ISHAK YASSIR

SOIL ORGANIC CARBON STOCKS AND CHANGES UPON FOREST REGENERATION IN EAST KALIMANTAN-INDONESIA

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Soil organic carbon stocks and changes upon forest regeneration in East Kalimantan-Indonesia

Ishak Yassir

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For my supervisors and mentors:

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Dr. Bram van Putten
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Abstract

Imperata grassland is a common vegetation type in Kalimantan (Indonesia), and other parts of South-East Asia. It indicates a high degree of degradation of the vegetation, and mostly occurs after slashing and burning of primary forest. Through secondary succession *Imperata* grassland is converted into new secondary forest and much of the original biodiversity is restored. The overall objective of the thesis was to study the regeneration of *Imperata* grasslands in East Kalimantan, and to measure the effects of regeneration on soil properties, with emphasis on the organic fraction. The research strategy was to compare plots of different regeneration stages, characterized by the period elapsed since the vegetation was last burned.

Results show that during regeneration of *Imperata* grasslands, both vegetation composition and soil properties change, including chemistry of soil organic matter. Soil carbon stocks are higher under *Imperata* grasslands than under primary forest, and increase further upon natural regeneration of grassland to secondary forest. Highest carbon stocks are found in the later regeneration phases. Lower carbon stocks under primary forests are due to extremely low fertility, combined with shallow soils and low root mass in the topsoil. Root density as observed in the field is much higher under the grass vegetation. Results show as well that soil organic matter decomposition is most advanced under forest, as indicated by lower amounts of plant derived compounds and higher contribution of microbial matter. The results indicate that decomposition efficiency is related to soil organic matter chemistry, but more to abundance of N-compounds than to that of potentially recalcitrant compounds.

In our case study, soil texture appears an important factor in the vegetation succession. On sandy soils, there is a strong increase with time of *Pteridium aquilinum* L., while the number of other species is lower. This slows down the development towards secondary forest. Canonical correspondence analysis (CCA) of environmental factors and vegetation show that pH, bulk density, sand and clay are the factors related to the distribution of species. The rapid secondary succession indicates that *Imperata* grasslands are not a final and stable stage of land degradation, but that frequent fires are necessary to maintain *Imperata* grasslands. If protected from fire and other intrusions such as shifting cultivation, *Imperata* grassland will readily develop into secondary forest. *Imperata* grasslands seem to be permanent because of human interference, especially through burning, and because so far few attempts have been made to sustainable rehabilitation.

List of Abbreviations

Abbreviations		Descriptions
BOS	:	Borneo Orangutan Survival Foundation
C	:	Carbon
CAM	:	Crassulacean Acid Metabolism
CCA	:	Canonical Correspondence Analysis
CDM	:	Clean Development Mechanism
DCA	:	Detrended Correspondence Analysis
ENSO	:	El Nino Southern Oscillation
FAO	:	Food and Agricultural Organization
FORDA	:	Forest Research and Development Agency
FWI/GFW	:	Forest Watch Indonesia/Global Forest Watch
g	:	Gram
GC/MS	:	Gas Chromatography/Mass Spectrometry
GD	:	Geometrical Design
GPS	:	Global Positioning System
Gt	:	Giga Ton (=1,000,000,000 ton)
Ha	:	Hectare
IPCC	:	Intergovernmental Panel on Climate Change
IRMS	:	Isotope Ratio-Mass Spectrometry
kg	:	Kilogram
LSD	:	Least Significant Difference

Abbreviations	Descriptions
MoF	: Ministry of Forestry
PDB	: Pee Dee Belemnite
Pg	: Petagram (10^{15} g)
RD	: Random Design
$REDD^+$	: Reducing Emission Deforestation and Degradation-Enhancing Forest Carbon Stock in Developing Countries
Sd	: Standard Deviation
SD	: Systematic Design
SOC	: Soil Organic Carbon
SOM	: Soil Organic Matter
SAP	: Spatial Average Procedure
SOP	: Spatial Optimal Procedure
SSAC	: Soil Survey Analytical Continuum
TD	: Transect Design
WD	: Wide Design

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Soil organic carbon stocks and changes upon forest regeneration in East Kalimantan-Indonesia

CHAPTER 1

General Introduction

1.1. General introduction

Indonesia is an essential play-actor in the global climate change arena. Indonesia has the last remaining large tracts of tropical forest in Asia, and is one of the key global centres of terrestrial biodiversity (Barber, 2002). There are about 130 million ha of designated forestlands, of which about 42 million ha are intact or primary forests (MoF, 2012). From the biodiversity point of view, Indonesia's forests contain about 10% of the world's flowering plant species, 12% of mammalian species, and 17% of the reptilian and amphibian species (Matthew et al. 2002). On the other hand, forests in Indonesia at the national level as well as at the local level are under increasing pressure from population growth, illegal logging, forest fire, land use change for agriculture, transmigration and estate crops such as timber and oil palm. The increasing pressure on forest has dramatically increased the rate of deforestation, from 800,000 ha year^{-1} in 1993 (Revilla, 1993), to 1.20 million ha year^{-1} in 1996 (MoF-FAO, 1996), to 1.67 million ha year^{-1} between 1984 and 1997 (World Bank, 2000), to above 2 million ha year^{-1} (FWI/GFW, 2002). The present level decreased to about 1.17 million ha year^{-1} between 2003 and 2006 (MoF, 2008).

Kalimantan, the Indonesian part of Borneo, covering about 73% of the island, has one of the most important tropical forests in the world, although large areas of primary forest have also been changed into secondary forest, oil palm plantation, timber estate plantation, slash-and-burn agriculture, and unproductive grasslands such as *Imperata cylindrica.* In Kalimantan, *Imperata* grasslands that include associations with ferns and shrubs, cover an estimated 2.2 million ha (Garrity et al. 1997). In East Kalimantan alone, the area of forest cover decreased from approximately 11.1 million ha in 1997 to approximately 9.3 million ha in 2003 (Harris et al. 2008).

In Indonesia, the name of *Imperata* grassland is alang-alang. Common names for *Imperata cylindrica* are cagongrass, japgrass, speargrass and bladygrass (MacDonald, 2004). Furthermore, MacKinnon et al. (1996) mention that *Imperata* grasslands are caused by human activities such as logging, forest clearing for shifting cultivation, agriculture and grazing, and also by fire. The latter is a frequent result of human interference. Older literature suggested that spontaneous regeneration did not occur in *Imperata* grasslands, but when *Imperata* grassland are abandoned and not burned regularly, they will undergo a process called secondary succession, encompassing a series of vegetation changes. In the early phase of secondary succession, ferns, herbs, lianas and young trees (pioneer species) rapidly colonize the site and gradually replace *Imperata* grassland. Leps (1987) mentions that this early stage of succession influences later stages of vegetation development, which in their turn determine the character of the secondary forest and the recovery of the original biodiversity.

Apart from affecting forest area and biodiversity, recuperation of *Imperata* grasslands affects soil properties, most notably the aboveground and belowground carbon stocks. While it is well understood that carbon in the standing biomass is much higher in forests than in grasslands, this is not necessarily the case for soil carbon stocks. Soil carbon stocks can equal or even exceed that of the standing biomass, and soil carbon stocks changes are therefore of direct importance for climate change. Accurate information on soil carbon stocks is a crucial issue concerning improved management of agricultural land or reforestation in relation to reduction of CO_2 emission, and in understanding the impact of regional land use change on the global carbon cycle. These information is also very important to provide baseline data related to carbon credits following the Clean Development Mechanism (CDM) or REDD$^+$ (Reducing Emission Deforestation and Degradation, and Enhancing Forest Carbon Stock in Developing Countries) schemes, particularly to provide or improve methodology for reliable estimation of soil carbon stocks.

1.2. Objectives of this thesis

The overall objective of this thesis was to study the regeneration of *Imperata* grasslands in East Kalimantan-Indonesia, and to measure the effects of regeneration on soil properties, with emphasis on the organic fraction. The research strategy was to compare plots of different regeneration stages, characterized by the last year that the vegetation was burned. Samples were taken in 2007, of soils on which the vegetation was last burned in 2004, in 2003, and in 1997/1998. Included were also a secondary forest with an approximate regeneration time of 15 years, and a primary forest. We investigated vegetation changes, soil properties including soil organic carbon stocks, SOM chemistry and dynamics (C-replacement) in relation to regeneration stages. The sequence of research questions is the following: (a) how does the species community develop after fire?; (b) how is the quantitative development of carbon contents and stocks under vegetation succession?; (c) how fast is decay and build-up of soil organic matter fraction contributed by C3 and C4 vegetation and do carbon dynamics throw a light on differences in carbon stocks?; (d) is there a relation between carbon stocks, SOM chemistry, and carbon dynamics?; (e) using geo-referenced information, how accurate are carbon stock estimates for larger area under *Imperata* grassland and which estimation procedure is optimal?

1.3. Study area

The study areas Sungai Wain and BOS Samboja Lestari are situated in East Kalimantan, Indonesia (Figure 1.1). Sungai Wain is a unique protected forest of about 10.000 hectares that contains one of the last unburned primary forests of the Balikpapan – Samarinda area (Whitehouse and Mulyana, 2004). Samboja Lestari is a 1850 hectares reforestation project owned by the Borneo Orangutan Survival Foundation (BOS). Plots selected for the analysis of regeneration impacts are situated at Samboja Lestari, whereas the primary forest plots from Sungai Wain function as a control. The Köppen system classifies the climate of the research area as Af (Tropical Rainforest). Average yearly precipitation is 2250 mm, with a wet period from December to May. The driest month has an average precipitation of 132 mm, and the wettest month of 231 mm. The daily maximum temperature varies from 23 °C to 31 °C and relative humidity is high. The soils in both areas are formed on marine sediments. Topsoils are generally slightly coarser than the deeper layers. In the FAO classification system (FAO, 2001) the soils of Samboja Lestari and Sungai Wain are classified as Acrisols. Nutrient levels are low, especially that of available phosphorus, pH values vary between 4.09 and 4.55 (Yassir and Omon, 2006).

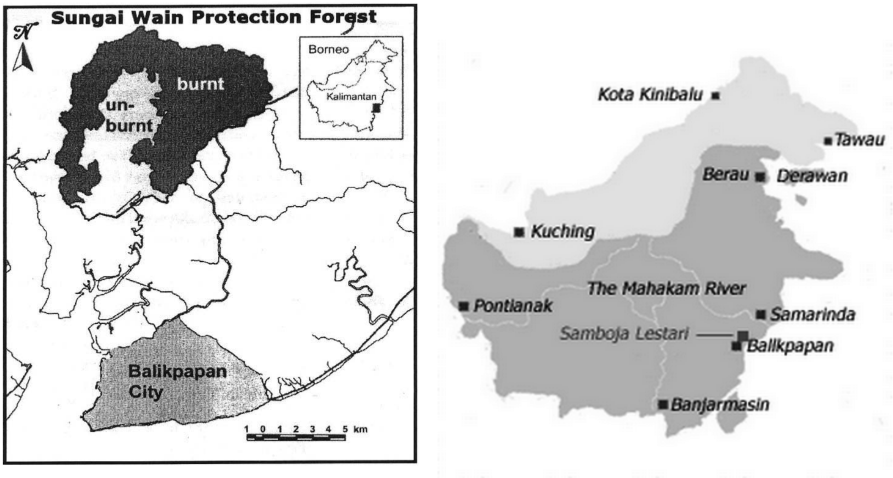

Figure 1.1 Location of Sungai Wain protection forest (Slik, 2006) and BOS Samboja Lestari (BOSF, 2007)

1.4. Outline of the thesis

This thesis is a collection of five major chapters that were written as independent papers. For this reason, some overlaps were unavoidable. The papers are structured in the sequence outlined above, which is also mainly their chronological sequence. Each paper forms a basis for the subsequent ones.

Chapter 1 presents a general introduction.

Chapter 2 analyses the pathways of secondary succession after fire in *Imperata* grasslands of East Kalimantan, Indonesia. The relation between community structure, pattern and environmental gradients or soil properties is also discussed.

Chapter 3 deals with the effect of forest regeneration after fire in *Imperata* grasslands on soil properties, with emphasis on soil carbon stocks. General chemical trends are explored by factor analysis and causes of change of carbon stocks are discussed.

Chapter 4 uses stable C isotopes to measure the influence of C3 and C4 litter on soil organic matter. A relation between accumulation and replacement is explored and discussed.

Chapter 5 explores changes in molecular chemistry of soil organic matter in relation to vegetation change, in order to decide whether changes in carbon stocks depend on chemical characteristics of litter. The used methodology, pyrolysis-gas chromatography/mass spectrometry (GC/MS) allows a detailed interpretation of SOM chemistry.

Chapter 6 deals with reliability of carbon stock estimate in *Imperata* grassland, using geo-referenced information. The chapter describes the spatial variability of soil carbon stocks using a sampling design consisting of transects, and resulting in geo-referenced information at various scales. Various carbon stock estimation procedures are discussed and recommendations for efficient sampling in similar areas to estimate carbon stocks are presented.

Chapter 7 finally presents conclusions and recommendations: a synthesis of soil carbon stocks and changes upon forest regeneration.

CHAPTER 2

Secondary succession after fire in *Imperata* grasslands of East Kalimantan, Indonesia

Chapter based on:

Yassir, I., Van der Kamp, J., Buurman, P. 2010. Secondary succession after fire in *Imperata* grasslands of East Kalimantan, Indonesia. Agriculture, Ecosystems and Environment, 137: 172-182.

Abstract

Regeneration of grassland areas is becoming increasingly important, not only to create new secondary forest and recover the original biodiversity, but also recover for agriculture. We studied an early succession in *Imperata* grasslands in East Kalimantan, Indonesia, using 47 plots that last burned 3 years previous, 94 that burned 4 years previous, 81 plots that burned 9 years previous, 41 plots of secondary forest, the age of which is more than 9 years, and 28 plots of primary forest. The species coverage data were analyzed using CANOCO. While *Imperata* decreases, the average percentage of shrubs and young trees clearly increases with time. In the burned plots, *Melastoma malabathricum*, *Eupatorium inulaefolium, Ficus* sp., and *Vitex pinnata L.* strongly increase with the age of regeneration, but these species were rare in the secondary forest. Texture strongly influenced regeneration: soils with more than 50% sand had a slower development towards secondary forest. The number of species was lower in the more sandy soils. The latter showed a stronger increase with time of *Pteridium aquilinum* L., which appears to slow down the subsequent vegetation development. Canonical correspondence analysis (CCA) of the environmental gradient and vegetation showed that pH, bulk density, sand and clay are the factors influencing the distribution of species. CCA showed also that soil properties had a strong influence on vegetation composition. *Melastoma malabathricum, Vitex pinnata* L.*, Lycopodium cernum, Vernonia arborea* Buch.-Ham., *Dicranopteris linearis* are all species associated with high levels of exchangeable Al and low pH. *Imperata* grasslands are not a final and stable stage of land degradation, but, when not maintained by frequent fires and human disturbances, regenerate spontaneously and rapidly to secondary forest. The introduction of native shrubs and trees will speed up this process. Recovery for agriculture has not been studied but should not pose major problems under management system without fire.

Keywords: *agriculture*, *early succession, human disturbances, Imperata grassland*, *Pteridium aquilinum.*

2.1. **Introduction**

Kalimantan, the Indonesian part of Borneo, covers about 73% of the land area of the island, and has one of the important tropical forests in the world. Nowadays, large areas of primary forest in Kalimantan have been changed into secondary forest, oil palm plantation, timber estate plantation, slash-and-burn agriculture, and also grasslands such as *Imperata cylindrica.* According to the latest estimate, *Imperata* grasslands in Kalimantan, which include associations of *Imperata* with ferns and shrubs cover an estimated 2.2 million ha (Garrity et al.1997). In East Kalimantan alone, the forest area decreased from approximately 11.1 million ha in 1997 to approximately 9.3 million ha in 2003 (Harris et al. 2008).

In Indonesia, the name of *Imperata* grassland is alang-alang. Common names for *Imperata cylindrica* are cagongrass, japgrass, speargrass and bladygrass (MacDonald, 2004). MacKinnon et al. (1996) mentioned that *Imperata* grasslands are caused by human activities such as logging, forest clearing for shifting cultivation, agriculture and grazing, and also by fire. The latter is a frequent result of human interference. When *Imperata* grassland are abandoned and not burned regularly, they will undergo a series of vegetation changes, a process called secondary succession. In the early phase of secondary succession, ferns, herbs, lianas and young trees (pioneer species) rapidly colonize the site. Leps (1987) mentioned that this early stage of succession influences the later stages of vegetation development, which in their turn determine the character of the secondary forest and the recovery of the original biodiversity.

Although the direction of the (early) secondary succession in *Imperata* grasslands is important, this aspect was hardly investigated in Indonesia. Most studies in Indonesia focused on tropical secondary forest (Brealey et al. 2004; Bischoff et al.

2005). Okimori and Matius (2000) described the secondary forest succeeding traditional slash-and-burn agriculture, and Kiyono and Hastaniah (2000) studied the role of slash-and-burn agriculture in transforming dipterocarp forest into *Imperata* grassland. Some studies described the effect of fire on tree species composition of lowland dipterocarp forest (Ohtsuka, 1999; Matius et al. 2000; Slik et al. 2002; Slik and Eichhorn, 2003; Hiratsuka et al. 2006). *Imperata* grasslands in Lampung, Indonesia consisted for 90-100% of *Imperata*, with the following associated plants: *Eupatorium odoratum, Bredelia monoica*, *Melastoma affine*, *Mimosa invisa* and *Saccharum spontaneum* (Eussen and Wirjahardja, 1973). Kiyono and Hastaniah (1997) reported in their study in East Kalimantan that one hectare of *Imperata* grassland contained up to 107 plant species, including trees such as *Vernonia arborea, Cratoxylum formusum* and *Vitex pinnata*. Hashimotio et al. (2000) reported that after 10-12 years of fallow, the dominant species in regenerated lowland forest in Borneo were *Piper aduncum*, *Ficus* sp., *Geunsia pentandra, Vernonia arborea, Melastoma malabathricum, Macaranga* sp., and *Bridelia glauca.* Hiratsuka et al. (2006) reported that after the forest fire of 1998 in East Kalimantan, the dominant pioneer species were *Homalantus populneus, Macaranga gigantea Macaranga hypoleuca*, *Mallotus paniculatus, Melastoma malabathricum, Piper aduncum* and *Trema orientalis*. All these species, described by Eussen and Wirjahardja (1973), Kiyono and Hastaniah (1997), Hashimotio et al. (2000) and Hiratsuka et al. (2006) were also identified during our field research.

Soil properties also change during secondary succession. Upon burning, pH initially increases due to production of carbonates upon ashing of vegetation. With time, the carbonates are leached and exchangeable cations (especially Ca) are lost, resulting in a decline of pH (Binkley et al. 1989; Cruz and del Castillo, 2005; Farley et al.

2008; Yamashita et al. 2008; Van der Kamp et al. 2009). Van der Kamp et al. (2009) described changes of soil carbon stocks under secondary succession, using the same plots as used in the present paper. The present paper describes the pathways of secondary succession in *Imperata* grasslands of East Kalimantan, Indonesia. We compared of plots that last burned in 2004, in 2003, in 1997/1998, secondary forest, and primary forest. Because the observations were made in early 2007, the regeneration time of the burned plots was 3, 4, 9, and more than 9 years. The objectives of this study were (a) to examine how the species community develops after fire and whether different directions are observed (b) to explore the relation between community structure and pattern and environmental gradients.

2.2. Materials and Methods

2.2.1. Study area

The study areas Sungai Wain and BOS Samboja Lestari are situated in East Kalimantan, Indonesia (Figure 2.1). Sungai Wain is a unique protected forest of about 10,000 ha that contains one of the last unburned primary forests of the Balikpapan – Samarinda area (Whitehouse and Mulyana, 2004). Samboja Lestari is a 1850 ha reforestation project owned by the Borneo Orangutan Survival Foundation (BOS). Plots selected for the analysis of regeneration impacts are situated at Samboja Lestari, whereas the primary forest plots chosen in the area of Sungai Wain function as controls. The Köppen system classifies the climate of the research area as Af (Tropical Rainforest). Average yearly precipitation is 2250 mm, with a wet period from December to May. The driest month has an average precipitation of 132 mm, and the wettest month of 231 mm. The daily maximum temperature varies from 23 °C to 31 °C and relative humidity is high. The soils in both areas are formed on marine sediments. Topsoils are generally slightly coarser than the deeper layers. In the FAO classification system (FAO, 2001) the soils of Samboja Lestari and Sungai Wain classify as Acrisols. Nutrient levels are low, especially that of available phosphorus; pH values vary between 4.09 and 4.55 (Yassir and Omon, 2006).

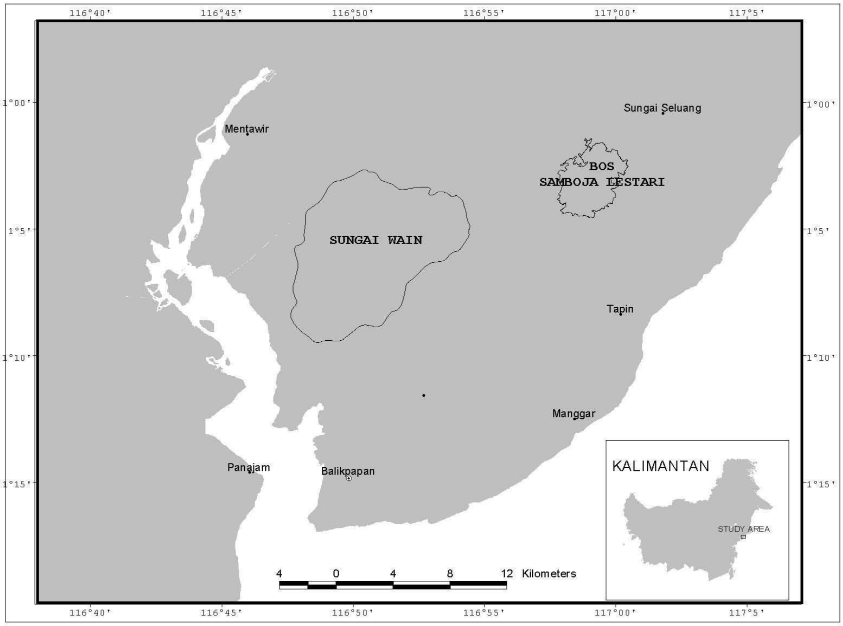

Figure 2.1 Location of Sungai Wain protection forest and BOS Samboja Lestari

2.2.2. Data collection

All field data were collected in the area of Samboja Lestari (secondary succession) and Sungai Wain (primary forest) from January until April 2007. In total, 291 plots were analyzed of which 28 in Sungai Wain and 263 in Samboja Lestari. The dataset contains 19 transects with a length varying from 200 to 1000 meters. All the plots in Sungai Wain belong to a single transect. The number of plots per transect varied from 6 to 24 and distance between the plots varied from 2 to 150 meters. The distance between plots in the transects was chosen in such a way that the data are adequate for studies of spatial variation. Vegetation was sampled in plots of 2*2 meter. The position of the plots was planned using GPS coordinates. When the planned location of a plot was in a planting row, it was shifted to the nearest possible place between the rows. The plots were placed in the middle of the rows with vegetation to minimize the effect of the plantation. Plots with big trunks, close to roads, or influenced by other factors that might have disturbed soil or vegetation were avoided. Using the coordinates of the plots, the fire incidence of every plot could be determined. This was done with the ARCMAP Software Package. Three categories were distinguished: plots that last burned in 2003 (3 years of regeneration); plots that last burned in 2004 (all of these plots also burned in 2003; 4 years of regeneration), plots that last burned in 1997/1998 (9 years of regeneration), and secondary forest that had regenerated during an unknown, longer time. Species abundance and species cover were estimated using a transformed fully numerical Braun-Blanquet scale with values varying from 1 to 9 (Van der Maarel, 1979). Wherever possible, the individual plants were identified to species level or to genus level when the species could not be identified. Vegetation higher than 3.5 meter was not studied, so in secondary and primary forests only the undergrowth was described. Species covering a plot but rooted outside were excluded, unless the species also occurred in the plot.

The soil profiles were shallow and consisted of an A, an AB and a B-horizon. The A and AB-horizon together amounted in general to less than 25 cm. All horizons were sampled, the B-horizon, which always had a very low C content, to a maximum depth of 50 cm (see Van der Kamp et al. 2009). Samples from each horizon consisted of a column covering the full depth. Samples were taken to the lab in labelled plastic bags. These samples were spread out and dried inside in open air.

After drying and mixing, the soil was sieved over a 2 mm sieve and packed in smaller labelled bags for transport to the laboratory.

2.2.3. Data analysis and statistical methods

The program WCanoImp was used to change the data into a condensed format. This enabled the use of the programs Canoco for Windows (ter Braak and Smilauer, 1998) and DCA (Detrended Correspondence Analysis) to analyse the community patterns. Changes in species distribution after last fire incidence were analyzed in spreadsheets of Microsoft Excel. For this purpose, data were transformed to percentage coverage. The modified Braun Blanquet classes 1-4, 5, 6, 7, 8 and 9 were transformed to cover percentages of, respectively, 2.5, 8.75, 18.75, 37.5, 62.5 and 87.5%.

Methods of soil analysis were listed by Van der Kamp et al. (2009). Bulk density of all horizons of both areas was measured at Samboja Lestari, using triplicate measurements with 100 cm^{-3} cylinders. Chemical properties were measured at the Soil Science Laboratory of the Agricultural University (IPB) of Bogor (Indonesia), while grain-size distribution was analyzed at the laboratory of the Earth System Science group of Wageningen University, the Netherlands. Methods were according to SSAC (1996). Chemical measurements included Exchangeable Al (extraction by 1M KCl; A-horizons only), total C determined by Walkley-Black (A-, AB- and B-horizon), available K determined by Bray I extraction and flame photometer (A- and B-horizon), total N determined by Kjeldahl/titrimetric (macro; A-horizons only), available P determined by Bray I extraction (A- and B-horizon), pH determined in 1:1 (soil: water) suspension with a pH meter (A-horizon only), H determined as titratable acidity of the KCl extract to pH 7. Texture analysis was done by laser diffraction, using a Coulter LS230 apparatus. This apparatus measures 116 fractions between

0.04 and 2000 microns (A-, AB- and B-horizon). Laser diffraction generally gives lower clay contents than field estimates or sieve-pipette method (Buurman et al. 2001). Because the vegetation cover depends most on the upper soil layer, some analyses were carried out for the A-horizon alone.

Canonical Correspondence Analysis (CCA) was applied to assess the relative importance of first and second major gradients of environmental variables in explaining the species distribution patterns. Nine properties of A-horizons were included in the analysis. The effects of all separate variables and whole variables on species composition in CCA were tested using global Monte-Carlo permutation test and forward selection in Canoco (ter Braak and Smilauer, 1998; Leps and Smilauer, 2003). The program CanoDraw was used to create bi-plot diagrams.

2.3. Results and discussion

2.3.1. Community structure

In the whole study (BOS Samboja Lestari and Sungai Wain), 252 plant species were identified (see Annex 1). The results of the DCA analysis are presented in Figure 2.2. This analysis does not include soil properties. The first DCA axis represents the secondary succession from *Imperata* grassland on the left, through secondary forest to primary forest. The second DCA axis largely represents the species variation in secondary and primary forest, which is obviously much larger than in the *Imperata* grasslands. The second axis seems to indicate the existence of two separate (undergrowth) communities, well established in the primary forest, and obviously re-initiated in the secondary forest. In DCA, the length of the longest axis provides an estimate of the beta diversity, which is the amount of compositional change represented in a set of samples (McCune and Grace, 2002). The length of the first axis is 7.218 and the second axis is 3.951, while the Eigenvalues of these axes were 0.874 and 0.543, respectively. The length of the longest axis suggests that the distribution of the species shows a unimodal response to the first DCA axis (ter Braak and Smilauer, 1998).

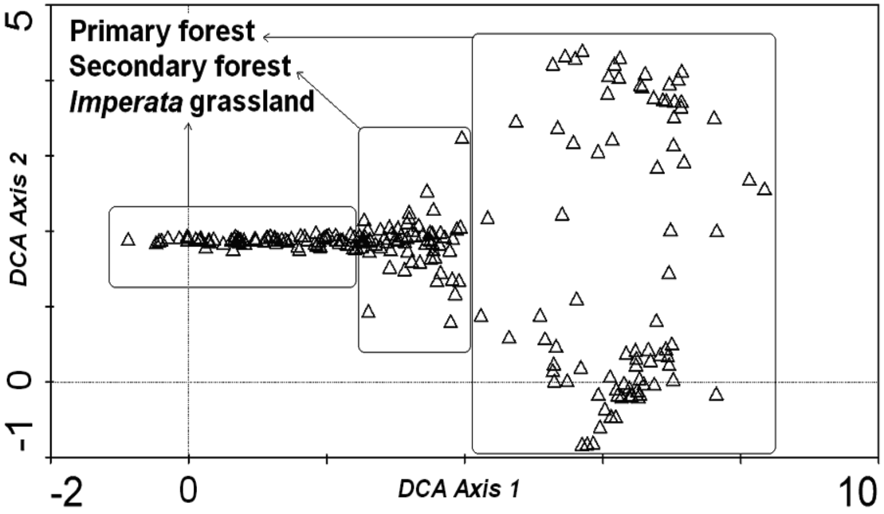

Figure 2.2. Plot of the species of *Imperata* grasslands, secondary forest, and primary forest on the first two DCA axes

2.3.2. Succession pathways in *Imperata* grasslands

Because of the much larger variation in the secondary and primary forest, Figure 2.2 does not show the changes with time in the *Imperata* grasslands, but these are also significant. Table 2.1 shows changes with time after burning of the cover of *Imperata cylindrica*, *Pteridium aquilinum*, and the percentage of shrubs and young trees, while Figure 2.3 illustrates changes in coverage for the most common shrub species. Both indicate significant changes with increasing time of regeneration. After three years of regeneration, *Imperata cylindrica* had the highest average coverage; it becomes less dominant from the fourth year on. The average cover of *Pteridium aquilinum* is initially low but increases after 4 and 9 years of regeneration. Also the average percentage of shrubs and young trees clearly increases with time. In the secondary forest other tree species take over, and both *Imperata* and *Pteridium* have disappeared.

