjected to a low severity fire (MF1) reached values of 8-21t ha^{-1} over a 4-year period. This range is comparable with the biomass values for twice burned PSF (15-21t ha^{-1}) reported by Jaya (2007). Locations that have been re-burnt less severely have an almost equal potential to recover biomass as areas subjected to single fires, on the basis that the woody-AGB of 4-year old regrowth is equal to half the average woody-AGB of 9-year old secondary forest (SF) (Figure 7.5). This result was confirmed by the study of Hiratsuka (2006) in lowland dipterocarp forest in East Kalimantan who reported that forest subjected to a single fire can recover AGB in the range 7-25t ha^{-1} over a 5-year period. The AGB of MF1 was dominated by a large number of saplings and trees, with 87% of trees <10cm in DBH; larger trees (with DBH>10cm) comprised 13%. These larger specimens are remnants of the original forest which survived the last low severity fire.

In contrast, second repeat fires of moderate (MF2) or high severity (MF3) had more profound effects on the ecosystem, with very slow or no recovery of woody-AGB and invasion by non-woody species of fern, sedge and grass. This represents a retrogressive successional phase (Page *et al.*, 2008). Woody-AGB in MF2 plots was greatly reduced (range 1-5t ha^{-1}); these values are four-times lower than the woody-AGB of MF1 plots. On this basis, the biomass would need at least several centuries to recover to pre-disturbance levels, assuming that there were no subsequent fires.

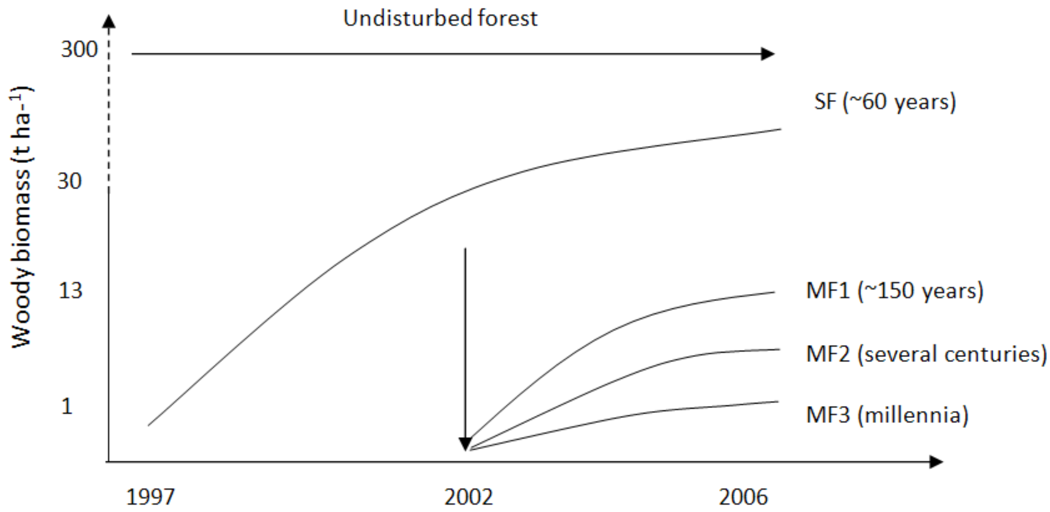


Figure 7.5 Rate of woody-AGB recovery in comparison to undisturbed forest following a single fire (SF) (1997) and multiple fires (1997 & 2002) characterised by different magnitudes of burn severity (i.e. low (MF1), moderate (MF2) and high (MF3)). Time required to recover to pre-disturbance level is given in brackets.

The lowest woody-AGB was recorded in the MF3 class (range 0-1t ha^{-1}) which represents locations exposed to a second fire of high severity. In these locations there was an almost complete absence of trees, with only 29 trees and around 638 saplings ha^{-1} compared with the MF2 sites (195 trees and 1295 saplings), which had a six-times greater tree density and two-times higher sapling density. Most of the regenerating trees in the locations subjected to multiple fires were multi-stemmed; of the total number of trees, 55%, 80% and 93% were made up by multi-stemmed trees in locations subjected to low, moderate and high severity second fires, respectively. The high proportion of multi-stemmed trees can be related to the high light levels in the burnt plots and the lack of competition with other woody species. In studies carried out in Caribbean dry forest, Dunphy *et al.* (2000) concluded that the presence of multi-stemmed trees was likely to be related to environmental stress associated with periodic droughts or hurricanes. They also pointed out the role of multi-stemmed trees in eliminating other species in the competition for light. Multi-stemmed trees apparently demonstrate tolerance to environmental perturbations, including fires, since they can survive indefinitely as a clone by producing new stems from ground level (Pugnaire and Valladares, 2007).

7.4.2 *Fern colonization*

The extensive colonization by fern-dominated vegetation at all sites subjected to multiple burns compensates in part for the low woody-AGB (Section 5.2). Non-woody-AGB (i.e. fern) on average reached 7.7t ha^{-1} in moderately and 8.5t ha^{-1} in high severity burned areas. Fern growth is limited by the presence of a tree canopy, thus non-woody-AGB was only 3t ha^{-1} in plots subjected to low severity burns where average tree canopy cover was 68%, compared to an average value of 28% for canopy cover in locations subjected to fire of moderate severity. For the same reason, ferns were not present in 9-year old secondary forest, where the tree canopy cover was on average 82%, which provided a high level of shade for the forest understory, thereby limiting fern species (van Nieuwstadt, 2002, Slik *et al.*, 2008). At sites subjected to multiple fires, fern cover increased considerably, from 29% in areas subjected to a low severity fire to 77% and 98% in locations experiencing fires of moderate and high severity, respectively. Several other studies have highlighted the invasion of ferns and grasses as a feature of repeatedly burned areas of tropical forest (Cleary and Priadjati, 2005, Slik *et al.*, 2008, van Nieuwstadt, 2002, Woods, 1989).

A high density of non-woody vegetation suppresses tree regrowth since it can overgrow and out-compete many seedlings and saplings in the early stages of their development (van Nieuwstadt, 2002, Cleary and Priadjati, 2005, Richards, 1996, van Nieuwstadt *et al.*, 2001). In addition, fire reduces seed availability and dispersal, leading to a decline in seedlings and saplings, and removes the vegetative regeneration potential of tree bases and roots, which are burned away (Cochrane, 2001, Cochrane *et al.*, 1999, van Nieuwstadt *et al.*, 2001, Cleary and Priadjati, 2005). Van Nieuwstadt *et al.* (2001) reported a loss of 85% of the dormant seeds in the litter layer and 60% in the upper layer of soil after fire in lowland forest in Borneo. A further decrease in density of woody strata occurs if fire returns to the same location within a short period of time, as was demonstrated in Section 5.2.3. This may explain why the density of saplings and seedlings remains so much lower after multiple fires than after a single fire.

The decline in the tree and sapling strata in conjunction with rapid colonisation by non-woody vegetation may lead to the irreversible transformation of burned tropical forests into herbaceous type habitats. The results of this study have shown that locations experiencing two fires of high severity enter a critical retrogressive phase of regeneration, where even four years after the last fire, the vegetation is largely dominated by non-woody species, particularly ferns, and with evidence of bare ground (Figure 7.4). In these heavily degraded locations, the capacity of woody vegetation to return to a pre-disturbance, mature condition appears to be extremely limited without human intervention (i.e. replanting).

The highly acidic peat soil in the study area appeared to be particularly favourable for the fern *Stenochlaena palustris* that formed a dense, impenetrable ground cover on burned surfaces, reaching a height of 1.5m. Clearly (2005) stressed that the variation in fern density in burned forest might be associated with a form of fire adaptation. He also pointed out that nutrient-poor, acidic soil provides an ideal environment for ferns but less so for woody species. Overall, the dominant fern in the study area provides the principle fire fuel on sites subjected to multiple burns, and its presence has greatly increased the threat of future fire owing to the high flammability of both dead and live fronds (van Nieuwstadt, 2002, Uhl and Kauffman, 1990).

Ironically, apart from the risk of fire, sites subjected to repeated fires are also at potential risk of flooding. The regenerating vegetation at sites which experienced high severity fires was dominated mainly by a combination of two species of fern, *Stenochlaena palustris* and *Blechnum indicum*, with a higher proportion of bare ground than at less severely burned sites. As illustrated in Table 5.7, the proportion of ground covered by *Blechnum indicum* doubled from 19% in moderately burned areas to 38% in highly degraded sites, where likely a greater depth of peat had been lost, hence the surface was lower and there would be an increased likelihood of flooding during the wet season. *Blechnum indicum* is a flood-tolerant species which grows in wet, high light conditions (Hoshizaki and Moran, 2001).

7.4.3 *Tree species composition modified by fire*

Even though vegetation structure and biomass have the potential to recover following multiple fires, the species composition may be permanently altered even by a single severe fire (Woods, 1989, Slik and Eichhorn, 2003). The process of tree species recovery in burned forest usually depends on the surviving saplings, sprouting trees, germination of seeds and the seed bank (van Nieuwstadt, 2002, Chazdon, 2003, Gould *et al.*, 2002). With increasing frequency and intensity of fire disturbance, as was demonstrated in Section 5.2, the number of tree species was greatly reduced from an average of 16 species in the plots exposed to a single fire to 3-9 species per plot in the re-burned locations, for fires of high and low severity respectively. The species richness of woody vegetation is enhanced by close proximity to undisturbed patches of natural unburned forest hence tree diversity following a single fire increased from 9 to 29 species per plot moving closer to the remaining patches of forest (around 500m) (Section 5.2.3). Remnant forests play a crucial role in promoting a more rapid rate of recovery, particularly in terms of the tree species composition (Chazdon, 2003).

Under natural, undisturbed conditions, a tropical forest is dominated by climax trees that can germinate only in canopy shade with small gaps or open areas occupied by light-tolerating species of pioneer tree (Whitmore, 1989). In disturbed forest, the number of potential re-sprouting stems is reduced and canopy openness is more favourable for pioneer species to establish (Cochrane *et al.*, 1999, Slik *et al.*, 2002, van Nieuwstadt *et al.*, 2001, Whitmore, 1989, Slik *et al.*, 2008, Chazdon, 2003). The same trend was observed in this study where, in the fire-degraded areas, the climax trees were replaced by pioneer species dominated mainly by *Combretocarpus rotundatus*, which has small, winged, wind-dispersed seeds. At the sites subjected to a single fire, the tree species composition revealed clear variation, which was influenced by proximity to the remaining forest. The plot located closest to the edge of the remaining forest contained a mixture of primary forest species, with *Cratoxylon spp.*, *Shorea spp.* and *Litsea spp.* representing 19%, 12% and 7% of the total number of all trees, respectively. By contrast, in the plots further away from the forest edge, primary forest species were replaced by pioneer species, dominated by *Combretocarpus rotundatus,* with this species comprising up to 88% of all trees recorded.

Combretocarpus rotundatus appears to have a number of adaptations that enable it to tolerate the environmental conditions in a fire-degraded ecosystem. This light-tolerating species is able to reproduce very easily both by quick production of seeds and re-sprouting from the roots (Giesen, 2004). As a result, it spreads relatively easily into burned areas and is the main coloniser. *Combretocarpus rotundatus* has also been confirmed as a fire-tolerating species that can survive a second less severe fire (i.e. it has a thick bark and hardwood that protect the underlying vascular tissues; Giesen, 2004). In this study, it was found that 88% of all survivor trees (trees>10cm DBH) at sites subjected to a low severity fire were represented by this particular species. Rapid invasion of pioneer species following fire disturbance has also been reported by a number of other researchers (Toma *et al.,* 2005, Cochrane and Schulze, 1999, Slik *et al.*, 2008, Slik and Eichhorn, 2003, Whitmore, 1989). Toma *et al*. (2005) noted that the replacement of pioneer species with primary species in dipterocarp forest in East Kalimantan required a (unspecified) long time period without fires.

The composition of saplings and seedlings varied in SF and MF1 plots, whereas the plots subjected to moderate and high severity burns were dominated by two pioneer species: *Combretocarpus rotundatus* and *Cratoxylon glaucum* (Table 5.7).

7.5 Fire as the main driver of carbon loss

The intensive deforestation and degradation of tropical peatlands demonstrated in this study and reported by several other authors (Langner *et al.,* 2007, Langner and Siegert, 2009, van der Werf *et al.,* 2008, Page *et al.,* 2008, Hooijer *et al.,* 2006) has led to on-going carbon losses from combustion of both vegetation and peat as well as from increasing peat decomposition (biological oxidation) (Hooijer *et al.*, 2006, Page *et al.*, 2002).

Estimation of carbon loss from tropical peatland subjected to repeated fires is not a straight forward procedure and requires many factors to be considered (e.g. those relating to the land degradation, fire regimes and hydrology). Inaccurate estimates may consequently contribute to the range of uncertainties related to carbon fluxes from peatland fires. The greatest area of uncertainty relates to the quantity (volume) of peat available for combustion. There are a very limited number of publications on peat carbon losses from tropical peatland fires. Page *et al.* (2002) estimated, using a combination of remote sensing and field measurements that between 0.06-0.07 GtC was released through peat combustion from Block C during the 1997 fires. Their study considered only one fire event, whereas currently tropical peatlands in Southeast Asia are subjected to frequent repeat fires.

This study used information from the comprehensive analyses of land cover and fire regime dynamics (Chapter 4 and 5), to propose a conservative and innovative approach to estimate the carbon losses from burning peat, taking into account a number of fire variables (i.e. fire extent, frequency, interval between subsequent fires), which determined the amount of available fuel, and fire intensity that was driven partly by El Niño droughts.

The more difficult and uncertain part of peat combustion analysis was to obtain accurate measures of the average depth of burned peat over a large area subjected to varying fire regimes. Currently, these measurements can either be acquired manually in the field using traditional survey equipment (Page *et al.*, 2002, Limin, unpublished data) or differential GPS equipment, or the use of Light Detection and Ranging (LIDAR) (Siegert, unpublished data). Each of these techniques has limitations associated with determining the spatial and temporal distribution of burn scars. In this study, all available data were combined to derive the depth of burned peat for various fire regime scenarios; in the worst case scenario the peat surface was lowered by 0.7-1.2m over four repeat fire events in one location (1997, 2002, 2004 and 2005). As was demonstrated in Chapter 6, the total amount of carbon lost from the burning of peat and vegetation in the study area over the 32-year period of investigation (1973-2005) was estimated to be in the range 79-113 Mt, of which 54-83 Mt came from peat and 26-29 Mt from vegetation combustion. This is equal to about 290-417 Mt of CO_2 emitted to the atmosphere, equivalent of 52-74% of annual CO_2 emissions for the UK (note that the study area covers only 0.6% of the UK).