Table 2.1. The cover (%) of *Imperata cylindrica, Pteridium aquilinum*, shrubs and young trees

Regeneration time and number of observations	*Imperata cylindrica* (%)	*Pteridium aquilinum* (%)	Shrubs and young trees (%)
3 years (n=47)	63	10	21
4 years (n=94)	40	18	31
9 years (n=81)	18	25	44
Secondary forest (n=41)	0	0	30

The relative importance of species of shrubs and young trees changes with the age of regeneration (Figure 2.3). After three years of regeneration, *Melastoma malabathricum* represented about 42%, *Eupatorium inulaefolium* 25%, and *Ficus* sp. 14% of the total shrubs and young trees. This changes little in the 4-year old growth, where *Melastoma malabathricum* represents about 36%, *Eupatorium inulaefolium* 25%, and *Ficus* sp. 10%. After nine years of regeneration, the pioneer *Melastoma malabathricum* reached about 42%, *Eupatorium inulaefolium* 9%, and *Vitex pinnata* 20%. In the plots with secondary forest, the pioneer *Syzigium lineatum* reached about 21%, *Fordia splendidissima* 19%, *Pternandra azurea* 8%, and *Macaranga* sp. 6%. Figure 2.3 also shows that *Melastoma malabathricum* and *Vitex pinnata* strongly increased with the age of regeneration but were rare in the secondary forest. This agrees with Nykvist (1996) who found *Eupatorium odoratum* the most common shrub after 2 years, while it completely disappeared after 3 years. Also *Melastoma malabathricum* and *Vitex pinnata* strongly increased with the age of regeneration but were rare in the secondary forest. The decrease of these species may be due to a negative effect of shade on the germination of their seeds, as was recorded by Faravani and Bakar (2007) for *Melastoma malabathricum* and by De Kok (2008) for *Vitex pinnata.*

In addition to *Imperata cylindrica* and *Melastoma malabathricum*, also *Scleria* sp., and *Lygodium flexuosum* were lacking in the primary forest, while *Connarus semidecandrus, Shorea laevis* and *Calamus* sp. were not found in the secondary succession. The secondary forest was characterized by the presence of *Syzygium lineatum, Psychotria* sp.*, Fordia splendisissima, Curculigo* sp.*,* and *Eugenia* sp.

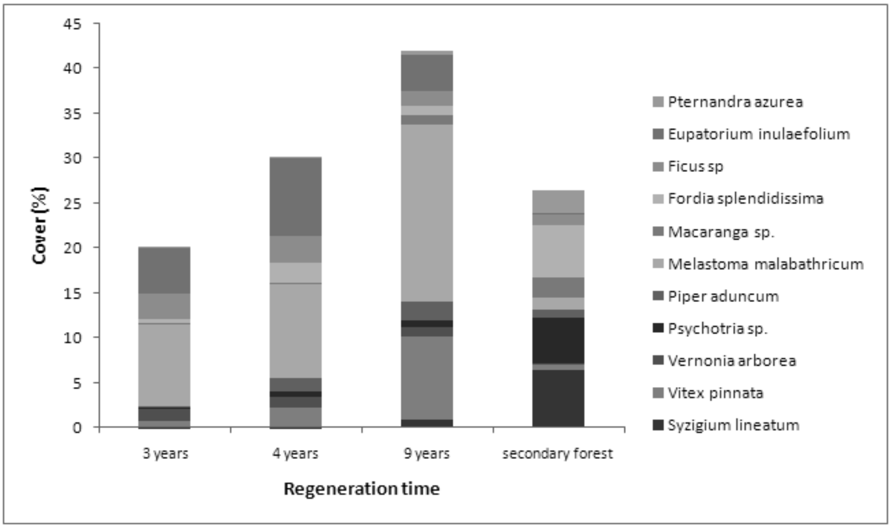

Figure 2.3. Cover (%) of shrubs and young trees in the different phases of regeneration

Observation in the field suggested that sandy textures might influence the secondary succession. This is not illogical; because sandy textures have lower bulk density, water binding capacity and available nutrients than more clayey or loamy ones. To test the effect of texture on the secondary succession, plots were separated into those having more and less than 50% sand in the topsoil. The chosen boundary of 50% was arbitrary. 118 plots were on soils with more than 50% sand in the A-horizon, and 104 plots had less than 50%. Soils with more than 50% of sand appear to have a slower development to secondary forest (Figure 2.4).

The decline of *Imperata* is very similar in the two texture groups, but there is a distinct difference in the covers of *Pteridium* and of shrubs. Although there is little difference after 3 and 4 years of regrowth, after nine years shrubs reached higher cover percentages on less sandy soils and *Pteridium* on the sandy soils. Figure 2.5 shows that, after 4 years, the number of species increases more rapidly on the less sandy soils. This indicates that soils with coarser texture have a slower development towards secondary forest. *Pteridium aquilinum* can reach a height of 2-3 meters and casts much more shade than *Imperata* grassland. In addition, it has thick and deep rhizomes and slowly decomposable litter, which may impede germination of seeds from other species. It is likely that seeds of shrubs and young trees such as *Melastoma malabathricum* and *Vitex pinnata* cannot germinate because of the shade. This is consistent with Den Oden (2000), who mentioned that *Pteridium aquilinum* can induce stagnation in succession through shading, smothering, the build up of a deep ectorganic soil layer and the support of a high density of herbivore and seed-eating rodents. Therefore, the lower number of shrub species on sandy soils may be more due to *Pteridium* than to the texture itself.

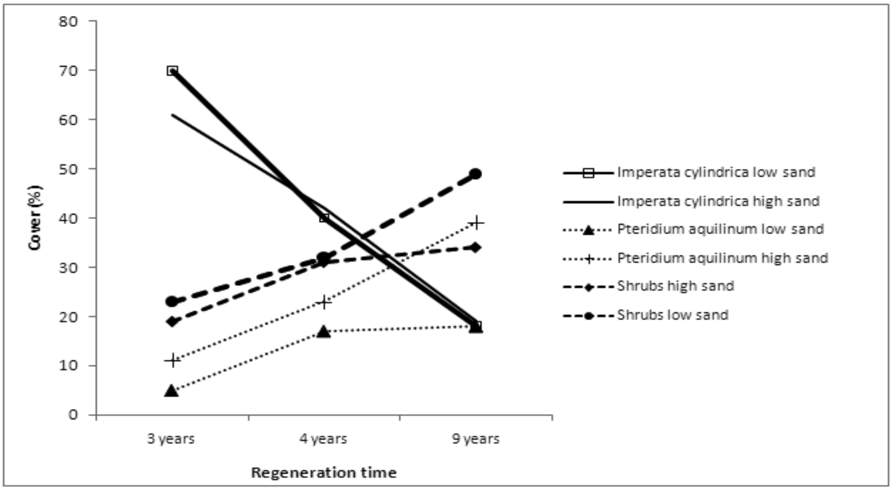

Figure 2.4. Cover (%) of *Imperata cyclindrica, Pteridium aquilinum* and shrubs + young trees in different phases of regeneration for high (> 50%) and low sand content (< 50%)

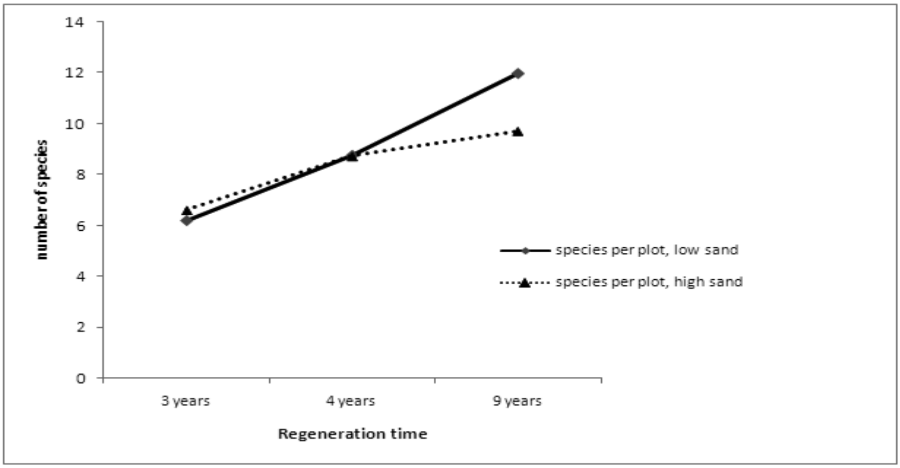

Figure 2.5. Number of species per plot in different phases of regeneration for high (> 50%) and low sand percentage (< 50%)

2.3.3.Vegetation pattern and environmental gradient in Imperata grasslands

To investigate the effect of soil properties on the secondary succession, the *Imperata* grasslands were analyzed separately. To this end, only the 40 species with sufficient abundance were included. Figure 2.1 shows that the length of Axis 1 for the *Imperata* grasslands (=3.38), which provides an estimate of the beta diversity, is sufficient to perform a constrained analysis with CCA (Leps and Smilauer, 2003). In CCA, the correlations between species and environmental variables for the first and two axes were 0.57 and 0.56, respectively, and species and environmental variables based on the first two axes explained 43.1% of the variance in the weighed averages and the class totals of the species. The Monte-Carlo global test indicated that the correlations of the selected environmental variables with the first and second axes were significant ($p=0.002$). The CCA diagram (Figure 2.6) shows that pH, bulk density (BD), sand and clay have a strong influence on the distribution of species, and also that soil carbon percentage (C) and Nitrogen (N) have little influence, probably because the latter are cause and effect at the same time. As expected, Aluminium (Al) and pH have a negative relation. Clay and Al show arrows in same quadrant, probably because Al depends both on pH and on the size of the exchange complex and the effects of regeneration on soil are strongest in the A-horizon especially pH showed a decrease with increasing time of secondary succession (Table 2.2).

Table 2.2 Means of soil parameters in the different phases of regeneration

	3 years (n=47)		4 years (n=94)		9 years (n=81)		Secondary forest (n=41)	
A-horizon	Mean	Sd[1]	Mean[2]	sd	Mean	sd	Mean	sd
Bulk density (g cm^{-3})	1.18b	0.10	1.16b	0.12	1.10a	0.10	1.10a	0.09
pH	5.29b	0.28	5.26b	0.24	5.09a	1.19	5.12a	1.14
C (g kg^{-1})	14.52a	3.24	15.32ab	5.31	15.96ab	4.82	16.71b	3.48
N total (g kg^{-}1)	1.43	0.32	1.47	0.50	1.54	0.44	1.58	0.31
P (mg kg^{-1})	4.04a	2.61	5.26b	3.24	4.47ab	2.98	4.08a	1.78
Al ($cmol^{+}$ kg^{-1})	1.10a	0.70	1.73b	0.93	1.78b	0.82	1.54b	0.85
K ($cmol^{+}$ kg^{-1})	0.16b	0.08	0.21c	0.09	0.16	0.07	0.18	0.06
AB-horizon								
Bulk density (g cm^{-3})	1.32	0.11	1.31	0.11	1.34	0.12	1.32	0.08
C (g kg^{-1})	8.99	2.26	9.31	3.27	9.10	2.44	8.93	1.71
B-horizon								
Bulk density (g cm^{-3})	1.38	0.10	1.37	0.11	1.39	0.11	1.41	0.08
C (g kg^{-1})	3.75a	1.19	4.46b	1.50	3.99ab	1.17	4.04ab	0.73
P (mg kg^{-1})	3.16a	2.82	4.63b	2.78	3.72a	2.07	3.60a	1.41
K ($cmol^{+}$ kg^{-1})	0.11a	0.09	0.13b	0.09	0.11ab	0.05	0.10a	0.04

[1] *Standard deviation*

[2] *Means followed by different letters within one soil parameter differ significantly ($P<0.05$) as established by the LSD- test*

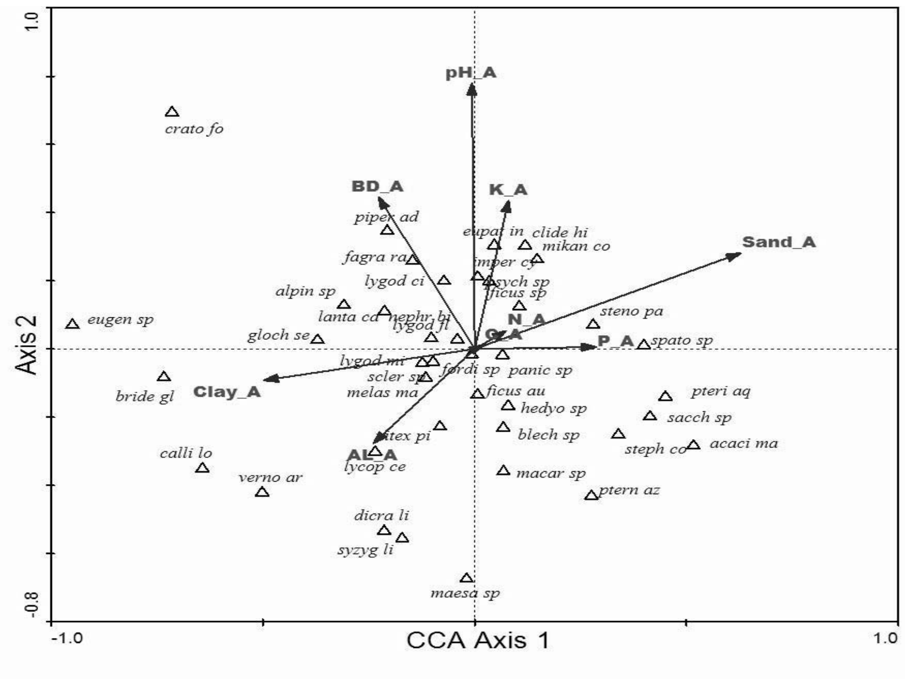

Figure 2.6. Plot of environmental variables and species (triangles) of *Imperata* grasslands systems (last burned 3 years, 4 years and 9 years previous plot on the first two CCA axes (only 40 species significantly abundant were analysed to CCA). Environmental variables in the A-horizon: (Clay_A=Clay; AL_A=Aluminium; BD_A=Bulk density; pH_A= pH; K_A=Potassium; C_A= Carbon percentage; N_A= Nitrogen total; P_A=Phosphor; Sand_A=Sand). Variable vegetation: Acaci ma= *Acacia mangium ;* Alpin sp= *Alpinia* sp; Bride gl= *Bridelia glauca* ; Calli lo= *Callicarpa longifolia ;* Clide hi= *Clidemia hirta ;* Crato fo= *Cratoxylum formosum ;* Dicra li= *Dicranopteris linearis ;* Eugen sp =*Eugenia* sp. ; Eupat in= *Eupatorium inulaefolium ;* Fagra ra= *Fagraea racemosa ;* Ficus sp= *Ficus* sp. ; Ficus au= *Ficus aurata ;* Fordi sp= *Fordia splendidissima ;* Gloch se= *Glochidion sericeum ;* Hedyo sp= *Hedyotis* sp. ; Imper cy= *Imperata cylindrica;* Lanta ca= *Lantana camara;* Lycop ce= *Lycopodium cernum;* Lygod ci= *Lygodium circinatum;* Lygod fl= *Lygodium flexuosum;* Lygod mi= *Lygodium microphyllum;* Macar sp= *Macaranga* sp.; Maesa sp= *Maesa* sp.; Melas ma= *Melastoma malabathricum;* Mikan co= *Mikania cordata;* Nephr bi= *Nephrolepis biserrata ;* Panic sp= *Panicum* sp.; Piper ad= *Piper aduncum;* Psych sp= *Psychotria* sp.; Pteri aq= *Pteridium aquilinum;* Ptern az= *Pternandra azurea;* Sacch sp= *Saccharum spontaneum;* Scler sp= *Scleria* sp.; Spato sp= *Spatholobus* sp.; Steno pa= *Stenochlaeana palustris;* Steph co= *Stephania corymbosa;* Syzyg li= *Syzygium lineatum;* Verno ar= *Vernonia arborea;* Vitex pi= *Vitex pinnata.*

Figure 2.6 shows that environmental variables have a strong influence on species composition. *Melastoma malabathricum, Vitex pinnata, Lycopodium cernum, Dicranopteris linearis, Syzygium lineatum, Vernonia arborea* are all associated with high concentrations of exchangeable Al and with low pH. These are species appearing in a later regeneration phase where pH values drop. The association of species such as *Melastoma malabathricum* with high concentrations of exchangeable Al or low pH values were also mentioned by Watanabe and Osaki (2001) and Osaki et al. (2003). *Bridelia glauca,* and *Callicarpa longifolia* are associated with finer-textured soils, contrary to *Stenochlaeana palustris*, *Pteridium aquilinum* and *Spatolobus* sp., which are associated with coarser textures. The relation between texture and vegetation has been described for widely different ecosystems (e.g., Cannone et al. 2004; El-Sheikh, 2005; Robertson, 2006).

Bridelia glauca, Eugenia sp. and *Callicarpa longifolia* are able to grow on soils very poor in Phosphorus, while *Stenochlaeana palustris*, *Pteridium aquilinum* and *Spatolobus* sp., prefer higher P contents. *Piper aduncum*, *Fagraea racemosa* and *Cratoxylum formosum,* finally, appear to tolerate high bulk density. Kooch et al. (2007) explained that soil texture and bulk density control the distribution of plant species by affecting moisture availability, ventilation and distribution plant roots. Schoenholtz et al. (2000) mentioned that the relation between bulk density, water and oxygen supply, and soil texture is the most of fundamental soil physical property controlling water, nutrient, and oxygen exchange, retention and uptake.

Figure 2.6 clearly shows the association of *Imperata* or *Pteridium* with *Eupatorium inulaefolium* and *Mikania cordata*, which are typical of the early secondary succession, and also the association of *Dicranopteris linearis* and *Lycopodium cernum*, which were always found together in the field. Hartemink (2001) supports the indication that *Piper aduncum* prefers high Potassium (K). CCA supports the farmers' knowledge that *Pteridium aquilinum* is an indicator of coarse soil texture and higher soil fertility, which implies that *Pteridium aquilinum* more easily invades soils that are sandy and have more Phosphorus.

2.4. Conclusions

Detrended Correspondence Analysis (DCA) resulted in a clear separation of the primary and secondary forest and *Imperata* grasslands. The first axis represented the succession gradient. Because the much larger variability of species in native forest, however, DCA of all plots together did not distinguish succession stages within the *Imperata* grassland. The succession within the *Imperata* grasslands was made visible by combining vegetation data with the fire history.

The results indicate two successional pathways in *Imperata* grasslands (Figure 2.7). The first pathway is dominated by *Imperata* grassland (model A) and the second by *Pteridium* (model B). In soils that contain more sand, the succession is more likely to follow model B, where *Pteridium* dominates. Shrubs and young trees in Model B will appear later than in plots where the regeneration follows model A where *Imperata* grassland dominates. Figure 2.7 shows that in mixtures of *Pteridium* and *Imperata*, overtopping may occur, and that the following stage depends on which species succeeds in overtopping the other. Scott (1977) described a similar system in the Montaña of Peru where *Imperata* and *Pteridium* were developing together in the beginning of the wet season. There, five months after fire, *Pteridium* managed to overtop *Imperata*.

Frequent fires maintain *Imperata* grasslands. If protected from fire and other intrusions such as shifting cultivation, they will readily develop into secondary forest.

Therefore, the assumption that *Imperata* grasslands are a final stage of land degradation and are very difficult to recover for more valuable land uses is not true. Neither is it true that *Imperata* grasslands always have low levels of available nutrients, high acidity, and low cation exchange capacity. In a previous paper, we have shown that, in Kalimantan, *Imperata* grasslands may have larger total soil carbon stocks than the native forest (Van der Kamp et al. 2009). *Imperata* grasslands seem to be permanent because of human induced disturbance, especially burning, and because so far few attempts have been made to sustainable rehabilitation. *Pteridium aquilinum* may induce stagnation in the regeneration of *Imperata* grasslands.

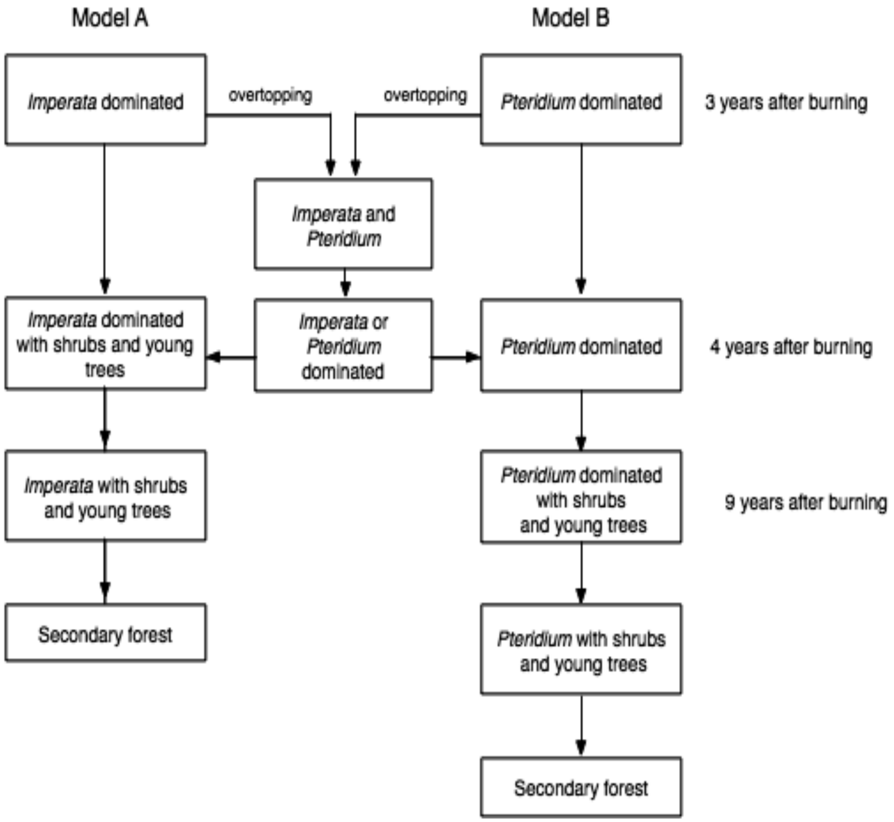

Figure 2.7. Hypothesized pathways of succession in *Imperata* grasslands of East Kalimantan

CHAPTER 3

Soil carbon changes upon secondary succession in *Imperata* grasslands (East Kalimantan, Indonesia)

Chapter based on:

Van der Kamp, J., Yassir, I., Buurman, P. 2009. Soil carbon changes upon secondary succession in I*mperata* grasslands (East Kalimantan, Indonesia). Geoderma, 149: 76-83.

Abstract

Soil carbon changes upon secondary succession in *Imperata* grasslands are important both for their effect on potential production and for possible implications of forest degradation and regeneration on global climate change. We studied the effect of forest regeneration after fire in *Imperata* (speargrass) grasslands of East Kalimantan on soil properties, using 47 plots that last burned in 2004, 94 plots that last burned in 2003, 126 plots last burned before 2003, 43 plots of secondary forest, and 28 plots of primary forest. Although soil carbon stocks increase upon natural regeneration from grassland to secondary forest, highest carbon stocks are found in the later regeneration phases and lowest under primary forest. This is contrary to the situation in other forest systems. Low C stocks under primary forests may be due to extremely low fertility, combined with shallow soils and low root mass in the topsoil. Root density – as observed in the field – is much higher under the grass vegetation. The effects of regeneration on soil are strongest in the A-horizon, where soil carbon content increases with 14%, from 14.5 g kg^{-1} in *Imperata* grassland to 16.5 g kg^{-1} in secondary forest, while carbon stocks in the A-horizon increase from 16.0 ton C ha^{-1} to 18.7 ton C ha^{-1}. This is accompanied by a decrease in pH and an increase in bulk density. The total soil carbon stocks in Kalimantan (fixed mass, approximate depth section 40 cm) are 36.2 ton C ha^{-1} in *Imperata* grassland, 38.9 ton C ha^{-1} in secondary forest and 33.2 ton C ha^{-1} in primary forest, which is considerably lower than in Sumatra. Aboveground C/belowground C ratios are higher in Kalimantan primary forest but lower in Kalimantan secondary forest than in Sumatra. Soil carbon stocks in *Imperata* grassland could be lower than previously thought. This has important consequences for carbon sequestration projects in East Kalimantan, because carbon storage potentials could be higher.

Keywords: *soil organic matter changes, Imperata grassland, regeneration, carbon sequestration.*

3.1. Introduction

Tropical forests occupy about 12% of the total terrestrial area and store about 17% of the total carbon. This represents a stock about 120 ton ha^{-1} of carbon for the vegetation and 123 ton ha^{-1} for the soil (IPCC, 2000). Indonesia has the last remaining large tracts of tropical forest in Asia, and is one of the key global centres of terrestrial biodiversity (Barber, 2002). There are about 90 million ha of designated forestlands, of which about 50 million ha are intact or primary forests (World Bank, 2003; FWI/GFW, 2002). From the biodiversity point of view, Indonesia's forests contain about 10% of the world's flowering plant species, 12% of mammalian species, 17% of the reptilian and 16% of the amphibian species (Primack et al. 1998). Therefore, Indonesia is an important player in the global climate change arena.

Also in Indonesia, forests are under increasing pressure of population growth, illegal logging, forest fire, and land use change for agriculture, transmigration and estate crops such as timber and oil palm. The rate of deforestation has dramatically increased from 800.000 ha year^{-1} in 1993 (Revilla, 1993), 1.2 million ha year^{-1} in 1996 (MoF-FAO, 1996), and 1.67 million ha year^{-1} between 1984 and 1997 (World Bank, 2000), to its present level of above 2 million ha year^{-1} (FWI/GFW, 2002).

According to the latest estimate, *Imperata* grasslands in Indonesia cover 8.5 million ha, or about 4.5% of Indonesia's total land area. In Kalimantan alone, *Imperata* grasslands cover an estimated 2.2 million ha (Garrity et al. 1997). *Imperata* grasslands are seen as a final stage of land degradation and are very difficult recover for more valuable land uses (Murniati, 2002). Regeneration of grassland areas is becoming increasingly important, not only to create new secondary forest, but also to recover the original biodiversity.

Information about soil carbon changes upon forest degradation and regeneration is very important, not only for production increase in converted areas, but also to get a better understanding effects of forest degradation on global climate change. Nevertheless, information on the effect of conversion of forests into *Imperata* grasslands, or of the recuperation of these grasslands on soil carbon is very scarce. Most studies focused on species composition (Ohtsuka, 1999; Hiratsuka et al. 2006) or on aboveground biomass estimates (Nykvist, 1996; Hiratsuka et al. 2006). In Sumatra (Indonesia), Van Noordwijk et al. (1997) found a decrease in soil carbon contents from primary forest, to secondary forest to areas used for tree crops and to slash-and burn areas. The difference in carbon organic content between land use types was about 0.5% C or 10 Mg ha^{-1} to a depth of 15 cm. Lal and Kimble (2000) show a decrease in soil carbon content after deforestation of about 1 kg m^{-2}, or 10 ton ha^{-1}, in the first 40 cm. Woomer et al. (2000) estimated soil organic carbon stocks of Dipterocarp forest in Sumatra, Indonesia, to be 48.1 ton ha^{-1} in the 0-15 cm layer, while *Imperata* grassland that followed after slash and burn contained approximately 36.8 ton ha^{-1}. However, Ohta et al. (2000) found in East Kalimantan - our study area - that levels of total soil carbon stocks did not decrease with forest degradation and in fact increased in every layer during the conversion of forest to grassland. Such contrary results confuse the issue of effects of forest degradation and regeneration on carbon contents and stocks. In East Kalimantan, it is possible to study soil carbon in relation to time or stage of regeneration because both the primary forest and the various degradation and regeneration stages can be studied. We compared soil organic carbon and soil properties of *Imperata* grassland plots that last burned in 2004, in 2003, and before 2003, with those of secondary forest (last burned in 1997/1998) and primary forest. Thus we hope to find, not just the differences between the various stages, but also trends with time in soil carbon and other soil properties upon secondary succession in East Kalimantan.

3. 2. Materials and Methods

3.2.1. Study area

The study areas Sungai Wain and Samboja Lestari (Figure 3.1) are situated in East Kalimantan, Indonesia. Sungai Wain is a unique protected forest of about 10,000 ha that contains one of the last unburned primary forests of the Balikpapan – Samarinda area (Whitehouse and Mulyana, 2004). Samboja Lestari is a 1850 ha reforestation project owned by the Borneo Orangutan Survival Foundation (BOS). Plots selected for the analysis of regeneration impacts are situated in Samboja Lestari, whereas plots chosen in the area of Sungai Wain function as control plots. The Köppen system classifies the climate of the research area as Af (Tropical Rainforest). Average yearly precipitation is 2250 mm with a wet period from December to May. The driest month has an average precipitation of 132 mm, and the wettest month of 231 mm. The daily maximum temperature varies from 23 °C to 31 °C and relative humidity is high. The soils in both areas are formed on marine sediments. Topsoils are generally slightly coarser than the deeper layers. In the FAO classification system (FAO, 2001) the soils of Samboja Lestari and Sungai Wain classify as Acrisols. Nutrient levels are low, especially that of available phosphorus; pH values vary between 4.09 and 4.55 (Yassir and Omon, 2006).

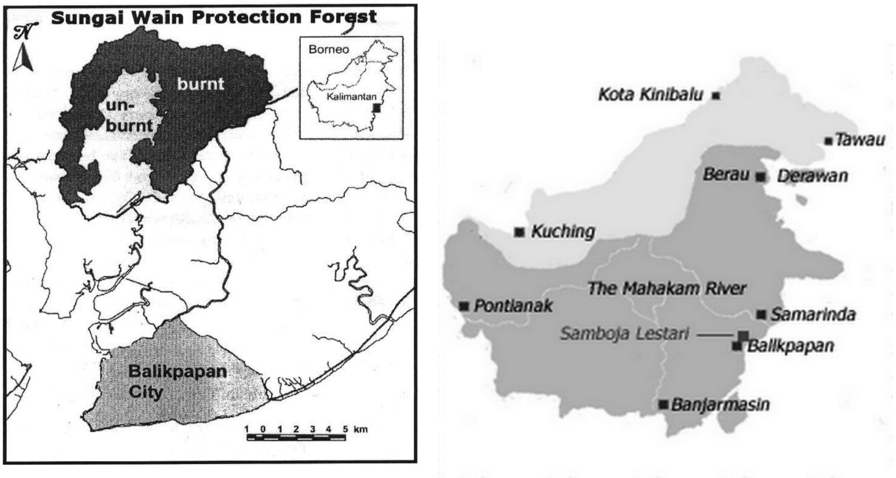

Figure 3.1 Location of Sungai Wain protection forest (Slik, 2006) and BOS Samboja Lestari (BOSF, 2007)

3.2.2. Data collection

All field data were collected in the areas of Samboja Lestari and Sungai Wain from January until April 2007. A classification of plots was based on the fire history. Using the coordinates of the plots, information on fire incidence in every plot could be gained using the ARCMAP software package. 338 plots were analyzed in Samboja Lestari, where four regeneration stages were distinguished: 47 plots last burned in 2004; 94 plots in 2003; and 126 plots before 2003: 43 plots were of secondary forest and last burned during the great Borneo fires of 1997/1998. In Samboja Lestari, there is no relation between regeneration stage and landscape position. The 28 plots in the area of Sungai Wain function as control plots representing primary forest. The dataset contains 23 transects with a length varying from 200 to 1000 meters. The number of plots per transect varied from 6 to 24 and distance between the plots varied from 2 to 150 meter. The position of plots in transects was chosen in such a way that the data can be used in studies of spatial variation of soil carbon. The position of the plots was planned using GPS coordinates. In Samboja Lestari, part of the *Imperata* grassland has been replanted with forest species. To this end, planting rows of about 3 m width have been cleared. These rows were avoided for sampling; when the planned location of a plot was in a planting row, it was shifted to the nearest possible place between the rows. Soil samples were taken of the A-, AB- and B-horizon. Plots that showed effects of poor drainage were not included in the present study. For chemical analyses, samples were taken from the full depth of each horizon. Samples were taken to the lab in labelled plastic bags. They were spread out and dried inside in open air. After drying and mixing, the soil was sieved over a 2 mm sieve and packed in smaller labelled bags for transport to the laboratory.