This loss of peat carbon represents a reduction of between 11-18% of the total peat carbon store in the study area (Table 6.5). The largest proportion of these losses took place over the period 1997-2005, with an estimated 65-94 Mt of carbon lost during this period. The greatest AGB-carbon stock reduction was observed following the 1997 fire, when AGB-carbon stock declined to 20 Mt in the study area (59% of the 1973-stock) and to 12 Mt (72% of the 1973-stock) in PSF. The changes in land cover over the study period not only reduced the AGB-carbon stock but also led to a significant reduction in the capacity of the vegetation to undergo natural recovery (Chapter 5), thus the carbon stocks of regrowing, secondary vegetation remained very low (1-3 MtC) in comparison to the carbon stocks of the primary PSF vegetation.

The result of this study showed that around 7-10 MtC yr^{-1} was emitted from peatlands in the study area (period 1997-2005). Hooijer *et al.* (2006) estimated that the rate of carbon loss from all peatland fires in Indonesia was currently 1400 Mt yr^{-1}, which equals to 0.006 MtC yr^{-1} km^2. According to the calculations by Hooijer *et al.* (2006) peatlands in the study area (4,500km^2) would have emitted twice as much carbon (28 Mt yr^{-1}) as they currently do (7-10 Mt yr^{-1} derived from this study). This confirm that the rate of carbon loss from fires derived by the Hooijer *et al.* (2006) study is considerably overestimated, mainly because the fire-affected areas were obtained from hotspot data and secondly they assumed that every fire burnt 0.5m of peat. The same sets of input data and assumptions were used by Spessa *et al.* (2009) to assess carbon loss from burning peat in Borneo. The limitations of hotspot data in quantitative estimation of burned areas were presented in Section 2.2.1. By contrast, in this study the depth of burned peat was calculated according to how many times and how often the area was affected by fire, which seems to be a more reliable and accurate approach.

It is also important to emphasise that the carbon loss estimates produced as a result of this study did not include an additional amount of carbon emitted from peat decomposition as a result of peatland drainage and subsequent biological oxidation of organic matter (Hooijer *et al.*, 2006, Jauhiainen *et al.*, 2008, Hirano *et al.*, 2008).

The results of this study indicate that, under a ‘business-as-usual’ scenario, the carbon losses in this area and similar areas of degraded tropical peatland, will continue to increase over time, due to both frequently repeated fires and on-going peat decomposition. Effective reduction of carbon emissions from degraded peatlands can be achieved only by reducing, and ideally preventing, further incidence of fire and by promoting more sustainable forms of land management, primarily activities that promote restoration of the peatland hydrology (e.g. through canal blocking in order to raise peatland water tables; Page *et al.*, 2008). Carbon emission reduction strategies can only be successful if they are based on reliable knowledge of the landscape degradation history and in particular the role of fire in GHG emissions.

8 CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER RESEARCH

8.1 Conclusions

The aim of this research was to demonstrate the ecological and climatic impacts of recently enhanced fire regimes in tropical peatlands through measures of land cover, biomass and carbon dynamics.

Research objectives:

1. **To quantify the land cover and fire regime dynamics over the period 1973-2005 in the study area.**

2. **To examine the effects of fire frequency and burn severity on the dynamics of post-fire vegetation recovery by studying four restoration indicators: aboveground biomass, vegetation ground cover, canopy cover and diversity of plant species.**

3. **To quantify carbon losses from burning aboveground (vegetation) and belowground (peat) biomass.**

In answer to objective 1, the following conclusions have been derived:

- The land cover dynamics of the study area are strongly driven by widespread fires that have become more frequent in recent years. During the entire period of investigation, 1973-2005, the land cover change revealed a gradual reduction in forest cover towards non-woody vegetation. Every year, around 2.2% of the primary was converted to some other land cover. A rapid acceleration of land cover dynamics with a significant increase in the rate of deforestation was documented for the period 1997-2005, with forest cover declining at a rate of 7.5% yr^{-1}.

- PSF was exposed to the highest levels of deforestation, with an average reduction rate of 2.5% yr^{-1} for the entire period of investigation (1973-2005) and up to 9% yr^{-1} for the post-MRP period (1997-2005). Over the 32-year study period, 80% of the PSF in the study area was lost. Most of the deforested land was converted to either non-woody vegetation or mosaic of trees and non-woody vegetation. Notably, the deforested land was not converted into cultivated land, but was left abandoned and exposed to further degradation and high risk of fire.

- Fire regimes have changed considerably over the last three decades (1973-2005), with an evident increase in fire frequency and a decline in the fire return interval after the implementation of the MRP (1997-2005). Up until 1997, fires had affected a relatively small area, in total 23% of the study area, and were largely related to land clearance. This situation changed significantly during the last decade (1997-2005), when the widespread, intensive fires of 1997 affected a much larger area (34%). Five years later, in 2002, extensive fires returned, affecting 22% of the study area. In 2004 and 2005, a further 14% and 12% was burnt respectively.

- Over the last decade (1997-2005) widespread and intensive fires have become an integral part of this peatland ecosystem and are now associated with most dry seasons. During the post-MRP period, around 45% of the study area was subject to multiple fires; with 37% burnt twice and 8% burnt three or more times. This confirms that once burned fragments have become more susceptible to future fires.

- Fire regime (i.e. fire intensity, recurrence and propagation) is driven by the availability, quality and amount of fire fuels. After 1997, the fire fuel shifted from mainly PSF biomass towards non-woody biomass, dominated by regenerating vegetation. Newly established vegetation has been shown to be fire prone, although fire propagation is slower than in forest and restricted by both low fuel quality and load.

- This study has highlighted that the majority of fire events were directly or indirectly associated with human activities (i.e. selective logging, land clearance, intensive drainage and transmigration re-settlement). The intensive drainage infrastructure associated with the MRP initiative greatly impaired the peatland hydrological system, increasing the risk of fire. In addition, the network of canals allowed easy access for people whose activities provided ignition sources. Hence, multiple fires (more than three repeated fires over a short period of time) were located within close proximity to canals and declined with distance away from canals.

- There is a clear relationship between extensive fire incidents and droughts associated with ENSO events (i.e. 1973, 1991, 1997 and 2002). There is also evidence to suggest that the incidence of fire has become uncoupled from ENSO-related drought conditions and that, in this highly degraded landscape, the risk of fire incidence is increasing even in non-El Niño years (i.e. 2004 and 2005).

Contribution of this study to the knowledge gaps identified in Chapter 2:

a) knowledge on the land cover and fire regimes dynamics improves our understanding of ecological processes associated with combustion and land degradation;
b) provides data essential to improve estimation of carbon losses and gains from and to the ecosystem.

In answer to objective 2, the following conclusions have been made:

- The character of the post-fire vegetation is an important factor defining the burning conditions for the next fire event whilst frequency, severity and return interval of fires influence the rate and nature of vegetation regrowth. An increase in fire frequency and burn severity reduce significantly the potential for recovery of forest vegetation. Secondary succession towards forest is possible following a single fire, but the regrowth potential decreases rapidly for areas affected by multiple fire disturbances over a short period of time.

- The woody biomass of PSF (on average 30t ha^{-1}) subjected to a single fire can recover to 10-15% of primary forest biomass over a 9 year period, with full recovery predicted to take about 60 years. The process of recovery of biomass and vegetation structure can be stimulated by proximity of burnt areas to remnant patches of primary forest. In areas affected by multiple fires, with a 5-year fire return interval, the potential of woody AGB to recover naturally to the level of pre-disturbance biomass decreased with the magnitude of burn severity. In areas affected by fires of low severity, the woody biomass accumulated to 8-21t ha^{-1} over a 4-year period following a second fire, but in areas affected by fires of moderate or high severity, biomass was reduced to only 1-5t $ha^{-1}$and 0-1t ha^{-1} respectively.

- Second fires of moderate or high severity had more profound effects on the ecosystem, with very slow or no recovery of woody-biomass and invasion by non-woody plant communities dominated by two species of fern, *Stenochlaena palustris* and *Blechnum indicum*. At this point, secondary succession back to forest becomes almost impossible since progressive succession is replaced by a retrogressive phase in which regeneration of woody species is impaired by the absence of both tree seeds and seed dispersers.

- Non-woody (i.e. fern) biomass on average reached 7.7t ha^{-1} and 8.5t ha^{-1} in areas subject to second fires of moderate and high severity, respectively. Fern growth is limited by the presence of a tree canopy, thus in areas subject to low severity second fires, fern cover and biomass were lower (29% and 3.2t ha^{-1}, respectively) since regenerating trees provided shade. The proportion of fern cover increased to 77% and 98% in areas subject to moderate and high severity burns, where canopy cover was only 28% and 5%, respectively. For the same reason, fern was not present in the 9-year old secondary forest, where the tree canopy cover was on average 82%.

- Sites subjected to repeat severe fires are also at potential risk of flooding; high proportion of bare ground and lack of decent tree cover increase peat decomposition and subsidence which consequently lead to flooding.

- With increasing frequency and intensity of fire disturbance the number of tree species was reduced from an average of 16 species in plots exposed to a single fire, to 3-9 species per plot in re-burned locations. Sites experiencing single fires have more potential for recovery of the original species complement only, if located in close proximity to remaining patches of forest.

- Recovery of vegetation in the study area is only possible if further fires are prevented and the remaining patches of natural forest are protected as a potential biodiversity source for re-colonisation.

- This study provided a novel approach to quantifying relative magnitude of burn severity through characteristic of the post-fire vegetation regrowth. A strong correlation was established between spectral indices (i.e. dNBR, NBR and NDWI) derived from pre/post satellite imagery and sets of vegetation parameters collected four years after the last fire. This demonstrates the long-lasting effect of multiple fires on vegetation structure and recovery. It confirms the hypothesis that the variation in the structure and species composition of post-fire vegetation regrowth (i.e. MF1, MF2 and MF3 plots) observed in the field could be explained by differences in the magnitude of burn severity.

Contribution of this study to the knowledge gaps identified in Chapter 2:

a) Provides unique sets of data on the post-fire secondary vegetation structure and composition in tropical peatlands (aboveground biomass can improve uncertainties in carbon flux estimations).
b) Provides sufficient knowledge of how single and multiple fires modify vegetation regeneration processes, and how these processes depend on the fire frequency and severity of disturbance.
c) Provides information on the capacity of vegetation to recover naturally under different fire regimes; this knowledge is essential to undertake restoration and rehabilitation actions.
d) This study used a different approach to estimate burn severity by looking at the pattern of vegetation regrowth. In addition, the burn severity analyses were conducted in the tropical ecosystem; to date there is a lack of tested methods to support analysis of burn severity in the tropics.

In answer to objective 3, the following conclusions have been derived:

- Due to a lack of appropriate methodology, this study proposed a conservative approach to estimating the carbon losses from burning peat, taking into account a number of fire variables (i.e. fire frequency, return interval, amount of available fuel, and fire intensity).

- The total amount of carbon lost from both burning vegetation and peat in the study area was estimated to be in the range of 79-113 Mt, with peat-derived carbon losses of 53-83 Mt and vegetation combustion losses of 26-30 Mt. This loss of peat carbon represents a reduction of between 11-18% of the total peat carbon store in the study area. The largest proportion of these losses took place between 1997 and 2005, with an estimated 65-94 Mt (~83% of total) of carbon lost over this period. Over the investigation period (1973-2005), the aboveground carbon stock in the study area declined by 66% from 50 Mt to 17 Mt. Most of this change occurred in the PSF carbon stock, which declined by

80% from 41 Mt to 8 Mt; these losses were largely related to recurrent and extensive fires.

- Under scenarios of decreasing rainfall, increasing intensity of El Niño, and ongoing pressure for cultivated land and settlement, the carbon losses in this area and similar areas of degraded tropical peatland, will continue to increase over time, due to both frequently repeated fires and on-going peat decomposition. Effective reduction of carbon emissions from degraded peatlands can be achieved only by reducing, and ideally preventing, further incidence of fire and by promoting more sustainable forms of land management, primarily activities that promote restoration of the peatland hydrology.

Contribution of this study to the knowledge gaps identified in Chapter 2:

a) Provides a conservative and unique methodology to estimate carbon losses from burning peat under various fire regimes.
b) Improves carbon flux estimations associated with fire in tropical peatland by incorporating both aboveground and surface biomass losses.

8.2 Recommendations for further research

Suggestions for further areas of research that build on the results of this study are described below.

- Use the comprehensive sets of data on land cover and fire regime dynamics derived from this study to develop a baseline methodology to monitor carbon losses from burning above and belowground biomass and carbon gains from regenerating vegetation in tropical peatlands. This would contribute to improved estimates of carbon fluxes in the tropics and could also be used in carbon emission reduction schemes (e.g. under the Voluntary Carbon Standard).

- Continued use of the established sampling plots to monitor post-fire vegetation regrowth and investigate longer-term trends and projections of vegetation dynamics (i.e. increment of biomass, tree density and diversity, fern colonization) under different fire regimes. This information would contribute to effective restoration and rehabilitation actions in areas subjected to similar fire degradation.

- Further burn severity analysis in order to improve understanding of the linkage between burn severity of initial and subsequent fires, taking into account different fire return intervals. The post-fire vegetation recovery model proposed in this study for multiple fires needs to be further developed for prediction of burn severity from a single fire. It is also important to examine the time frame over which the above model can be used.

- The innovative approach to estimate carbon losses from burned peatland, proposed in this study, can be implemented on a larger scale to improve current quantification of carbon losses in the whole Southeast Asian region. This methodology can be additionally enhanced by developing more reliable and precise remote sensing techniques to measure the depth of burned peat over a large area subjected to varying fire regimes.

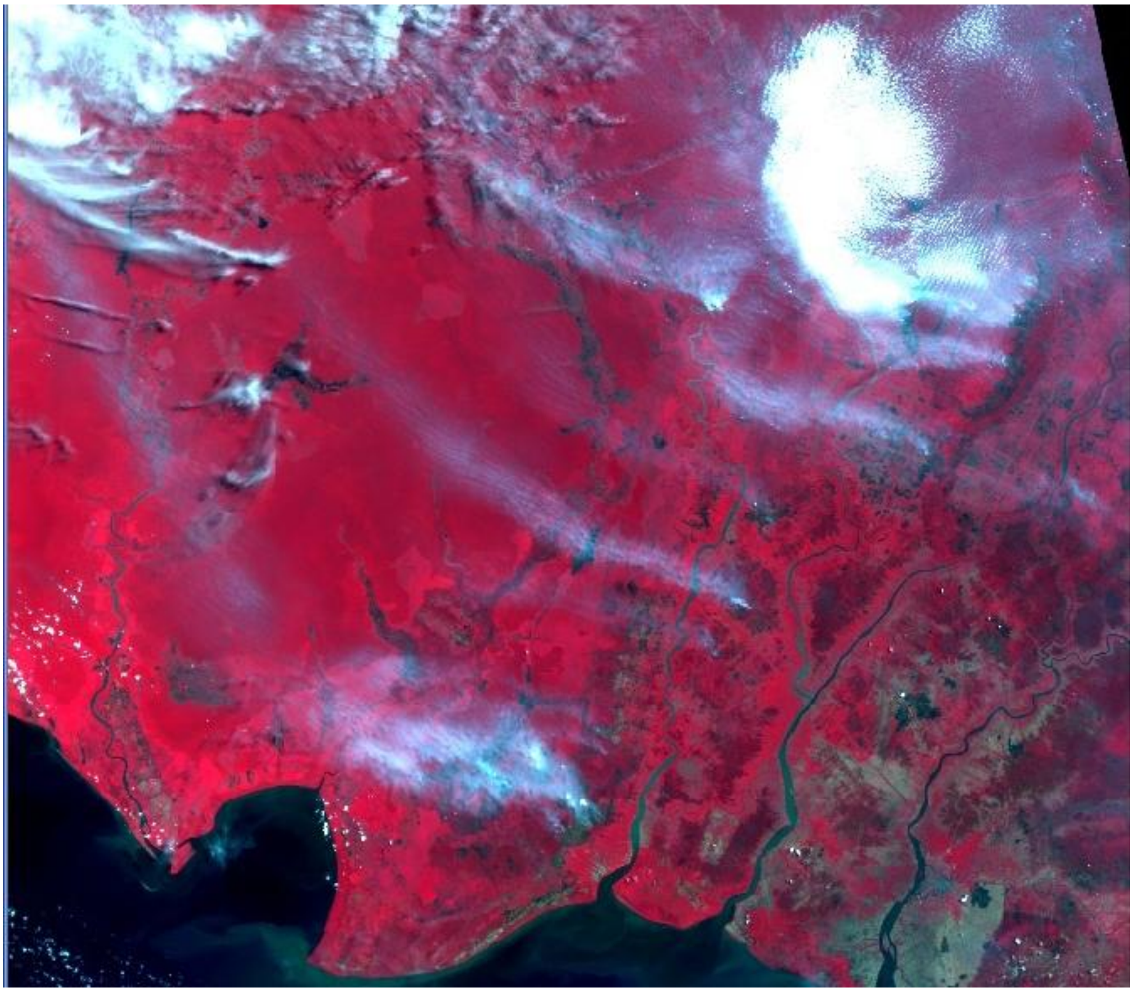
- Fly the area with a hyperspectral instrument. Hyperspectral sensors seem to be very useful for vegetation study due to data sampling using much narrower band widths. This type of data could be valuable for detecting the stages of succession and in addition to improve biomass and carbon estimations, which are difficult to assess from Landsat.

9 APPENDICES

Appendix 1 *Examples of the DMC images used in this study.*

The DMC image enquired at the beginning of fire season (2005-07-30) covering the study area (band combination: 1 (IR), 2 (R), 3 (G)). Plumes of first fires can clearly be seen.

The DMC image enquired almost at the end of fire season (2005-09-11) showing several peat fires and burn scars; band combination 1 (IR), 2 (R), 3 (G).



Appendix 2 *Vegetation inventory forms.*

Date: Area/transect:……………………..

Research name ……………………………….

Site ID (20x20m) …………………………………….
Subplot of tree ID (10X10m)……………………………
Subplot of saplings ID (5x5m) …………………………
Subplot of seedlings ID (2.5x2.5m) ……………………

GPS centre position: Lat………………….. Long …………………………

Fire history: SF- single fire or MF- multiple fires

Secondary succession:

MF1- advance
MF2- intermediate
MF3- the least advance

Vegetation structure:

Avg. canopy height (m): ………………………………..

Avg. vegetation height (m):……………………………..

Avg. DBH trees: <5cm ; 5-10cm ; 10-20cm ; >20cm

Dominant tree species: …………………………………

Dominant saplings species …………………………….

Canopy cover:
ID of photo: ……………………………..

Additional measurements/observations:

Within each subplot 10x10m:

- To drew a sketch of spatial distribution of fern, bare ground and canopy;

- To estimate the proportion of ground covered by ferns: *Stenochlacena palustris* and *Blechnum indicum* and bare ground;
- To measure and write down the GPS coordinates at each corner and in the plot centre;
- To take photographs at each corner and in the plot centre, clockwise at every 45 degrees;
- To indicate the North Arrow.

To draw the vertical Vegetation Profile:

Single fire: Tree inventory

Observer name: ………………………….. …. Date: ………………………….. Side ID ……… Subplot ID…………………
Plant type: Trees (1)
Tree structure: Single-stemmed tree (1) or Multi-stemmed tree (2)

Plant type	Tree structure	No of stems if multi-stemmed tree	Species (Local name)	DBH (cm) (measure for each stem)	Height (cm)	Observation

Single fire: Inventory for Saplings (1) and Seedlings (2)

Observer name: ……………………………. Date: ………………………….. Side ID ……… Subplot ID ………………..
Plan type: Saplings (1) – plot 5x5m
Seedlings (2) – plot 2.5x2.5m
Canopy structure: Single-stemmed (1) or Multi-stemmed (2)

Plant type	Canopy structure	No of stems if multi-stemmed	Species (Local name)	DBH (cm) (measure for each stem)	Height (cm)	Observation

Multiple fires: Inventory for trees, sapling and seedlings

Observer name: ………………………….. Data: ……………………… Side ID ………... Subplot ID……………

Plant type: Tree (1)
Saplings (2)
Seedlings (3)

Canopy structure: Single-stemmed (1) or Multi-stemmed (2)

Plant type	Canopy structure	No of stems if multi-stemmed	Species (Local name)	Crown diameter (cm)	DGH (cm) (measure for each stem)	Distance to the tree (cm)	Angle	Height (cm)	Observation

Tree DBH >5cm
Saplings DBH 1.0-4.9 cm
Seedlings DBH <1cm; Height>50cm

Appendix 3 *List of woody species identified in the field (Latin names were provided by CIMTROP).*

Inventory name	Family	Genus	Species	Local Name Sebangau
Tumih	Anisophyllaceae	*Combretocarpus*	*rotundatus*	Tumih
Geronggang	Guttiferae	*Cratoxylon*	*arborescens*	Geronggang
Alulup	Annonaceae	*Polyalthia*	*hypoleuca*	Rewoi/Alulup
Asam Asam	Magnoliaceae	*Magnolia*	*bintulensis*	Asam Asam
Bungaris	Fabaceae (Leguminosae)	*Koompassia*	*malaccensis*	Bungaris/Kempas
Bintangor	Clusiaceae (Guttiferae)	*Calophyllum*	*hosei*	Jinjit/Bintangor
Blawan Punai	Myrtaceae	*Tristaniopsis*	*grandifolia*	Blawan Punai
Ehang	Ebenaceae	*Diospyros*	*siamang*	Ehang
Gemur	Leucaena	*Alseodaphne*	*coriacea*	Gemur
Gulung Haduk	Ebenaceae	*Diospyros*	*evena*	Gulung Haduk
Hangkang	Sapotaceae	*Palaquium*	*leiocarpum*	Hangkang
Jambu	Myrtaceae	*Syzygium/Eugenia*		Jambu Burung
Jelutong	Apocynaceae	*Dyera*	*costulata*	Jelutong
Kenari	Euphorbiaceae	*Blumeodendron*	*elateriospermum*	Kenari
Lalas (Galam Tikus)	Myrtaceae	*Syzygium*	*spicata*	Kayu Lalas Daun Kecil/Galam Tikus
Lunuk	Moraceae	*Ficus*		Lunuk
Malam Malam	Ebenaceae	*Diospyros*	*bantamensis*	Aring Pahe/Malam Malam
Mangkinang	Elaeocarpaceae	*Elaeocarpus*	*mastersii*	Mangkinang
Mata Undang (Tampohot)	Euphorbiaceae	*Antidesma*	*phanerophleum*	Tanundang/Matan Undang
Medang	Lauraceae	*Litsea*	*elliptica*	Medang
Mendarahan	Myristicaceae	*Myristica*	*crassifolia*	Mendarahan daun besar
Meranti		*Shorea*	*marantha*	
Pampaning	Fagaceae	*Lithocarpus*		Pampaning
Patanak	Elaeocarpaceae	*Elaeocarpus*	*acmocarpus*	Patanak
Pendu	Unknow			
Ponak	Tetrameristaceae	*Tetramerista*	*glabra*	Ponak
Rambai Hutan	Euphorbiaceae	*Baccaurea*	*bracteata*	Rambai Hutan
Rambangun	Rutaceae	*Tetractomia*	*tetrandra*	Rambangun
Rambutan	Sapindaceae	*Nephellium*		
Sangkuak	Sapotaceae	*Planchonella*	*maingayi*	Sangkuak
Sumpung	Aquifoliaceae	*Ilex*	*hypoglauca*	Sumpung

Inventory name	**Family**	**Genus**	**Species**	**Local Name Sebangau**
Tabitik	Lauraceae	*Phoebe*	*grandis*	Tabitik
Tampohot	Myrtaceae	*Syzygium.sp*		Tampohot/Tengelam
Tapanggang	Fabaceae (Leguminosae)	*Adenanthera*	*pavonina*	Tapanggang
Terontang	Anacardiaceae	*Camnosperma*	*auriculata*	Terontang
Kajalaki	Meliaceae	*Aglaia*	*rubignosa*	
Kambasira	Aquifoliaceae	*Ilex*	*hypoglauca*	
Karipak	Annonaceae	*Mezzettia*	*leptopoa*	
Kasak Undang	Unknow			
Niatugagas	Unknow			
Balangeran	Dipterocarpaceae	*Shorea*	*balangeran*	Kahui/Balangeran

Appendix 4 *Analysis of variance (one-way ANOVA) for all vegetation variables; df –degrees of freedom, F–F-test of significance, Sig.–level of significance; N=20.*

ANOVA		Sum of Squares	df	Mean Square	F	Sig.
No of tree species	Between Groups	514.867	3	171.622	7.098	.003
	Within Groups	386.883	16	24.180		
	Total	901.750	19			
Density of single-stemmed trees	Between Groups	1.023E7	3	3411461.806	89.190	.000
	Within Groups	611989.583	16	38249.349		
	Total	1.085E7	19			
Density of multi-stemmed trees	Between Groups	955968.750	3	318656.250	15.902	.000
	Within Groups	320625.000	16	20039.062		
	Total	1276593.750	19			
Density of single- and multi-stemmed trees	Between Groups	1.624E7	3	5412118.056	118.178	.000
	Within Groups	732739.583	16	45796.224		
	Total	1.697E7	19			
Density of single-stemmed saplings	Between Groups	129447.900	3	43149.300	7.061	.003
	Within Groups	97768.300	16	6110.519		
	Total	227216.200	19			
Density of multi-stemmed saplings	Between Groups	228604.167	3	76201.389	7.585	.002
	Within Groups	160739.583	16	10046.224		
	Total	389343.750	19			
Ratio of multi- to single-stemmed tree	Between Groups	32.435	3	10.812	1.050	.398
	Within Groups	164.792	16	10.300		
	Total	197.227	19			
Density of single- and multi-stemmed saplings	Between Groups	8.090E7	3	2.697E7	7.061	.003
	Within Groups	6.111E7	16	3819074.219		
	Total	1.420E8	19			
Ration (density of sapling to tree)	Between Groups	996.696	3	332.232	.761	.532
	Within Groups	6988.308	16	436.769		
	Total	7985.005	19			
Density of seedlings	Between Groups	1.850E8	3	6.165E7	75.893	.000
	Within Groups	1.300E7	16	812339.844		
	Total	1.979E8	19			
Total woody biomass	Between Groups	2410.529	3	803.510	16.444	.000
	Within Groups	781.832	16	48.864		
	Total	3192.361	19			
Biomass of trees	Between Groups	2070.816	3	690.272	17.574	.000
	Within Groups	628.464	16	39.279		
	Total	2699.279	19			

ANOVA		Sum of Squares	df	Mean Square	F	Sig.
Biomass of saplings	Between Groups	12.627	3	4.209	3.890	.029
	Within Groups	17.314	16	1.082		
	Total	29.941	19			
Biomass of fern	Between Groups	225.325	3	75.108	14.060	.000
	Within Groups	85.473	16	5.342		
	Total	310.798	19			
Canopy cover	Between Groups	18884.959	3	6294.986	32.381	.000
	Within Groups	3110.454	16	194.403		
	Total	21995.413	19			
No of trees with DBH<10cm	Between Groups	23958.300	3	7986.100	37.716	.000
	Within Groups	3387.900	16	211.744		
	Total	27346.200	19			
No of trees with DBH >10;15<cm	Between Groups	699.250	3	233.083	7.465	.002
	Within Groups	499.550	16	31.222		
	Total	1198.800	19			
No of trees with DBH <15cm	Between Groups	6.600	3	2.200	1.166	.354
	Within Groups	30.200	16	1.888		
	Total	36.800	19			
DBH of trees	Between Groups	82.443	3	27.481	7.854	.002
	Within Groups	55.980	16	3.499		
	Total	138.423	19			
Total Biomass (woody +fern)	Between Groups	1190.375	3	396.792	8.129	.002
	Within Groups	780.951	16	48.809		
	Total	1971.326	19			
DBH of tree	Between Groups	82.445	3	27.482	7.855	.002
	Within Groups	55.981	16	3.499		
	Total	138.426	19			
Basal area of trees	Between Groups	563.446	3	187.815	46.101	.000
	Within Groups	65.184	16	4.074		
	Total	628.629	19			
Basal area of saplings	Between Groups	13.242	3	4.414	3.449	.042
	Within Groups	20.476	16	1.280		
	Total	33.718	19			
Basal area – total	Between Groups	727.240	3	242.413	37.078	.000
	Within Groups	104.608	16	6.538		
	Total	831.848	19			
Ratio of basal area of saplings to total basal area	Between Groups	93.570	3	31.190	8.019	.002
	Within Groups	62.236	16	3.890		
	Total	155.805	19			