3.2.3. Data analyses and statistical methods

Bulk density of all horizons was measured at Samboja Lestari, using triplicate measurements with 100 cm^{-3} cylinders. Chemical properties were measured at the Soil Science Laboratory of the Agricultural University (IPB) of Bogor (Java), while grain-size distribution was analyzed at the laboratory of the Earth System Science group of Wageningen University, the Netherlands. Methods were according to SSAC (1996). Chemical measurements included Exchangeable Al (extraction by 1M KCl; A-horizons only), total C determined by Walkey-Black (A-, AB- and B-horizon), available K determined by Bray I extraction and flame photometer (A- and B-horizon), total N determined by Kjeldahl / titrimetric (macro; A-horizons only), available P determined by Bray I extraction (A- and B-horizon), pH determined in 1:1 (soil: water) suspension with a pH meter (A-horizon only), H determined by titratable acidity of the KCl extract to pH 7. Texture analysis was done by laser diffraction, using a Coulter LS230 apparatus. This apparatus measures 116 fractions between 0.04 and 2000 microns (A-, AB- and B-horizon). Laser diffraction generally gives lower clay contents than field estimates or sieve-pipette method (Buurman et al. 2001). Data were entered in a Microsoft Excel spreadsheet and analysed using SPSS. Least Significant Difference (LSD) tests were performed to establish differences in soil parameters between the various stages of secondary succession. Our modal assumption is that the stage of secondary succession is the only factor affecting the C content. This is realistic, because sampling procedure and measurements were uniform in all the stadia, and carried out over a short period of time. If the p value was lower than 0.05, the difference was seen as significant. Because changes in soil carbon stocks during regeneration are very small, stocks need to be calculated accurately. As any land use may cause a change in bulk density, using a fixed depth as criterion for comparison of carbon stocks between land uses (or regeneration stages) creates an error because a fixed depth is not equivalent to an equal amount of soil. We therefore based our comparisons on columns of fixed soil mass. A good description of this method can be found in Ellert et al. (2001, 2002). Some examples of applications of the fixed soil mass method to situations similar to the present research are given in Mannetje et al. (2008). Factor analysis was performed to get a better insight in the relation between variables, including regeneration stage, that determine carbon stocks in *Imperata* grassland systems.

3.3. Results

3.3.1. The effect of regeneration on soil parameters

Mean values and standard deviations of soil parameters in different phases of regeneration are listed in Table 3.1. Many differences appear small, but are significant at $p<0.05$ level because the number of observations is large. There is a significant difference in soil profile between the primary forest and the various regeneration phases. Most striking is the absence of an AB-horizon under primary forest. This is probably due to a less developed rooting system induced by the virtual absence of understory vegetation. On the other hand, the A-horizon under primary forest is significantly thicker than that under the various regeneration phases.

Bulk density generally increases with depth. Bulk density of the A-horizon was fairly high in the most recently burned fields. It decreased during the first phases of succession but then increased again, possibly upon disappearance of the undergrowth. The carbon content of the A-horizon was lowest in the most recently burned plots. It increased in the first phases of regeneration but then declined again, while carbon stocks in the A-horizon are highest under primary forest, largely because of the slightly thicker A-horizon. Carbon contents of the deeper horizons follow those in the A-horizon, which implies that they decrease under forest.

Table 3.1. Means of soil parameters in the different phases of regeneration

	Last burned 2004 (n=47)		Last burned 2003 (n=94)		Burned before 2003 (n=126)		Secondary forest (n=43)		Primary forest (n=28)	
horizon	Mean	Sd[1]	Mean[2]	sd	Mean	sd	Mean	sd	Mean	sd
A-horizon										
Depth (cm)	9.77ab	1.77	9.63ab	2.02	9.30a	2.53	10.20b	2.12	12.22c	2.44
BD (g cm^{-3})	1.18b	0.10	1.16b	0.12	1.09a	0.10	1.11ab	0.09	1.20b	0.13
pH	5.29c	0.28	5.26c	0.24	5.05b	0.20	5.11b	0.14	4.82a	0.12
C (g kg^{-1})	14.52a	3.24	15.32ab	5.31	17.07c	4.88	16.54bc	3.50	14.33ab	4.48
C stc (ton ha^{-1})	16.51ab	4.28	16.54a	5.75	16.87ab	5.54	18.70bc	6.06	21.15c	6.79
N total (g kg^{-}1)	1.43b	0.32	1.47b	0.50	1.67c	0.47	1.56bc	0.31	1.19a	0.31
P (mg kg^{-1})	4.04a	2.61	5.26b	3.24	3.82a	2.73	4.05a	1.77	5.31b	2.45
Al (cmol^{+} kg^{-1})	1.10a	0.70	1.73b	0.93	1.83b	0.76	1.58b	0.85	2.39c	0.65
K (cmol^{+} kg^{-1})	0.16b	0.08	0.21c	0.09	0.19bc	0.08	0.18b	0.06	0.11a	0.04
C/N ratio	10.22a	0.75	10.55a	1.45	10.20a	0.75	10.61a	1.37	12.17b	2.75
AB-horizon										
Depth (cm)	7.70	1.93	7.79	2.60	7.65	1.74	7.46	1.05		
BD (g cm^{-3})	1.32	0.11	1.31	0.11	1.30	0.13	1.32	0.08		
C (g kg^{-1})	8.99a	2.26	9.31a	3.27	10.33b	2.99	8.93a	1.67		
C stc (ton ha^{-1})	9.15ab	3.38	9.31ab	3.27	9.97b	2.86	8.72a	1.84		
B-horizon										
Depth (cm)	27.53a	2.84	27.58a	3.45	28.05a	3.55	27.34a	2.45	32.78b	2.44
BD (g cm^{-3})	1.38ab	0.10	1.37a	0.11	1.37a	0.12	1.41b	0.08	1.43b	0.11
C (g kg^{-1})	3.75a	1.19	4.46b	1.50	4.32b	1.32	4.06ab	0.74	3.43a	1.13
C Stc (ton ha^{-1})	14.13a	4.28	16.55b	5.98	16.57b	5.33	15.62ab	3.31	16.03ab	5.00
P (mg kg^{-1})	3.16a	2.82	4.63b	2.78	3.24a	1.98	3.62a	1.41	7.27c	3.57
K (cmol^{+} kg^{-1})	0.11a	0.09	0.13b	0.09	0.12ab	0.05	0.11a	0.04	0.13ab	0.08
Total C stc (ton ha^{-1}) F. soil depth 45 cm	39.79ab	7.75	42.40b	10.7	43.14b	8.87	43.04b	5.74	37.18a	8.36
C stc (ton ha^{-1}) F. soil mass	36.19ab	7.69	38.36b	10.2	39.02b	8.82	38.98b	5.92	33.19a	8.39

[1] Standard deviation

[2] Means followed by different letters within one soil parameter differ significantly ($P<0.05$) as established by the LSD- test ; (BD=bulk density; C stc= C stock; F.soil depth=fixed soil depth; F. soil mass=fixed soil mass).

Total carbon stocks were calculated on both fixed depth and a fixed mass basis. Although the second method is scientifically correct for comparison between systems, most publications still use a fixed depth approach. In Table 3.1, we report both to show the differences, but we will discuss only the fixed-mass data. Lowest soil carbon stocks in the reference section are found under primary forest, followed by the most recently burnt plots. During regeneration there is a significant increase of soil carbon stock that appears to reach is maximum in the (early stages of) secondary forest. The maximum difference amounts to no more than 5.8 ton C ha^{-1}.

Due to production of carbonates upon ashing of organic matter, the pH value of the A-horizon increases almost one unit when the vegetation is burned, and the soil gets significantly more acid again during secondary succession. The plots under the primary forest had a significantly lower pH value than all the plots at Samboja Lestari. Burned plots apparently need more than ten years to revert to the original pH value. Exchangeable Al decreases with increasing pH. For a number of Indonesian soil groups, this relation was studied by Buurman and Rochimah (1980). Due to its relation with pH, exchangeable Al in the A-horizon significantly increases with the time of secondary succession after fire (Table 3.1) and is highest under primary forest.

N-content in the A-horizon shows a small increase with regeneration in Samboja Lestari data, but N-contents decrease under forest. Especially the Sungai Wain forest was very poor in N, which is also reflected in a higher C/N ratio than in the other plots. K-contents show and increase upon burning, which is due to the accumulation of this element in the vegetation. Even in the secondary forest, K-contents in the topsoil are still higher than in the primary forest (see also the factor analysis). There was no clear relation between secondary succession and P-content of the topsoil. Subsoil P-contents were significantly higher under primary forest.

3.3.2. Factor analysis with soil variables

In the factor analysis of the total data set, four factors explained 69% of all variation, while factors 1 and 2 explained 32.7% and 18%, respectively. In the plot of samples in Factor 1- Factor 2 space (Figure 3.2), the samples from primary forest (bottom) are clearly separated from the rest of the population. In the regeneration plots, there is a general tendency to shift towards the bottom with age of regeneration, but the groups are not clearly separated. The group of plots last burned in 2003 shows the largest variation in properties (largest spread on the F1 axis). The plots last burned in 2004, the youngest stage of regeneration, plot somewhat more towards the upper left quadrant, while the plots last burned before 2003 tend toward the right side of the distribution.

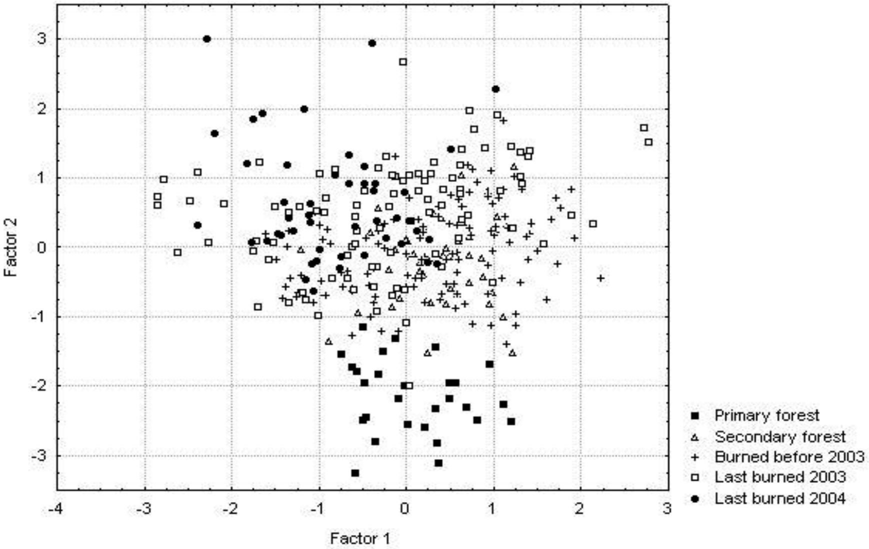

Figure 3.2. Factor scores of all A-horizon samples in F1F2 space

The factor loadings underlying Figure 3.2 are given in Figure 3.3a. The arrow indicates the effect of regeneration. This overall direction of regeneration, including the primary forest, seems to deviate slightly from the first stages as indicated by the samples from Samboja. As seen in Figure 3.3a, regeneration has a negative influence on pH (upper left quadrant) and a positive on exchangeable Al and H (lower right quadrant). The factor loadings indicate a negative correlation between pH and C content, but his correlation is very weak (r=0.40). There is an even weaker (positive) correlation between Al and C content (r=0.24). Phosphate and C/N ratios have low loadings on both axes and therefore hardly contribute to the differentiation between samples.

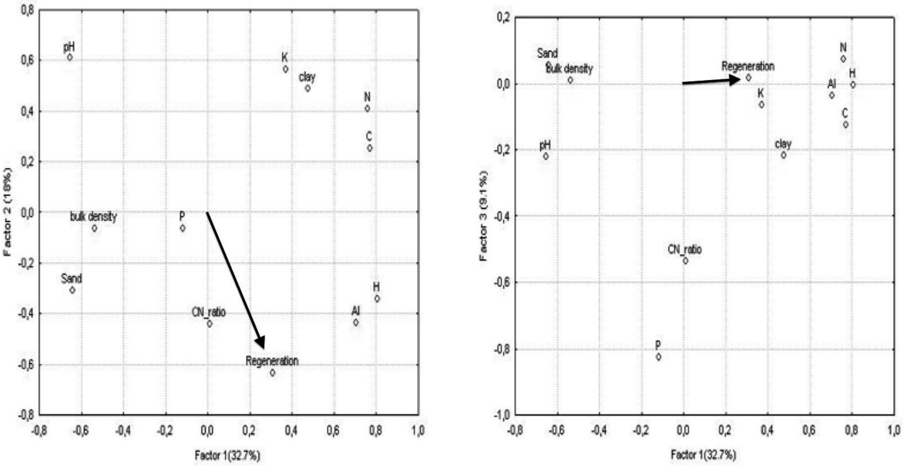

Figure 3.3. Loadings of soil variables for A-horizon samples (a) on Factor 1-Factor 2, belonging to Figure 3.2; (b) on Factor 1-Factor 3. Values between brackets are explained variance. The arrows indicate the direction of regeneration.

As shown in Table 3.1, regeneration does not have an overall positive effect on C and N contents. K and clay plot are close together, which may indicate that the clay itself is a source of K. The graph indicates that clay is lost upon regeneration, which suggests superficial losses through water erosion after fire and before the new vegetation has been established. K may be lost through dissolution and leaching of vegetation ashes. Sand and bulk density appears to be negatively correlated with C, N and clay. This is probably due to the fact that organic matter and clay stabilize aggregation and stronger aggregation increases pore size and thus reduces bulk density. A similar explanation was given by Buurman et al. (2008) for soils in Colombia. When the C/N ratio is close to constant, as in the present data set, C and N should plot close together. Clay content is supposed to influence carbon content, but the correlation between both properties is weak (r=0.54), and virtually absent under secondary forest (Figure 3.4).

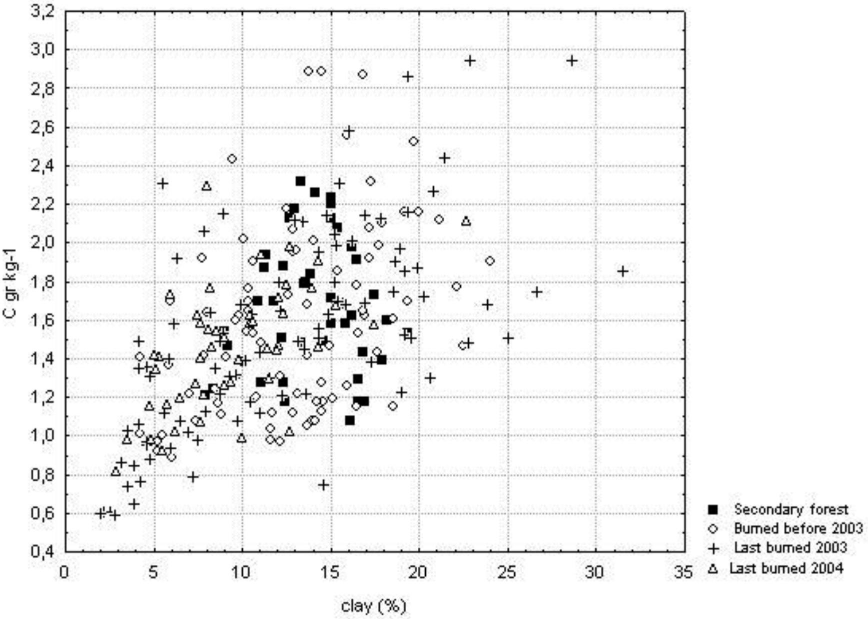

Figure 3.4. Relation between clay and carbon content by regeneration stage.

In Factor 1-Factor 3 space (Figure 3.3b) regeneration is not a major driving force. The major separation on Factor 3 is caused by Phosphate content and somewhat less by C/N ratio. The other variables have low loadings on F3. The connected plot of sample scores in F1F3 space (not shown) does not show any separation of groups, and it is likely that the differences reflect parent material rather than regeneration stage, as already suggested by the wide spread of samples in Figure 3.2. In the plot of Figure 3.2, the samples from Sungai Wain strongly dominate the pattern. Because Sungai Wain has significantly higher sand contents and lower clay contents than Samboja Lestari and both clay and K have a high loading on Factor 2, we also carried out the factor analysis for the Samboja Lestari samples alone, but neither the loadings of the variables nor the plot of the samples changed significantly.

3.4. Discussion

The primary forest of Sungai Wain is used as a control because it is the nearest untouched primary forest. It was assumed that Samboja Lestari had been covered with forest similar to that of Sungai Wain, and that also the soil was similar. The data indicate that this is not completely true. Significant differences in soil texture are found between Sungai Wain and Samboja Lestari. The soils of Sungai Wain were coarser than those of Samboja Lestari. The comparison of soil carbon stocks between areas with different soil textures is difficult, even though the correlation between carbon and clay contents was weak. Bulk densities, which are negatively correlated with carbon content, were higher in the Sungai Wain area, but this is probably a secondary effect due to texture differences (bulk density is positively correlated with sand content).

For the calculation of carbon stocks based on fixed mass (Ellert et al. 2001, 2002) we took the profile with the lowest mass of the specified depth section (45 cm in this case) as a reference, and therefore, the depth section of most profiles is less than the specified 45 cm. It varies from 33.4 to 50 cm, with a mean value of 39.9 cm. This results in lower calculated stocks than for the fixed depth method. Because the profiles do not show a wide variation in bulk densities, as is sometimes found under different land uses, and carbon contents of the B-horizons are low, the differences between the two methods are minor. Although the fixed mass method is always superior to the fixed depth method if different land uses on similar soils need to be compared, also the fixed mass method surmises identical soils. In our study, the variation in soil properties, notably texture, is considerable and, as a result, the properties of each treatment (succession stage) are also quite variable. The differences are largely levelled out by the large number of observations, but the factor analysis suggests that the soil variation is not equally divided between the various succession stages.

The carbon pool under *Imperata* grassland is 37.3 ton ha^{-1} in the reference mass (approximately the first 40 cm of the soil, mean value of the 141 measurements for plots last burned in 2003 and 2004). Woomer et al. (2000) found 36.8 ton C ha^{-1} in the upper 15 or 20 cm in *Imperata* grassland of Sumatra. The total soil carbon pool in the upper 15 cm under *Imperata* grasslands of Samboja Lestari is much lower than what Woomer found. In East Kalimantan, a significant part of the carbon stock is found below 15 cm, which is in accordance with the observations of Lal and Kimble (2000) about carbon allocation in soil profiles.

We found a lower carbon stock under primary forest (about 33.2 ton C ha^{-1} in the first 40 cm) than under *Imperata* grassland. This is consistent with Ohta et al. (2000), who found that, in East Kalimantan, the stock of total carbon did not decrease with forest degradation and in fact increased in every layer upon conversion of forest to grassland. It is contrary, however, to the data of Woomer et al. (2000) who found a higher carbon stock under primary Dipterocarp forest in Indonesia (48.1 ton ha^{-1} in the first 20 cm) than under *Imperata* grassland. Most authors find a decrease in the carbon pool after burning. Ekinci (2006) found that organic carbon in a burned soil decreased from 7.74 to 7.14% in Canakkale, Turkey. Alauzis et al. (2004) found a strong decrease in organic carbon content of the 0-10 cm layer after fire in a broadleaf deciduous forest on volcanic ash soils in Patagonia, Argentina. The effect of burning on carbon stock, however, will depend on the vegetation and on the carbon determination method. A forest fire, while it may decrease carbon contents in the upper millimetres of the soil, leaves a lot of charcoal that can be incorporated into the soil. Depending on the carbon determination method and on the time of incorporation, this fraction may contribute significantly to the soil carbon pool. Fires of open vegetations, where temperatures remain lower, have less effect on soil carbon, because both the oxidation of soil C and the contribution of charred material are smaller.

Carbon stocks under primary forest in East Kalimantan appear lower than in other Indonesian locations. Buurman (1980) calculated stocks of 90-130 ton C ha^{-1} to a depth of 1 m for forest soils of West Sumatra and West Kalimantan, where carbon contents stayed relatively high at greater depths. Although clay percentage may have a positive relation to carbon content, for instances through its effect on soil aggregation and carbon turnover time (e.g., Wattel Koekkoek, 2002), the low correlation between these properties in the present research suggest that the differences in carbon content between Sungai Wain and Samboja Lestari should not be ascribed to this factor. More likely, the higher C content under *Imperata* grassland could be due to high production of root biomass and also to accumulation of stable organic matter supplied through burning (Ohta et al. 2000). Our results support Mori et al. (2000), who stated that in the tropics, the hypothetical rapid decrease in soil carbon with forest loss and its slow recovery under secondary vegetation are misconceptions.

Differences in microclimate between the ecosystems of primary forest and *Imperata* grassland may also affect carbon stocks. In the primary forest, the atmosphere is very moist and temperatures are relatively low and constant. In *Imperata* grassland the humidity is lower and the day temperature is relatively high. These differences will probably affect the decomposition rate of soil organic matter. Decomposition will probably be faster under primary forest than under the *Imperata* grassland, where periodic drought is an impediment to decay. Figure 3.5 illustrates how the soil carbon pool could behave during the transition of primary forest through *Imperata* to secondary forest, showing changes in microclimate, aboveground and belowground C stocks. We ascribe the initial increase in soil C upon conversion from forest to grassland to the incorporation of charcoal. Assuming that the plots under secondary forest were not burned since the "Borneo fire" of 1997/1998, the increment of the soil carbon stock from *Imperata* grassland (last burned 2004) to secondary forest (2007) is 0.32 ton C ha^{-1} per year, which is almost similar to values of Lugo and Brown (1992), who found a rate of soil C accumulation in forest succession of 0.45-1.8 ton C ha^{-1} per year, and 0.27-0.38 ton C ha^{-1} per year upon conversion of crop land to pasture. However, these values are very low compared to sequestration rates in other tropical ecosystems. Amezquita et al. (2008) list increments of 1.6-9.1 ton C ha^{-1} per year for silvo-pastoral systems in Colombia and Costa Rica.

System	Primary forest	*Imperata* Grassland	*Imperata* & shrub/tree	Secondary Forest	Regrown Primary forest
Micro climate	Moist	Dry	Dry	Dry/Moist	Moist
C Vegetation	Very high	Low	Increasing	Increasing	High
C soil	Low	Increasing	Increasing	Increasing	Decreasing

Figure 3.5. Hypothesized relations between the carbon pools of soil, vegetation and micro-climate during regeneration of *Imperata* grassland especially in East Kalimantan.

The bulk density of the A-horizon is significantly higher in plots that burned in 2004 compared to plots that burned before 2003 and under secondary forest. This is consistent with Andreu et al. (2001), who found that bulk density increases due to deterioration of soil structure after fire. Boyer and Miller (1994) found an increase of bulk density from 1.22 to 1.26 g cm^{-3} in a loamy soil after fire in a longleaf pine forest on the coastal plain of Alabama, USA. However, Sakalauskas et al. (2001) found no change of bulk density after fire in tall bunchgrass (*Paspalum quadrifarium*) in Argentina. Here again, the type of vegetation and the intensity of the fire appear to play a role.

pH and total N showed a decrease with increasing time of secondary succession. This is consistent with Cruz and del Castillo (2005), who found that pH decreases with secondary succession in Mexico, but for N this is contrary to Feldpausch et al.

(2003), who found that soil N stocks (and C stocks) increased with forest age in Central Amazonia. Ohta et al. (2000) suggested that higher total N in *Imperata* grassland might be due to its higher production of underground biomass than in the primary forest. For P and K there was no significant increase with time of secondary succession, which may indicate either a limited stock in the soil or leaching from the system. Leaching of Phosphorus is unlikely, and therefore a limited supply is probably the best explanation.

Because the carbon stocks measured by us are rather low, we compared our data with those of other researchers in Kalimantan and Sumatra. Table 3.2 lists aboveground biomass and C stocks, soil carbon stocks, and the ratios of biomass C to total C for various land uses in Indonesia. Table 3.2 shows that, in general, soil carbon stocks and the ratio of standing biomass C to total C are different for Sumatra and East Kalimantan. Standing biomass carbon stocks and soil carbon stocks are much higher in Sumatran forests (220 versus 155 ton ha^{-1}). The relative amount of standing biomass in Sumatra is lower than in Kalimantan (0.72 versus 0.82). This proportion declines in the secondary forest, which are difficult to compare because the time of regrowth is not known for the Sumatra systems. On the other hand, biomass C is and its fraction of total C are similar in Sumatra and Kalimantan *Imperata* grasslands, although soil C stocks are again higher in Sumatra. The cause of the differences in biomass and soil carbon stocks between Sumatra and Kalimantan is not completely clear, but the soils in East Kalimantan are generally shallow, poor in nutrients, and very prone to erosion, while those in Sumatra tend to be deeper and more developed.

The factor analysis shows how the soil properties vary together. In Figure 3.3, bulk density and sand percentage have a positive relation. This is consistent with Askin and Ozdemir (2003), who found that bulk density showed a positive relation with sand and very find sand and a negative one with silt, clay, and organic matter content. The weak positive relation between soil carbon and clay content is consistent also with Van

Noordwijk et al. (1997) and Plante et al. (2006) who found that the soil carbon concentration was positively related to silt and clay content.

Table 3.2. Aboveground biomass, aboveground C stocks, and soil carbon stocks, in forests and *Imperata* grasslands

Land use Location	Aboveground biomass (ton ha^{-1})	C stocks aboveground (ton ha^{-1}) A	Soil carbon stocks (ton ha^{-1})	Total carbon stocks (ton ha^{-1}) B	Ratio A/B
Papua New Guinea [a]	-	6.7	85.7	92.4	0.07
Sumatra					
Original Forest [b]	-	219.6	84.4	305	0.72
Secondary forest [b]	-	133.6	85.4	219	0.61
Imperata grassland [b]	-	2.4	44.6	47	0.05
East Kalimantan					
Bukit Soeharto (East Kalimantan, Indonesia) [c]	309.3	154.7 [g]	33.19 [j]	187.9	0.82
Secondary forest (East Kalimantan) 33 years after fallow [d]	97.4	43.83 [h]	38.98 [j]	82.81	0.53
Secondary forest (East Kalimantan) 10-12 years after fire [e]	50.5	22.72 [h]	38.98 [j]	61.7	0.40
Imperata grasslands (East Kalimantan) [f]	7.5	3.45 [i]	36.19 [j]	39.64	0.09

a) Hartemink (2004), upper 15 cm
b) Woomer et al. (2000), source data including Sumatra
c) Toma et al. (2000)
d) Kiyono and Hastaniah (1997)
e) Hashimotio et al. (2000)
f) Kiyono and Hastaniah (2000)
g) Conversion factor form aboveground biomass to C stocks in primary forest is 0.5 (Houghton, 2005).
h) Conversion factor form aboveground biomass to C stocks in secondary forest is 0.45 (Fearnside, 2000).
i) Conversion factor form aboveground biomass to C stocks in *Imperata* grassland is 0.46 (CATIE and Guelph, 2000).
j) C stock based on fixed soil mass, this paper.

3.5. Conclusions

During regeneration of *Imperata* grasslands, both vegetation composition and soil properties change. The effects of regeneration on soil are strongest in the A-horizon, where an increase in carbon stock, N, and C/N ratio, and a decrease in bulk density and pH are observed. Regeneration does not affect the status of P and K of the soil. In East Kalimantan, soil carbon stock increases upon regeneration of the vegetation after fire, but it is higher in the degraded *Imperata* grassland than under primary forest. This is contrary to the generally higher soil carbon stocks under primary forest as found in other areas, e.g. in Sumatra (Indonesia). The cause of this discrepancy is not completely clear, but the soils in East Kalimantan are generally shallow, poor in nutrients, with shallow rooting systems and very prone to erosion, while those in Sumatra tend to be deeper, more developed, and deeply rooted. The aboveground C stocks and also the ratio of biomass C stock to total C stock are different in Sumatra and East Kalimantan, especially under primary and secondary forest. Differences under *Imperata* grasslands are much smaller. Our results indicate that soil carbon stocks in *Imperata* grasslands of East Kalimantan are lower than previously thought. This may have important consequences for carbon sequestration projects in this area, because carbon storage in soil potentials might be higher.

CHAPTER 4

Soil organic matter dynamics upon secondary succession in *Imperata* Grasslands, East Kalimantan

Chapter based on:

Yassir, I., Buurman, P. 2012. Soil organic matter dynamics upon secondary succession in *Imperata* Grasslands, East Kalimantan. Submitted in Soil Science Journal. Under review.

Abstract

Soil organic matter (SOM) dynamics upon secondary succession in *Imperata* grasslands was investigated by stable carbon isotope analysis. The data of litter and soil samples of twenty plots in four different stages of succession were compared. These different stages were represented by plots that were; (1) last burned 3 years before (*Imperata* grassland), (2) last burned 9 years before, (3) a secondary forest (≥15 years) and (4) a primary forest. Isotopic signatures of all soil horizons of the regeneration stages were statistically different from those of the primary forest. The A-horizon under the 3-years *Imperata* plot still contained 23% forest (C3) carbon, and this fraction increased to 51% in the-B-horizon. In the 9-years plot and in the secondary forest, the C3 carbon on the A-horizon increased to 51% and 96%, respectively. In the topsoil, the loss of C4-C between the 3-years and the 9-years plot was significant, while it appeared negligible in the AB-horizon. The strong decay in the topsoil under *Imperata* grassland may be due to the rather high carbohydrate content of the SOM, which is considered easily decomposable. Further research is needed especially to determine the relation between carbon stocks and chemical of SOM composition. Such insight may help to better understand and predict soil carbon changes in relation to climate and vegetation change.

Keywords: *Carbon isotopes, Imperata grasslands, secondary succession, and soil organic matter.*

4.1. Introduction

Imperata grassland is a common vegetation type in Kalimantan, Indonesia and in adjacent parts of Southeast Asia. It indicates a high degree of degradation of the vegetation, and mostly occurs after slashing and burning of primary forests. Through secondary succession, *Imperata* grassland is converted into new secondary forest and most of the original biodiversity is restored. During this succession, plant species composition and soil properties change, as discussed in previous publications (Van der Kamp et al. 2009; Yassir et al. 2010). It was shown that soil carbon stocks were lower under primary forests than under *Imperata* grasslands and this is contrary to the situation in other forest systems.

In order to better understand the process of succession in *Imperata* grassland, more should be known about its SOM dynamics. A common way to explore SOM dynamics in soils that have been under consecutive vegetation (succession) with a different photosynthetic pathway is by using the relative abundance of the stable isotopes ^{13}C and ^{12}C, expressed as $\delta^{13}C$ (Balesdent and Mariotti, 1996; Roscoe et al. 2000).

Plants with difference photosynthetic pathway have different ^{13}C discrimination; for instance C3 are more efficient in discriminating ^{13}C than C4 plants. Therefore, C4 plants have significantly more ^{13}C in their tissue and residues than C3 plants. Plants take up carbon from the atmosphere, but ^{13}C contents in the atmosphere have changed over time, and atmospheric ^{13}C is therefore not a good reference. ^{13}C discrimination is therefore expressed with respect to a geological standard, the PeeDee belemnite, or its gas equivalent (Vienna-PeeDee). The discrimination is expressed as $\delta^{13}C$ and calculated as $\delta^{13}C$ (per mil) = $1000 * (^{13}R_{sample} - {}^{13}R_{standard}) / {}^{13}R_{standard}$, in which ^{13}R is the $^{13}C/^{12}C$ ratio. Plants with a C3 photosynthetic pathway have $\delta^{13}C$ values ranging from -32 to -22 $^{0}/_{00}$ (mean of -27$^{0}/_{00}$), and C4 plants have $\delta^{13}C$ values of -16 to -9$^{0}/_{00}$ (mean of -13$^{0}/_{00}$), while the present $\delta^{13}C$ value of the atmosphere is -8$^{0}/_{00}$ (Boutton, 1996; Balesdent and Mariotti, 1996; Roscoe et al. 2000). Plant litter and the SOM derived from it inherits the ^{13}C signature of the living plant. Hence, the isotopic signature of SOM can be used to explore its dynamics, when C4 plants are replaced by C3 plants or vice-versa. As *Imperata* is a C4 grass, the contribution of *Imperata*-derived SOM and its disappearance upon secondary succession, as well as the new input of C3 SOM from trees and shrubs can be studied this way.

$\delta^{13}C$ values have used analysed in many studies on SOM dynamics in a variety of vegetation and land use types, including tropical ones (Boutton et al. 1989; Vitorello et al. 1989; Martin et al. 1990; Veldkamp, 1994; Roscoe et al. 2000; Magnusson et al. 2002; Qiming et al. 2003; Wilcke and Lilienfein, 2004; Marin-Spiotta et al. 2009; Katsuno et al. 2010). Nevertheless, such studies have not been carried out on the effect of a conversion of forests into *Imperata* grasslands or the succession from *Imperata* grassland to secondary forest. So far in Indonesia, most of the studies on *Imperata* grasslands focussed on soil carbon stocks changes in relation to changing land use (Van Noordwijk et al. 1997; Lal and Kimble, 2000; Woomer et al. 2000; Ohta et al. 2000; Van der Kamp et al. 2009; Yonekura et al. 2010).

The present study describes SOM dynamics upon secondary succession in *Imperata* grassland using stable carbon isotopes. The data of litter and soil samples of twenty plots in four different stages of succession were compared. We compared the effect of succession on proportions and absolute amounts of C3- and C4-derived SOM in order to determine differences in accumulation and decay. Such information may indicate why carbon stocks under primary forest are lower than under *Imperata* grassland.

4.2. Materials and Methods

4.2.1. Study area

The study areas Sungai Wain and BOS Samboja Lestari are situated in East Kalimantan, Indonesia (Figure 4.1). Sungai Wain is a unique protected forest of about 10,000 ha that contains one of the last primary forests of the Balikpapan – Samarinda area (Whitehouse and Mulyana, 2004). Samboja Lestari is a 1850 ha reforestation project owned by the Borneo Orangutan Survival Foundation (BOS). Plots selected for the analysis of regeneration impacts are situated at Samboja Lestari, whereas the primary forest plots chose in the area of Sungai Wain function as controls. The Köppen system classifies the climate of the research area as Af (Tropical Rainforest). Average annually precipitation is 2250 mm. The daily maximum temperature varies from 23 °C to 31 °C and relative humidity is high. The soil in both areas, Samboja Lestari and Sungai Wain, is formed on marine sediments and it is classified as Acrisol according to the FAO classification system (FAO, 2001). This soil shows a low level of nutrients, especially that of available phosphorus. Its pH value varies between 4.09 and 4.55 (Yassir and Omon, 2006).

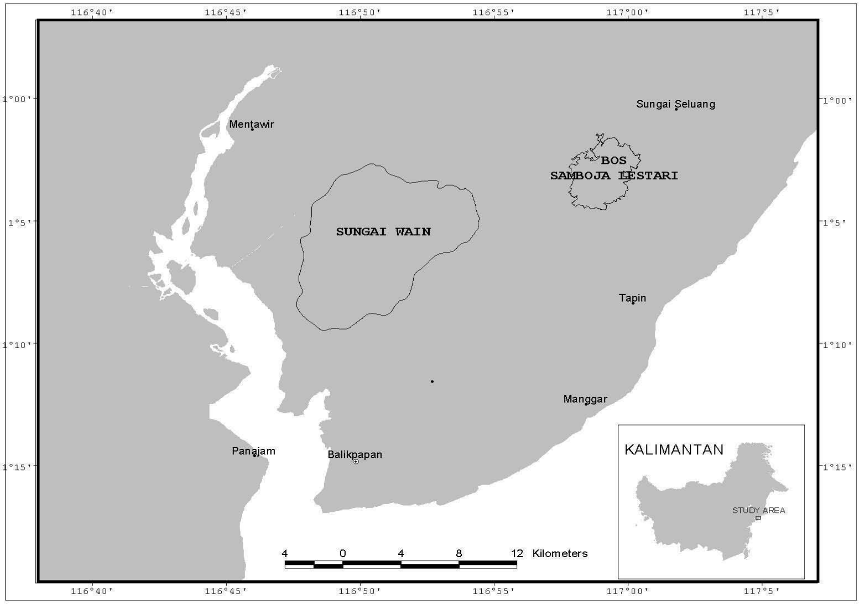

Figure 4.1. Location of Sungai Wain protection forest and BOS Samboja Lestari

4.2.2. Data collection and analysis

All the field data and soil samples were collected in the areas of Samboja Lestari and Sungai Wain. The classification of the plots in Samboja Lestari was based on the fire history and previous studies (Van der Kamp et al. 2009). The general descriptions of the soil properties and vegetation dominances in each of the sampling plots are summarized in Table 4.1.

In total twenty plots were analysed, representing four different stages of succession. Five plots were sampled of each stage. The defined stages were found in using plot areas that were burned 3 years and 9 years before sampling, a secondary forest of at least 15 years growth, and a primary forest. Logging in the area started around 1970 and was followed by continued slashing and burning. In 1982 to 1983 after droughts and forest fires, induced by the El Nino Southern Oscillation (ENSO) event, the area was fully covered by *Imperata* grassland, which was burnt again virtually every year afterwards, except in the protected area of Samboja Lestari. When the 3-years plots were sampled in 2007, there were about 25 years for the decay of C3 SOM in these plots. Soil samples were taken of the A-, AB- and B-horizon, except under the primary forest where AB- horizons were lacking. The analysis of the litter and soil samples using isotope ratio mass spectrometry (IRMS) was conducted at the stable isotope laboratory, University of California at Davis.

As the $\delta^{13}C$ value of any soil sample is a linear mixture of the contributions of a vegetation dominated by C_3 and C_4 plants, the relative contribution of each can be calculated from $\delta^{13}C_{sample} = x * \delta^{13}C_{C3} + (1-x) * \delta^{13}C_{C4})$, in which x is the fraction of C_3-derived, and (1-x) the fraction of C_4-derived soil organic carbon, while the $\delta^{13}C$ values denote the typical means of the relevant C_3 and C_4 litter. Data were analysed using SPSS. Least Significant Difference (LSD) tests were performed to determine statistically significant differences between $\delta^{13}C$ values of the various stages of succession.

Table 4.1. Soil properties and dominant species in sampled plots

Regeneration			Means					
Stage	Bd$^{a)}$ (g cm^{-3})	pH	C (g kg^{-1})	N (g kg^{-1})	C/N	P (mg kg^{-1})	K (cmol^{+} kg^{-1})	Dominant species
3 years (n=47)								
A-horizon	1.18	5.29	14.52	1.43	10.53	4.04	0.16	*I. cylindrica*
AB-horizon	1.32		8.99					*E. inulaefolium*
B-horizon	1.38		3.75			3.16	0.11	
9 years (n=126)								
A-horizon	1.10	5.09	15.96	1.54	10.36	4.47	0.16	*M. malabathricum*
AB-horizon	1.34		9.10					*E. inulaefolium*
B-horizon	1.39		3.99			3.72	0.11	*Vernonia arborea*
S. Forest (> 15 years) (n=43)								
A-horizon	1.10	5.11	16.71	1.58	10.58	4.08	0.18	*Syzigium lineatum*
AB-horizon	1.32		8.93					*F. splendidissima*
B-horizon	1.41		4.04			3.60	0.10	*Pternandra azurea*
								Macaranga sp.
								Vernonia arborea
								Vitex pinnata
P. Forest (n=28)								
A-horizon	1.2	4.82	14.33	1.19	12.04	5.31	0.11	*Shorea* sp.
AB-horizon								*Madhuca* sp.
B-horizon	1.43		3.43			7.27	0.13	*Macaranga* sp.
								Syzigium sp.
								Gironniera sp.

*) Bd$^{a)}$ = bulk density

4.3. Results and Discussion

4.3.1. Soil properties in different phases of regeneration

In our case study, carbon and nitrogen content in the A-horizon showed a small increased with regeneration stage, but C and N contents decrease in the primary forest (Table 4.1). The plots of the primary forest showed a low N content, which was also reflected in a higher C/N ratio than in the other plots. The pH of the A-horizon was highest in *Imperata* grassland and lowest in the primary forest samples. When the vegetation was reduced to ashes through burning, as happened in the grassland plots, the pH increases due to the formation of carbonates. With time, the carbonates are leached and exchangeable cations (especially calcium) were lost, resulting in a decline of the pH (Binkley et al. 1989; Cruz and del Castillo, 2005; Farley et al. 2008; Yamashita et al. 2008). Table 4.1 also indicates vegetation structure and floral composition change under vegetation succession from *Imperata* grassland to primary forest. Grasslands dominated by *Imperata cylindrica* were replaced by shrubs and young growth of trees. In the plots that were burned 3 years and 9 years before sampling, *Melastoma malabathricum*, *Eupatorium inulaefolium, Ficus* sp. and *Vitex pinnata* became the dominant species, but these species were rarely found in the secondary forest. More detailed information related to Table 4.1 is described by Van der Kamp et al. (2009) and Yassir et al. (2010).

4.3.2. ^{13}C signature

Table 4.2 indicates that the ^{13}C signatures of the four litters are significantly different. This difference is reflected in the signatures of the soil horizons, which reflect the differences of the litters. The largest difference is between the 3-years plots and the other stages. The mean $\delta^{13}C$ value of the litter samples from the *Imperata* grassland plots ($-19.83^0/_{00}$) was lower than that of C_4 plants in general ($-13^0/_{00}$). These low values reflect that the grasslands, in addition to *Imperata*, also contain the C_3 plants *Eupatorium inulaefolium* and *Melastoma malabathricum.* The common occurrence of these C_3 species and their variation in abundance explains the exceptionally high variability (standard deviation = 3.75) of the $\delta^{13}C$ value in the 3-years litter samples compared to the other plots (Table 4.2).

In all the plots, the $\delta^{13}C$ value increased gradually with soil depth (Table 4.2). It also increased from litter to topsoil (A-horizon) except in the 3-years plot. The causes for an increase with depth of $\delta^{13}C$ in systems that have not undergone a C3-C4 change have been discussed in detail by other authors (Nadelhoffer and Fry 1988; Martin et al. 1990; Ehleringer et al. 2000; Balesdent et al. 1993; Balesdent and Mariotti, 1996; Roscoe et al. 2000; Garten et al. 2007; Chen et al. 2008). As selective decay, which has been suggested as a cause, would result in relative accumulation of ^{13}C-depleted compounds such as aromatic and aliphatics (compare Boutton, 1996), this cannot be an explanation. We consider the decay of primary plant material and the admixture of microbial SOM as the most likely cause, as microbial matter has higher $\delta^{13}C$ values than the corresponding SOM (Dijkstra at al. 2006).

Table 4.2. Means of $\delta^{13}C$ parameters in litter and soil samples following the different phases of succession

Descriptions	3 years (n=5)		9 years (n=5)		Secondary forest (n=5)		Primary forest (n=5)	
	Mean	Sd[1]	Mean[2]	sd	Mean	sd	Mean	sd
Litters								
$\delta^{13}C$ $(^0/_{00})$	-19.83a	3.75	-27.72b	0.83	-29.91bc	0.67	-31.07c	0.32
A-horizon								
(0-10 cm)								
$\delta^{13}C$ $(^0/_{00})$	-22.14a	1.00	-25.09b	1.11	-28.19c	0.84	-29.86d	0.33
AB-horizon								
(10-18 cm)								
$\delta^{13}C$ $(^0/_{00})$	-23.00a	0.75	-25.12b	1.16	-27.63c	0.55		
B-horizon								
(18-45 cm)								
$\delta^{13}C$ $(^0/_{00})$	-24.54a	0.43	-26.09b	0.68	-27.30c	0.52	-29.14d	0.24

[1] *Standard deviation*

[2] *Means followed by different letters within one soil parameter differ significantly ($P<0.05$) as established by the LSD-test*

4.3.3. C4-C3 replacement during the succession

The vegetation sequence started with primary forest, which was followed – through logging and slashing and burning – by *Imperata* grassland. The latter, if its turn, was replaced by shrubs and trees. To calculate the remaining forest C in the 3-years plot, we used the mean values of the δ^{13}C soil samples under primary forest for the initial stage, and the δ^{13}C of the 3-years litter to calculate the relative contribution of the *Imperata* and the forest SOM. This implies that the '*Imperata* contribution' includes that of the C3 plants in that vegetation. No correction was applied for a possible increase in δ^{13}C signature from *Imperata* litter to SOM.

Table 4.3 shows that the fractions of C3 carbon (and thus of C4 carbon) in the A-horizon were significantly different in the three regeneration stages. In the deeper horizons, there is no significant difference between the 3-years and the 9-years stage. The 3-years stage retained 23% of forest C in the topsoil, which increased to 51% in the B-horizon. The 9-years stage shows a significant increase in the C3 fraction in the A-horizon, but not in the deeper horizons. The secondary forest has the highest C3-C fractions (82-96%) in all horizons. In the 9-years plot, the increase in C3-C is due to (1) input of the C3-plants (*Melastoma malabathricum*, *Eupatorium inulaefolium*, and *Vernonia arborea*), and (2) loss of the C4 fraction through decomposition. The further increase in C3-C fraction from the 9-years plot to the secondary forest is due to a different set of C3 species (*Syzigium lineatum, Fordia splendidissima, Pternandra azurea, Macaranga* sp., *Vernonia arborea, Vitex pinnata*), and to further decay of the C4-C fraction.

To distinguish between the effects of accumulation and decay, we will consider the absolute amounts of C3- and C4-derived carbon. Table 4.4 shows that the absolute amounts of C4 and C3-carbon in the A-horizons differ significantly between all plots.

The amount of C4-C decreased from 12.1 to 7.2 g kg^{-1} between the 3-and 9-years plots, while the secondary forest had only 0.8 g kg^{-1}. For the AB-horizon, there was a significant difference between the secondary forest and the other two stages, while no significant differences in quantities were found in the B-horizons. Figure 4.2 depicts the absolute amounts of C3 and C4 carbon in the three succession stages, while in Figure 4.2A also the contents of the primary forest are included.

Table 4.3. Means and standard deviations of the C3 carbon fraction in soils under the various succession stages

Depth	C-fraction from C3 [1]					
	3 years (n=5)		9 years (n=5)		Secondary forest (n=5)	
	Mean	Sd [2]	Mean [3]	Sd	Mean	Sd
A-horizon (± 0-10 cm)	0.23a	0.10	0.51b	0.19	0.96c	0.09
AB-horizon (± 10-18 cm)	0.31a	0.07	0.43a	0.23	0.93b	0.10
B-horizon (±18-45 cm)	0.51a	0.04	0.47a	0.20	0.82b	0.15

[1] Imperata grassland fraction; [2] Standard deviation; [3] Means followed by different letters within one soil parameter differ significantly (P<0.05) between regeneration stages as established by the LSD- test

Figure 4.2 shows that under secondary forest, the absolute amount of C4 carbon was very low throughout. Considering that there was no significant decrease in C4-C in the AB-horizon from the 3-years to the 9-years plot, it is unlikely that decay could explain the very low contents of C4-C in the secondary forest plot. It is therefore likely, that the secondary forest plot represented a regeneration of the primary forest after logging and burning, but without a significant *Imperata* phase. This implies that the secondary forest plot cannot be used to estimate decay of C4-C. In that case, the calculated (low) C4-C contributions in the soil under secondary forest are due to the choice of parameters (δ^{13}C of *Imperata* grassland and secondary forest litter) rather than to actual *Imperata* input.

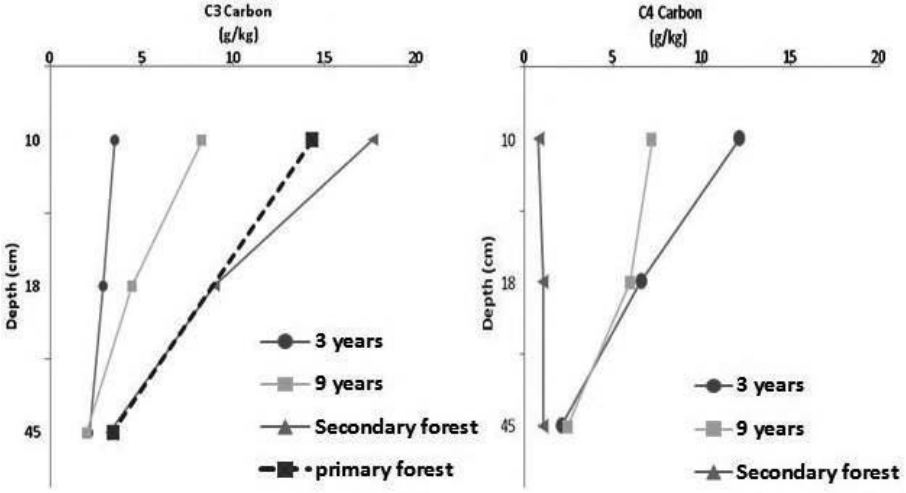

Figure 4.2. Absolute amounts of C3 (2A) and C4 (2B) carbon in the various succession stages

Figure 4.2b and Table 4.4 indicate that the loss of C4-C in the topsoil between the 3-years and the 9-years plot is considerable, while it appears negligible in the AB-horizon. The decay of C4-C in the topsoil amounted to approximately 4.9 g kg^{-1} in approximately 6 years or a decay rate of C4-C is 0.8 g C kg^{-1} per year. In the primary forest, the in A-horizon contained 14.3 g kg^{-1} C3-C, of which only 3.6 g kg^{-1} remained in the 3-years plot. This is a loss of 10.7 g kg^{-1} in approximately 25 years, or a decay rate of 0.4 g C kg^{-1} per year. The differences between decay rate of C4-C and C3-C in A-horizon should be associated with litter quality and substrate quality. The strong decay C4-C in topsoil under vegetation succession indicates that C4-C (*Imperata cylindrica*) –derived SOM is more easily removed by decomposition than C3-SOM.

The strong decay of C4-C in the topsoil may be due to rather high carbohydrate contents in both of the litter and soil organic matter.

Table 4.4. The mean and standard deviations of absolute amounts of C3 and C4 carbon in the various succession stages

Depth	C4[1] (g kg^{-1})						C3[2] (g kg^{-1})					
	3 years (n=5)		9 years (n=5)		Secondary forest (n=5)		3 years (n=5)		9 years (n=5)		Secondary forest (n=5)	
	Mean	Sd[3]	Mean[4]	Sd	Mean	Sd	Mean	Sd	Mean	Sd	Mean	Sd
A-horizon (0-10 cm)	12.1a	2.8	7.2b	2.8	0.8c	1.0	3.6c	1.4	8.3b	4.1	17.6a	7.8
AB-horizon (10-18 cm)	6.5a	1.4	5.9a	2.6	1.0b	0.8	2.9b	0.7	4.5b	2.3	8.9a	3.0
B-horizon (18-45 cm)	2.1a	0.7	2.4a	1.2	1.0a	0.7	2.1a	0.9	2.0a	0.7	3.3a	1.0

1) absolute amounts of C from Imperata grassland; 2) absolute amounts of C from primary forest; 3) Standard deviation; 4) Means followed by different letters within one soil parameter differ significantly (P<0.05) between regeneration stages as established by the LSD- test

Figure 4.2 also indicates that from the 3-years to the 9-years stage the increase in C3-C approximately equals the loss of C4-C. It further suggests that the differences in stock between primary forest and the three succession stages is largely due to the insertion of an AB-horizon (not present under primary forest) which, as observed in the field, has a large volume of *Imperata* roots. The dense root system under *Imperata* grassland might cause higher potential carbon storage in soil than surface litter. The largest difference between profiles under primary forest and under *Imperata* grassland is the insertion of an AB-horizon under the latter. This horizon contains a large amount of roots, and Figure 4.2 clearly suggests that this horizon causes the increase in carbon stocks. Whether the differences in carbon stocks are also related to different chemical composition of the litter input and the resulting SOM will be investigated later. Such insight may help to better understand and predict of soil carbon changes in relation to climate and vegetation change.

4.4. Conclusions

Isotopic signatures of all soil horizons of the regeneration stages were statistically different from those of the primary forest. The A-horizon under the 3-years *Imperata* plot still contained 23% forest (C3) carbon, and this fraction increased to 51% in the-B-horizon. In the 9-years plot and in the secondary forest, the C3 carbon on the A-horizon increased to 51% and 96%, respectively. Our data show that the decay rate in the topsoil of C4-C is 0.8 g C kg^{-1} per year whereas; primary forest is 0.4 g C kg^{-1} per year. Under equal external circumstances, the differences between decay rate of C4-C and C3-C in the A-horizon should be associated with litter quality and substrate quality. However, external factors also change in response to vegetation change. Upon clearing of the forest and establishment of the *Imperata* vegetation, fluctuations in soil temperature will increase, and especially maximum temperature will be considerably higher. As a result, soil moisture will be strongly reduced during the dry season. This combined effect leads to a reduction in SOM decay because both higher temperature and lower moisture availability reduce microbial activity (Cortez, 1998; Liang et al. 2003; Risch et al. 2007). During the secondary succession, the process is reversed and soil temperature and moisture gradually revert to those of the primary forest. Higher microbial activity increases the amount of soil-nitrogen and thus also stimulates decomposition. The combined effects of production and decay on soil-C stocks are summarized in Table 4.5. Although too little information is available for further evaluation of the decomposition speed and stock changes, it is possible that the final C-stock equilibrium will not revert to that of the primary forest. Because both the removal of the forest and the maintenance of the *Imperata* grassland involve burning, the soils of the secondary succession should have some charcoal. As this is a very stable fraction, it is likely that soil-C equilibrium stocks under secondary forest will be somewhat higher than those under primary forest.

Table 4.5. Factors determining carbon stocks in East Kalimantan

Imperata grassland	Secondary forest	Primary forest
Seasonally high temperature	Temperature maxima decreasing with time	Moderate temperature
Seasonally low moisture	Moisture deficit decreasing with time	Permanent moisture
High root litter production	High litter production	Moderate litter deposition
Less efficient SOM decay	Increasingly efficient SOM decay	Efficient SOM decay
Higher carbon stocks	Carbon stock first increases further and then decreases when primary forest equilibrium is regained	Low carbon stocks

CHAPTER 5

Soil organic matter chemistry upon secondary succession in *Imperata* Grasslands: A Pyrolysis-GC/MS study

Chapter based on:

Yassir, I., Buurman, P. 2012. Soil organic matter chemistry upon secondary succession in *Imperata* Grasslands, Indonesia: A Pyrolysis-GC/MS Study. Geoderma, 173-174: 94-103.

Abstract

The chemical composition of soil organic matter (SOM) following secondary succession in *Imperata* grassland was investigated by Pyrolysis-Gas Chromatography/Mass Spectrometry (GC/MS). We studied 46 samples from different stages of succession using plots that last burned 3 and 9 years previously, secondary forest (≥15 years), primary forest and *Acacia mangium* plantation (9 years). During regeneration of *Imperata* grasslands the chemical composition of SOM changes considerably. Differences between litters and SOM were larger than within SOM, which is mainly due to a rapid degradation of lignin in the soil. Both litter and SOM under *Imperata* contain larger amounts of carbohydrates and fewer lignin moieties, aliphatics and N-compounds than those under secondary and primary forest. Nevertheless, SOM degradation under grassland is less efficient because of scarcity of N-compounds. SOM decomposition is most advanced under forest, as indicated by lower amounts of plant derived compounds and higher contribution of microbial matter. Decomposition efficiency appears to be related to SOM chemistry, but more to abundance of N-compounds than to that of potentially recalcitrant compounds. C stocks were linked to decomposition efficiency and litter production.

Keywords; *Imperata grassland, decomposition, pyrolysis GC/MS, secondary succession.*

5.1. Introduction

Regeneration of grassland areas is becoming increasingly important, both to create new secondary forest and to recover the original biodiversity. In previous papers was shown that, during regeneration of *Imperata* grasslands, both vegetation composition and soil properties change, including carbon stocks. We found that the effects of regeneration on soil were strongest in the A-horizon, where an increase in carbon stock, N stock, and C/N ratio, and a decrease in bulk density and pH were observed. Soil carbon stocks increased upon natural regeneration from grassland to secondary forest; highest carbon stocks were found in the later regeneration stages and lowest under primary forest (Van der Kamp et al. 2009; Yassir et al. 2010). Because soil carbon stocks were lower under primary forest than under *Imperata* grasslands, the question arises whether such differences are due to chemically different litter inputs and/or different decomposition efficiency. Such insight may help to predict soil carbon changes in relation to climate and vegetation change.

In Indonesia, most studies focussed on soil carbon quantity changes in relation to land use change (Van Noordwijk et al. 1997; Lal and Kimble, 2000; Woomer et al. 2000; Ohta et al. 2000; Van der Kamp et al. 2009). No studies exist on the chemical composition of soil organic matter in relation to land use changes in Indonesia, but some studies using ^{13}C NMR or pyrolysis-GC/MS exist for temperate areas (Römkens et al. 1998; Nierop et al. 2001a; Mendham et al. 2002; Dorado et al. 2003). Studies for tropical areas are scarce. Buurman and Roscoe (2011) studied soil organic matter chemistry in cultivated fields and Brazilian Cerrado, and compared cultivation practices. Such studies have been linked to SOM dynamics, through which relations with SOM stocks should be expected.

Pyrolysis-GC/MS provides a large amount of structural information on SOM (Saiz-Jimenez and De Leeuw, 1986; Hatcher et al. 2001; Parsi et al. 2007; De la Rosa et al. 2008), and better identification and comparison of the SOM produced under different environments (Saiz-Jimenez et al. 1996; Leinweber and Schulten, 1999; Nierop et al. 2001b; Perobelli Ferreira et al. 2009). It also provides information on, e.g., selective decomposition and preservation of plant-derived material, admixture of microbial products, and addition of charred material (Buurman et al. 2007b; Kaal et al. 2008; Buurman and Roscoe, 2011). We have therefore chosen to use this method to describe the chemical composition of SOM following secondary succession in *Imperata* grasslands. We collected and analyzed samples from different stage of secondary succession, primary forest, and *Acacia mangium* plantation (9 years). The objective of this study was to improve our understanding of the relations between vegetation, SOM chemistry, and SOM stock.

5.2. Materials and Methods

5.2.1. Study area

The study areas Sungai Wain and BOS Samboja Lestari are situated in East Kalimantan, Indonesia (Figure 5.1). Sungai Wain is a unique protected forest of about 10,000 ha that contains one of the last unburned primary forests of the Balikpapan – Samarinda area (Whitehouse and Mulyana, 2004). Samboja Lestari is a 1850 ha reforestation project owned by the Borneo Orangutan Survival Foundation (BOS). Plots selected for the analysis of regeneration impacts are situated at Samboja Lestari, whereas the primary forest plots from Sungai Wain function as a control. The Köppen system classifies the climate of the research area as Af (Tropical Rainforest). Average yearly precipitation is 2250 mm, with a wet period from December to May. The driest month has an average precipitation of 132 mm, and the wettest month of 231 mm. The daily maximum temperature varies from 23°C to 31°C and relative humidity is high. The soils in both areas are formed on marine sediments. Topsoils are generally slightly coarser than the deeper layers. In the FAO classification system (FAO, 2001) the soils of Samboja Lestari and Sungai Wain classify as Acrisols. Nutrient levels are low, especially that of available phosphorus; pH values vary between 4.09 and 4.55 (Yassir and Omon, 2006).

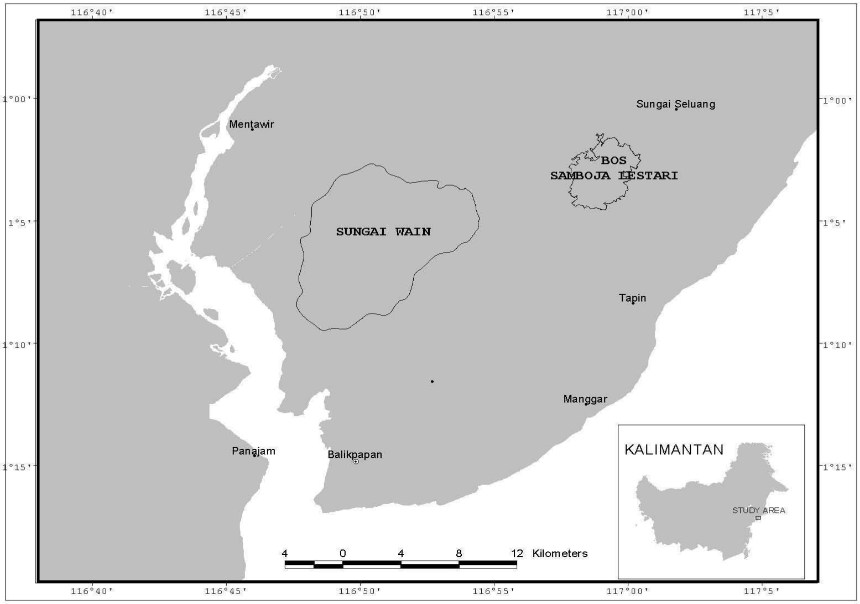

Figure 5.1. Location of Sungai Wain protection forest and BOS Samboja Lestari

5.2.2. Data collection

All field data were collected in the areas of Samboja Lestari and Sungai Wain. A classification of plots in Samboja Lestari was based on the fire history and succession stage (Van der Kamp et al. 2009). 46 samples from different stages of succession were analyzed, using plots that last burned 3 and 9 years previously, secondary forest (≥ 15 years of regeneration), and primary forest. *Acacia mangium* plantations are the main production forests in this area, and at Semboja Lestari we observed a profound change of soil colour under this vegetation. We therefore included soil and litter samples of a 9-year old *A. mangium* plantation. In addition to litter, soil samples were taken of the A-, AB- and B-horizon. The AB-horizon was absent under primary forest and under *A. Mangium.* Methods of soil analysis were listed detail by Van der Kamp et al. (2009). The descriptions of soil properties and vegetation dominances in sampling plot are summarized in Table 5.1.

5.2.3. Organic matter extraction

Soil organic matter was extracted according to the standard protocol of the International Humic Substances Society. In short, 5 g of air-dried soil sample (<2 mm) was extracted with 50 mL of 0.1 M NaOH and shaken during 24 hours under N_2 atmosphere to prevent oxidation. The suspension was centrifuged at 4000 rpm during one hour and the extract was decanted. The extraction was repeated. The two extracts were mixed and the residue was discarded. The extracts were acidified to pH 1 with concentrated HCl to protonate SOM. One mL of concentrated HF was added to dissolve silicates and increase the relative content of organic C of the extracted fraction. The acid mixture was shaken for 48 hours, after which it was dialyzed to neutral pH against distilled water to remove excess salt, using dialysis membranes with a pore diameter of 10000 Da. The solution was then freeze-dried. Litter samples were washed to remove adhering soil, dried at 40°C during 48 hours, finely crushed in a mill, and pyrolysed.