ANOVA		Sum of Squares	df	Mean Square	F	Sig.
Ratio of basal area of trees to total basal area	Between Groups	14338.301	3	4779.434	8.348	.001
	Within Groups	9160.228	16	572.514		
	Total	23498.530	19			
Percentage of multiple trees to total no of trees	Between Groups	7649.234	3	2549.745	2.109	.139
	Within Groups	19347.916	16	1209.245		
	Total	26997.150	19			
Percentage of trees < 10cm DBH	Between Groups	7735.250	3	2578.417	2.591	.089
	Within Groups	15921.950	16	995.122		
	Total	23657.200	19			
Percentage of trees from 10 to 15cm DBH	Between Groups	815.000	3	271.667	6.679	.004
	Within Groups	650.750	16	40.672		
	Total	1465.750	19			
Percentage of trees >15cm	Between Groups	6.750	3	2.250	1.053	.397
	Within Groups	34.200	16	2.138		
	Total	40.950	19			
Percentage of *Combretocarpus.sp* to total no of trees	Between Groups	10138.373	3	3379.458	3.393	.044
	Within Groups	15934.559	16	995.910		
	Total	26072.932	19			
Fern cover (%)	Between Groups	28398.267	3	9466.089	46.623	.000
	Within Groups	3248.533	16	203.033		
	Total	31646.800	19			
Bare ground (%)	Between Groups	53.167	3	17.722	3.221	.051
	Within Groups	88.033	16	5.502		
	Total	141.200	19			
Percentage of *Combretocarpus.sp* to total no of saplings	Between Groups	13578.991	3	4526.330	5.266	.010
	Within Groups	13752.771	16	859.548		
	Total	27331.762	19			

Appendix 5 *List of woody species identified in each of the four different successional stages, SF- single fire, MF-multiple fires.*

Inventory name	SF	MF1	MF2	MF3
Alulup		x		x
Asam Asam	x	x		x
Balangeran	x			
Bintangor	x			
Blawan Punai	x	x		
Bungaris	x			
Ehang				x
Gemur		x		
Geronggang	x	x	x	x
Gulung Haduk	x			
Hangkang	x			
Jambu	x			x
Jelutong	x	x		x
Kajalaki	x			
Kambasira	x	x	x	x
Karipak	x			
Kasak Undang	x			
Kenari	x			
Lalas (Galam Tikus)	x	x		x
Lunuk	x	x	x	
Malam Malam	x	x	x	
Mangkinang	x			
Mata Undang (Tampohot)	x	x		
Medang	x	x		
Mendarahan	x	x	x	
Meranti	x			
Niatugagas		x		
Pampaning	x		x	
Patanak	x	x	x	x
Pendu	x	x		
Ponak	x			
Rambai Hutan	x	x		x
Rambangun	x	x		
Rambutan	x			
Sangkuak	x			
Sumpung	x			
Tabitik	x			
Tampohot (Tengelam)		x	x	
Tapanggang	x	x		
Terontang	x			x
Tumih	x	x	x	x

Appendix 6 *Results of the Tukey multiple comparisons of means test; 1–SF, 2–MF1, 3–MF2 and 4–MF3.*

Multiple Comparisons								
Dependent Variable		(I) class	(J) class	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
							Lower Bound	Upper Bound
NBR	Tukey HSD	1	2	.216805*	.056011	.007	.05656	.37705
			3	.480961*	.056011	.000	.32071	.64121
			4	.738818*	.053897	.000	.58462	.89302
		2	1	-.216805*	.056011	.007	-.37705	-.05656
			3	.264156*	.052808	.001	.11307	.41524
			4	.522013*	.050560	.000	.37736	.66667
		3	1	-.480961*	.056011	.000	-.64121	-.32071
			2	-.264156*	.052808	.001	-.41524	-.11307
			4	.257857*	.050560	.001	.11320	.40251
		4	1	-.738818*	.053897	.000	-.89302	-.58462
			2	-.522013*	.050560	.000	-.66667	-.37736
			3	-.257857*	.050560	.001	-.40251	-.11320
dNBR	Tukey HSD	1	2	-.217120*	.062854	.016	-.39695	-.03729
			3	-.471557*	.062854	.000	-.65138	-.29173
			4	-.768346*	.060481	.000	-.94138	-.59531
		2	1	.217120*	.062854	.016	.03729	.39695
			3	-.254438*	.059259	.003	-.42398	-.08490
			4	-.551227*	.056736	.000	-.71355	-.38890
		3	1	.471557*	.062854	.000	.29173	.65138
			2	.254438*	.059259	.003	.08490	.42398
			4	-.296789*	.056736	.000	-.45911	-.13447
		4	1	.768346*	.060481	.000	.59531	.94138
			2	.551227*	.056736	.000	.38890	.71355
			3	.296789*	.056736	.000	.13447	.45911
NDVI	Tukey HSD	1	2	.118699	.047499	.098	-.01720	.25460
			3	.259001*	.047499	.000	.12310	.39490
			4	.375248*	.045706	.000	.24448	.50601
		2	1	-.118699	.047499	.098	-.25460	.01720
			3	.140303*	.044783	.029	.01218	.26843
			4	.256549*	.042876	.000	.13388	.37922
		3	1	-.259001*	.047499	.000	-.39490	-.12310
			2	-.140303*	.044783	.029	-.26843	-.01218
			4	.116247	.042876	.066	-.00642	.23892
		4	1	-.375248*	.045706	.000	-.50601	-.24448
			2	-.256549*	.042876	.000	-.37922	-.13388
			3	-.116247	.042876	.066	-.23892	.00642

Dependent Variable		(I) class	(J) class	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
							Lower Bound	Upper Bound
dNDVI	Tukey HSD	1	2	-.120874	.054944	.165	-.27807	.03632
			3	-.244997*	.054944	.002	-.40219	-.08780
			4	-.384081*	.052870	.000	-.53534	-.23282
		2	1	.120874	.054944	.165	-.03632	.27807
			3	-.124124	.051802	.118	-.27233	.02408
			4	-.263207*	.049597	.000	-.40510	-.12131
		3	1	.244997*	.054944	.002	.08780	.40219
			2	.124124	.051802	.118	-.02408	.27233
			4	-.139083	.049597	.056	-.28098	.00281
		4	1	.384081*	.052870	.000	.23282	.53534
			2	.263207*	.049597	.000	.12131	.40510
			3	.139083	.049597	.056	-.00281	.28098
NDWI	Tukey HSD	1	2	.218941*	.049458	.002	.07744	.36044
			3	.433115*	.049458	.000	.29162	.57461
			4	.637525*	.047591	.000	.50137	.77368
		2	1	-.218941*	.049458	.002	-.36044	-.07744
			3	.214174*	.046629	.002	.08077	.34758
			4	.418584*	.044644	.000	.29086	.54631
		3	1	-.433115*	.049458	.000	-.57461	-.29162
			2	-.214174*	.046629	.002	-.34758	-.08077
			4	.204410*	.044644	.002	.07668	.33214
		4	1	-.637525*	.047591	.000	-.77368	-.50137
			2	-.418584*	.044644	.000	-.54631	-.29086
			3	-.204410*	.044644	.002	-.33214	-.07668
dNDWI	Tukey HSD	1	2	-.221400*	.068237	.024	-.41663	-.02617
			3	-.433053*	.068237	.000	-.62828	-.23782
			4	-.676327*	.065662	.000	-.86419	-.48847
		2	1	.221400*	.068237	.024	.02617	.41663
			3	-.211653*	.064335	.022	-.39572	-.02759
			4	-.454927*	.061596	.000	-.63115	-.27870
		3	1	.433053*	.068237	.000	.23782	.62828
			2	.211653*	.064335	.022	.02759	.39572
			4	-.243274*	.061596	.006	-.41950	-.06705
		4	1	.676327*	.065662	.000	.48847	.86419
			2	.454927*	.061596	.000	.27870	.63115
			3	.243274*	.061596	.006	.06705	.41950

Dependent Variable		(I) class	(J) class	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	Upper Bound
GV	Tukey HSD	1	2	.296880*	.073350	.005	.08702	.50674
			3	.686480*	.073350	.000	.47662	.89634
			4	.884217*	.070581	.000	.68228	1.08615
		2	1	-.296880*	.073350	.005	-.50674	-.08702
			3	.389600*	.069155	.000	.19175	.58745
			4	.587337*	.066211	.000	.39790	.77677
		3	1	-.686480*	.073350	.000	-.89634	-.47662
			2	-.389600*	.069155	.000	-.58745	-.19175
			4	.197737*	.066211	.039	.00830	.38717
		4	1	-.884217*	.070581	.000	-1.08615	-.68228
			2	-.587337*	.066211	.000	-.77677	-.39790
			3	-.197737*	.066211	.039	-.38717	-.00830
Shade	Tukey HSD	1	2	-.008980	.035689	.994	-.11109	.09313
			3	-.012500	.035689	.985	-.11461	.08961
			4	.066783	.034342	.249	-.03147	.16504
		2	1	.008980	.035689	.994	-.09313	.11109
			3	-.003520	.033648	1.000	-.09979	.09275
			4	.075763	.032216	.128	-.01641	.16793
		3	1	.012500	.035689	.985	-.08961	.11461
			2	.003520	.033648	1.000	-.09275	.09979
			4	.079283	.032216	.105	-.01289	.17145
		4	1	-.066783	.034342	.249	-.16504	.03147
			2	-.075763	.032216	.128	-.16793	.01641
			3	-.079283	.032216	.105	-.17145	.01289
NPV	Tukey HSD	1	2	-.431155*	.074010	.000	-.64290	-.21941
			3	-.642655*	.074010	.000	-.85440	-.43091
			4	-.817425*	.071217	.000	-1.02118	-.61367
		2	1	.431155*	.074010	.000	.21941	.64290
			3	-.211500*	.069778	.036	-.41114	-.01186
			4	-.386270*	.066807	.000	-.57741	-.19513
		3	1	.642655*	.074010	.000	.43091	.85440
			2	.211500*	.069778	.036	.01186	.41114
			4	-.174770	.066807	.079	-.36591	.01637
		4	1	.817425*	.071217	.000	.61367	1.02118
			2	.386270*	.066807	.000	.19513	.57741
			3	.174770	.066807	.079	-.01637	.36591

*. The mean difference is significant at the 0.05 level.

10 BIBLIOGRAPHY

Achard, F., Eva, H. D., Stibig, H. J., Mayaux, P., Gallego, J., Richards, T. & Malingreau, J. P. (2002) Determination of deforestation rates of the world's humid tropical forests. *Science*, 297, 999-1002.

Achard, F., Eva, H. D., Mayaux, P., Stibig, H. J. & Belward, A. (2004) Improved estimates of net carbon emissions from land cover change in the tropics for the 1990s. *Global Biogeochemical Cycles,* 18.

Aiken, S. R. (2004) Runaway fires, smoke-haze pollution, and unnatural disasters in Indonesia. *The Geographical Review*, 94, 55-79.

Alleaume, S., Hely, C., Le Roux, J., Korontzi, S., Swap, R. J., Shugart, H. H. & Justice, C. O. (2005) Using MODIS to evaluate heterogeneity of biomass burning in southern African savannahs: a case study in Etosha. *International Journal of Remote Sensing,* 26, 4219-4237.

Allen, J. L. & Sorbel, B. (2008) Assessing the differenced Normalized Burn Ratio's ability to map burn severity in the boreal forest and tundra ecosystems of Alaska's national parks. *International Journal of Wildland Fire*, 17, 463-475.

Anderson, J. A. R. (1976) Observations on the ecology of five peat swamp forests in Sumatra and Kalimantan. *Peat and podzolic soils and their potential for agriculture in Indonesia, Proceedings of the ATA 106 Midterm Seminar, Tugu,13.-14.10.1976 Soil Research Institute, Bogor*.

Arellano, A. F., Jr., Kasibhatla, P. S., Giglio, L., van der Werf, G. R., Randerson, J. T. & Collatz, G. J. (2006) Time-dependent inversion estimates of global biomass-burning CO emissions using Measurement of Pollution in the Troposphere (MOPITT) measurements. *Journal of Geophysical Research,* 111, D09303.

Arino, O., Picolini, I., Kasischke, E., Siegert, F., Chuvieco, E., Martin, P. , Li, Z., Fraser, R., Eva, H., Stroppiana, D., Pereira, J., Silva, J.M.N., Roy D. Barbosa,P.M. (2001) Methods of mapping surfaces burned in vegetation fires. *In: F.J. Ahern, J.G. Goldammer and C.O. Justice, Editors, Global and Regional Vegetation Fire Monitoring from Space: Planning a Coordinated International Effort, SPB Academic Publishing, The Hague, The Netherlands*, 227-255.

Arino, O., Gross, D., Ranera, F., Leroy, M., Bicheron, P., Brockman, C., Defourny, P., Vancutsem, C., Achard, F., Durieux, L., Bourg, L., Latham, J., Di Gregorio, A., Witt, R., Herold, M., Sambale, J., Plummer, S. & Weber, J. L. (2007) GlobCover: ESA service for Global land cover from MERIS. *Igarss: 2007 IEEE International Geoscience and Remote Sensing Symposium,* 1-12, 2412-2415.

Arroyo-Mora, J. P., Sanchez-Azofeifa, G. A., Kalacska, M. E. R., Rivard, B., Calvo-Alvarado, J. C. & Janzen, D. H. (2005) Secondary forest detection in a neotropical dry forest landscape using Landsat 7 ETM+ and IKONOS imagery. *Biotropica,* 37, 497-507.

Asner, G. P., Keller, M. & Silva, J. N. M. (2004) Spatial and temporal dynamics of forest canopy gaps following selective logging in the eastern Amazon. *Global Change Biology,* 10, 765-783.

Asner, G. P., Nepstad, D., Cardinot, G. & Ray, D. (2004) Drought stress and carbon uptake in an Amazon forest measured with spaceborne imaging spectroscopy. *Proceedings of the National Academy of Sciences of the United States of America,* 101, 6039-6044.

Baker, T. R., Phillips, O. L., Malhi, Y., Almeida, S., Arroyo, L., Di Fiore, A., Erwin, T., Killeen, T. J., Laurance, S. G., Laurance, W. F., Lewis, S. L., Lloyd, J., Monteagudo, A., Neill, D. A., Patino, S., Pitman, N. C. A., Silva, J. N. M. & Martinez, R. V. (2004) Variation in wood density determines spatial patterns in Amazonian forest biomass. *Global Change Biology,* 10, 545-562.

Baker, J. C. & Williamson, R. A. (2006) Satellite imagery activism: Sharpening the focus on tropical deforestation. *Singapore Journal of Tropical Geography,* 27, 4-14.

Balch, J., Nepstad, D. C., Brando, P. M., Curran, P., Portela, O., de Carvelho, O. & Lefebvre, P. (2008) Negative fire feedback in a transitional forest of southeastern Amazonia. *Global Change Biology,* 14, 2276-2287.

Bechteler, A. & Siegert, F. (2004) Recurrent fires in tropical peatlands in Central Kalimantan. *Proceedings of the 12th International Peat Congress: Wise Use of Peatlands,* 1, 607-614.

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

Bontemps, S., Bogaert, P., Titeux, N. & Defourny, P. (2008) An object-based change detection method accounting for temporal dependences in time series with medium to coarse spatial resolution. *Remote Sensing of Environment,* 112, 3181-3191.

Broadbent, E. N., Zarin, D. J., Asner, G. P., Pena-Claros, M., Cooper, A. & Littell, R. (2006) Recovery of forest structure and spectral properties after selective logging in lowland Bolivia. *Ecological Applications,* 16, 1148-1163.

Brünig, E. F. (1973) Some further evidence on the amount of damage attributed to the lightning and wind-throw in Shorea albida forest in Sarawak. *Commonwealth Forestry Review,* 133-144.

Castro, K. L., Sanchez-Azofeifa, G. A. & Rivard, B. (2003) Monitoring secondary tropical forests using space-borne data: implications for Central America. *International Journal of Remote Sensing,* 24, 1853-1894.

Chafer, C. J. (2008) A comparison of fire severity measures: An Australian example and implications for predicting major areas of soil erosion. *Catena,* 74, 235-245.

Charron, I. & Greene, D. F. (2002) Post-wildfire seedbeds and tree establishment in the southern mixedwood boreal forest. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere,* 32, 1607-1615.

Chave, J., Condit, R., Aguilar, S., Hernandez, A., Lao, S. & Perez, R. (2004) Error propagation and scaling for tropical forest biomass estimates. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences,* 359, 409-420.

Chave, J., Andalo, C., Brown, S., Cairns, M. A., Chambers, J. Q., Eamus, D., Folster, H., Fromard, F., Higuchi, N., Kira, T., Lescure, J. P., Nelson, B. W., Ogawa, H., Puig, H., Riera, B. & Yamakura, T. (2005) Tree allometry and improved estimation of carbon stocks and balance in tropical forests. *Oecologia,* 145, 87-99.

Chazdon, R. L. (2003) Tropical forest recovery: legacies of human impact and natural disturbances. *Perspectives in Plant Ecology, Evolution and Systematics,* 6, 51-71.

Chuvieco, E., Riano, D., Aguado, I. & Cocero, D. (2002) Estimation of fuel moisture content from multitemporal analysis of Landsat Thematic Mapper reflectance data: applications in fire danger assessment. *International Journal of Remote Sensing,* 23, 2145-2162.

Chuvieco, E., Rian'o, D., Danson, F. M. & Martin, P. (2006) Use of a radiative transfer model to simulate the postfire spectral response to burn severity. *Journal of Geophysical Research-Biosciences,* 111, 1-15.

Chuvieco, E., De Santis, A., Riaño, D. & Halligan, K. (2007) Simulation approaches for burn severity estimation using remotely sensed images. *Fire Ecology Special Issue,* 3, 129-149.

Cihlar, J. (2000) Land cover mapping of large areas from satellites: status and research priorities. *International Journal of Remote Sensing,* 21, 1093-1114.

Cleary, D. F. R. & Priadjati, A. (2005) Vegetation responses to burning in a rain forest in Borneo. *Plant Ecology,* 177, 145-163.

Cochrane, M. A. & Souza, C. M. (1998) Linear mixture model classification of burned forests in the Eastern Amazon. *International Journal of Remote Sensing,* 19, 3433-3440.

Cochrane, M. A., Alencar, A., Schulze, M. D., Souza, C. M., Nepstad, D. C., Lefebvre, P. & Davidson, E. A. (1999) Positive feedbacks in the fire dynamic of closed canopy tropical forests. *Science*, 284, 1832-1835.

Cochrane, M. A. & Schulze, M. D. (1999) Fire as a recurrent event in tropical forests of the eastern Amazon: Effects on forest structure, biomass, and species composition. *Biotropica,* 31, 2-16.

Cochrane, M. A. (2001) Synergistic interactions between habitat fragmentation and fire in evergreen tropical forests. *Conservation Biology,* 15, 1515-1521.

Cochrane, M. A. (2003) Fire science for rainforests. *Nature*, 421, 913-919.

Cochrane, M. A. (2009) Tropical Fire Ecology, Climate Change, Land Use and Ecosystem Dynamics. *Springer Praxis Books*.

Cocke, A. E., Fulé, P. Z. & Crouse, J. E. (2005) Comparison of burn severity assessments using Differenced Normalized Burn Ratio and ground data. *International Journal of Wildland Fire,* 14, 189-198.

Coppin, P., Jonckheere, I., Nackaerts, K., Muys, B. & Lambin, E. (2004) Digital change detection methods in ecosystem monitoring: a review. *International Journal of Remote Sensing,* 25, 1565-1596.

Crowley, G. (2008) DMC Data Product Manual. *DMC International Imaging Ltd., WMCO-0115056-01,* 1-113.

Crutzen, P. J., Heidt, L. E., Krasnec, J. P., Pollock, W. H. & Seiler, W. (1979) Biomass burning as a source of atmospheric gases: CO, H_2, N_20, NO, CH_3C1, and COS. *Nature,* 282, 253-256.

Crutzen, P. J. & Andreae M.O. (1990) Biomass burning in the tropics: impact on atmospheric chemistry and biochemical cycle. *Science,* 250, 1669-1678.

Curran, L. M., Trigg, S. N., McDonald, A. K., Astiani, D., Hardiono, Y. M., Siregar, P., Caniago, I. & Kasischke, E. (2004) Lowland forest loss in protected areas of Indonesian Borneo. *Science,* 303, 1000-1003.

Dabrowska-Zielinska, K., Gruszczynska, M., Lewinski, S., Hoscilo, A., Bojanowski, J. (2009) Application of remote and in situ information to the management of wetlands in Poland. *Journal of Environmental Management*, 90, 2261-2269.

Danielsen, F., Beukema, H., Burgess, N. D., Parish, F., Bruhl, C. A., Donald, P. F., Murdiyarso, D., Phalan, B., Reijnders, L., Struebig, M. & Fitzherbert, E. B. (2009) Biofuel Plantations on Forested Lands: Double Jeopardy for Biodiversity and Climate. *Conservation Biology,* 23, 348-358.

Dasgupta, S., Qua, J. J., Haoa, X. & Bhoi, S. (2007) Evaluating remotely sensed live fuel moisture estimations for fire behavior predictions in Georgia, USA. *Remote Sensing of Environment,* 108, 138-150.

Davies, S. J. & Unam, L. (1999) Smoke-haze from the 1997 Indonesian forest fires: effects on pollution levels, local climate, atmospheric CO2 concentrations, and tree photosynthesis. *Forest Ecology and Management*, 124, 137-144.

De Santis, A. & Chuvieco, E. (2006) Burn severity estimation from remotely sensed data: Performance of simulation versus empirical models. *Remote Sensing of Environment,* 108, 422-435.

De Alwis, D. A., Easton, Z. M., Dahlke, H. E., Philpot, W. D. & Steenhuis, T. S. (2007) Unsupervised classification of saturated areas using a time series of remotely sensed images. *Hydrology and Earth System Sciences Discussions*, 4, 1663-1696.

De Fries, R. S., Houghton, R. A., Hansen, M. C., Field, C. B., Skole, D. & Townshend, J. (2002) Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s. *Proceedings of the National Academy of Sciences of the United States of America*, 99, 14256-14261.

De Fries, R., Achard, F., Brown, S., Herold, M., Murdiyarso, D., Schlamadinger, B. & Souza, C. (2007) Earth observations for estimating greenhouse gas emissions from deforestation in developing countries. *Environmental Science & Policy*, 10, 385-394.

Dempewolf, J., Trigg, S., DeFries, R. S. & Eby, S. (2007) Burned-area mapping of the Serengeti-Mara region using MODIS reflectance data. *IEEE Geoscience and Remote Sensing Letters,* 4, 312-316.

Dennis, R. A., Mayer, J., Applegate, G., Chokkalingam, U., Pierce Colfer, C. J., Kurniawan, I., Lachowski, H., Maus, P., Pandu Permana, R., Ruchiat, Y., Stolle, F., Suyanto & Tomich, T. P. (2005) Fire, People and Pixels: Linking Social Science and Remote Sensing to Understand Underlying Causes and Impacts of Fires in Indonesia. *Human Ecology,* 33, 465-504.

Dennis, R. A. & Colfer, C. P. (2006) Impacts of land use and fire on the loss and degradation of lowland forest in 1983-2000 in East Kutai District, East Kalimantan, Indonesia. *Singapore Journal of Tropical Geography,* 27, 30-48.

Diaz-Delgado, R., Llorett, F. & Pons, X. (2003) Influence of fire severity on plant regeneration by means of remote sensing imagery. *International Journal of Remote Sensing,* 24, 1751-1763.

Duncan, B. N., Bey, I., Chin, M., Mickley, L. J., Fairlie, T. D., Martin, R. V. & Matsueda, H. (2003) Indonesian wildfires of 1997: Impact on tropospheric chemistry. *Journal of Geophysical Research-Atmospheres,* 108.

Dunphy, B. K., Murphy, P. G. & Lugo, A. E. (2000) The tendency for trees to be multiple-stemmed in tropical and subtropical dry forests: Studies of Guanica forest, Puerto Rico. *Tropical Ecology,* 41, 161-167.

Edwards, D. P., Emmons, L. K., Hauglustaine, D. A., Chu, D. A., Gille, J. C., Kaufman, Y. J., Petron, G., Yurganov, L. N., Giglio, L., Deeter, M. N., Yudin, V., Ziskin, D. C., Warner, J., Lamarque, J. F., Francis, G. L., Ho, S. P., Mao, D., Chen, J., Grechko, E. I. & Drummond, J. R. (2004) Observations of carbon monoxide and aerosols from the Terra satellite: Northern Hemisphere variability. *Journal of Geophysical Research-Atmospheres,* 109, 1-17.

Eichhorn, K. A. O. (2006) Plant diversity after rain-forest fires in Borneo. *Blumea Supplement 18. Backhuys, Leiden*.

Engelbrecht, B. M. J. & Herz, H. M. (2001) Evaluation of different methods to estimate understorey light conditions in tropical forests. *Journal of Tropical Ecology,* 17, 207-224.

Epting, J., Verbyla, D. & Sorbel, B. (2005) Evaluation of remotely sensed indices for assessing burn severity in interior Alaska using Landsat TM and ETM+. *Remote Sensing of Environment,* 96, 328-339.

Escuin, S., Navarro, R. & Fernandez, P. (2008) Fire severity assessment by using NBR (Normalized Burn Ratio) and NDVI (Normalized Difference Vegetation Index) derived from LANDSAT TM/ETM images. *International Journal of Remote Sensing,* 29, 1053-1073.

Euroconsult (2008) Master Plan for the Rehabilitation and Revitalisation of the Ex-Mega Rice Project Area in Central Kalimantan. *Euroconsult Mott MacDonald and Deltares, Delft Hydraulics in association with DHV, Wageningen UR, Witteveen and Bos, PT MLD and PT INDEC*.

FAO (2005) Global Forest Resources Assessment 2005; Progress towards sustainable forest management. *Food and Agriculture Organization of the United Nations, Rome, 2005, FAO Forestry,* Paper 147.

Fearnside, P. M. (1997) Transmigration in Indonesia: Lessons from Its Environmental and Social Impacts. *Environmental Management*, 21, 553-570.

Fearnside, P. M. & Laurance, W. F. (2004) Tropical deforestation and greenhouse-gas emissions. *Ecological Applications,* 14, 982-986.

Feeley, K. J., Gillespie, T. W. & Terborgh, J. W. (2005) The utility of spectral indices from Landsat ETM+ for measuring the structure and composition of tropical dry forests. *Biotropica,* 37, 508-519.

Foody, G. M., Cutler, M. E., McMorrow, J., Pelz, D., Tangki, H., Boyd, D. S. & Douglas, I. (2001) Mapping the biomass of Bornean tropical rain forest from remotely sensed data. *Global Ecology and Biogeography,* 10, 379-387.

Foody, G. M. & Cutler, M. E. J. (2003) Tree biodiversity in protected and logged Bornean tropical rain forests and its measurement by satellite remote sensing. *Journal of Biogeography,* 30, 1053-1066.

Fraser, R. H., Li, Z. & Cihlar, J. (2000) Hotspot and NDVI Differencing Synergy (HANDS) A New Technique for Burned Area Mapping over Boreal Forest. *Remote Sensing of Environment,* 74, 362-376.

Fraser, R. H., Li, Z. & Landry, R. (2000) SPOT VEGETATION for characterizing boreal forest fires. *International Journal of Remote Sensing,* 21, 3525-3532.

Freeborn, P. H., Wooster, M. J., Hao, W. M., Ryan, C. A., Nordgren, B. L., Baker, S. P. & Ch., I. (2008) Relationships between energy release, fuel mass loss, and trace gas and aerosol emissions during laboratory biomass fires. *Journal of Geonhvsical Research,* 113, 1-17.

Fuller, D. O. & Fulk, M. (2001) Burned area in Kalimantan, Indonesia mapped with NOAA-AVHRR and Landsat TM imagery. *International Journal of Remote Sensing,* 22, 691-697.

Fuller, D. O. (2006) Tropical forest monitoring and remote sensing: A new era of transparency in forest governance? *Singapore Journal of Tropical Geography,* 27, 15-29.

Fuller, D. O. & Murphy, K. (2006) The ENSO-fire dynamic in insular Southeast Asia. *Climatic Change,* 74, 435-455.

Gao, B. C. (1996) NDWI—A normalized difference water index for remote sensing of vegetation liquid water from space. *Remote Sensing of Environment,* 58, 257-266.

George, C., Rowland, C., Gerard, F. & Balzter, H. (2006) Retrospective mapping of burnt areas in Central Siberia using a modification of the normalised difference water index. *Remote Sensing of Environment,* 104, 346-359.

Gerwing, J. (2002) Degradation of forests through logging and fire in the eastern Brazilian Amazon *Forest Ecology and Management,* 157, 131-141.

Gibbs, H. K., Brown, S., Niles, J. O. & Foley, J. A. (2007) Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environment Research Letter*, 2, 1-13.

Giesen, W. (2004) Causes of peatswamp forest degradation in Berbak NP, Indonesia and recommendations for restoration. *Part of the project on Promoting the river basin and ecosystem approach for sustainable management of SE Asian lowland peat swamp forests: Case study Air Hitam Laut river basin, Jambi Province, Indonesia*, ARCADIS Euroconsult.