5.2.4. Pyrolisis –GC/MS

Curie-point pyrolysis was carried out using a Horizon Instruments Curie Point Pyrolyser (Horizon Instruments, Heathfield, UK). Freeze-dried SOM extractsand litter samples were pressed on Curie point alloy rods and heated for 5 s at 600°C. The pyrolysis unit was connected to a Carlo-Erba GC8000 gas chromatograph (Thermo Fisher Scientific, Milan, Italy). The pyrolysis products were separated on a fused silica column (Chrompack, 25 m, 0.25 mmi.d.) coated with CP-Sil-5 (film thickness 0.40 mm) using helium as a carrier gas. The initial oven temperature was 40°C, and then rose to 320°C at a rate of 7°C minute^{-1}. The final temperature of 320°C was maintained for up to 20 minutes. The column was coupled to a Fisons MD-800 mass spectrometer (Fisons, Ipswich, UK; mass range m/z 45–650, ionisation energy 70 eV, cycle time 1 s). 46 chromatograms were produced: 13 from A-horizons, 7 from AB-horizons, 13 from B-horizons and 13 from litters). The pyrolysis products were identified using identifications of the NIST library and those published by Ralph and Hatfield (1991); Raven et al. (1997); Van Bergen et al. (1997, 1998), and Nierop et al. (1999, 2001b). Pyrolysis products were quantified using the surface of two characteristic ion fragments of each product (see Table 5.2). The sum of all peak surfaces was set at 100% and relative amounts were calculated. The resulting quantification gives a relative fragment abundance and, because the molecular weights of the individual fragments vary widely, not a weight percentage.

Table 5.1 Soil properties, dominant vegetation and sample numbers

Regeneration Stage	Samples	Bd[a)] (g cm^{-3})	pH	C (g kg^{-1})	N (g kg^{-1})	C/N	P (mg kg^{-1})	Dominant vegetation
3 years								
Litter	1L, 2L, 3L							
A-horizon	1A, 2A, 3A	1.18	5.29	14.52	1.43	10.5	4.04	*I. cylindrica*
AB-horizon	1AB ,2AB, 3AB	1.32		8.99				
B-horizon	1B, 2B, 3B	1.38		3.75			3.16	
9 years								
Litter	11L, 12L							
A-horizon	11A, 12A	1.10	5.09	15.96	1.54	10.4	4.47	*M. Malabathricum*
AB-horizon	11AB, 12AB	1.34		9.10				*E. inulaefolium*
B-horizon	11B, 12B	1.39		3.99			3.72	*V. arborea*
S. Forest								
Litter	16L, 17L							
A-horizon	16A, 17A	1.10	5.11	16.71	1.58	10.6	4.08	*S. lineatum*
AB-horizon	16AB, 17AB	1.32		8.93				*F. splendidissima*
B-horizon	16B, 17B	1.41		4.04			3.60	*P. azurea*
								Macaranga sp.
								V. arborea
								V. pinnata
P. Forest								
Litter	26L, 27L, 28L							
A-horizon	26A, 27A, 28A	1.2	4.82	14.33	1.19	12.0	5.31	*Shorea sp.*
B-horizon	26B, 27B, 28B	1.43		3.43			7.27	*Madhuca sp.*
								Macaranga sp.
								Syzigium sp.
								Gironniera sp
A.mangium								
Liiter	33L							
A-horizon	21A, 22A, 23A	1.13	4.98	18.8	1.55	12.1	4.46	*A. mangium*
B-horizon	21B, 22B, 23B	1.43		3.90			5.42	*N. biserrata*
Imperata Litter	31L							
Pteridium Litter	32L							

[a)]Bd[a]= bulk density (S. Forest=Secondary Forest;= P. Forest=Primary Forest)

5.2.5. Factor Analysis

Because the data set contains 46 cases (samples) with 122 variables each (see Table 5.3), data evaluation is based on factor analysis, using Statistica® Version 8 (Statsoft, Tulsa). Factor analysis is a statistical approach that can be used to analyze interrelationships among a large number of variables and to explain these variables in terms of their common underlying dimensions (factors). It is a prime method to explain the intercorrelations among input variables and to detect structures. Each variable is modeled as a linear combination of the factors plus error terms. Factors are selected by grouping related items and are independent (orthogonal). Consecutive factors account for less and less variance. The degree of correlation between the variables and the extracted factor is called the factor loading, while the values of samples (cases, which each contain all of the variables but in different amounts) for each factor are called factor score. Factor loadings diagrams of organic fragments can be interpreted in terms of different plant contributions, microbial contribution, degradation, etc. Projections of samples in factor space can be understood by examining the projection of factor loadings in the same space. Data sets that have fewer cases (samples) than variables (organic compounds) may result in lower explained variation, but this does not invalidate the observed patterns.

5.3. Results and Discussion

5.3.1. General Chemistry

Almost 350 different pyrolytic compounds were recognized in the 26 samples that were interpreted in detail. After omitting compounds that could not be identified and those that occurred in one sample only, 122 compounds were selected for quantification. The following is based on these 122 compounds, which are listed in

Table 5.2. For convenience, the compounds were grouped according to source and chemical similarity into: *n*-alkanes and *n*-alkenes, other alkenes and alkanals, aromatics, fatty acids, lignin compounds, an ester, carbohydrate compounds, polyaromatics, phenols, N -compounds, a phosphate and a terpenoid. The variation in contents per compound group is given in Table 5.3.

Table 5.2. List of pyrolysis–GC/MS products found in the studied SOM samples

Name	Codes	M/z[a)]	RT[b)]	Name	Code	M/z[a]	RT[b]
n-Alkenes				pyrrole	N3	67	0.34
C_{12} – C_{28} alkenes	12:1-28:1	55+69	1.25-3.61	1H-pyrrole, 2 methyl	N4	80+ 81	0.47
n-Alkanes				1H pyrrole, 3 methyl	N5	81+81	0.49
C_{11}-C_{31} alkanes	11 - 31	57+71	1.06-3.89	pyridine, x-methyl	N6	66+93	0.51
Other Alkenes				benzylamine	N7	106+107	0.65
alkene	Al1	55+69	2.16	benzeneacetonitrile	N8	90+117	1.07
pristene	Pr	56+57	2.26	indole	N9	90+117	1.44
alkene	Al2	55+69	3.10	diketodipyrrole	N10	93+186	2.19
alkanal	Al3	57+82	3.19	***Phenols***			
alkenal	Al4	57+82	3.31	Phenol	Ph1	66+94	0.81
Aromatics				2-methylphenol	Ph2	107+108	0.97
benzene	Ar1	77+78	0.24	3/4-methylhenol	Ph3	107+108	1.00
toluene	Ar2	91+92	0.36	dimethylphenol	Ph4	107+122	1.18
ethylbenzene	Ar3	91+106	0.51	phenol, 4-(1,1,3,3-tetramethylbutyl)	Ph5	107+135	2.05
xylene	Ar4	91+106	0.54	***Polyaromatics***			
styrene	Ar5	78+104	0.57	naphthalene, 1,2-dihydro	Pa1	115+130	1.16
xylene	Ar6	91+106	0.58	naphthalene	Pa2	128	1.21
trimethylbenzene	Ar7	105+120	0.76	x-methyl Naphthalene	Pa3	141+142	1.46
indene	Ar8	115+116	0.91	naphthalene, dimethyl	Pa4	141+156	1.71
ethyl-methyl benzene	Ar9	77+105	0.94	1H-inden-1-one, 2,3-dihydro-7-hydroxy-3-methyl	Pa5	147+162	1.81
Fatty acids				***Polysaccharide compounds***			
C12 fatty acid	F12	60+73	1.98	2-Methylfuran	Ps1	53+82	2.05
C14 fatty acid	F14	60+73	2.31	Acetic acid	Ps2	45+60	0.19
C16 fatty acid	F16	60+73	2.60	(2H)-Furan-3-one(?)	Ps3	54+84	0.21
C18 fatty acid	F18	60+73	2.87	2-furaldehyde	Ps4	95+96	0.38
pentanoic acid, 4-methyl-, methyl ester	ME	74+87	1.63	3-furaldehyde	Ps5	95+96	0.41

Name	Codes	M/z[a)]	RT[b)]	Name	Code	M/z[a]	RT[b]
Lignins				2(3H) furanone, 5-methyl	Ps6	55+98	0.44
guaiacol	Lg1	109+124	1.00	2,3-dihydro-5-methylfuran-2-one	Ps7	55+98	0.50
4-methylguaiacol	Lg2	123+138	1.23	5-methyl-2 furaldehyde	Ps8	109+110	0.62
4-vinylphenol	Lg3	91+120	1.32	3-hydroxy-2-penteno-1,5-lactone	Ps9	58+114	0.70
4-ethylguaiacol	Lg4	137+152	1.41	3,4-dihydropyran-2,5-dione	Ps10	55+112	0.78
4-vinylguaiacol	Lg5	135+150	1.48	3-hydroxy-2/3-methyl-2-cyclopenten-1-one	Ps11	69+112	0.82
syringol	Lg6	139+154	1.55	dianhydrorhamnose	Ps12	113+128	0.86
4-propylguaiacol	Lg7	137+166	1.59	2,5-dimethyl-4-hydroxy-3(2H)-furanone	Ps13	57+128	0.88
4-formylguaiacol, vanillin	Lg8	151+152	1.64	Levoglucosenone	Ps14	68+98	1.01
trans 4-(prop-1-enyl) guaiacol	Lg9	149+164	1.75	3-hydroxy-2-methyl-(2H)-pyran-4-one	Ps15	71+126	1.05
4-methylsyringol	Lg10	153+168	1.73	7-methyl benzofuran	Ps16	131+132	1.05
4-acetylguaiacol	Lg11	151+166	1.80	1,4:3,6-dianhydro-alpha-D-glucopyranose	Ps17	57+69	1.18
4-(propan-2-one) guaiacol	Lg12	137+180	1.88	3,5-dihydroxy-2-methyl-(4H)-pyran-4-one	Ps18	142	1.24
4-formic acid methyl ester	Lg13	151+182	1.86	1,4:3,6-dianhydro-alpha-D-glucopyranose	Ps19	57+69	1.27
4-vinylsyringol	Lg14	165+180	1.95	Dimethylbenzofuran	Ps20	145+146	1.29
4-hydroxy-3methoxy benzoic acid	Lg15	153+168	2.01	1,5-anhydro-D-altitrol	Ps21	74+103	1.56
4-(prop-1-enyl) syringol, trans	Lg16	91+194	2.18	1,5-anhydro-D-altitrol	Ps22	74+103	1.59
4-acetylsyringol	Lg17	181+196	2.23	Sugar compound	Ps23	74+101	1.68
4-(propan-2-one) syringol	Lg18	167+210	2.28	Levoglucosan	Ps24	60+73	1.94
ethylsyringol	Lg19	181+196	2.36	***Phospate***			
N-Compounds				Phosphonic acid, methyl-bis(trimethylsilyl) ester	P	225+240	1.94
1,3/4-dimethyl pyrazole	N1	95+96	0.28	***Terpene***			
pyridine	N2	52+79	0.32	Squalene	Tp	69+81	1.40

**) Codes as used in the Figures; [a] M/z = specific masses used for quantification; [b] RT = Retention time relative to guaiacol*

Table 5.3. Relative abundances (fragment %) of the main chemical groups in relation to regeneration phase*

Groups	3 years	9 years	Secondary forest	Primary forest	Acacia plantation	Imperata litter	Pter-litter
			Litters				
Alkanes and alkenes	2.2-3.2	3.1-3.6	4.3-6.1	4.2-5.3	4.9	1.0	0.7
Aromatics	3.3-7.1	6.8-12.5	8.0-10.1	3.8-5.4	3.8	1.4	1.8
Fatty acids	0.4-0.9	0.7	0.6-1.4	0.8-1.0	1.6	0.7	1.0
Lignins	27.9-34.0	28.5-32.2	25.1-26.7	23.9-26.9	26.6	45.0	4.9
N-compounds	1.1-3.6	0.1	3.4-4.6	0.0-0.1	2.9	0.7	0.4
Polyaromatics	0.4-0.9	0.9	1.1	0.4-0.9	0.4	0.4	0.1
Phenols	5.1-10.3	12.7-15.0	11.0-21.5	10.0-23.7	5.5	4.2	5.7
Carbohydrates	41.1-59.5	36.5-37.1	30.8-43.3	39.5-47.1	53.9	46.4	85.3
			A horizons				
Alkanes and alkenes	4.3-8.0	2.5-3.8	2.9-3.6	7.6-11.2	4.9-7.1		
Aromatics	11.9-18.0	13.3-15.5	11.7-13.9	11.4-15.0	14.9-20.8		
Fatty acids	0.7-2.0	0.5-1.2	1.0-1.1	1.1-1.8	0.3-0.7		
Lignins	7.3-9.9	9.1-10.0	6.8-9.4	9.4-11.3	4.0-9.4		
N-compounds	6.5-10.5	8.0-8.2	8.6-9.4	9.1-10.0	12.2-14.2		
Polyaromatics	0.9-1.2	0.8-0.9	0.8-0.9	1.4-1.7	0.8-1.2		
Phenols	7.9-15.7	10.8-11.7	9.7-12.4	15.8-21.9	10.2-15.3		
Carbohydrates	38.0-55.3	47.7-49.7	50.9-51.3	29.4-37.5	29.5-46.9		
			AB horizons				
Alkanes and alkenes	4.6-7.9	3.1-4.3	3.9-5.6	-	-		
Aromatics	13.6-14.3	13.2-13.7	9.3-13.9	-	-		
Fatty acids	0.5-1.5	0.5-1.0	0.4-1.0	-	-		
Lignins	4.1-4.4	4.0	2.0-3.1	-	-		
N-compounds	7.3-8.9	8.8-9.5	6.9-7.4	-	-		
Polyaromatics	0.8-1.4	0.9	0.5-0.9	-	-		
Phenols	7.2-10.7	8.0-10.3	5.4-7.1	-	-		
Carbohydrates	52.9-56.5	55.5-55.9	61.8-62.7	-	-		

Groups	3 years	9 years	Secondary forest	Primary forest	Acacia plantation	Imperata litter	Pter-litter
			B-horizons				
Alkanes and alkenes	4.6-7.1	4.7-6.9	4.2-6.3	6.1-17.3	4.3-8.3		
Aromatics	8.9-16.5	14.5-16.5	12.2-12.7	14.5-18.4	16.4-22.6		
Fatty acids	0.7-1.4	0.6	1.5-1.6	0.7-2.5	0.7-1.3		
Lignins	0.9-1.9	0.4-2.0	1.8	4.0-6.3	1.4-4.7		
N-compounds	5.4-8.3	7.3-10.1	6.7-8.3	9.7-10.5	11.3-14.8		
Polyaromatics	0.1-1.3	0.5-1.3	0.8-0.9	0.8-1.7	1.1-1.5		
Phenols	3.8-8.9	4.5-9.3	5.8-6.4	10.9-20.0	6.6-16.3		
Carbohydrates	54.9-70.4	52.6-60.5	58.7-56.5	29.3-43.9	32.8-53.7		

** The data are not presented as means and standard deviations because very dissimilar compounds have been summed in each group. (Pter-litter=Pteridium litter).*

The alkanes (C_{11}-C_{31}) and alkenes (C_{12}-C_{28}) constitute only a small part of the pyrolysis products. In all litters, the C_{29} and C_{31} alkanes, pyrolysis products of protective waxes, dominate. In the soils, there is no clear predominance in chain length except for the C_{14} alkane in some samples. Of the alkenes, the C_{12} members dominate in some soil samples. In *Pteridium* litter, the C_{21} alkene dominates. Mid-chain n-alkanes and n-alkenes originate from plant biopolymers (cutin, suberin) as well as from microorganisms (Saiz-Jimenez, 1996: Nierop, 1998; Augris et al, 1998; Kögel-Knabner, 2000; Lorenz et al. 2007), but usually the plant contribution dominates.

In all litters and soils, toluene (Ar2, mainly a pyrolysis product of proteins) is the dominant aromatic compound (up to 4.5% in litters and 13% in soils). In one single litter sample of the 9-years plots, styrene (Ar5) is also very high (5%). Aromatic compounds are largerly ascribed to proteins (Chiavari and Galletti, 1992). Whereas of the fatty acids, the C_{16} member clearly dominates in litters (up to 1.1%), in soils the C_{12} member may reach almost the same abundance. The C_{14}, $C_{16,}$ and C_{18} fatty acids can be both plant-derived and microbial, while the C_{12} member is usually attributed to microbes (Perobelli Ferreira et al. 2009).

Lignin compounds are especially abundant in litters. 4-vinyphenol (Lg3) and 4-vinylguaiacol (Lg5) dominate the litter lignins (3-14%), indicating a rather undegraded lignin. 4-Vinylphenol is a major compound in grass lignin (Saiz-Jimenez and de Leeuw, 1986). In soils, guaiacol (Lg1) is usually the dominant product (up to 3%), accompanied by Lg3 and Lg5 (0.5-2%), and also by 4-acetylguaiacol (Lg11) and vanillic acid (Lg15). This indicates, as expected, a generally more degraded lignin in the soil samples.

Individual N-compounds constitute up to 1.5% in litters and to 3% in soils. Pyridine (N2) dominates in most soil and litter samples, but pyrrole (N3) is dominant in litter of primary forest and *A. mangium* plantation. Diketodipyrrole (N10) is more abundant in primary forest litter than in the other litters. In soils, all N-compounds are more abundant under *A. mangium.* A large abundance of N-compounds in SOM is generally linked with a high degree of decomposition and a significant contribution of microbial SOM (Nierop et al. 2001b; Chefetz et al. 2002; Vancampenhout et al. 2009).

Phenol (Ph1) and 3/4-methylphenol (Ph3) are the most abundant phenol compounds in both litter and soils. Phenol, which is a pyrolysis product with various sources (proteins, lignins) constituted 2-13% of the pyrolysis products in both litters and soils, while Ph3 reached 1-6% in soils and 1-8 % in litters. Polyaromatics are largely due to the incorporation of burned organic matter associated with forest- and grassland fires (Kaal et al. 2008). In the present samples, individual compounds did not exceed 0.5% in litters and 0.8% in soils. In soils, naphthalene (Pa1) and dimethylnaphthalene (Pa4) dominated, in litters the inden-one (Pa5) compound.

Because the litter samples were fresh, it is unlikely that the latter compound is due to burning.

A large number of carbohydrate pyrolysis products are encountered. Litter samples generally showed a predominance of levoglucosan (Ps24), a pyrolysis product of relatively fresh cellulose (5-35%, highest in *Pteridium* and lowest in *Imperata*). Other major compounds (3-15%) were 3-furaldehyde (Ps5) and acetic acid (Ps2). The soil samples showed the same predominance with somewhat less levoglucosan (2-27%) and additional 5-methyl-2-furaldehyde (Ps8, 2-7%). The carbohydrate products methylbenzofuran (Ps16) and dimethylbenzofuran (Ps20) are usually due to burning of wood (Kaal et al. 2008).

5.3.2. Factor Analysis

Factor analysis was carried out for all samples together (Figure 5.2 and 5.3) and for the litter and soil samples separately (Figure 5.4, 5.5, 5.6 and 5.7). In the factor analysis of the total data set, four factors explained 66.7% of all variation, while factors 1 and 2 explained 32.5% and 19.3%, respectively. Because each of the subsequent factors explained less than 10% of the total variability, the discussion will be limited only to the first two factors. The explained variation by two factors (52%) may seem low with respect to other similar studies, but this can be explained by the choice of regeneration stage and soil horizon as the units of comparison. This implies that differences in vegetation community are neglected. Such differences are considerable already in the various regeneration stages, and very large under primary forest with its large number of species. In addition, the soils under secondary vegetations presumably still contain a considerable amount of the original forest SOM. Nevertheless, the chosen units of comparison explain more than 50% of the total chemical variation.

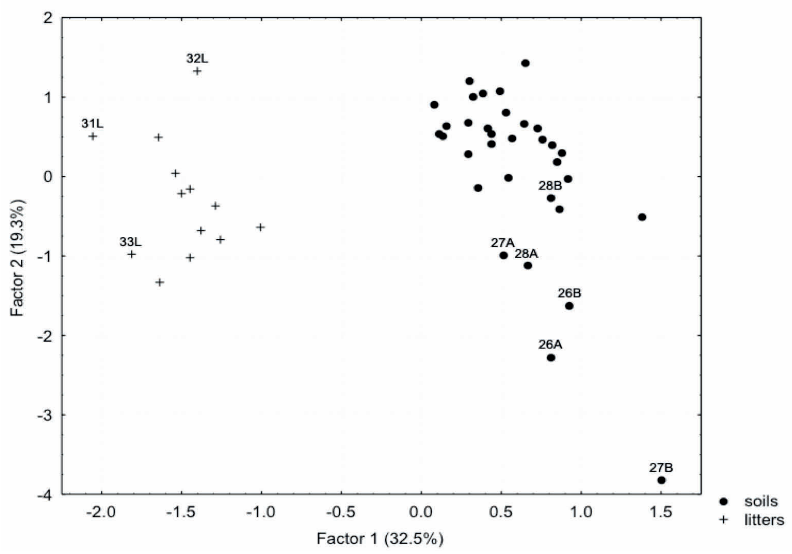

Figure 5.2. Factor scores of litter and soil samples in F1F2 factor space

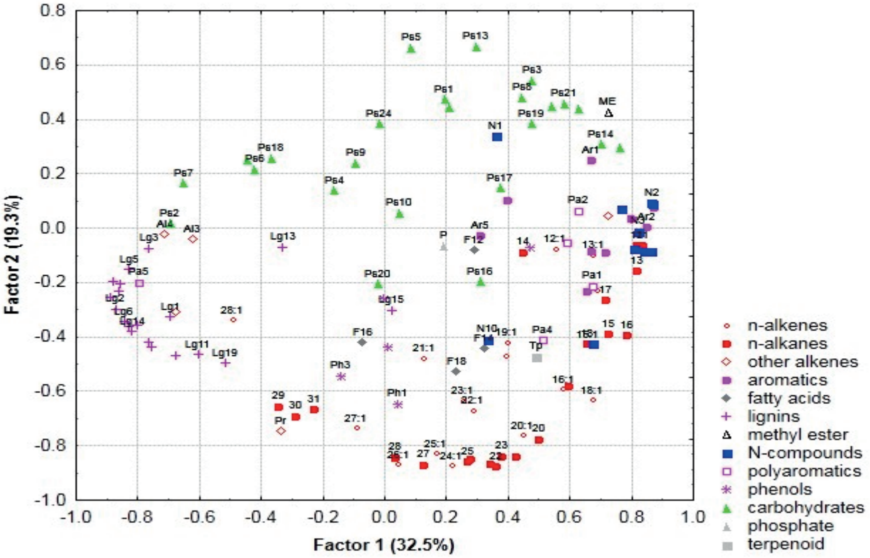

Figure 5.3. Factor loadings underlying the factor scores for litter and soil samples

In the plot of samples in Factor 1–Factor 2 spaces (Figure 5.2), the litter and soil samples are clearly separated: litters on the left and soil samples on the right. Extremes in litter composition are *Imperata* (31L), *Pteridium* (32L), *A. mangium* (33L) and the secondary forest litter 16L. The primary forest litters (26-28) plot towards the base of the cluster. In the soils cluster, the primary forest samples also plot towards the bottom of the cluster. The factor loadings underlying Figure 5.2 are given in Figure 5.3.

Figure 5.3 indicates that all lignin fragments cluster in the bottom left quadrant. Associated with this cluster are alkene (Al2) and alkanals (Al3 and Al4), which occur in the litters alone. Further towards the bottom of the lower left quadrant are the long-chain alkanes (29-31). These are derived from protective waxes and usually not stable in soils. Pristene (Pr), a pyrolysis product of, for instance chlorophyll and therefore also indicative of relatively fresh plant material, plots with the latter group.

All alkanes and alkenes plot in the lower right quadrant, with longer chains towards the bottom and shorter ones towards the far right. The chain-length shortening is generally interpreted as an effect of decomposition (e.g. Buurman et al. 2006, 2007a). The fact that the C_{14} alkane plots further to the left suggests that it has a different origin, while the 21:1 alkene plots outside the cluster because it is especially abundant in *Imperata* litter. Most N-compounds plot close to the shortest alkane/alkene chains on the far right. In the same cluster plots toluene (Ar2), while other aromatic compounds are not far removed. When toluene and N-compounds plot close together, this probably represents microbial matter, as found in decomposing peat (Schellekens et al. 2009). Indole (N9) and diketodipyrrole (N10), which are usually plant-derived, plot in the lower right quadrant. In the upper right quadrant is a large cluster of carbohydrate pyrolysis products. Levoglucosan (Ps24), which represents fresh cellulose, does not plot with this cluster, which probably means that the upper right quadrant represents somewhat degraded plant material. The polyaromatics, except for Pa5 due to burning and therefore associated with the soil samples and not with the (fresh) litter, finally, plot on the far right of the diagram.

From this general distribution of compounds we can conclude that the far left represents fresh plant material, and the far right a predominance of microbial material and surface addition of burned litter. The lower right corner represents a (residual) accumulation of relatively recalcitrant aliphatics resulting from a depletion of plant-derived carbohydrates (cellulose) and lignin. Combining Figures 5.2 and 5.3, we can conclude that the main change from litter to soil is a strong loss of lignins and, in the case of *Pteridium* (32L, upper left quadrant in Figure 5.2) a loss of specific carbohydrates. The rapid degradation of lignin is consistent with the results of Kögel (1986), who found that in forests, the lignin fraction is readily degraded in the litter layer, while the most significant decrease occurs at the transition to the mineral soil. The soil of the various regeneration stages shows a predominance of carbohydrate pyrolysis products that are related to somewhat degraded litter, while the forest soils are characterized by a larger amount of aliphatics. This suggests that SOM in the forest soils is more decomposed than in the regeneration profiles.

To observe the chemical changes in litters and soils more clearly, these sets were analysed separately. Factor scores of the litter samples are presented in Figure 5.4. This figure shows that the SOM in the 3-years plots (1L-3L) is closely related to *Imperata* (31L) and somewhat less to *Pteridium* (32L). Upon regeneration, the chemistry moves to the lower right corner and from there, with the disappearance of the grass vegetation, to the upper right quadrant, where *A. mangium* litter (33L) represents the extreme of the sequence. One should be aware, though, that *Imperata, Pteridium*, and *A. mangium* litters are taken from fresh material, while in the other cases it is a mixture of forest floor material.

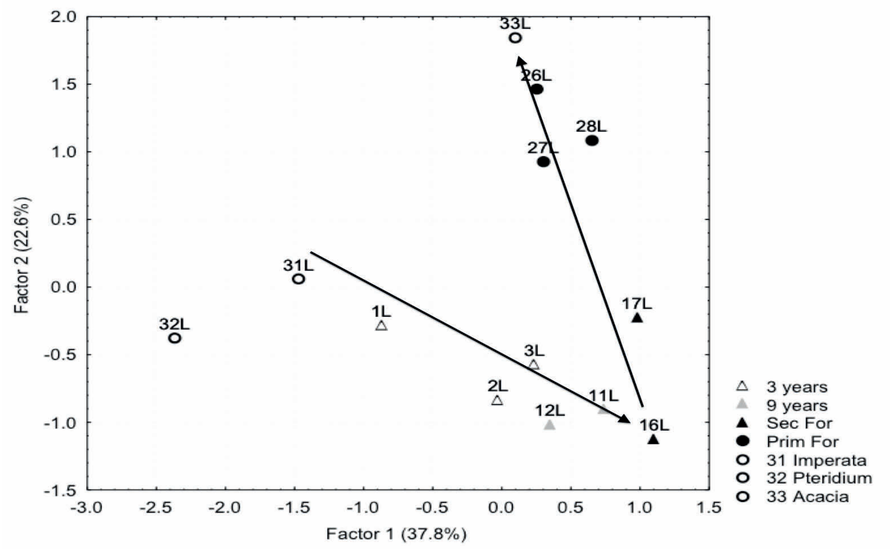

Figure 5.4. Factor scores of litter samples in F1F2 factor space

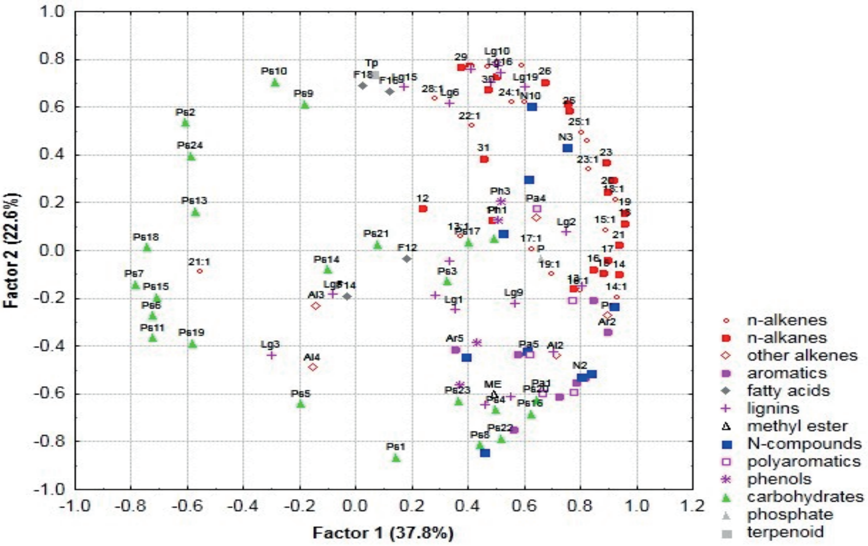

Figure 5.5. Factor loadings underlying the factor scores for litter samples

The chemical changes related to this regeneration can be read from the factor loadings in Figure 5.5. It appears that the left half of Figure 5.4 is associated mainly with a number of carbohydrate pyrolysis products and with the *Imperata*-specific C_{21} alkene. No N-compounds plot in the left hand side of Figure 5.5, which indicates that this is a relatively N-poor system. The shift towards the lower right corner in Figure 5.4 is accompanied by a change in carbohydrate pyrolysis products and an increase in guaiacyl lignin moieties, aromatics, polyaromatics and N-compounds. In this first phase of regeneration, the lipid fraction is still relatively unimportant: most alkanes, alkenes and fatty acids plot in the upper right quadrant. Litter under primary forest appears to be much more aliphatic than that of secondary forest, and also lignin moieties, predominantly syringyls (Lg 6,10,14,16-19), increase under primary forest. The abundance of syringyls with intact C3 side-chain and long-chain aliphatics in litter of primary forest indicates that this litter is relatively undecomposed. Again the association of diketodipyrrole (N10) and $C_{16,18}$ fatty acids with this primary material is noted. The litters in the upper right quadrant are essentially depleted of carbohydrates and indicate selective decomposition and a relative but small (Table 5.3) accumulation of unpalatable compounds.

The fact that polyaromatics are associated with the secondary forest and not with the *Imperata*-litters is due to the fact that the forest floor contains remnants of previous fires while the *Imperata* litter is pure plant material. The paucity of polyaromatics in litter of the primary forest indicates absence of recent fire effects. Furthermore, the combination of aromatics such as toluene (Ar2) with N-compounds in the lower right quadrant suggests that the litter of the secondary forest is somewhat more decomposed – and contains more microbial products - than that of the primary forest.