Giglio, L., Kendall, J. D. & Tucker, C. J. (2000) Remote sensing of fires with the TRMM VIRS. *International Journal of Remote Sensing*, 21, 203-207.

Giglio, L., Kendall, J. D. & Mack, R. (2003) A multi-year active fire dataset for the tropics derived from the TRMM VIRS. *International Journal of Remote Sensing*, 24, 4505-4525.

Giglio, L., Csiszar, I., Restás, Á., Morisette, J. T., Schroeder, W., Morton, D. & Justice, C. O. (2008) Active fire detection and characterization with the advanced spaceborne thermal emission and reflection radiometer (ASTER). *Remote Sensing of Environment*, 112, 3055-3063.

Giglio, L., Loboda, T., Roy, D. P., Quayle, B. & Justice, C. O. (2009) An active-fire based burned area mapping algorithm for the MODIS sensor. *Remote Sensing of Environment*, 113, 408-420.

Glantz, M. H. (2001) Currents of Change: Impacts of El Niño and La Niña on Climate and Society (Paperback). *Second edition. Cambridge University Press, Cambridge, UK.*

Goldammer, J. G. (1997) The Role of Fire on Greenhouse Gas and Aerosol Emissions and Land Use and Cover Change in Southeast Asia: Ecological Background and Research Needs. *International Conference on Science and Technology for the Assessment of Global Environmental Change and its Impacts on the Indonesian Maritime Continent, Jakarta, Indonesia, 10-12 November 1997*, 1-32.

Goldammer, J. G. (1999) Forests on fire. *Science*, 284, 1782-1783.

Gonzalez-Alonso, F., Merino-De-Miguel, S., Roldan-Zamarron, A., Garcia-Gigorro, S. & Cuevas, J. M. (2007) MERIS Full Resolution data for mapping level-of-damage caused by forest fires: the Valencia de Alcantara event in August 2003. *International Journal of Remote Sensing*, 28, 797-809.

Gould, K. A., Fredericksen, T. S., Morales, F., Kennard, D., Putz, F. E., Mostacedo, B. & Toledo, M. (2002) Post-fire tree regeneration in lowland Bolivia: implications for fire management. *Forest Ecology and Management,* 165, PII S0378-1127(01)00620-X.

Grassi, G., Monni, S., Federici S., Achard F. and Mollicone D. (2008) Applying the conservativeness principle to REDD to deal with the uncertainties of the estimates. *Environment Research Letter*, 3, 1-12.

Griscom, H. P., Griscom, B. W. & Ashton, M. S. (2009) Forest Regeneration from Pasture in the Dry Tropics of Panama: Effects of Cattle, Exotic Grass, and Forested Riparia. *Restoration Ecology*, 17, 117-126.

Guevara-Escobar, A., Tellez, J. & Gonzalez-Sosa, E. (2005) Use of digital photography for analysis of canopy closure. *Agroforestry Systems,* 65, 175-185.

Guild, L. S., Cohen, W. B. & Kauffman, J. B. (2004) Detection of deforestation and land conversion in Rondonia, Brazil using change detection techniques. *International Journal of Remote Sensing,* 25, 731-750.

Gullison, R. E., Frumhoff, P. C., Canadell, J. G., Field, C. B., Nepstad, D. C., Hayhoe, K., Avissar, R., Curran, L. M., Friedlingstein, P., Jones, C. D. & Nobre, C. (2007) Tropical Forests and Climate Policy. *Science,* 316, 985-986.

Gupta, A. (2005) The Physical Geography of Southeast Asia. *Oxford University Press*.

Hammill, K. A. & Bradstock, R. A. (2006) Remote sensing of fire severity in the Blue Mountains: influence of vegetation type and inferring fire intensity. *International Journal of Wildland Fire,* 15, 213-226.

Hardy, C. C. (2005) Wildland fire hazard and risk: Problems, definitions, and context. *Forest Ecology and Management,* 211, 73-82.

Hayes, D. J. & Sader, S. A. (2001) Comparison of change-detection techniques for monitoring tropical vegetation clearing and vegetation regrowth in a time series. *Photogrammetric Engineering and Remote Sensing*, 67, 1067–1075.

Hirano, T., Jauhiainen, J., Inoue, T. & Takahashi, H. (2008) Controls on the Carbon Balance of Tropical Peatlands. *Ecosystems,* 10, 1-15.

Hirano, T., Segah, H., Harada, T., Limin, S., June, T., Hirata, R. & Osaki, M. (2007) Carbon dioxide balance of a tropical peat swamp forest in Kalimantan, Indonesia. *Global Change Biology,* 13, 412-425.

Hiratsuka, M., Toma, T., Diana, R., Hadriyanto, D. & Morikawa, Y. (2006) Biomass recovery of naturally regenerated vegetation after the 1998 forest fire in East Kalimantan, Indonesia. *Jarq-Japan Agricultural Research Quarterly,* 40, 277-282.

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

Hoscilo, A., Page, S. E. & Tansey, K. (2008a) Spatial and temporal alterations of tropical peatlands (Indonesia) due to widespread and repeated fires over the last 30 years. *Proceedings of the Remote Sensing and Photogrammetry Society Conference, Exeter 2008*.

Hoscilo, A., Page, S. E. & Tansey, K. (2008b) Development of post-fire vegetation in the tropical ecosystem of Central Kalimantan, Indonesia. *Proceedings of the 13th International Peat Congress, Tullamore, Ireland, June 2008*, 202-205.

Hoshizaki, B. J. & Moran, R. C. (2001) Fern Grower's Manual. *Published by Timber Press, 2001*, 221-224.

Houghton, R. A. (2000) A new estimate of global sources and sinks of carbon from land-use change. *Eosystems,* 81, S281.

Houghton, R. A. (2001) Counting terrestrial sources and sinks of carbon. *Climatic Change,* 48, 525-534.

Houghton, R. A. (2003) Why are estimates of the terrestrial carbon balance so different? *Global Change Biology,* 9, 500-509.

Houghton, R. A. (2005) Aboveground Forest Biomass and the Global Carbon Balance. *Global Change Biology,* 11, 945-958.

Houghton, R. A. (2007) Balancing the Global Carbon Budget. *Annual Review of Earth and Planetary Sciences,* 35, 313-347.

Hudak, A. T., Robichaud, P.R., Evans, J.B., Clark, J., Lannom, K., Morgan, P., Stone, C. (2004) Field validation of Burned Area Reflectance Classification (BARC) products for post fire assessment. *Paper presented at Remote sensing for field users - Tenth Forest Service Remote Sensing Applications Conference in Salt Lake City, Utah. April 5-9*.

Hughes, R. F., Kauffman, J. B. & Jaramillo, V. J. (1999) Biomass, carbon, and nutrient dynamics of secondary forests in a humid tropical region of Mexico. *Ecology,* 80, 1892-1907.

Ichoku, C. & Kaufman, Y. J. (2005) A method to derive smoke emission rates from MODIS fire radiative energy measurements. *IEEE Transactions on Geoscience and Remote Sensing*, 43, 2636-2649.

Idris, M. H., Kuraji, K. & Suzuki, M. (2005) Evaluating vegetation recovery following large-scale forest fires in Borneo and northeastern China using multi-temporal NOAA/AVHRR images. *The Japanese Forest Society and Springer-Verlag Tokyo,* 10, 101-111.

Immirzi, C. P., Maltby, E., Clymo, R.S.; (1992) The Global Status of Peatlands and their Role in Carbon Cycling - A Report for Friends of the Earth by the Wetland. *Ecosystems Research Group, University of Exeter, Report,* 11, 1-145.

Ingram, J. C., Dawson, T. P. & Whittaker, R. J. (2005) Mapping tropical forest structure in southeastern Madagascar using remote sensing and artificial neural networks. *Remote Sensing of Environment,* 94, 491-507.

IPCC (2006) Guidelines for National Greenhouse Gas Inventories.

IPCC (2007) Climate Change 2007 Synthesis Report, Impacts, Adaptation and Vulnerability. *Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*.

Isaev, A. S., Korovin, G.N., Bartalev, S.A., Ershov, D.V., Janetos, A., Kasischke, E.S., Shugart, H.H., French, N.H., & Orlick, B. E., Murphy, T.L. (2002) Using remote sensing for assessment of forest wildfire carbon emissions. *Climatic Change,* 55, 231-255.

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

Jaya, A. (2007) Biomass content and biodiversity of tropical peat swamp forest under various land cover conditions. *International Symposium and workshop on tropical peatlands, Yogyakarta, Indonesian, August 27-31*.

JRC (2004) GLC2000. *JRC, homepage: http://www-gvm.jrc.it/glc2000/defaultGLC2000.htm*

Justice, C. O., Giglio, L., Korontzi, S., Owens, J., Morisette, J. T., Roy, D., Descloitres, J., Alleaume, S., Petitcolin, F. & Kaufman, Y. (2002) The MODIS fire products. *Remote Sensing of Environment,* 83, 244-262.

Kalacska, M., Sanchez-Azofeifa, G. A., Calvo-Alvarado, J. C., Quesada, M., Rivard, B. & Janzen, D. H. (2004) Species composition, similarity and diversity in three successional stages of a seasonally dry tropical forest. *Forest Ecology and Management,* 200, 227-247.

Kasischke, E. S. & Hoy, E. E. (2007) Post-Fire Evaluation of the Effects of Fire on the Environment using Remotely-Sensed Data. *Forest Fire, EARSeL Workshop, 2007,* 44-52.

Kasperson, J. X., Kasperson, R. E. & Turner, B. L. (1995) Regions at risk: comparisons of threatened environments *edited by: The United Nations University Press*.

Kauffman, Y. J., Kleidman, R. G. & King, M. D. (1998) SCAR-B fires in the tropics: properties and remote sensing from EOS-MODIS. *Journal of Geophysical Research - Atmospheres,* 104, 1955-1968.

Kaufman, Y. J., Ichoku, C., Giglio, L., Korontzi, S., Chu, D. A., Hao, W. M., Li, R. R. & Justice, J. O. (2003) Fire and smoke observed from the Earth Observing System MODIS instrument—products, validation, and operational use. *International Journal of Remote Sensing*, 24, 1765-1781.

Keeley, J. E., Fotheringham, C. J. & Baer-Keeley, M. (2005) Factors affecting plant diversity during post-fire recovery and succession of mediterranean-climate shrublands in California, USA. *Diversity and Distributions,* 11, 525-537.

Keeley, J. E., Brennan, T. & Pfaff, A. H. (2008) Fire severity and ecosytem responses following crown fires in California shrublands. *Ecological Applications,* 18, 1530-1546.

Keeley, J. E. (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire,* 18, 116-126.

Kettering, Q. M., Coe, R., Noordwijk, M., Ambagau, Y. & Palm, C. A. (2001) Reducing uncertenty in the use of allometric biomass eqations for predicting above-ground tree biomass in mixed secondary forests. *Forest Ecology and Management,* 146, 199-209.

Key, C. H. & Benson, N. C. (2002) Measuring and remote sensing of burn severity. *US Geological Survey Wildland FireWorkshop, 31 October to 3 November 2000, Los Alamos, NM. USGS Open-File Report 02-11.*

Key, C. H. & Benson, N. C. (2004) Ground Measure of Severity, The Composite Burn Index. . *In D. C. Lutes, R. E. Keane, J. F. Caratti, C. H. Key, N. C. Benson, & L. J. Gangi (Eds).FIREMON: Fire Effects Monitoring and Inventory System. Gen. Tech Rep. RMRS-GTR-XXX, Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station*.

Key, C. H. & Benson, N. C. (2006) Landscape Assessment: Sampling and Analysis Methods. *USDA forest Servis, General Technical Reort. RMRS-GTR-164-CD,* 1-55.

Kodandapani, N., Cochrane, M. A. & Sukumar, R. (2008) A comparative analysis of spatial, temporal, and ecological characteristics of forest fires in seasonally dry tropical ecosystems in the Western Ghats, India. *Forest Ecology and Management,* 256, 607-617.

Kokaly, R. F., Rockwella, B. W., Haireb, S. L. & Kinga, T. V. V. (2007) Characterization of post-fire surface cover, soils, and burn severity at the Cerro Grande Fire, New Mexico, using hyperspectral and multispectral remote sensing. *Remote Sensing for Environment,* 106, 305-325.

Komiyama, A., Moriya, H., Prawiroatmodjo, S., Toma, T. & Ogino, K. (1988) Primary productivity of mangrove forest. *Ogino K, Chihara M (eds), Biological system of mangroves. A report of east Indonesian mangrove expedition 1986, Ehime University,* 97-117.

Lamb, D., Erskine, P. D. & Parrotta, J. A. (2005) Restoration of Degraded Tropical Forest Landscapes. *Science,* 310, 1628-1632.

Langenfelds, R. L., Francey, R. J., Pak, B. C., Steele, L. P., Lloyd, J., Trudinger, C. M. & Allison, C. E. (2002) Interannual growth rate variations of atmospheric CO_2 and its σ ^{13}C, H_2, CH_4, and CO between 1992 and 1999 linked to biomass burning. *Global Biogeochemical Cycles,* 16, 1-21.

Langmann, B. & Heil, A. (2004) Release and dispersion of vegetation and peat fire emissions in the atmosphere over Indonesia 1997/1998. *Atmospheric Chemistry and Physics Discussions,* 4, 2117-2159.

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

Langner, A. & Siegert, F. (2007) The role of fire on land cover changes in Borneo. *Proceedings of the 4th International Wideland Fire Conference, Wildfire 2007, Seville, Spain, 13-17May*.

Langner, A. & Siegert, F. (2009) Spatiotemporal fire occurrence in Borneo over a period of 10 years. *Global Change Biology,* 15, 48-62.

Lecomte, N., Simard, M., Fenton, N. & Bergeron, Y. (2006) Fire severity and long-term ecosystem biomass dynamics in coniferous boreal forests of eastern Canada. *Ecosystems,* 9, 1215-1230.

Lentile, L. (2006) Remote sensing techniques to assess active fire characteristics and post-fire effects. *International Journal of Wildland Fire,* 15, 319-345.

Lepers, E., Lambin, E.F., Janetos, A.C., DeFries, R., Achard, F., Ramankutty, N. & Scholes, R.J. (2005) A synthesis of information on rapid land-cover change for the period 1981-2000. *Bioscience,* 55, 115-124.

Levang, P. (1984) Shifting cultivation for transmigration projests? How "primitive" techniques could help to solve development problems in Central Kalimantan Transmigration Areas. *Ilmu Pert. (Agriculture Science),* 3, 275-283.