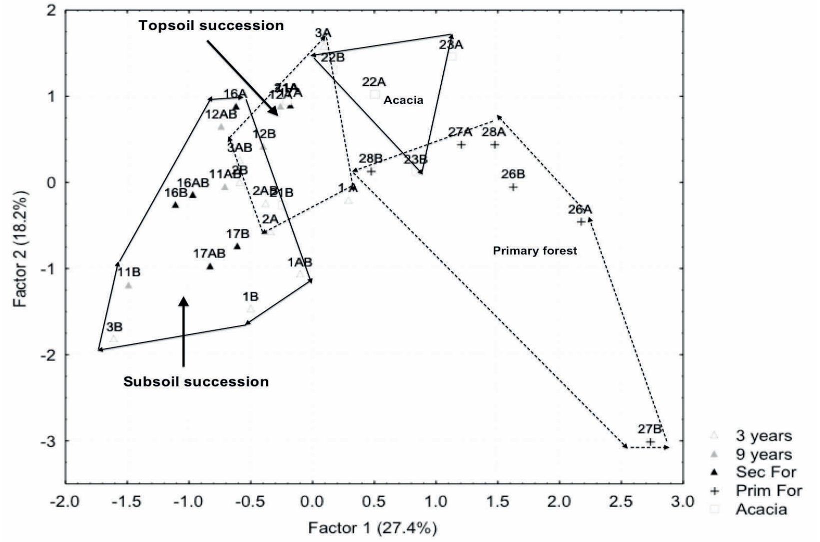

Figure 5.6. Factor scores of soil samples in F1F2 factor space

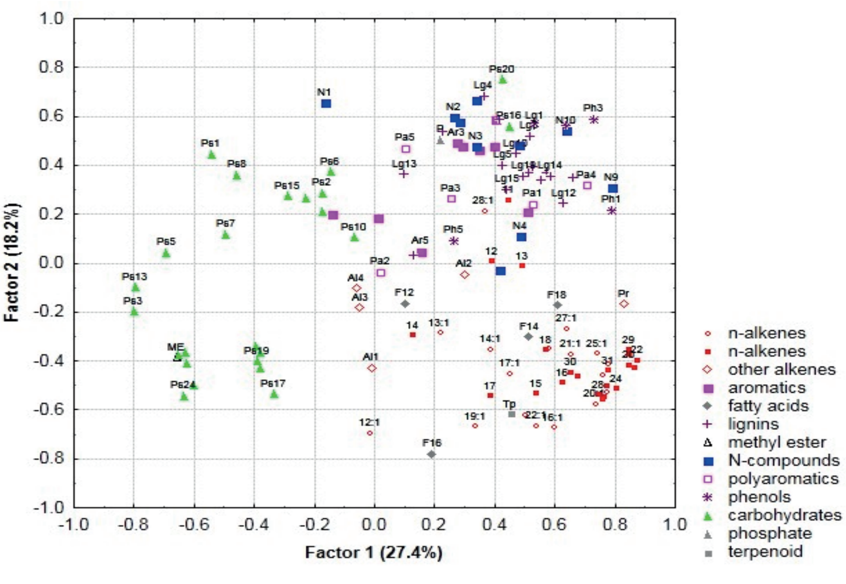

Figure 5.7. Factor loadings underlying the factor scores for soil samples

A separate analysis of the soil samples should indicate whether litter chemistry has an effect on SOM chemistry. Figure 5.6 depicts the factor scores of the soil samples. This picture is more complex than that of the litter samples, because it contains A, AB, and B-horizons. The *A.mangium* samples are found in the upper right quadrant. All horizons of all soils of the 3-years, 9-years, and secondary forest stages are found in a cluster to the left of *A. mangium*. All topsoils and subsoils from the primary forest string out towards the lower right, with A-horizons projecting somewhat more to the top right. The B-horizons of the regeneration stages, finally, plot towards the lower left quadrant. The chemical changes causing these shifts can be read from Figure 5.7. Most N-compounds cluster in the top right quadrant and are predominantly associated with *A. mangium* and with topsoils. Also all lignin and aromatic moieties are found in the top right quadrant. This indicates that the forest soils (native + *A. mangium*) contain more lignin and N-compounds than the regeneration soils. Only one of the forest B-horizons (27B) shows appreciable accumulation of aliphatics (lower right quadrant).

Most of the carbohydrate moieties in the upper left quadrant of Figure 5.7 are those that were found to be related to *Imperata* and *Pteridium* in Figure 5.5, while some in the lower left quadrant, with the notable exception of Ps24, were found in the lower right quadrant of Figure 5.5. This suggests, as expected, that the carbohydrate chemistry of the topsoils of the regeneration stages is more related to present litter chemistry than that of the subsoils. Clearly, the subsoils of the regeneration profiles are characterized by a predominance of (microbial) sugars and a strong relative depletion of lignins, N-compounds, and aliphatics.

The soils under *A. mangium* exhibit the largest abundance of aromatic compounds. This was also observed by Muñoz et al. (2008) and may be due to a larger amount of proteinaceous material under *A. mangium.* The distribution of polyaromatics in the soil samples is peculiar. They appear to be more related to primary forest and *Acacia* plantation than to the *Imperata* grasslands, while in the litter samples they were associated with secondary forest. This can perhaps be explained by the nature of the vegetation. Upon burning, the *Imperata* grassland produces char that is easily carried away by the wind, while large amounts of charcoal are produced when forests burn. Although the technique used is not ideal for quantifying polyaromatics (according to Kaal et al. 2009, an ideal pyrolysis temperature would be 750°C), the present results seem to indicate that charred material is not a major component of the total C stock. Comparing litter chemistry with SOM chemistry, we note that the obvious enrichment in N-compounds of SOM under *A. mangium* is not related to N in surface litter, but presumably in root litter with associated *Rhizobium* (which was not analyzed). Under primary forest, litters are N-depleted, but soils – through microbial activity – have relatively high contents of N-compounds. The rather large aliphatic contents in litter of primary forest are reflected in the related soils.

Carbohydrates in the topsoils of regeneration profiles are related to those of the litter, but this relation is largely lost in the subsoils, where, supposedly, carbohydrates are predominantly microbial. The source of levoglucosan (Ps24) in this case is not quite clear. In most soils, it is clearly related to relatively fresh cellulosic material, but its projection in Figure 5.7 suggests that this is not the case here. In some cases (Vancampenhout and Buurman, unpublished material), it may be a pyrolysis product of microbial carbohydrates, and this would fit better in the present picture. The carbohydrates Ps16 and Ps20, which are both benzofurans, are presumably due to burning and are therefore found in the upper right quadrant, with the polyaromatics. In the soil samples, specific compounds, such as the C_{21}-alkene in *Imperata* litter do not show a relation with vegetation, presumably because they are rapidly decomposed.

5.3.3. Soil carbon stocks and SOM chemistry

An analysis of soil carbon stocks (excluding litter) of the same regeneration sequence (Van der Kamp et al. 2009) indicates that the primary forest has the lowest soil C stock. It is somewhat higher under *Imperata* grassland and gradually increases during the succession towards secondary forest. The whole of the carbon stocks in all these soils is contained in the upper 50 cm of the soil. C stocks are governed mainly by three factors: yearly litter deposition (F_l), litter decomposition rate (k_f), and SOM decomposition rate (k_h). The pool size can be expressed as Pool = F_l $(1-k_f)/k_h$ (Van Breemen and Buurman, 2002). Thus, fast decomposition rates of litter and SOM will decrease the equilibrium C pool, while large yearly litter deposition will increase it. Decomposition rates depend on litter quality and substrate quality. Figures 5.4 and 5.5 indicate that litter chemistry changes considerably upon secondary succession. *Imperata*- and *Pteridium*-related litters are rich in carbohydrates and relatively low in N-compounds. This means that their decomposition is probably N-limited. Change of grass-dominated to tree-dominated vegetation increases the litter N-compounds, but also its lignin and aliphatic biopolymer content, although the latter remains very low. As lignin and aliphatics are not necessarily a stable fraction in tropic soils without severe decomposition restrictions (e.g. Buurman and Roscoe, 2011), litter decomposition under forest is presumably more rapid than under *Imperata.*

Similar considerations are valid for the soil organic matter. Figure 5.6 indicates that, in the secondary succession, SOM in A-horizons is chemically different from that of the subsoil, but there are no systematic differences between the 3-years, 9-years and secondary forest plots. SOM in the A-horizons under primary forest and under *Acacia*, however, is clearly different from that under secondary succession.

Figure 5.7 indicates that, compared to the soils under secondary succession (*Imperata* to secondary forest), those under primary forest have a higher content of N-compounds, more lignin in the topsoil, somewhat higher aliphatic content, and considerably less carbohydrate compounds. This combination probably indicates that under primary forest: (1) the contribution of microbial matter to the surface horizons is larger: (2) concomitantly there is a more efficient decomposition of carbohydrates, including the microbial ones, and (3) this results in a slight residual accumulation of more recalcitrant aliphatics and lignin. The more efficient decay under primary forest may thus be the cause for a lower carbon stock. The C stock under *Imperata* grassland may be further enhanced by its dense root system, the litter of which has a higher potential to be stored in soil than surface litter (Rumpel et al. 2009).

5.4. Conclusions

In the present system, the first decomposition step when litter is converted to soil organic matter (SOM) includes a strong decrease of lignin. Lignin degradation in this step is even more pronounced than that of cellulose (represented by levoglucosan). This indicates that, in agreement with other authors (Kiem and Kögel Knabner, 2003; Dignac et al., 2005; Mikutta et al., 2006; Marschner et al., 2008), lignin is not the recalcitrant compound it was thought to be. SOM under primary forest is generally more degraded than that under *Imperata* or secondary succession, presumably because the soil under primary forest is less N-deficient and there is less input of fresh root litter than under *Imperata*. The grasslands have larger amounts of carbohydrates in both litter and SOM, but degradation appears to be hampered by low N-compounds. Under primary forest, there is a slight relative accumulation of recalcitrant compounds (notably aliphatic biopolymers), which is also related to contents of these compounds in the litter. Plant-derived SOM under primary forest is more strongly degraded than under other vegetations, and microbial SOM contributes significantly. Decomposition under primary forest appears to be efficient and keeps pace with litter production, as was also observed by Fujii et al. (2009).

CHAPTER 6

Reliability of carbon stock estimates in *Imperata* grasslands (East Kalimantan, Indonesia), using geo-referenced information

Chapter based on:

Yassir, I., Van Putten, B., Buurman, P., 2012. Reliability of carbon stock estimates in *Imperata* grasslands (East Kalimantan, Indonesia), using geo-referenced information. Soil Science. 177: 22-30

Abstract

Knowledge of the spatial distribution of total carbon is important for understanding the impact of regional land use change on the global carbon cycle. We studied spatial total carbon variability using transect sampling in *Imperata* grassland area. Spatial variability was modelled following an isotropic stationary process with Spherical and Exponential variogram functions. Range and sill were estimated at 100 m and 82.29 ton^2 ha^{-2}, respectively. Nugget: sill ratio was estimated at about 24%, implying a rather strong spatial dependence. In a subsequent total carbon stock inventory based on the sampling design mentioned above, we applied three types of estimators, namely 'naive average procedure' (NAP), 'spatial average procedure' (SAP) and 'spatial optimal procedure' (SOP). Estimation of total carbon stock (in ton ha^{-1}) following NAP (which erroneously ignores the spatial dependence), resulted in a considerably too narrow 95%-confidence interval of (37.52, 39.75), whereas the outcomes using SAP and SOP were (36.54, 40.73) and (37.14, 40.78), respectively using the Spherical model, and (36.63, 40.64) and (37.07, 40.64), respectively using the Exponential model. Our research indicated that, when total carbon stock estimation is the main goal, random sampling is optimal, whereas wide design sampling (i.e., shortest distance between sampling locations not less than the range) can be preferred in some cases.

Keywords: *Imperata grassland, spatial dependence, total carbon stock, variogram functions.*

6.1. Introduction

Soil organic carbon (SOC) is the largest carbon reservoir in many terrestrial ecosystems including grasslands, savannah, boreal forests, tundras, temperate forests, boreal forests, tropical forests, wetlands and cultivated systems. The Intergovernmental Panel on Climate Change (IPCC, 2000) reported that tropical forests occupy about 12% of the total terrestrial area and store about 17% of the total carbon. This represents a stock about 120 ton ha^{-1} of carbon for the vegetation and 123 ton ha^{-1} for the soil. In wetlands, the estimates are about 43 ton ha^{-1} of carbon for the vegetation and 643 ton ha^{-1} for the soil and in temperate grassland about 7 ton ha^{-1} of carbon for the vegetation and 236 ton ha^{-1} for the soil.

Considering the large variation in SOC, more accurate information is needed at the sub regional scale to better understand the role of this reservoir in global climate and environmental issues (Meersmans et al. 2008). Furthermore, Venteris et al. (2004) mentioned that measuring quantity and spatial distribution of SOC is important also for evaluating and understanding the influence of management history and landscape variables. Moreover, SOC is an essential factor in soil quality. The soil carbon stock strongly depends on factors such as temperature, moisture, vegetation structure/land use, plant input chemistry, texture, soil type and topography. Consequently, it shows a large spatial variability. In spite of its importance, Goidts et al. (2009) stated that uncertainties in SOC stock assessments are rarely quantified even though they are critical in determining the significance of the results.

Tropical forests are among the largest carbon sinks in the world. In Kalimantan (Indonesia), large areas of primary forest have been transformed into secondary forest, oil palm plantation, timber estate plantation, slash-and-burn agriculture, and also into degraded grasslands dominated by *Imperata cylindrica.* According to the latest estimate, *Imperata* grasslands in Kalimantan, which include associations of *Imperata* with ferns and shrubs (e.g., *Chromolaena* sp., *Lantana* sp., and *Melastoma* sp.), cover an estimated 2.2 million hectares (Garrity et al. 1997). MacKinnon et al. (1996) mentioned that *Imperata* grasslands are caused by logging, by forest clearing for shifting cultivation, agriculture and grazing, and also by fire. The latter is a frequent result of human interference.

Because of their large area, *Imperata* grasslands have become increasingly important for the carbon stock of tropical areas. Accurate information on these stocks becomes a crucial issue concerning improved management of agricultural land or reforestation to reduce CO_2 emission, and in understanding the impact of regional land use change on the global carbon cycle.

Soil carbon stocks are usually calculated to a depth of one meter, using carbon contents and bulk densities of a number of pedogenic horizons or of layers of fixed depth. Although this is not a good basis for comparison of carbon stocks in relation to different land use systems (e.g. Ellert et al. 2001, 2002), it is a valid method for calculating stocks of larger areas. For reliable estimates of stocks in larger areas, however, information on spatial variation of properties such as horizon depth, carbon content, and bulk density are required. Information on spatial variability of carbon is very limited. Bonmati et al. (1991), Conant and Paustian (2002), and Zang and McGrath (2004) studied spatial variability of SOC in grassland. Schoning et al. (2006) studied small scale spatial variability of organic carbon stocks in forested Luvisol and Jiang-bing et al. (2008) studied topography and land use effects on the spatial variation of SOC. Other studies have also been carried out under conifer forests (e.g., Palmer et al. 2002; Oliver et al. 2004). Bonmati et al. (1991) and Jiang-bing et al. (2008) found that in grassland SOC had a strong spatial dependence, but information on spatial variability of soil carbon stocks under *Imperata* grasslands is altogether absent.

The present paper's aim is three-fold. The first aim is to study the spatial variability of soil carbon stock in an *Imperata* grassland area of East Kalimantan, using a sampling design consisting of transects, resulting in geostatistical information at various scales. The second aim is to estimate the total carbon stock, based on the information obtained through the sampling design mentioned above and to quantify the quality of the estimator used. The third aim is to derive recommendations for efficient sampling in similar areas.

Our paper differs from many other papers on spatial variability in the sense that our goal is to estimate the carbon quantity globally over an area, instead of estimating it locally at an unvisited location. A possible solution could be sought into the direction of 'block kriging' technique, as discussed e.g. by Isaaks and Srivastava (1989), taking our plot as the area of interest. However, the discretization process causes unnecessary subjective elements in the estimation procedure. Instead, we advocate a more natural type of global estimation, based on best linear unbiased estimation (BLUE), see section total carbon stock estimation. Related studies using different approaches are Schwager and Mikhailova (2002), Falloon et al. (2002), Janik et al. (2002), Ogle et al. (2003), VandenBygaart et al. (2004), Kravchenko et al. (2006), Falloon et al. (2006), Goidts and van Wesemael (2007), Meersmans et al. (2008), Panda et al. (2008), Post et al. (2008), Goidts et al. (2009), Li (2010), VandenBygaart et al. (2011), Kravchenko and Robertson (2011).

6.2. Materials and Methods

6.2.1. Study area

The study area of Samboja Lestari (Figure 6.1) is situated in East Kalimantan, Indonesia. It is an 1850 ha reforestation project owned by the Borneo Orangutan Survival Foundation (BOS). The Köppen system classifies the climate of the research area as Af (Tropical Rainforest). Average yearly precipitation is 2250 mm with a wet period from December to May. The driest month has an average precipitation of 132 mm, and the wettest month of 231 mm. The daily maximum temperature varies from 23 °C to 31°C and relative humidity is high. The soils are formed on marine sediments of Tertiary age. Topsoils are generally slightly coarser than the deeper layers. In the FAO classification system (FAO, 2001) the soils of Samboja Lestari classify as Acrisols. Nutrient levels are low, especially that of available phosphorus; pH values vary between 4.09 and 4.55 (Yassir and Omon, 2006).

6.2.2. Data collection

All field data were collected via systematic sampling along transects. The dataset contains 18 transects with a length varying from 200 to 1000 meters. The number of observations per transect varied from 6 to 24 and distance between observations varied from 2 to 150 meters. In total 250 soil pits were analyzed with dominant vegetation varying from *Imperata* grassland to shrubs. The position of the observations was planned using GPS coordinates. In Samboja Lestari, part of the *Imperata* grassland has been replanted with forest species. Soil samples were taken of the A-, AB- and B-horizons and the depths of these horizons were carefully recorded. For chemical analyses, samples were taken from the full depth of each horizon. Samples were taken to the lab in labelled plastic bags. After air-drying and mixing, the soil was sieved over a 2 mm sieve and packed in smaller labelled bags for transport to the laboratory.

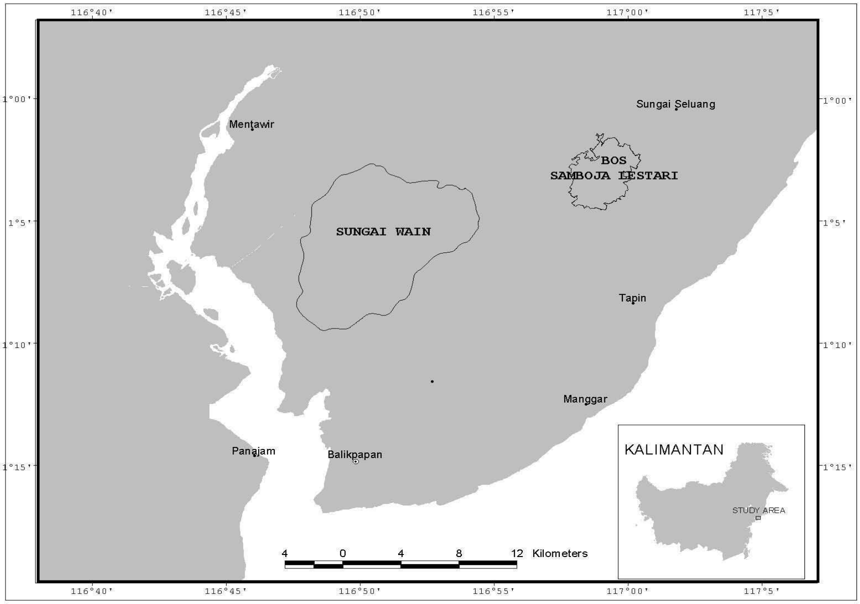

Figure 6.1. Location of BOS Samboja Lestari in East Kalimantan, Indonesia.

6.2.3. Physical and chemical analyses

Methods of soil analysis were listed by Van der Kamp et al. (2009). Bulk density of all horizons was measured at Samboja Lestari, using triplicate measurements with 100 cm^{-3} cylinders. Chemical properties were measured at the Soil Science Laboratory of the Agricultural University (IPB) of Bogor (Indonesia). Total C was determined by wet combustion according to Walkley-Black (A-, AB- and B-horizon). Because changes in soil carbon stocks during regeneration are very small, stocks need to be calculated accurately. As any land use may cause a change in bulk density, using a fixed depth as criterion for comparison of carbon stocks between land uses (or regeneration stages) creates an error because a fixed depth is not equivalent to an equal amount of soil. We therefore based our comparisons on columns of fixed soil mass (see e.g., Van der Kamp et al. 2009). A good description of this method can be found in Ellert et al. (2001, 2002). Some examples of applications of the fixed soil mass method to situations similar to the present research are given in Mannetje et al. (2008). In our case, the equivalent depth section was about 45 cm (see Van der Kamp et al. 2009). Carbon contents in deeper layers were very low.

6.2.4. Data analysis

We used the SURFER, EXCEL 2003/2007 and MATLAB 7.1 programs in the statistical data analysis. A crucial part of the estimation is a careful spatial analysis of the data, followed by an appropriate model choice. We refer to Isaaks and Srivastava (1989), Chilès and Delfiner (1999), and Fortin and Dale (2005) for more details on the subject of spatial statistics.

6.3. Results and Discussion

6.3.1. Data analysis for an appropriate data reduction

In order to avoid rescaling when the observations are integrated to one total carbon stock estimate (in ton ha^{-1}) for the region as a whole, the stock value for all $n = 250$ soil pits was expressed in ton ha^{-1}, instead of the more common value of kg m^{-2} for point data. We produced variograms to investigate and quantify the spatial variation of total carbon. Visual inspection of a contour plot of the data did not reveal any spatial trends. Many experimental variograms were tested; using different lag classes and angular tolerances, omni-directional as well as directional, based on all pairs, and based on pairs within transects and/or parts of the plot only. An appropriate data reduction of the spatial information appeared the omni-directional variogram taken over the whole plot and including all pairs.

In order to obtain proper values for the experimental sill and range, we calculated, for several values of lag distance *h* , the quantity s_h^2 defined by;

$$s_h^2 = \frac{1}{2N_h} \sum_{i<j} {}^{!} (y_i - y_j)^2$$

In this summation, the '!'- sign indicates that the summation is only taken over pairs fulfilling the condition that lag distance $h_{ij} > h$. Integer N_h is the number of pairs involved (pairs not counted twice). With reference to its later use in total carbon stock estimation section, we point out that, at the lag of zero, we have $s_0^2 = s^2$ where s^2 , defined by $s^2 = \frac{1}{n-1} \sum_{i=1}^{n} (y_i - \bar{y})^2$ with $n = 250$, is the so-called dispersion variance. Generally − we do not consider special cases like e.g. 'hole effect' (Cressie, 1991) − the s_h^2 tends to increase with *h* until a stabilized value is reached. The corresponding lag value is the experimental range, whereas the stabilized value itself is the experimental sill. Based on Table 6.1 that summarizes the results, range and sill are estimated at 100 m and 82.29 ton^2 ha^{-2}, respectively. Estimates are based on many observations and therefore quite accurate, in contrast to nugget for which the assignment of a proper value is rather subjective, in accordance with Heuvelink and Pebesma (1999) who observed that the nugget is typically a parameter that is often poorly estimated. This is inherent to the definition of nugget effect, especially because our data set contains a relatively limited number of short-distance pairs despite of the transect sampling scheme along (more or less) logarithmic distances. From the experimental variogram presented in Figure 6.2, we concluded that nugget values roughly between 10 and 40 ton ha^{-1} are appropriate. We come back to this point when we deal with total carbon stock estimation.

Table 6.1. Some estimates of the sill, based on pairs with lags larger than *h*, for several values of *h.*

Lag *h* (meters)	Sill estimate s_h^2 , only based on pairs with lag > *h*	pairs involved
0	81.32	31125
10	81.39	31071
20	81.50	30959
50	81.92	30572
75	82.20	30211
100	82.29	29823
150	82.25	29144
200	82.32	28559
250	82.28	28135

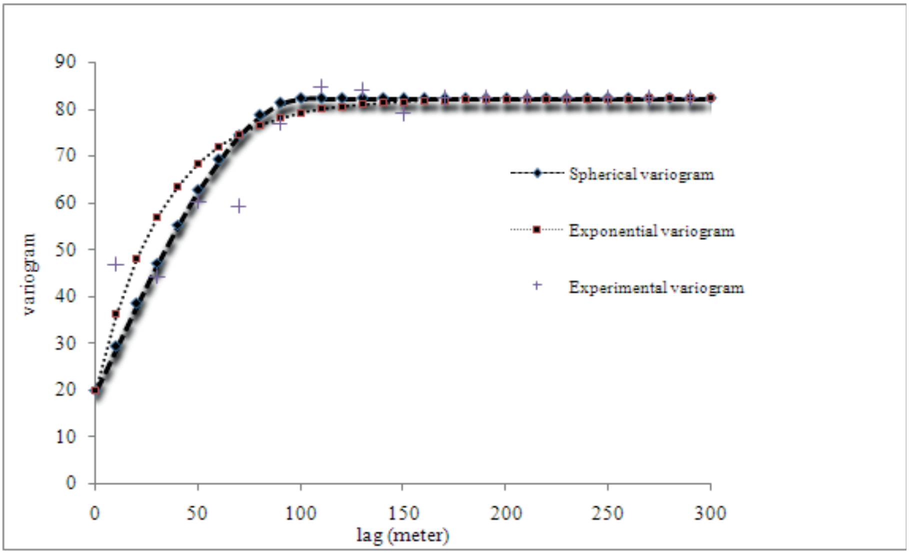

Figure 6.2. Plot of experimental variogram together with the theoretical Spherical and Exponential function (sill σ^2 =82.29, range=100 and nugget=20).

6.3.2. Choice of model

Given the systematic, *i.e.* non-random, sampling design, spatial dependence needs to be accounted for. Results of the data analysis support the choice of a so-called *stationary model* for the carbon stock $Y(\mathbf{p})$ in location $\mathbf{p}$ (visited or unvisited) of the plot. From the data analysis we conclude that the spatial dependence structure can be modelled following a covariance function that is directionally-independent, so in the following we assume an *isotropic* covariance function for the spatial distribution of carbon stock, describing the covariance as a function of the lag vector $\mathbf{h} = (h_x, h_y)$ through its magnitude $|\mathbf{h}| = \sqrt{h_x^{\ 2} + h_y^{\ 2}}$ only.

Our model assumptions therefore will read, for all locations $\mathbf{p}$ and all lag vectors $\mathbf{h}$;

$$\begin{cases} \mathrm{E}Y(\mathbf{p}) = \mu \\ \mathrm{cov}(Y(\mathbf{p}), Y(\mathbf{p}+\mathbf{h})) = \sigma^2 - \gamma(\mathbf{h}) \end{cases}$$

where $Y(\mathbf{p})$ is the value of total carbon at location $\mathbf{p}$, μ and σ^2 are the expectation and the (theoretical) sill, respectively, and γ is an isotropic variogram function such that $\lim_{|\mathbf{h}|\to\infty} \gamma(\mathbf{h}) = \sigma^2$.

In order to quantify the spatial dependence structure, we fitted to our data the two variogram functions that are most commonly used in soil research, namely the Spherical function and the Exponential function. Parametric variogram functions, such as the functions mentioned above, are necessary to avoid negative variances. The forms of the functions are, respectively;

$$Sph(\mathbf{h}) = \begin{cases} 0 & \text{if } |\mathbf{h}| = 0 \\ w + (\sigma^2 - w)[\frac{3}{2}(\frac{|\mathbf{h}|}{r}) - \frac{1}{2}(\frac{|\mathbf{h}|}{r})^3] & \text{if } 0 < |\mathbf{h}| < r \\ \sigma^2 & \text{if } |\mathbf{h}| \geq r \end{cases}$$

and

$$Exp(\mathbf{h}) = \begin{cases} 0 & \text{if } |\mathbf{h}| = 0 \\ w + (\sigma^2 - w)(1 - \exp(-3|\mathbf{h}|/r)) & \text{if } |\mathbf{h}| > 0 \end{cases}$$

Both functions contain the three-parameter sill (σ^2), nugget (*w*) and range (*r*).

Especially the Spherical function appears appropriate in modelling physical and chemical soil properties (e.g. Odeh et al. 1990; Cambardella et al. 1994; Shukla et al. 2004). Because our experimental variograms confirm this, in our next calculations we will take the Spherical model as our preferred model. We also carried out calculations for Gaussian model but decided not to include these results, as the model did not fit the experimental variogram. Many other studies similarly observed that Gaussian model is “too smooth” (e.g., Isaaks and Srivastava, 1989). We briefly come back to disadvantages of Gaussian modelling in the total carbon stock estimation section.

6.3.3. Estimation of the theoretical variogram

The three parameters in each of the Spherical and Exponential models can be estimated from their experimental analogue. Therefore, the theoretical sill σ^2 is estimated by its experimental counterpart, for which we found the value 82.29. Range parameter *r* is estimated at 100 m, conform our data analysis. From the data analysis, we found a likely value of 10-40 for the theoretical nugget, and a value of 20 could serve as an acceptable point estimate. In our calculations of total carbon stock, we will account for uncertainty in the nugget by varying its value.

The relative nugget, defined by the ratio of nugget: sill, is a measure that reflects the strength of the (short distance) dependence (Cambardella et al. 1994; Kravchenko, 2003). Our estimate for the relative nugget is 0.24 that suggests a rather strong spatial dependence.

Simulation studies with sample size 225 by Webster and Oliver (1992), revealed that sampling along transects leads to reliable well-balanced variogram estimation, giving confidence in our model fitting. The range of 100 m we found for total carbon is comparable to that found in other case studies. Cambardella et al. (1994) in Central Iowa observed an organic carbon range of 104 m. Yasreby et al. (2008) found a range of 64 m for organic matter in Southern Iran, and Balasundram et al. (2008) reported a range of 43.5 m for total carbon in Johor, Malaysia.

6.3.4. Total carbon stock estimation

To estimate the total carbon stock *globally* over the plot, we compared three estimation procedures for expectation μ. The first one, which we will call 'naive average procedure' (NAP), applies "classical" sampling techniques developed for independent observations and so, erroneously, does not account for the spatial dependence of the observations in our systematic sampling scheme. Estimator in NAP is the average $\overline{Y} = \frac{1}{n}\sum_{i=1}^{n} Y_i$ of all $n = 250$ observations that obviously is an unbiased estimator of μ. The corresponding standard error $se_{NAP}(\overline{Y}) = \sqrt{\mathrm{var}(\frac{1}{n}\sum_{i=1}^{n} Y_i)}$ is (incorrectly) estimated by $\sqrt{s^2/n}$ where $s^2 = \frac{1}{n-1}\sum_{i=1}^{n}(Y_i - \overline{Y})^2$ is the dispersion variance (see Cochran, 1977, for a correct derivation of this result in the case of uncorrelated observations). Based on the Normal approximation, a 95%-confidence interval for μ is given by $\overline{Y} \pm t_{n-1}(R2.5\%) * \sqrt{s^2/n}$ where $t_{n-1}(R2.5\%)$ is the right hand sided 2.5%-quantile of Student's *t*-distribution with *n*-1 degrees of freedom, which, if *n* is sufficiently large, approaches $\overline{Y} \pm 1.96 * \sqrt{s^2/n}$.