Levine, J. S. (1994) Biomass burning and the production of greenhouse gases. *Climate Biosphere Interaction: Biogenic Emissions and Environmental Effects of Climate Change Edited by Righard G.Zepp ISBN 0-471-58943-3 Copyright 1994 John Wiley and Sons, Inc.*

Lewis, S. L. (2006) Tropical forests and the changing earth system. *Philosophical Transactions of the Royal Society B-Biological Sciences,* 361, 195-210.

Lewis, S. L., Phillips, O. L. & Baker, T. R. (2006) Impacts of global atmospheric change on tropical forests. *Trends in Ecology & Evolution*, 21, 173-174.

Li, W. H., Dickinson, R. E., Fu, R., Niu, G. Y., Yang, Z. L. & Canadell, J. G. (2007) Future precipitation changes and their implications for tropical peatlands. *Geophysical Research Letters,* 34, 1-6.

Li, Z., Nadon, S., Cihlar, J. & Stocks, B. (2000) Satellite-based mapping of Canadian boreal forest fires: evaluation and comparison of algorithms. *International Journal of Remote Sensing,* 21, 3071 - 3082.

Loboda, T., O'Neal, K. J. & Csiszar, I. (2007) Regionally adaptable dNBR-based algorithm for burned area mapping from MODIS data. *Remote Sensing of Environment,* 109, 429-442.

Lu, D., Batistella, M., Moran, E. & Mausel, P. (2004) Application of spectral mixture analysis to Amazonian land-use and land-cover classification. *International Journal of Remote Sensing,* 25, 5345-5358.

Lu, D. (2005) Integration of vegetation inventory data and Landsat TM image for vegetation classification in the western Brazilian Amazon. *Forest Ecology and Management*, 213, 369-383.

Lucas, R. M., Honzak, M., Curran, P. J., Foody, G. M., Milne, R., Brown, T. & Amaral, S. (2000) Mapping the regional extent of tropical forest regeneration stages in the Brazilian Legal Amazon using NOAA AVHRR data. *International Journal of Remote Sensing*, 21, 2855-2881.

Lucas, R. M., Honzák, M., Do Amaral, I., Curran, P. J. & Foody, G. M. (2002) Forest regeneration on abandoned clearances in central Amazonia. *International Journal of Remote Sensing,* 23, 965-988

Lucas, R. M., Bunting, P., Cronin, N., Armston, J., Moghaddam, M., Lee, A. & Witte, C. (2005) Integration of radar and Landsat-derived foliage projected cover for woody regrowth mapping, Queensland, Australia. *Remote Sensing of Environment*, 100, 388-406.

Lucas, R., Rowlands, A., Brown, A., Keyworth, S. & Bunting, P. (2007) Rule-based classification of multi-temporal satellite imagery for habitat and agricultural land cover mapping. *Isprs Journal of Photogrammetry and Remote Sensing,* 62, 165-185

MacKinnon, K., G. Hatta, H. Halim, and A. Mangalik (1996) The ecology of Kalimantan. Indonesian Borneo. *Singapore: Periplus Editions*.

Malhi, Y. & Grace, J. (2000) Tropical forests and atmospheric carbon dioxide. *TREE,* 15, 332-337.

Malingreau, J. P., Stephens, G. & Fellows, L., 314-321 (1985). (1985) Remote sensing of forest fires: Kalimantan and North Borneo in 1982-83. *Ambio,* 14, 314-321.

Mansor, S., Tai Hong, W., Rashid, A. & Shariff, M. (2001) Object-oriented classification for land cover mapping. *eCognition Application Notes, 2, 2*.

Miettinen, J. & Liew, S. C. (2005) Connection between fire and land cover change in Southeast Asia: a remote sensing case study in Riau, Sumatra. *International Journal of Remote Sensing,* 26, 1109-1126.

Miettinen, J., Langner, A. & Siegert, F. (2007) Burnt area estimation for the year 2005 in Borneo using multi-resolution satellite imagery. *International Journal of Wildland Fire,* 16, 45-53.

Miettinen, J. & Liew, S. C. (2008) Comparison of multitemporal compositing methods for burnt area detection in Southeast Asian conditions. *International Journal of Remote Sensing,* 29, 1075-1092.

Miettinen, J., Wong, C. M. & Liew, S. C. (2008) New 500m spatial resolution land cover map of the western insular Southeast Asia region. *International Journal of Remote Sensing,* 29, 6075-6081.

Miles, L. & Kapos, V. (2008) Reducing greenhouse gas emissions from deforestation and forest degradation: Global land-use implications. *Science,* 320, 1454-1455.

Miller, J. D. & Thode, A. E. (2007) Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). *Remote Sensing of Environment,* 109, 66-80.

Miyamoto, K., Rahajoe, J. S. & Kohyama, T. (2007) Forest Structure and Primary Productivity in a Bornean Heath Forest. *Biotropica,* 39, 35-42.

Moran, E. F., Brondizio, E. S., Tucker, J. M., da Silva-Forsberg, M. C., McCracken, S. & Falesi, I. (2000) Effects of soil fertility and land-use on forest succession in Amazonia. *Forest Ecology and Management,* 139, 93-108.

Muhamad, N. Z. & Rieley, J. O. (2002) Management of tropical peatlands in Indonesia: mega reclamation project in Central Kalimantan. . *In: J.O. Rieley and S.E. Page, Editors, Peatlands for people: natural resource functions and sustainable management, Proceedings of the International Symposium on tropical peatland, Jakarta, 22-23 August 2001, Agency for the Assessment and Application of Technology and Indonesian Peat Association, Jakarta, Indonesia,* 155-162.

Nascimento, H.E.M. & Laurance, W.F. (2002) Total aboveground biomass in central Amazonian rainforests: a landscape-scale study. *Forest Ecology and Management*, 168, 311-321.

Nasi, R., R., D., Meijaard, E., Applegate, G. & Moore, P. (2001) Forest fire and biological diversity. *FAO Corporate Document Repository*.

Neary, D. G., Klopatek, C. C., DeBano, L. F. & Ffolliott, P. F. (1999) Fire effects on belowground sustainability: a review and synthesis. *Forest Ecology and Management*, 122, 51-71.

Nepstad, D. C., Âssimo, A. V., Alencar, A., Nobre, C., Lima, E., Lefebvre, P., Schlesinger, P., Potterk, C., Moutinho, P., Mendoza, E., Cochrane, M. & Brooks, V. (1999) Large-scale impoverishment of Amazonian forests by logging and fire. *Nature*, 398, 505-508.

Nepstad, D., Lefebvre, P., Da Silva, U. L., Tomasella, J., Schlesinger, P., Solorzano, L., Moutinho, P., Ray, D. & Benito, J. G. (2004) Amazon drought and its implications for forest flammability and tree growth: a basin-wide analysis. *Global Change Biology*, 10, 704-717.

Nightingale, J. M., Phinn, S. R. & Held, A. A. (2004) Ecosystem process models at multiple scales for mapping tropical forest productivity. *Progress in Physical Geography*, 28, 241-281.

Nishimua, T. & Suzuki, E. (2001) Allometric differentiation among tropical tree seedlings in heath and peat-swamp forests. *Journal of Tropical Ecology*, 17, 667-681.

Nishimua, T., Kohyama, T. & Tsuyuzaki, S. (2007) Mortality and growth of trees in peat-swamp and heath forests in Central Kalimantan after severe drought. *Plant Ecology*, 188, 165-177.

NWCG (2008) National Wildfire Coordinating Group Glossary of wildland fire terminology. (http://www.nwcg.gov/).

Nykvist, N. (1996) Regrowth of secondary vegetation after the 'Borneo fire' of 1982-1983. *Journal of Tropical Ecology*, 12, 307-312.

Olschewski, R. & Benitez, P. C. (2005) Secondary forests as temporary carbon sinks? The economic impact of accounting methods on reforestation projects in the tropics. *Ecological Economics*, 55, 380-394.

Osunkoya, O. O., Sheng, T. K., Mahmud, N. A. & Damit, N. (2007) Variation in wood density, wood water content, stem growth and mortality among twenty-seven tree species in a tropical rainforest on Borneo Island. *Austral Ecology*, 32, 191-201.

Page, S. E., Rieley, J.O., Doody, K., Hodgson, S., Husson, S., Jenkins, P.M., Morrough-Bernard, H., Otway, S., Wilshaw, S., (1997) Biodiversity of tropical peat swamp forest: a case study of animal diversity in the Sungai Sebangau catchment in Central Kalimantan, *In: Rieley, J.O. and Page, S.E (eds),* 231-242.

Page, S. E., Rieley, J. O., Shotyk, O. W. & Weiss, D. (1999) Interdependence of peat and vegetation in a tropical peat swamp forest. *Philosophical Transactions of the Royal Society of London Series Biological Sciences,* 354, 1885-1897.

Page, S. E., Siegert, F., Rieley, J. O., Hans-Dieter, V., Boehm, W., Adi, J. & Limin, S. (2002) The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, 420, 61-65.

Page, S. E., Wst, R. A. J., Weiss, D., Rieley, J. O., Shotyk, W. & Limin, S. H. (2004) A record of Late Pleistocene and Holocene carbon accumulation and climate change from an equatorial peat bog (Kalimantan, Indonesia): implications for past, present and future carbon dynamics. *Journal of Quaternary Science*, 19, 625-635.

Page, S. E., Banks, C. J., Rieley, J. & Wust, R. (2008) Extent, significance and vulnerability of the tropical peatland carbon pool: past, present and future prospects. *Proceedings of the 13th International Peat Congress, Tullamore, Ireland, June 2008. International Peat Society*, 233-236.

Page, S. E., Hoscilo, A., Wosten, H., Jauhiainen, J., Silvius, M., Rieley, J., Ritzema, H., Tansey, K., Graham, L., Vasander, H. & Limin, S. (2008) Restoration ecology of lowland tropical peatlands in Southeast Asia: current knowledge and future research directions. *Ecosystems,* online, 1432-9840.

Page , S. E., Hoscilo, A., Langner, A., Tansey, K., Siegert, F., Limin, S. & Rieley, J. (2009) Tropical peatland fires in Southeast Asia. *In: Cochrane et al. (eds) Fire in Tropical Ecosystems. Springer Verlag*.

Page, S. E., Banks, C. J. & Rieley, J.O. Carbopeat; Technical Report 1: Extent and Global Significance of Tropical Peat Carbon Pools. Unpublished report.

Phua, M., Tsuyuki, S., Lee, J. & Sasakawa, H. (2007) Detection of burned peat swamp forest in a heterogeneous tropical landscape: A case study of the Klias Peninsula, Sabah, Malaysia. *Landscape and Urban Planning*, 82, 103-116.

Phua, M. H., Tsuyuki, S., Furuya, N. & Lee, J. S. (2008) Detecting deforestation with a spectral change detection approach using multitemporal Landsat data: A case study of Kinabalu Park, Sabah, Malaysia. *Journal of Environmental Management,* 88, 784-795.

Pugnaire, F. I. & Valladares, F. (2007) Functional plant ecology, Edition 2 *Published by CRC Press*, 114.

Ramankutty, N., Gibbs, H. K., Achard, F., Defries, R., Foley, J. A. & Houghton, R. A. (2007) Challenges to estimating carbon emissions from tropical deforestation. *Global Change Biology,* 13, 51-66.

Reijnders, L. & Huijbregts, M. A. J. (2008) Palm oil and the emission of carbon-based greenhouse gases. *Journal of Cleaner Production,* 16, 477-482.

Riano, D., Moreno Ruiz, J. A., Isidoro, D. & Usin, S. L. (2007) Global spatial patterns and temporal trends of burned area between 1981 and 2000 using NOAA-NASA Pathfinder. *Global Change Biology,* 13, 40-50.

Richards, P. W. W (1996) The tropical rain forest and ecological study, Second edition *The university Press, Cambridge.*

Rieley, J. O. & Page, S. E. (2005) Wise Use of Tropical Peatlands: Focus on Southeast Asia. *Synthesis of results and conclusions of the UK Darwin Initiative and the EU INCO EUTROP, STRAPEAT AND RESTORPEAT.*

Rieley, J., Wüst, R. A. J., Jauhiainen, J., Page, S. E., Wösten, H., Hooijer, A., Siegert, F., Limin, S., Vasander, H. & Stahlhut, M. (2008) Tropical Peatlands: Carbon stores, Carbon Gas Emissions and Contribution to Climate Change Processes *Peatlands and Climate Change;13th International Peat Congress. Tullamore, Ireland 8-13 June 2008*, 1-30.

Roberts, G., Wooster, M. J. & Perry, G. W. (2004) High temporal resolution fire radiative energy and biomass combustion estimates derived from MSG SEVIRI *Proc. Second MSG RAO Workshop, Salzburg, Austria 9 - 10 September 2004, ESA SP-582, November 2004*.

Roberts, G., Wooster, M. J., Perry, G. L. W., Drake, N., Rebelo, L. M. & Dipotso, F. (2005) Retrieval of biomass combustion rates and totals from fire radiative power observations: Application to southern Africa using geostationary SEVIRI imagery. *Journal of Geophysical Research,* 110, D21111.

Robichaud, P. R., Lewis, S. A., Laes, D. Y. M., Hudak, A. T., Kokaly, R. F. & Zamudio, J. A. (2007) Postfire soil burn severity mapping with hyperspectral image unmixing. *Remote Sensing of Environment,* 108, 467-480.

Rogan, J. & Chen, D. (2004) Remote sensing technology for mapping and monitoring land-cover and land-use change *Progress in Planning,* 61, 301-325.

Roldan-Zamarron, A., Merino-De-Miguel, S., Gonzalez-Alonso, F., Garcia-Gigorro, S. & Cuevas, J. M. (2006) Minas de Riotinto (south Spain) forest fire: Burned area assessment and fire severity mapping using Landsat 5-TM, Envisat-MERIS, and Terra-MODIS postfire images. *Journal of Geophysical Research-Biogeosciences,* 111.

Roy, D. P., Jin, Y., Lewis, P. E. & Justice, C. O. (2005) Prototyping a global algorithm for systematic fire-affected area mapping using MODIS time series data. *Remote Sensing of Environment,* 97, 137-162.

Roy, D., Boschetti, L. & Trigg, S. (2006) Remote Sensing of Fire Severity: Assessing the Performance of the Normalized Burn Ratio. *IEEE Geoscience and Remote Sensing Letters,* 3, 112-116.

Roy, D., Boschetti, L. & Justice, C. (2008) The collection 5 MODIS burned area product - Global evaluation by comparison with the MODIS active fire product *Remote Sensing of Environment*, 112, 3690-3707.

Schroeder, W., Csiszar, I. & Morisette, J. (2008) Quantifying the impact of cloud obscuration on remote sensing of active fires in the Brazilian Amazon. *Remote Sensing of Environment,* 112, 456-470.

Siegert, F. & Ruecker, G. (2000) Use of multitemporal ERS-2 SAR images for identification of burned scars in south-east Asian tropical rainforest. *International Journal of Remote Sensing,* 21, 831-837.

Siegert, F., Ruecker, G., Hinrichs, A. & Hoffmann, A. A. (2001) Increased damage from fires in logged forests during droughts caused by El Nino. *Nature,* 414, 437-440.

Silva, J. M. N., Sá, A. C. L. & Pereira, J. M. C. (2005) Comparison of burned area estimates derived from SPOT-VEGETATION and Landsat ETM+ data in Africa: Influence of spatial pattern and vegetation type. *Remote Sensing of Environment,* 96, 188-201.

Simon, M., Plummer, S., Fierens, F., Hoelzemann, J. J. & Arino, O. (2004) Burnt area detection at global scale using ATSR-2: The GLOBSCAR products and their qualification. *Journal of Geophysical Research-Atmospheres,* 109.

Skutsch, M., Bird, N., Trines, E., Dutschke, M., Frumhoff, P., de Jong, B. H. J., van Laake, P., Masera, O. & Murdiyarso, D. (2007) Clearing the way for reducing emissions from tropical deforestation. *Environmental Science & Policy,* 10.

Slik, J. W. F., Verburg, R. W. & Keßler, P. J. A. (2002) Effects of fire and selective logging on the tree species composition of lowland dipterocarp forest in East Kalimantan, Indonesia. *Biodiversity and Conservation,* 11, 85-98.

Slik, J. W. F. & Eichhorn, K. A. O. (2003) Fire survival of lowland tropical rain forest trees in relation to stem diameter and topographic position. *Oecologia*, 137, 446-455.

Slik, J., Bernard, C., Beek, M., Breman, F. & Eichhorn, K. (2008) Tree diversity, composition, forest structure and aboveground biomass dynamics after single and repeated fire in a Bornean rain forest. *Oecologia,* 158, 579-588.

Slik, J. W. F., Bernard, C. S., Breman, F. C., Van Beek, M., Salim, A. & Sheil, D. (2008) Wood Density as a Conservation Tool: Quantification of Disturbance and Identification of Conservation-Priority Areas in Tropical Forests. *Conservation Biology,* 22, 1299-1308.

Smith, A. M., Wooster, M. J., Drake, N. A., Dipotso, F. M., Falkowski, M. J. & Hudak, A. T. (2005) Testing the potential of multi-spectral remote sensing for retrospectively estimating fire severity in African Savannahs. *Remote Sensing of Environment,* 97, 92-115.

Smith, A. M., Wooster, M. J., Drake, N. A., Dipotso, F. M. & Perry, G. L. (2005) Fire in African savanna: Testing the impact of incomplete combustion on pyrogenic emissions estimates. *Ecological Applications,* 15, 1074-1082.

Smith, A. M. S. & Wooster, M. J. (2005) Remote classification of head and backfire types from MODIS fire radiative power and smoke plume observations. *International Journal of Wildland Fire,* 14, 249-254.

Smith, A. M. S., Lentike, L. B., Hudak, A. T. & Morgan, P. (2007) Evaluation of linear spectral unmixing and DNBR for predicting postfire recovery in a North American ponderosa pine forest. *International Journal of Remote Sensing,* 28, 5159-5166.

Smith, R. M., Adams, S., Maier, R., Craig, A. K. & Maling, I. (2007) Estimating the area of stubble burning from the number of active fires detected by satellite. *Remote Sensing of Environment*, 109, 95-106.

Sobrino, J. A. & Romaguera, M. (2004) Land surface temperature retrieval from MSG1-SEVIRI data. *Remote Sensing of Environment,* 92, 247-254.

Song, C. & Woodcock, C. E. (2002) The spatial manifestation of forest succession in optical imagery: The potential of multiresolution imagery. *Remote Sensing of Environment,* 82, 271-284.

Souza, C., Firestone, L., Silva, L. M. & Roberts, D. (2003) Mapping forest degradation in the Eastern Amazon from SPOT 4 through spectral mixture models. *Remote Sensing of Environment,* 87, 494-506.

Spessa, A., Weber, U., Langner, A., Siegert, F. & Heil, A. (2009) Fire in the vegetation and peatlands of Borneo, 1997-2007: Patterns, Drivers and Emissions. *EGU General Assembly, 2009.*

Stibig, H. J., Beuchle, R. & Janvier, P. (2002) Forest cover map of Insular Southeast Asia at 1:5 500 000, derived from SPOT-VEGETATION satellite images, TREES Tropical Ecosystem Environment Observations by satellites. *TREES Publications Series D: Thematic outputs,* 3.

Stibig, H. J., Beuchle, R. & Achard, F. (2003) Mapping of the tropical forest cover of insular Southeast Asia from SPOT4-Vegetation images. *International Journal of Remote Sensing,* 24, 3651-3662.

Stibig, H. J., Belward, A. S., Roy, P. S., Rosalina-Wasrin, U., Agrawal, S., Joshi, P. K., Hildanus, R., Beuchle, S., Fritz, S., Mubareka, C. & Giri (2007) A land-cover map for South and Southeast Asia derived from SPOT-VEGETATION data. *Journal of Biogeography,* 34, 625-637.

Sulistiyanto, J. (2004) Nutrient dynamics in different sub-types of peat swamp forest in Central Kalimantan, Indonesia. *PhD Thesis, University of Nottingham.*

Suzuki, E. (1999) Diversity in specific gravity and water content of wood among Borean tropical rainforest trees. *Ecological Research,* 14, 211-224.

Tacconi, L. (2006) Slash and bum and fires in Indonesia: A comment. *Ecological Economics,* 56, 1-4.

Tansey, K. J., Gregoire, J. M., Stroppiana, D., Sousa, A., Silva, J., Pereira, J. M. C., Boschetti, L., Maggi, M., Brivio, P. A., Fraser, R., Flasse, S., Ershov, D., Binaghi, E., Graetz, D. & Peduzzi, P. (2004) Vegetation burning in the year 2000: Global burned area estimates form SPOT VEGETATION data. *Journal of Geophysical Research-Atmospheres,* 109, 1-22.

Tansey, K., Beston, J., Hoscilo, A., Page, S. E. & Paredes-Hernández, C. U. (2008) The relationship between MODIS fire hotspot count and burned area in a degraded tropical peat swamp forest in Central Kalimantan, Indonesia. *Journal of Geophysical Research - Atmospheres,* 113, 1-8.

Tansey, K., Gregoire, J. M., Defourny, P., Leigh, R., Pekel, J.-F., van Bogaert, E. & Bartholome, E. (2008) A new, global, multi-annual (2000–2007) burnt area product at 1 km resolution. *Geophysical Research Letters* 35, 1-6.

Toma, T., Ishida, A. & Matius, P. (2005) Long-term monitoring of post-fire aboveground biomass recovery in a lowland dipterocarp forest in East Kalimantan, Indonesia. *Nutrient Cycling in Agroecosystems,* 71, 63-72.

Trabucco, A., Zomer, R. J., Bossio, D. A., van Straaten, O. & Verchot, L. V. (2008) Climate change mitigation through afforestation/reforestation: A global analysis of hydrologic impacts with four case studies. *Agriculture Ecosystems & Environment,* 126, 81-97.

Trigg, S. N., Curran, L. M. & McDonald, A. K. (2006) Utility of Landsat 7 satellite data for continued monitoring of forest cover change in protected areas in Southeast Asia. *Singapore Journal of Tropical Geography,* 27, 49-66.

Tutin, C. E. G., White, L. J. T. & Mackanga-Missandzou, A. (1996) Lightning strike burns large forest tree in the Lope Reserve, Gabon *Global Ecology and Biogeography Letters,* 5, 36-41.

Uhl, C. & Kauffman, J. B. (1990) Deforestation, Fire Susceptibility, and Potential Tree Responses to Fire in the Eastern Amazon. *Ecology,* 71, 437-449.

UNFCCC (2007) Reducing emissions from deforestation in developing countries: approaches to stimulate action. *Decision 2/CP,*13.

Usup, A., Hashimoto, Y., Takahashi, H. & Thayasaka, H. (2004) Combustion and thermal characteristics of peat fire in the tropical peatlands in Central Kalimantan, Indonesia. *Tropics,* 14, 1-18.

van der Lee, G. & Haasnoot, M. (2006) Manual HABITAT. *RWS, WL/Delft Hydraulics*.

van der Werf, G. R., Randerson, J. T., Giglio, L., Collatz, G. J., Kasibhatla, P. K. & Arellano, A. F. J. (2006) Interannual variability of global biomass burning emissions from 1997 to 2004. *Atmospheric Chemistry and Physics Discussions,* 6, 3175-3226.

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

van Eijk, P. & Leenman, P. H. (2004) Regeneration of Fire Degraded Peat Swamp Forest in Berbak National Park and Implementing Replanting Programmes. Water for Food & Ecosystems Programme on: Promoting the river basin and ecosystem approach for sustainable management of SE Asian lowland peat swamp forests. http://www.waterfoodecosystems.nl.

van Leeuwen, W. J. D. (2008) Monitoring the effects of forest restoration treatments on post-fire vegetation recovery with MODIS multitemporal data. *Sensors,* 8, 2017-2042.

van Nieuwstadt, M. G. L., Sheil, D. & Kartawinata, K. (2001) The ecological consequences of logging in the burned forests of East Kalimantan, Indonesia. *Conservation Biology,* 15, 1183-1186.

van Nieuwstadt, M. G. L. (2002) Trial by fire, Postfire development of a tropical dipterocarp forest. *PhD Thesis, Faculty of Biology, University of Utrecht, The Netherlands,* 45-66.

van Nieuwstadt, M. G. L. & Sheil, D. (2005) Drought, fire and tree survival in a Borneo rain forest, East Kalimantan, Indonesia. *Journal of Ecology,* 93, 191-201.

van Wagtendonk, J. W., Rootb, R. R. & Key, C. H. (2004) Comparison of AVIRIS and Landsat ETM+ detection capabilities for burn severity. *Remote Sensing of Environment* 92, 397-408.

Vieira, I. C. G., de Almeida, A. S., Davidson, E. A., Stone, T. A., Reis de Carvalho, C. J. & Guerrero, J. B. (2003) Classifying successional forests using Landsat spectral properties and ecological characteristics in eastern Amazonia. *Remote Sensing of Environment,* 87, 470-481.

Waldes, N. J. L. & Page, S. E. (2001) Forest structure and tree diversity of a peat swamp forest in Central. Kalimantan, Indonesia. *Jakarta symposium proceeding on peatlands for people natural resources function and sustainable management, Rieley, J. and Page S. (Eds),* 16-22.

Walz, Y. (2007) Classification of burn severity using Moderate Resolution Imaging Spectroradiometer (MODIS): A case study in the jarrah-marri forest of southwest Western Australia. *Journal of Geophysical Research-Biosciences,* 112, G02002.

Wang, Y. H., Field, R. D. & Roswintiarti, O. (2004) Trends in atmospheric haze induced by peat fires in Sumatra Island, Indonesia and El Nino phenomenon from 1973 to 2003. *Geophysical Research Letters,* 31.

Whiteside, T. & Ahmad, W. (2005) A comparison of object-oriented and pixel-based classification methods for mapping land cover in Northern Australia. *Proceedings of SSC2005 Spatial intelligence, innovation and praxis: The national biennial Conference of the Spatial Sciences Institute September 2005. Melbourne*.

Whitmore, T. C. (1989) Canopy gaps and the two major groups of forest trees. *Ecology,* 70, 536-538.

Wimberly, M. & Reilly, M. (2007) Assessment of fire severity and species diversity in the southern Appalachians using Landsat TM and ETM plus imagery. *Remote Sensing of Environment,* 108, 189-197.

Wolter, K. & Timlin, M. S. (1993) Monitoring ENSO in COADS with a seasonally adjusted principal component index. *Proc. of the 17th Climate Diagnostics Workshop, Norman OK, NOAA/N MC/CAC, NSSL, Oklahoma Clim. Survey, CIMMS and the School of Meteorology, University of Oklahoma*, 52-57.

Wolter, K. & Timlin, M. S. (1998) Measuring the strength of ENSO events - how does 1997/98 rank? *Weather*, 53, 315-324.

Woods, P. (1989) Effects of logging, drought, and fire on structure and composition of tropical forest in Sabah, Malaysia. *Biotropica*, 21, 290-298.

Wooster, M. J., Roberts, G., Perry, G. L. W. & Kaufman, Y. J. (2005) Retrieval of biomass combustion rates and totals from fire radiative power observations: FRP derivation and calibration relationships between biomass consumption and fire radiative energy release. *Journal of Geophysical Research-Atmospheres,* 110.

Wosten, J. H. M., van Denberg, J., van Eijk, P., Gevers, G. J. M., Giesen, W. B. J. T., Hooijer, A., Idrisi, A., Leenman, P. H., Rais, D. S., Siderius, C., Silvius, M. J., Suryadiputra, N. & Wibisono, I. T. (2006) Interrelationships between Hydrology and Ecology in Fire Degraded Tropical Peat Swamp Forests. *Water Resources Development,* 22, 157-174.

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

Wotawa, G., De Geer, L. E., Becker, A., D'Amours, R., Jean, M., Servranckx, R. & Ungar, K. (2006) Inter- and intra-continental transport of radioactive cesium released by boreal forest fires. *Geophysical Research Letters,* 33.

WWF (2008) The contribution made by emissions from tropical peatland fires is frequently omitted from global carbon flux models. Similar situation seems to be with the carbon gain in the secondary regrowth. This is due to lack of understanding of fire behaviour and sufficient data on post-fire regeneration. *WWF Indonesia technical Report*.

Wyrtki, K. (1975) El Niño - The Dynamic Response of the Equatorial Pacific Ocean to Atmospheric Forcing. *Journal of Physical Oceanography,* 5, 572-584.

Xiao, X., Boles, S., Frolking, S., W., S., Moore, B., Li, C., He, L. & Zhao, R. (2002) Observation of flooding and rice transplanting of paddy rice fields at the site to landscape scales in China using VEGETATION sensor data *International Journal of Remote Sensing,* 23, 3009-3022.

Yeager, C. P., Marshall, A. J., Stickler, C. M. & Chapman, C. A. (2003) Effects of fires on peat swamp and lowland dipterocarp forests in Kalimantan, Indonesia. *Tropical Biodiversity,* 8, 121-138.

Yijun, C. & Ali Hussin, Y. (2003) Object-oriented classifier for detection Tropical Deforestation using Landsat ETM+ in Berau, East Kalimantan, Indonesia. *Map Asia Conference 2003*.