However, as spatial correlations between sampling points for soil characteristics are normally positive, NAP leads to (pretended) 95%-confidence intervals that are too narrow, see e.g. Cressie (1991). Despite this serious setback – see also Dormann (2007) and Haining (2003) – NAP approach is rule rather than exception.

The second procedure, which we will call 'spatial average procedure' (SAP), also applies as an estimator $\overline{Y} = \frac{1}{n}\sum_{i=1}^{n} Y_i$, but correctly takes into account the spatial dependence of the observations in our systematic sampling design. The procedure depends on the model assumptions of isotropy and stationarity that are supported by the findings in the data analysis for an appropriate data reduction section. Estimator $\overline{Y} = \frac{1}{n}\sum_{i=1}^{n} Y_i$ is an unbiased estimator of the expected value μ of total carbon in the stationary model. We obtain a (correct) standard error equal to;

$$se_{SAP}(\overline{Y}) = \sqrt{\operatorname{var}(\overline{Y})} = \sqrt{\operatorname{var}(\tfrac{1}{n}\sum_{i=1}^{n} Y_i)} = \sqrt{\tfrac{1}{n^2}\operatorname{var}(\sum_{i=1}^{n} Y_i)} = \sqrt{\tfrac{1}{n^2}\sum_{i=1}^{n}\sum_{j=1}^{n}\operatorname{cov}(Y_i, Y_j)} =$$

$$\sqrt{\tfrac{1}{n^2}[(n^2\sigma^2 - \sum_{i=1}^{n}\sum_{j=1}^{n}\gamma(\mathbf{h}_{i,j})]} = \sqrt{\sigma^2 - \tfrac{2}{n^2}\sum_{i<j}\gamma(\mathbf{h}_{i,j})}$$

where $\mathbf{h}_{i,j}$ is the distance vector between the locations of observations Y_i and Y_j, respectively, γ is the variogram function and σ^2 is the theoretical sill. If $\gamma(\mathbf{h})$ is increasing sufficiently fast with lag vector magnitude $|\,\mathbf{h}\,|$, a Normal approximation for the distribution of estimator $\overline{Y}$ is quite reasonable when the number *n* of observations is sufficiently large (Janžura, 1994). In that case, (under the very mind conditions for the normal approximation, e,g, finite variance) the corresponding 95%-confidence interval approaches $\overline{Y} \pm 1.96 * \sqrt{\sigma^2 - \frac{2}{n^2}\sum_{i<j} \gamma(\mathbf{h}_{i,j})}$.

Although SAP is correct, it is not optimal. The procedure that applies unbiased linear estimation of μ with the smallest standard error is the 'spatial optimal procedure' SOP, which is the best linear unbiased estimation (BLUE) of expectation μ in the underlying isotropic stationary model with variogram function γ. The BLUE of μ, which from now on is denoted by $\widetilde{Y}$, takes the form $\widetilde{Y} = \sum_{i=1}^{n} w_i Y_i$ where the weights *wi* are to be solved from;

$$\begin{pmatrix} w_1 \\ w_2 \\ \ldots \\ w_n \\ \eta \end{pmatrix} = \begin{pmatrix} \rho_{1,1} & \rho_{1,2} & \cdots & \rho_{1,n} & 1 \\ \rho_{2,1} & \rho_{2,2} & \cdots & \rho_{2,n} & 1 \\ \ldots & \ldots & \ldots & \ldots & \ldots \\ \rho_{n,1} & \rho_{n,2} & \cdots & \rho_{n,n} & 1 \\ 1 & 1 & \ldots & 1 & 0 \end{pmatrix}^{-1} \begin{pmatrix} 0 \\ 0 \\ \ldots \\ 0 \\ 1 \end{pmatrix}$$

In this equation, $\rho_{i,j}$ is the correlation coefficient between observations *Yi* and *Yj*, equal to $\rho_{i,j} = 1 - \gamma(\mathbf{h}_{i,j}) / \sigma^2$ where $\mathbf{h}_{i,j}$ is the distance vector between the locations of observations Y_i and Y_j, respectively, and σ^2 is the theoretical sill. The quantity η, which is included in the above matrix-vector equation to meet the constraint $\sum_{i=1}^{n} w_i = 1$, is a so-called dummy variable. Nevertheless, it has an interesting meaning through $\eta = -\frac{1}{\sigma^2} \operatorname{var} \widetilde{Y}$. In principle, negative weights are possible, underlining the flexibility of the estimation procedure as they can give rise to estimates that are outside the range of the observations Y_i involved. Moreover, we point out that the weights w_i in BLUE $\widetilde{Y} = \sum_{i=1}^{n} w_i Y_i$ are not a function of σ^2. More details about this kind of 'global kriging' technique, including proofs, can be found in Van Putten et al. (2010).

Because the estimator is unbiased, the standard error equals the square root of the (global) kriging variance, so $se_{SOP}(\widetilde{Y}) = \sqrt{-\eta\sigma^2}$. When the number n of observations is sufficiently large and $\gamma(\mathbf{h})$ is increasing sufficiently fast with $|\mathbf{h}|$, the corresponding 95%-confidence interval (under the very mild conditions for the normal approximation, e,g, finite variance) approaches $\overline{Y} \pm 1.96 * \sqrt{-\eta\sigma^2}$.

Based on our dataset, Table 6.2 gives stock estimates and pretended (estimated) standard error following NAP. For SAP and SOP, both for Spherical and Exponential model, estimates and standard errors are produced. In these calculations, we used estimates σ^2=82.29 and r=100 as obtained above. For nugget w we applied various values. In Figure 6.3, corresponding approximate (pretended) 95%-confidence intervals for μ are given. For the nugget, we applied the quite reasonable fixed value of $w = 20$. The calculation for the approximate pretended 95%-confidence interval for μ, following NAP, is;

$$\overline{Y} \pm 1.96 \cdot se(\overline{Y}) = \overline{Y} \pm 1.96 \cdot \sqrt{s^2/250} = 38.634 \pm 1.96 \cdot \sqrt{81.3214/250} = 38.634 \pm 1.96 \cdot 0.5703 = (37.52, 39.75)$$

In accordance to the BLUE property, standard errors for SOP are always smaller than or equal to those for SAP, whereas from Table 6.2 it appears that standard errors for Exponential model are smaller than for Spherical model. In both the Spherical and the Exponential case, higher values of the nugget imply smaller standard errors, tending to the (pretended) NAP standard error. Sizes of standard errors do not critically depend on accurate estimation of the nugget; so total carbon 95%-confidence intervals are rather robust against deviations of the nugget *w*, at least for the relevant values of *w* between 10 and 40.

Table 6.2. Estimates of carbon stocks (ton ha^{-1}) and (pretended) standard errors (*se*) for NAP, SAP and SOP, for Spherical and Exponential models with sill σ^2=82.29 and range *r*=100 both fixed, and for various values of the nugget.

nugget	NAP		SAP			SOP			
	Est.	*se*	Est.	*se* (Sph)	*se* (Exp)	Est. (Sph)	Est. (Exp)	*se* (Sph)	*se* (Exp)
-	38.63	0.57	38.63						
0				1.18	1.13	39.21	39.00	0.99	0.98
10				1.13	1.08	39.07	38.92	0.96	0.94
20				1.07	*1.02*	***38.96***	*38.85*	***0.93***	*0.91*
30				1.00	0.97	38.86	38.79	0.89	0.87
40				0.94	0.90	38.77	38.73	0.85	0.83
50				0.87	0.84	38.70	38.67	0.80	0.79
60				0.79	0.77	38.64	38.63	0.75	0.74
70				0.70	0.69	38.60	38.61	0.68	0.67
80				0.60	0.60	38.62	38.62	0.60	0.60

**) NAP= naïve average procedure; SAP= spatial average procedure; SOP= spatial optimal procedure; the values for nugget equal to 20 are given in italics. Outcomes in the case of SOP and Spherical model are given in bold italics*

We compare the standard errors of NAP on the one hand, and both spatial estimators SAP and SOP in the case of Spherical model on the other hand (parameters $\sigma^2 = 82.29, w = 20$ and $r = 100$), we see that the estimated value of $se_{SAP}(\bar{Y}) / se_{NAP}(\bar{Y})$ is equal to $1.0675/0.5703 = 1.87$ and that the estimated value of $se_{SOP}(\widetilde{Y})/se_{NAP}(\overline{Y})$ is equal to 0.9269/0.5703=1.63, which implies that the 95%-confidence interval produced by NAP is a factor 3.74 (for SAP), and a factor 3.25 (for SOP) times narrower than it should be when spatial dependence is adequately accounted for. Similar calculations can be carried out for Exponential model.

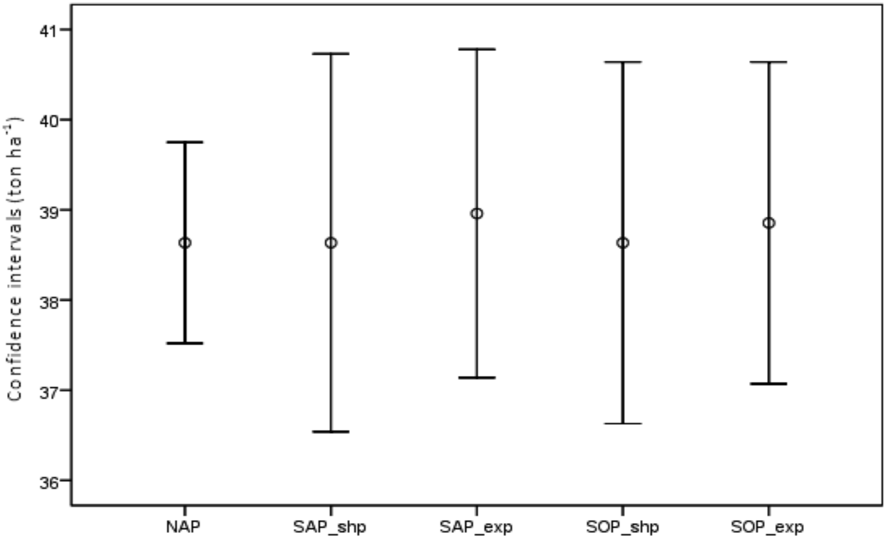

Figure 6.3. Approximate (pretended) 95% confidence intervals for carbon stock μ (ton ha^{-1})

(*NAP= Naive Average Procedure; SAP_shp= Spatial Average Procedure for Spherical model; SAP_exp= Spatial Average Procedure for Exponential model; SOP_shp= Spatial Optimal Procedure for Spherical model; SOP_exp= Spatial Optimal Procedure for Exponential model*; σ^2 =82.29, range *r*=100 and nugget = 20).

Taking the outcome of SOP using Spherical model with $\sigma^2 = 82.29, w = 20$ and $r = 100$, our final conclusion is that for the Samboja Lestari area, the 95% confidence interval of total carbon stock, is (37.14, 40.78) ton ha^{-1}. The 95%

confidence interval (37.52, 39.75) resulting from the (erroneous) ‘naive average procedure’ NAP, is clearly too narrow, especially its upper limit is too small.

Note that estimates and confidence intervals obtained from SAP and SOP are based on models in which estimated parameters for sill, nugget and range are used as if they were equal to their true theoretical values; this is common practice in kriging procedures. So far, no technique seems to be available to correctly account for this additional source of variation. Due to the large number of observations, the effect is small in our case.

We conclude this section with some remarks on the weights. Within SOP, effects of weights w_i can be quite different. Using parameters $\sigma^2 = 82.29, w = 20$ and $r = 100$, we found $\max(w_i) = 0.0104$ and $\min(w_i) = 0.0006$ (Spherical model), and $\max(w_i) = 0.0100$ and $\min(w_i) = 0.0003$ (Exponential model). Despite the poor fit of the Gaussian model – see section choice of model – we applied SAP and SOP also for the Gaussian model (with parameters $\sigma^2 = 82.29, w = 20$ and $r = 100$). Standard errors were considerably higher than in the Spherical and Exponential models. With respect to SOP, we observed that outcomes of weights w_i exhibited a huge variation, especially for values of the nugget in the neighbourhood of zero. In the case of nugget $w = 0$, the maximum weight was equal to 23.22, whereas the minimum weight was equal to -18.45 (with sum of the weights still equal to 1.00). Not surprisingly, we even obtained a negative estimate for total carbon stock, which underlines the conclusion in choice of model section that a Gaussian model is less appropriate in our case, with disastrous outcomes when SOP is applied.

6.3.5. Adequacy of various sampling schemes and accuracy of estimators

There are two main types of sampling design, namely (unstratified) random design (RD) in which sample locations are pre-determined by a random selection procedure, and systematic design (SD) in which the assignment of sample locations goes without any random procedure. A special case of SD is the geometrical design (GD) in which sampling is carried out following a (regular) geometric pattern, e.g. a square grid or a honey comb pattern. Statistical analysis after RD and SD differ greatly. Statistical analysis based on RD can be carried out using classical techniques as described in Cochran (1977). In this so-called ‘sample design based’ strategy, no assumptions need to be made on the spatial distribution of the observations. In contrast, any SD needs to be analysed following a model-based approach, in which spatial dependence plays a crucial role. More information on model and design based approach can be found in e.g. Särndal (1992) and Brus and de Gruijter (1997).

As stated in the introduction, our paper’s first aim was to study the spatial variability of soil carbon stock. Within the limitations of our study, a number of 250 observations were feasible, amply fulfilling Webster and Oliver’s (1992) recommendation to sample at least at some 150 to 200 points. Due to the (expected) lack of sufficient neighbouring pairs at smaller lag distances, allocation of these 250 observations on, e.g., a square grid or according random sampling, would have resulted in a poorly estimated variogram. For that reason, we followed a further recommendation given by Webster and Oliver’s (1992), viz. application of a transect design (TD), in which is sampled along transects (with distances within each transect more or less logarithmic), spread over the region.

In the following, for the spatial distribution of total carbon, we assume an isotropic stationary Spherical model with parameters $\sigma^2 = 82.29, w = 20$ and $r = 100$, the appropriateness of which has been established in estimation of the theoretical section. We will demonstrate that the allocation of sample points following a TD is less efficient when estimate of total carbon stock rather than variogram estimation would have been our primary goal (although TD revealed the appropriateness of this model and its corresponding parameters). In particular, a 'wide design' (WD), in which the shortest distance between locations is not less than the range, needs far fewer observations to reach the same level of accuracy as our TD did in which *n*=250 observations Y_i were involved. In passing, we note that a WD is always of the SD type, but not necessarily geometrical. In WD, covariances between different observations are all zero, so procedures SAP and SOP coincide. The BLUE takes the form $\frac{1}{m}\sum_{j=1}^{m} Y'_j$ where $Y'_1,\ldots,Y'_m$ are the observations, and *m* is the number of observations in the WD. Compared to our SAP calculations of total carbon stock estimation section, we conclude that, in WD, a number of $n_{wsd} = 73$ observations Y'_j would have been sufficient to reach at least the same level of estimation accuracy, following from

$$se_{WD}\left(\tfrac{1}{73}\sum_{j=1}^{73} Y'_j\right) = \sqrt{\frac{82.29}{73}} = 1.0617 \cong 1.0675 = se_{SAP}(\bar{Y}).$$

The similar calculation for SOP:

$se_{WD}(\frac{1}{96}\sum_{j=1}^{96} Y_j') = \sqrt{\frac{82.29}{96}} = 0.9258 \cong 0.9269 = se_{SOP}(\widetilde{Y})$, leads to a number of $n_{wsd} = 96$ required observations instead of our current $n = 250$, without any loss of accuracy. The size of our research area would indeed have allowed a WD with $n_{wsd} = 73$ cq. $n_{wsd} = 96$ observations, so a reduction of the number of observations with a factor of 2.6 to 3.5 would have been feasible.

In a spatial dependence model in which covariances between observations cannot be negative (which is a very common model assumption for soil characteristics), estimation of μ based on a WD of sample size m is always more accurate than estimation based on any other SD of the same sample size, as no linear unbiased estimator can beat WD's BLUE $\frac{1}{m}\sum_{j=1}^{m} Y_j'$ in having a variance lower than σ^2 / m. Note that in a sufficiently large field there are many WDs, and each of the corresponding procedures is equally optimal (although leading to different estimates).

Given a WD of sample size m, under the model assumption of an isotropic stationary field with expectation μ and sill σ^2 and the additional assumption of (multi-) normality of observations $Y_1',\ldots,Y_m'$, estimation of μ is as follows. The variance of the unbiased estimator $\overline{Y}_{WD} = \frac{1}{m}\sum_{j=1}^{m} Y_j'$ is estimated by s_{WD}^2 / m

where s_{WD}^2 is the sample (dispersion) variance. The latter estimation is unbiased, as it holds $s_{WD}^2 = \frac{1}{2m(m-1)} \sum_{i=1}^{m} \sum_{j \neq i}^{m} (Y_i' - Y_j')^2$ and each of the $m(m-1)$ summands fulfils $E(Y_i' - Y_j')^2 = 2\sigma^2$.

Multi-normality of uncorrelated observations $Y_1', \ldots, Y_m'$ implies their independence, so that from classical statistical theory it follows that a 95%-confidence interval for the expected value μ of total carbon in a WD is $\overline{Y}_{WD} \pm t_{m-1}(R2.5\%) * \sqrt{s_{WD}^2 / m}$ where $\overline{Y}_{WD}$ is the average of the observations, m is the sample size, s_{WD}^2 is the (dispersion) variance, and $t_{m-1}(R2.5\%)$ the right hand sided 2.5%-quantile of Students *t*-distribution with *m*-1 degrees of freedom.

If, under the same model conditions, one wants to obtain a confidence interval for μ of a certain pre-described width $2d$, a two-step sampling procedure, adapted from Stein (1945) classical formulas, would be appropriate. In the first step, take a certain number of m_1 initial observations $Y_1', Y_2', \ldots, Y_{m_1}'$ following a WD, from which σ^2 is to be estimated as $\hat{\sigma}^2 = \frac{1}{m_1 - 1} \sum_{i=1}^{m_1} (Y_i' - \overline{Y}')^2$ where $\overline{Y}' = \frac{1}{m_1} \sum_{i=1}^{m_1} Y_i'$. If m is the smallest integer greater than $t_{m_1-1}^2(\alpha/2) \cdot \hat{\sigma}^2 / d^2$, an additional sample of $m_2 = m - m_1$ observations is needed in order to obtain a coverage probability of at least 95% for the interval $(\overline{Y} - d, \overline{Y} + d)$, where $\overline{Y}$ is the average of the pooled sample. An important condition is that all m observations need to be sampled following a WD.

For a WD, formally the validity of an isotropic stationary model is assumed. Knowledge about the functional form of the corresponding variogram is not needed. Only needed is information about the range, e.g. from concomitant research, whereas an upper bound for the range suffices. On the basis thereof, if the dimensions of the area under study are sufficiently large, a WD can be constructed. The stationarity assumption can be relaxed in various ways, for instance by incorporating a linear trend term in the model; however we do not process this here.

An alternative to wide design WD is random design RD. The following construction of a random sampling is applicable for a plot of any form. Let $X_{\min}$ and $X_{\max}$ be the minimum and maximum *X*-coordinate of points within the plot, and let $Y_{\min}$ and $Y_{\max}$ be the minimum and maximum *Y*-coordinate, respectively. The Excel program (and many other programs) can provide (pseudo) independent uniform random numbers X and Y from interval $[X_{\min}, X_{\max}]$ and $[Y_{\min}, Y_{\max}]$, respectively. If the location with coordinates (X,Y) is within the field, put it in the sample, otherwise skip it, and process selecting locations until the required sample size has been obtained.

Nor WD neither RD is a suitable tool for investigating the spatial structure, as it is to be expected that spatial information gained at small lag distances will be poor, especially when the field is extensive and the sample size is limited. For estimation of μ, however, the RD is very well suited. No model assumptions are needed, apart from the very mild assumption that the random variable representing total carbon at a randomly selected location has a finite variance, to validate normal approximation. Its expected value μ coincides with its counterpart of the same name in the spatial model when stationarity is assumed.

Because of the independence of observations in the corresponding design-based model, classical formulas can be applied, so that for observations $Y_1',\ldots,Y_m'$ taken from a RD, an approximate 95%-confidence interval for μ can be constructed, equal to $\overline{Y}_{RD} \pm t_{m-1}(R2.5\%) * \sqrt{s_{RD}^2 / m}$ where $\overline{Y}_{RD}$ is the average of the observations, m is the sample size, s_{RD}^2 is the (dispersion) variance, and $t_{m-1}(R2.5\%)$ the right hand sided 2.5%-quantile of Students *t*-distribution with *m*-1 degrees of freedom. Moreover, Stein's (1945) two-stage approach is applicable as well, where in both stages sampling needs to be carried out randomly. Formulas are literally the same as in the WD case, so we do not repeat them here.

We conclude this section by a comparison between WD (and more general also SD) and RD, both in case estimation of μ is the main goal. First, we consider the situation in which an isotropic stationary model is assumed. No specification about the variogram γ are needed, except (an upper bound for) the sill σ^2, so that some designs can be identified as WD. For the expected value of the dispersion variance s^2 of a sample $Y_1',\ldots,Y_m'$ with distances $\mathbf{h}_{ij}$ between the locations it holds

$$Es^2 = \frac{1}{2m(m-1)} \sum_{i=1}^{m} \sum_{j \neq i}^{m} E(Y_i' - Y_j')^2 = \frac{1}{m(m-1)} \sum_{i=1}^{m} \sum_{j \neq i}^{m} \gamma(\mathbf{h}_{ij}),$$

which replaces a lengthy derivation by Barnes (1991). From this, the formula $Es^2 \leq \sigma^2$ is easily derived by observing that each of the $m(m-1)$ summands in the last expression is less than or equal to the sill (under the rather natural assumption that negative correlations, e.g. in hole effects, do not occur; Cressie, 1991), where the equality sign in the formula holds true if and only if the observations are taken from a WD. Now let be given two samples, both of size m, one from a WD and one from a RD, with dispersion variances s_{WD}^2 and s_{RD}^2, respectively. As $Es_{RD}^2 < Es_{WD}^2 (= \sigma^2)$, the (approximate) 95%-confidence interval $\overline{Y}_{RD} \pm t_{m-1}(R2.5\%) * \sqrt{s_{RD}^2 / m}$ based on RD, in expectation will be narrower than the 95%-confidence interval $\overline{Y}_{WD} \pm t_{m-1}(R2.5\%) * \sqrt{s_{WD}^2 / m}$ based on WD. Thus, estimation based on RD is more accurate than based on WD, and WD in turn provides the most accurate estimation among all type of SD, it can be concluded that RD is more accurate than each of SD. In the case a stationary model assumption has not been made, SD (and WD) is not a feasible way, as covariances between observations cannot be quantified. However, in that case RD is still a correct approach, as it does not require any model assumption (see above).

Although thus RD, in terms of accuracy and model choice, appears to be optimal, WD nevertheless can be preferred for good reasons, e.g. in case the m sample points are evenly distributed over an areal extent many times larger than the range of the variogram. In that case, Barnes (1991) correctly observes that s^2 is a reasonable first estimate for sill σ^2. The consequence is that, under isotropic and stationary model assumption, estimation based on a WD, for instance in the form of a square grid or honey comb pattern, can be an attractive alternative to RD, as it provides (almost) optimal estimation accuracy and, due to its survey ability, in practice is more conveniently carried out.

6.4. Conclusions

The spatial dependence of total carbon, as summarized in the variogram of Figure 6.2, does not exhibit surprising results, when compared with literature on the same subject. A range of about 100 m is also found in other case studies. Total carbon that has a strong spatial dependence may be controlled by changes in texture, mineralogy, topography and vegetation/land use. Our results indicate that, in estimating the total carbon stock, the spatial dependence cannot be ignored. The ‘Naive Average procedure’ NAP leads to 95%-confidence intervals that are considerably too narrow. With respect to the correct procedures SAP and SOP, the latter is recommended because of its smaller standard error.

The following recommendations are derived for research in similar areas. When insight in the spatial dependence structure is the main goal, a transect design is the best sampling strategy. Subsequent carbon stock estimation should be carried out following the SOP procedure, at least when an isotropic stationary model can be assumed. If carbon stock estimation is the main goal, the expected confidence interval of random sampling is smaller than that of a systematic sampling design. In the important case that the areal extent is many times larger than the variogram range, a wide sampling design can be preferred instead, because of its (near-) optimality and its practical survey ability.

CHAPTER 7

Conclusions and Recommendations: a synthesis of soil carbon stocks and changes upon forest regeneration

7.1. Introduction

This chapter reviews the results in relation to specific objectives stated in the general introduction of Chapter 1. This synthesis is divided into six topics. The first deals with secondary succession in *Imperata* grassland. The second deals with soil carbon stocks and changes under secondary succession. The third deals with chemical composition of SOM in the various stages of secondary succession. The fourth deals with changes in carbon dynamics composition under secondary succession. The fifth deals with reliability of soil carbon stock estimates using geo-referenced information. Finally, the sixth deals with implications of this research and recommendations based on its results.

7.2. Secondary succession pathway in *Imperata* grassland

Imperata grassland is a common vegetation type in Kalimantan, Indonesia. It indicates a high degree of degradation of the vegetation, and mostly occurs after slashing and burning of primary forests. Through secondary succession *Imperata* grassland is converted into new secondary forest and most of the original biodiversity is restored. I studied the pathways of secondary succession from *Imperata* grassland to secondary forest in the Borneo Orangutan Survival Foundation project area.

Chapter 2 documents an early succession in *Imperata* grassland, which shows that the percentage of shrubs and young trees increases from *Imperata* grassland to secondary forest, whereas *Imperata* decreases. After three years of regeneration, *Melastoma malabathricum* represented about 42%, *Eupatorium inulaefolium* 25%, and *Ficus* sp. 14% of the total shrubs and young trees. This changes little in the 4-year old growth, where *Melastoma malabathricum* represents about 36%, *Eupatorium inulaefolium* 25%, and *Ficus* sp. 10%. After nine years of regeneration, the pioneer *Melastoma malabathricum* reached about 42%, *Eupatorium inulaefolium* 9%, and *Vitex pinnata* 20%. In the plots with secondary forest, the pioneer *Syzigium lineatum* reached about 21%; *Fordia splendidissima* 19%; *Pternandra azurea* 8%, and *Macaranga* sp. 6%.

In this area soil texture appears an important factor in the vegetation succession. On sandy soils, there was a strong increase with time of *Pteridium aquilinum* L., while the number of other species was lower than on clay soils. This slows down the development towards secondary forest. The lower number of shrub species on sandy soils might be more due to *Pteridium* than to the texture itself. *Pteridium aquilinum* can reach a height of 2-3 meters and casts much more shade than *Imperata* grassland. In addition, it has thick and deep rhizomes and slowly decomposable litter, which may impede germination of seeds from other species. This hypothesis is supported by a statement of Den Ouden (2000) that *Pteridium aquilinum* can induce stagnation in succession through shading, smothering, the build up of a deep ectorganic soil layer and the support of a high density of herbivore and seed-eating rodents.

The secondary succession pathway in *Imperata* grassland indicated two successional pathways. The first pathway is dominated by *Imperata* grassland (model A) and the second by *Pteridium* (model B). In soils that contain more sand, the succession is more likely to follow model B, where *Pteridium* dominates. Shrubs and young trees in Model B will appear later than in plots where the regeneration follows model A where *Imperata* grassland dominates.

In Chapter 2, I used Canonical correspondence analysis (CCA) to explore the relation between community structure and environmental gradients. Result showed that pH, bulk density, sand and clay were the factors related to the distribution of species. Furhermore, CCA indicates that *Melastoma malabathricum, Vitex pinnata, Lycopodium cernum, Dicranopteris linearis, Vernonia arborea* were all species associated with high levels of exchangeable Al and low pH.

7.3. Soil carbon stocks and changes in the succession of *Imperata* grassland

In Chapter 3, I studied soil carbon stocks and changes under the succession of *Imperata* grassland, using plot conditions as in Chapter 2. I studied soil organic carbon because it is important both for its effect on potential production and for possible implications of forest degradation and regeneration on CO_2 storage and thus on global climate change.

Results show that during regeneration of *Imperata* grassland not only vegetation composition but also soil properties change. The effects of regeneration on soil were strongest in the A-horizon, where an increase in carbon stock, N, and C/N ratio, and a decrease in bulk density and pH were observed. Soil carbon content increased with 14%, from 14.5 g kg^{-1} in *Imperata* grassland to 16.5 g kg^{-1} in secondary forest, while carbon stocks in the A-horizon increase from 16.01 ton C ha^{-1} to 18.69 ton C ha^{-1}. The total soil carbon stocks (fixed mass, approximate depth section 40 cm) were 36.19 ton C ha^{-1} under *Imperata* grassland, 38.98 ton C ha^{-1} under secondary forest and 33.19 ton C ha^{-1} under primary forest. The latter is considerably lower than result obtained in Sumatra.

My results are consistent with those of Ohta et al. (2000), who found that in East Kalimantan; the stock of total carbon did not decrease with forest degradation and in fact increased in every layer upon conversion of forest to grassland. It is different result from Woomer et al. (2000) who found a higher soil carbon stocks under primary Dipterocarp forest in Sumatra (48.1 ton ha^{-1} in the first 20 cm) than under *Imperata* grassland. The cause of this discrepancy is not completely clear, but the soils in East Kalimantan are generally shallow, poor in nutrients, with shallow rooting systems and very prone to erosion, whereas those in Sumatra tend to be deeper, more developed, and deeply rooted. Besides differences of soil conditions between Sumatra and East Kalimantan, the differences between the ecosystems of primary forest and *Imperata* grassland itself may also affect carbon stocks. In the primary forest, the atmosphere is very moist and temperatures are relatively low and constant. In *Imperata* grassland the humidity is lower and the day temperature is relatively high. These differences will probably affect the decomposition rate of soil organic matter.

7.4. Carbon dynamics changes in the succession of *Imperata* grassland

In Chapter 4, to understand why carbon stocks increase under secondary succession, and to have an impression of build up of SOM fractions contributed by C4 and C3 vegetation, I studied SOM dynamics using stable carbon isotope ratio mass spectrometry (IRMS), using plots in the previous paper. The results show that during regeneration of *Imperata* grasslands the isotope signature changes significantly. This change during the development of the succession is closely related to vegetation changes, when C3 plants replace C4 plants.

The A-horizon under the 3-years *Imperata* plot still contained 23% forest (C3) carbon, and this fraction increased to 51% in the B-horizon. In the 9-years plot and in the secondary forest, the C3 carbon on the A-horizon increased to 51% and 96%, respectively. In the topsoil, the loss of C4-C between the 3-years and the 9-years plot was significant, while it appeared negligible in the AB-horizon. Result also indicates that from the 3-years to the 9-years stage the increase in C3-C approximately equals the loss of C4-C. It further suggests that the differences in stock between primary forest and the three succession stages are largely due to the insertion of an AB-horizon (not present under primary forest), which, as observed in the field, has a large volume of *Imperata* roots. The dense root system under *Imperata* grassland might cause higher potential carbon storage in soil than surface litter.

7.5. Chemical SOM, - relation to carbon stocks

In Chapter 5, to better understand whether the differences in carbon stocks are related to different chemical composition of the litter input and SOM, I studied the chemistry composition of extractable SOM using pyrolisis-gas chromatography/mass spectrometry (Py-GC/MS). Results showed that during regeneration of *Imperata* grasslands chemistry of SOM changes in relation to vegetation composition. Lignin compounds were abundant in litters. Largest differences between litter and SOM are caused by rapid lignin degradation. Both litter and SOM under *Imperata* contain larger amounts of carbohydrates and fewer lignin moieties, aliphatics and N-compounds than those under secondary and primary forest.

Soil C stocks are governed mainly by three factors: yearly litter deposition (F_l), litter decomposition rate (k_f), and SOM decomposition rate (k_h). The pool size can be expressed as Pool = $F_l\ (1-k_f)/k_h$ (Van Breemen and Buurman, 2002). Thus, fast decomposition rates of litter and SOM will decrease the equilibrium C pool, while large yearly litter deposition will increase it. Decomposition rates depend on litter quality and substrate quality.

Results indicate that litter chemistry changes considerably upon secondary succession. *Imperata*- and *Pteridium*-related litters are rich in carbohydrates and relatively low in N compounds. This means that their decomposition was probably N-limited. Change of grass-dominated to tree-dominated vegetation increases the litter N content, but also its lignin and aliphatic biopolymer content, although the latter remains very low. As lignin and aliphatics are not necessarily a stable fraction in tropical soils without severe decomposition restrictions (e.g. Buurman and Roscoe, 2010), litter decomposition under forest is presumably more rapid than under *Imperata.*

Compared to the soils under secondary succession, however, the topsoils under primary forest have lower content of N-compounds, more lignin in the topsoil, somewhat higher aliphatic content, and considerably less carbohydrate compounds. This combination probably indicates that under primary forest: (1) the contribution of microbial matter to the surface horizons is larger: (2) concomitantly there is a more efficient decomposition of carbohydrates, including the microbial ones, and (3) this results in a slight residual accumulation of more recalcitrant aliphatics and lignin. The more efficient decay under primary forest may thus be the cause for a lower carbon stock. The C stock under *Imperata* grassland may be further enhanced by its dense root system, the litter of which has a higher potential to be stored in soil than surface litter (Rumpel et al. 2009).

7.6. Reliability of soil carbon stocks estimates

In chapter 6, I studied the spatial variability of soil carbon stocks using transect sampling and the methodology of estimating carbon stocks in *Imperata* grassland. Considering the large spatial variability in soil organic carbon, more accurate information is needed at the sub regional scale to better understand the role of this reservoir in global climate and environmental issues (Meersmans et al., 2008). Moreover, Venteris et al., (2004) mentioned that measuring quantity and spatial distribution of soil organic carbon is important also for evaluating and understanding the influence of management history and landscape variables. Applying geo-referenced information, spatial variability was modelled following an isotropic stationary process with Spherical and Exponential variogram functions. Range and sill were estimated at 100 m and 82.29 ton^2 ha^{-2}, respectively. Nugget: sill ratio was estimated at about 24%, implying a rather strong spatial dependence.

Based on the transect sampling data, I applied three types of estimators, namely 'naive average procedure' (NAP), 'spatial average procedure' (SAP) and 'spatial optimal procedure' (SOP). Estimation of total carbon stock (in ton ha^{-1}) following NAP (which erroneously ignores the spatial dependence), resulted in a considerably too narrow 95%-confidence interval of (37.52, 39.75), whereas the outcomes using SAP and SOP were (36.54, 40.73) and (37.14, 40.78), respectively using the Spherical model, and (36.63, 40.64) and (37.07, 40.64), respectively using the Exponential model. With respect to the correct procedures SAP and SOP, the latter is recommended because of its smaller standard error.

7.7. Implications and Recommendations

The Intergovernmental Panel on Climate Change (IPCC, 2000) reported that tropical forests occupy about 12% of the total terrestrial area and store about 17% of the total carbon. This represents a stock about 120 ton ha^{-1} of carbon for the vegetation and 123 ton ha^{-1} for the soil. In wetlands, the estimates are about 43 ton ha^{-1} of carbon for the vegetation and 643 ton ha^{-1} for the soil and in croplands about 2 ton ha^{-1} of carbon for the vegetation and 20 ton ha^{-1} for the soil.

Compared to the Intergovernmental Panel on Climate Change (IPCC, 2000) data, my research showed that soil carbon stocks increase upon natural regeneration from grassland to secondary forest, and that highest carbon stocks are found in the later regeneration phases and lowest under primary forest. This implies that soil carbon stocks in *Imperata* grasslands of East Kalimantan are lower than previously thought. My research supports the statement by Mori et al. (2000) that the hypothetical rapid decrease in soil carbon as a result of forest loss and its slow recovery under secondary vegetation are misconceptions.

Soil carbon stocks strongly depend on factors such as temperature, moisture, composition of vegetation, plant input chemistry, soil texture, soil type and topography. Consequently, it shows a large spatial variability. Because of the large spatial variability in soil organic carbon, generalisation even for similar conditions in the tropics would result in misconceptions. Thus, further research is still needed at the sub-regional scale in order to better understand the role of soil carbon reservoirs in global climate and environmental issues.

Our results suggest that in order to estimate carbon stock using systematic sampling, the spatial dependence cannot be ignored. Recommendations are given for estimation of carbon stocks in similar areas. When insight in the spatial dependence structure is the main goal, a transect design using different lag distances is an adequate sampling strategy. Subsequent carbon stock estimation should be carried out following the ‘spatial optimal procedure’ (SOP), at least when an isotropic stationary model can be assumed. If carbon stock estimation is the only main goal, random sampling is preferred to systematic sampling design. In the case the areal extent is many times larger than the variogram range, of the underlying stationery model a wide sampling design should be preferred instead, because of its (near-) optimality and its practical survey ability.

References

Alauzis, M.V., Mazzarino, M.J., Raffaele, E., Roselli, L., 2004. Wildfires in NW Patagonia: long term effects on a Nothofagus forest soil. Forest Ecology and Management, 192: 131-142.

Amézquita, M.C., Amézquita, E., Casasola, F., Ramirez, B.L., Giraldo, H., Gómez, M.E., Llanderal, T., Velázquez, J., and Ibrahim, M.A., 2008. C stocks and sequestration. In: ‘t Mannetje, L., Amezquita, M.C., Buurman, P., Ibrahim, M.A. (Eds.), Carbon sequestration in tropical grassland ecosystems. Wageningen Academic Publishers, Wageningen, pp. 49-67.

Andreu, V., Imeson, A.C. and Rubio, J.L., 2001. Temporal changes in soil aggregates and water erosion after a wildfire in a Mediterranean pine forest. Catena, 44: 69-84.

Askin, T. and Özdemir, N., 2003. Soil bulk density as related to soil particle size distribution and organic matter content. Agriculture Scientific and Professional Review, 9: 52-55.

Augris, N., Balasdent, J., Mariotti, A., Derenne, S., Largeau, C., 1998. Structure and origin of insoluble and non-hydrolyzable, aliphatic organic matter in a forest soil. Organic. Geochemistry, 28: 119-124.

Balasundram, S.K., Husni, M.H.A. & Ahmed, O.H., 2008. Application of geostatistical tools to quantify spatial variability of selected soil chemical properties from a cultivated Tropical Peat. Journal of Agronomy, 7: 82-87.

Balesdent, J., Girardin, C., Mariotti, A., 1993. Site related $\delta^{13}C$ of tree leaves and soil organic matter in a temperate forest. Ecology, 74: 1713-1721.

Balesdent, J. and Mariotti, A. 1996. Measurement of soil organic matter turnover using $\delta^{13}C$ natural abundance. In: T.W. Boutton and S. Yamasaki, Ed. Mass Spectrometry of Soils, Marcel-Dekker, New York, pp. 83–111.

Baber, C.V., 2002. Forests, fires and confrontation in Indonesia. In: Matthew, R., Halle, M. and Switzer, J. (Eds.), Conserving the peace: resource, livelihoods and security. The International Institute for Sustainable Development, Winnipeg, Manitoba, Canada, pp. 99-169.

Barnes, R.J., 1991. The variogram sill and the sample variance. Mathematical Geology, 23: 673-678.

Binkley, D., Valentine, D., Wells, C., Valentine, U., 1989. An empirical analysis of the factor contributing to 20-year decrease in soil pH in an old-field plantation of loblolly pine. Biogeochemistry, 8: 39-54.

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.

Bonmati, M., Ceccanti, B., Nanniperi, P., 1991. Spatial variability of Phosphatase, Urease, Protease, organic Carbon and total Nitrogen in soil. Soil Biology and Biochemistry, 23: 391-396.

BOSF., 2007. The Borneo orangutan survival foundation - BOS Deutschland e.V.: http://**www.bos-deutschland.de.**

Boutton, T.W., 1996. Stable carbon isotope ratio of soil organic matter and their uses as indicators of vegetation and climate changes. In: Boutton, T.W., Yamasaki, S., (Eds). Mass spectrometry of soil. Marcel Dekker, New York. pp. 47-82.

Boutton, T.W., Archer, S.R., Midwood, A.J., Zitzer, S.F., Bol, R., 1989. δ^{13}C values of soil organic carbon and their use in documenting vegetation change in a subtropical savanna ecosystem. Geoderma, 85: 5-41.

Boyer, W. D. and Miller, J.H., 1994. Effect of burning and brush treatments on nutrient and soil physical properties in young longleaf pine stands. Forest Ecology and Management, 70: 311-318.

Brearley, F.Q., Prajadinata, S., Kidd, P.S., Proctor, J., Suriantata., 2004. Structure and floristics of an old secondary rain forest in Central Kalimantan, Indonesia, and a comparison with adjacent primary forest. Forest Ecology and Management, 195: 385-397.

Brus, D.J. and de Gruijter, J.J., 1997. Random sampling or geostatistical modelling? Choosing between design-based and model-based sampling strategies for soil (with Discussion). Geoderma, 80: 1-59.

Buurman, P. (Ed), 1980. Red soils in Indonesia. Agricultural Research Reports 889, Wageningen.

Buurman, P. and Rochimah, L., 1980. Mathematical correlations between exchange characteristics of soil and clay samples. In: Buurman, P. (Eds). Red Soils in Indonesia. Agricultural Research Reports 889, Wageningen, pp. 141-152.

Buurman, P., Pape, Th., Reijneveld, J.A., de Jong, F., van Gelder, E., 2001. Laser-diffraction and pipette-method grain sizing of Dutch sediments: correlations for fine fractions of marine, fluvial, and loess samples. Netherlands Journal of Geosciences, 80: 49-57.

Buurman, P., Nierop, K.G.J., Pontevedra Pombal, X., Martínez Cortizas, A., 2006. Molecular chemistry by pyrolysis-GC/MS of selected samples of the Penido Vello peat deposit, Galicia, NW Spain. In: Martini, L.P., Martínez Cortizas, A., Chesworth, W. (Eds) Peatlands; Evolution and Records of Environmental and Climate Change. Elsevier, Amsterdam, pp. 217-240.

Buurman, P., Schellekens, J., Fritze, H., Nierop, K.G.J., 2007a. Selective depletion of organic matter in mottled podzol horizons. Soil Biology and Biochemistry, 39: 607–621.

Buurman, P., Peterse, F., Almendros Martín, G., 2007b. Soil organic matter chemistry in allophanic soils: a pyrolysis-GC/MS study of a Costa Rican Andosol Catena. European Journal of Soil Science, 58: 1330-1347.

Buurman, P., Amezquita, M.C. and Ramirez, H.F., 2008. Factor affecting soil C stock: a multivariate analysis approach. In: ‘t Mannetje, L., Amezquita, M.C., Buurman, P., Ibrahim, M.A. (Eds.), Carbon Sequestration in Tropical Grassland Ecosystems. Wageningen Academic Publishers, Wageningen, pp. 91-101.

Buurman, P. and Roscoe, R., 2011. Different chemical composition of free light, occluded light and extractable SOM fractions in soils of Cerrado, tilled and untilled fields, Minas gerais, Brazil – pyrolysis-GC/MS study. European Journal of Soil Science, 62: 253-266.

Cambardella, C.A., Moorman, T.B., Novak, J.M., Parkin, T.B., Karlen, D.L., Turco, R.F., Konopka, A.E., 1994. Field-scale variability of soil properties in Central Iowa soils. Soil Science Society of America Journal, 58: 1501-1511.

Cannone, N., Guglielmin, M., Gerdol, R., 2004. Relationships between vegetation pattern and periglacial landforms in Northwest Svalbard. Polar Biology, 27: 562-571.

CATIE. and Guelph., 2000. Evaluaciones de C en sistemas silvopastoriles. Publicacion interna, Proyecto Agroforesteria Tropical, CATIE, Diciembre 2000.

Chefetz, B., Tarchitzky, J., Deshmukh, A.P., Hatcher, P.G., Chen, Y., 2002. Structural characterization of soil organic matter and Humic acids in particle-size fractions of an agricultural soil. Soil Science Society of America Journal, 66: 129-141

Chen, Q., Shen, C., Sun, Y., Peng, S., Yi, W., Li., Z., Jiang, M., 2008. Soil carbon dynamics in a Subtropical Mountainous Region, South China: Results Based on Carbon Isotopic Tracing. In: Huang, H., Huang, P.M., Violante, A., (Eds). Soil Mineral-Microbe-Organic Interactions. Theories and Applications. pp. 233-258.

Chiavari, G. and Galletti, G.C., 1992. Pyrolysis-gas chromatography/mass spectrometry of amino acids. Journal of Analytical and Applied Pyrolisis, 24: 123-137.

Chilès, J.P. and Delfiner, P., 1999. Geostatistics. Modeling Spatial Uncertainty. Wiley Series in Probability and Statistics. John Wiley and Sons Inc. New York.

Cochran, W.G. 1977. Sampling techniques. 3rd edition. John Wiley and Sons, New York.

Conant, R.T. and Paustian, K., 2002. Spatial variability of soil organic Carbon in grasslands: implications for detecting change at different scales. Environmental Pollution, 116: 127-135.

Cortez, J., 1998. Field decomposition of leaf litters: Relationships between decomposition rates and soil moisture, soil temperature and earthworm activity. Soil Biology and Biochemistry, 30: 783-793.

Cressie, N.A.C. 1991. Statistics for saptial data. John Wiley and Sons, Inc. New York, etc.

Cruz, A.B. and del Castillo, R.F., 2005. Soil change during secondary succession in a tropical Montane cloud forest area. Soil Science Society of America Journal, 69: 906-914.

De Kok, R., 2008. The genus *Vitex (Labiatae)* in the flora Malesiana region, excluding New Guinea. KEW Bulletin, 63: 17-40.

De la Rosa, J.M., Gonzalez-Perez, J.A., González-Vázquez, R., Knicker, H., Lopez-Capel, E., Manning, D.A.C., González-Vila, F.J., 2008. Use of pyrolysis/GC–MS combined with thermal analysis to monitor C and N changes in soil organic matter from a Mediterranean fire affected forest. Catena, 74: 296–303.

Den Ouden, J., 2000. The role of bracken (*Pteridium aquilinum*) in forest dynamic. PhD Thesis, Wageningen University, Wageningen.

Dignac, M.F., Bahri, H., Rumpel, C., Rasse, D.P., Bardoux, G., Balesdent, J., Girardin, C., Chenu, C., Mariotti, A., 2005. Carbon-13 natural abundance as a tool to study the dynamics of lignin monomers in soil: an appraisal at the Closeaux experimental field (France). Geoderma, 128: 3-17.

Dijkstra, P., Ishizu, A., Doucett, R., Hart, S.C., Schwartz, E., Menyailo, O.V., Hungate, B.A., 2006. ^{13}C and ^{15}N natural abundance of the soil microbial biomass. Soil Biology and Biochemistry, 38: 3257-3266.

Dorado, J., Gonzalez-Vila, F.J., Zancada, M.C., Almendros, G., Lopez-Fando, C., 2003. Pyrolytic descriptors responsive to changes in humic acid characteristics after long term sustainable management of dryland farming systems in Central Spain. Journal of Analytical and Applied Pyrolisis. 68-69: 299-314.

Dormann, C. F., 2007. Effects of incorporating spatial autocorrelation into the analysis of species distribution data. Global Ecology and Biogeography, 16: 129-138.

Ehleringer, J.R., Buchmann, N., Flanagan, L.B., 2000. Carbon isotope ratios in belowground carbon cycle process. Ecological Applications, 10: 412-422.

Ekinci, H., 2006. Effect of forest fire on some physical, chimical and biological properties of soil in Canakkale, Turkey. International Journal of Agriculture and Biology, 8:102-106.

Ellert, B.H., Janzen, H.H. and McConkey, B.G., 2001. Measuring and comparing soil carbon storage. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), Assessment Method For Soil Carbon. CRC Press, Boca Raton, pp. 131-146.

Ellert, B.H., Janzen, H.H. and Entz, T., 2002. Assessment of a method to measure temporal change in soil carbon storage. Soil Science Society of America Journal, 66: 1687-1695.

El-Sheikh, M.A., 2005. Plant succession on abandoned fields after 25 years of shifting cultivation in Assuit, Egypt. Journal of Arid Environments, 61: 461-481.

Eussen, J.H.H. and Wirjahardja, S., 1973. Studies of an alang-alang (*Imperata cylindrica* (L.) Beauv.) vegetation. Biotrop Bulletin. No 6.

Falloon, P., Smith, P., Szabó, J., Pásztor, L., 2002. Comparison of approaches for estimating carbon sequestration at the regional scale. Soil Use and Management, 18: 164-174.

Falloon, P., Smith, P., Bradley, R., Milne, R., Tomlinson, R., Viner, D., Livermore, M. and Brown, T., 2006. RothCUK – a dynamic modelling system for estimating changes in soil C from mineral soils at 1 km resolution in the UK. Soil Use and Management, 22: 274-288.

FAO., 2001. Lecture notes on major soils of the world. In: Driessen, P. and Deckers, J. (Eds.), World soil resources reports-94. Rome, Italy.

Faravani, M. and Bakar, B.B., 2007. Effects of light on seed germination, growth pattern of straits rhododendron (*Melastoma Malabatricum L.*). Journal of Agricultural and Biological Science. Vol 2 No.3.

Farley, K.A., Pineiro, G., Palmer, S.M., Jobbagy, E.G., Jackson, R.B., 2008. Stream acidification and base cation losses with grassland afforestation. Water Resources Research. Vol. 44.

Fearnside, P. M., 2000. Global warning and tropical land-use change: Green House gas emissions from biomass burning, decomposition and soil in forest conversion, shifting cultivation and secondary vegetation. Climatic Change, 46:115-158.

Feldpausch, T.R., Rondon, M.A., Fernandes, E.C.M., Riha, S.J., Wandelli, E., 2003. Carbon and nutrient accumulation in secondary forests regenerating on pastures in Central Amazonia. The Ecological Society of America, Ecological Applications, 14: 164-176.

Fortin, M.J. and Dale, M., 2005. Spatial analysis, a guide for ecologists. Cambridge University Press, Cambridge.

Fujii, K., Uemura, M., Hayakawa, C., Funakawa, S., Sukartiningsih, KJosaki, T., Ohta, S., 2009. Fluxes of dissolved organic carbon in two tropical forest ecosystems of East Kalimantan, Indonesia. Geoderma, 152: 127-136.

FWI/GFW., 2002. The State of the Forest Indonesia. Forest Watch Indonesia (FWI) and Global Forest Watch (GFW). Editor: Emily Matthews, Bogor and Washington D.C.

Garrity, D., Soekardi, M., Noordwijk, M.V., De la Cruz, R., Pathak, P.S., Gunasena, H.P.M., Van So. N., Huijun, G., Majid, N.M., 1997. The *Imperata* grasslands of tropical Asia: area, distribution, and typology. Agroforestry Systems, 36: 3-29.

Gaten, C.T., Hanson, P.J., Todd, D.E., Lu, B.B., Brice, D.J., 2007. Natural 14N- and 12C-abudance as indicators of forest nitrogen status and soil carbon dynamics. In: Michener, R., Lajtha, K. (Eds.), Stable isotopes in Ecology and Environmental Science. Australia. Blackwellpublishing. pp. 61-82.

Goidts, E. and Van Wesemael, B., 2007. Regional assessment of soil organic Carbon changes under agriculture in Southern Belgium (1955–2005). Geoderma, 141: 341–354.

Goidts, E., Van Wesemael, B., Crucifix, M., 2009. Magnitude and sources of uncertainties in soil organic Carbon (SOC) stock assessments at various scales. European Journal of Soil Science, 60**:** 723–739.

Haining, R., 2003. Spatial data analysis – theory and practice. Cambridge University Press, Cambridge.

Harris, N.L., Petrova, S., Stolle, F., Brown, S., 2008. Identifying optimal areas for REDD intervention: East Kalimantan, Indonesia as a case study. Environ. Res. Lett. 3, 035006.

Hartemink, A.E., 2001. Biomass and nutrient accumulation of *Piper aduncum* and *Imperata cylindrica* fallows in the humid lowlands of Papua New Guinea. Forest Ecology and Management, 144: 19-32.

Hashimotio, T., Katsumi, K., Takeshi, T., Satohiko, S., 2000. Changes in Carbon storage on fallow forests in the tropical lowlands of Borneo. Forest Ecology and Management, 126: 331-337.

Hatcher, P.G., Dria, K.J., Kim, S., Frazier, S.W., 2001. Modern analytical studies of humic subtances. Soil Science, 16: 770-794.

Heuvelink, G.B.M. and Pebesma, E.J., 1999. Spatial aggregation and soil process modelling. Geoderma, 89: 47–65.

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

Houghton, R.A., 2005. Aboveground forest biomass and the global carbon balance. Global Change Biology, 11: 945-958.

IPCC., 2000. Land use, land-use change and forestry. In: Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D., Dokken, D. (Eds.), A Special Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Isaaks, E.H. and Srivastava, R.M., 1989. An introduction to applied geostatistics. Oxford University Press, New York and Oxford.

Janik, L.J., Spouncer, L., Correll, R.L., Skjemstad, J., 2002. Sensitivity analysis of the RothC Carbon model (Ver. 26.3 Excel Version). National Carbon Accounting System. Technical report No 30. Canberra: Australian Greenhouse Office.

Janžura, M., 1994. The central limit theorem for random fields. Kybernetika, 30: 77-86.

Jiang-bing, W., Du-ning, X., Xing-yi, Z., Xiao-yu, L., 2008. Topography and land use effects on the spatial variation of soil organic carbon: A case study in a typical small watershed of the balck soil region in Northeast China. Eurasian Soil Science, 41: 44-53.

Kaal, J., Brodowski, S., Baldock, J.A., Nierop, K.G.J., Martínez-Cortizas, A., 2008. Characterisation of aged black carbon using pyrolysis-GC/MS, thernally assisted hydrolysis and methylation (THM), direct and cross-polarisation 13C nuclear magnetic resonance (DP/CP NMR) and the benzenepolycarboxylic acid (BPCA) method. Organic Geochemistry, 39:1415-1426.

Kaal, J., Martínez Cortizas, A., Nierop, K.G.J., 2009. Characterisation of aged charcoal using a coil probe pyrolysis-GC/MS method optimised for black carbon. Journal of Analytical and Applied Pyrolisis, 85: 408-416.

Katsuno, K., Miyairi, Y., Tamura, K., Matsuzaki, H., Fukuda, K., 2010. A study of the Carbon dynamics of Japanese grassland and forest using 14C and 13C. Nuclear Instruments and Methods in Physics Research, 268: 1106-1109.

Kiem, R. and Kogel-Knabner, I., 2003. Contribution of lignin and polysaccharides to the refractory carbon pool in C-depleted arable soils. Soil Biology and Biochemistry, 35: 101-118.

Kiyono, Y. and Hastaniah., 1997. Slash-and-burn agriculture and succeeding vegetation in East Kalimantan. PUSREHUT Special Publication vol. 6, Mulawarman University, Samarinda, Indonesia.

Kiyono, Y. and Hastaniah., 2000. The role of slash-and-burn agriculture in transforming Dipterocarp forest into *Imperata* grassland. In: Guharja, E., Fatawi, M., Sutisna, M., Ohta, S. (Eds.), Rain Forest Ecosystem of East Kalimantan (El Nino, Drought, Fire and Human Impacts). Japan, Springer-Verlag. Ecological Studies 140, pp. 199-208.

Kogel, I., 1986. Estimation and decomposition pattern of the lignin component in forest humus layers. Soil Biology and Biochemistry, 18: 589-594.

Kögel-Knabner, I., 2000.The macromolecular organic composition of plant and microbial residues as inputs to soil organic matter. Soil Biology and Biochemistry, 34: 139-162.

Kooch, Y., Jalilvand, H., Bahmanyar, M.A., Pormajidian, M.R., 2007. Ecological distribution of indicator species and effective edaphical factor on the Northern Iran lowland forests. Journal of Applied Science, 7: 1475-1483.

Kravchenko, A.N., 2003. Influence of spatial structure on accuracy of Interpolation methods. Soil Science Society of America Journal, 67: 1564-1571.

Kravchenko, A.N., Robertson, G.P., Snap, S.S., Smucker, A.J.M., 2006. Using information about spatial variability to improve estimates of total soil Carbon. Agronomy Journal, 98:823-829.

Kravchenko, A.N. and Robertson, G.P., 2011. Whole-Profile soil Carbon stocks: The danger of assuming too much from analyses of too little. Soil Science Society of America Journal, 75:235–240.

Lal, R. and Kimble, J.M., 2000. Tropical ecosystems and the global C cycle. In: Lal, R., Kimble, J.M., Stewart, B. A. (Eds.), Global Climate Change and Tropical Ecosystems. CRC Press, Boca Raton, pp. 3-32.

Leinweber, P. and Schulten, H.R., 1999. Advance in analytical pyrolysis of soil organic matter. Journal of Analytical and Applied Pyrolysis, 49: 359-383.

Leps, J., 1987. Vegetation dynamics in early old field succession: a quantitative approach. Vegetatio, 72: 95-102.

Leps, J. and Smilauer, P., 2003. Multivariate analysis of ecological data using Canoco. New York: Cambridge University Press.

Liang, C., Das, K.C., McClendon, R.W., 2003. The influence of temperature and moisture contents regimes on the aerobic microbial activity of a biosolids composting blend. Bioresource Technology, 86: 131–137.

Li, Y., 2010. Can the spatial prediction of soil organic matter contents at various sampling scales be improved by using regression kriging with auxiliary information? Geoderma, 159: 63-75.

Lorenz, K., Lal, R., Preston, C.M., Nierop, K.G.J., 2007. Strengthening the soil organic Carbon pool by increasing contributions from recalcitrant aliphatic bio (macro) molecules. Geoderma, 142: 1-10.

Lugo, A.E. and Brown, S., 1992. Tropical forest as sinks of atmospheric Carbon. Forest Ecology and Management, 54: 239-255.

MacDonald, G.E., 2004. Cogongrass (*Imperata cylindrica*)-Biology, ecology, and management. Critical Reviews in Plant Sciences, 23: 367-380.

MacKinnon, K., Hatta, G., Halim, H., Mangalik, A., 1996. Ecology of Kalimantan. The ecology of Indonesia Seri Vol. III.

Magnusson, W.E., Sanaiotti, T.M., Lima, A.P., Martinelli, L.A., Victoria, R.L., C de Arau'jo, M.C., Albernaz, A.L., 2002. A comparison of δ^{13}C ratios of surface soils in savannas and forests in Amazonia. Journal of Biogeography, 29: 857 – 863.

Mannetje, L.'t, Amezquita, M.C., Buurman, P., Ibrahim, M.A. (Eds)., 2008. Carbon sequestration in tropical grassland ecosystems. Wageningen Academic Publishers, Wageningen.

Marin-Spiotta, E., Silver, W.L., Swanston, C. W., Ostertag, R., 2009. Soil organic matter dynamics during 80 years of reforestation of tropical pastures. Global Change Biology, 15: 1584–1597.

Martin, A., Mariotti, A., Balesdent, J., Lavelle, P., Vuattoux, R., 1990. Estimate of organic matter turnover rate in a savannah soil by $\delta^{13}C$ natural abundance measurement. Soil Biology and Biochemistry, 22: 517-523.

Marschner, B., Brodowski, S., Dreves, A., Gleixner, G., Gude, A., Grootes, P.M., Hamer, U., Heim, A., Jandl, G., Ji, R., Kaiser, K., Kalbitz, K., Kramer, C., Leinweber, P., Rethemeyer, J., Schäffer, A., Schmidt, M.W.I., Schwark, L., Wiesenberg, G.L.B., 2008. How relevant is recalcitrance for the stabilization of organic matter in soils? Plant Nutrion and Soil Science, 171: 91-110.

Matius, P., Toma, T., Sutisna, M., 2000. Tree species composition of a burned lowland diptrocarp forest in Bukit Soeharto, East Kalimantan. In: Guhardja E, Fatawi M, Sutisna M, Mori T, Ohta S (Eds.) Rain Forest Ecosystem of East Kalimantan (El Nino, Drought, Fire and Human Impacts). Japan, Springer-Verlag. Ecological Studies 140, pp 99-119.

Matthew, R., Halle, M., Switzer, J., 2002. Conserving the peace: resource, livelihoods and security. IISD, IUCN, CEESP, Foreign and Commonwealth Office, Wiinipeg.

McCune, B. and Grace, J.B., 2002. Analysis of ecological communities. Gleneden Beach, Oregon.

Meersmans, J., De Ridder, F., Canters, F., De Baets, S., Van Molle, M., 2008. A multiple regression approach to assess the spatial distribution of soil organic Carbon (SOC) at the regional scale (Flanders, Belgium). Geoderma, 143: 1-13.

Mendham, D.S., Mathers, N.J., O'Connell, A.M., Grove, T.S., Saffigna, P.G., 2002. Impact of land-use on soil organic matter quality in South-Western Australia—characterization with 13C CP/MAS NMR spectroscopy. Soil Biology and Biochemistry, 34: 1669–1673.

Mikutta, R., Kleber, M.S., Torn, M., Jahn, R., 2006. Stabilization of soil organic matter: association with minerals of chemical recalcitrance? Biogeochemistry, 77: 25-26.

MoF-FAO., 1996. Report of the national forest inventory for Indonesia. DG of Forest Inventory and Land use Planning of Forestry, Ministry of Forestry, Government of Indonesia-Food and Agriculture Organization of the United Nation, Jakarta.

MoF., 2008. Perhitungan deforestasi Indonesia 2008. Pusat Inventarisasi dan Perpetaan Hutan. Badan Planologi Kehutanan, Ministry of Forestry, Indonesia.

Mof., 2012. Indonesia's forest management policy. . Pusat Inventarisasi dan Perpetaan Hutan. Badan Planologi Kehutanan, Ministry of Forestry, Indonesia.

Mori, T., Ohta, S., Ishida, A., Toma, T., Oka, T., 2000. Overview of the changing forest ecosystems in East Kalimantan. In: Guharja, E., Fatawi, M., Sutisna, M., Ohta, S. (Eds.), Rain Forest Ecosystem of East Kalimantan (El Nino, Drought, Fire and Human Impacts). Japan, Springer-Verlag. Ecological Studies 140, pp. 309-317.

Munoz, C., Monreal, C.M., Schitzer, M., Zagal, E., 2008. Influence of *Acacia caven* (Mol) coverage on carbon distribution and its chemical composition in soil organic carbon fractions in Mediterranean-type climate region. Geoderma, 144: 352-360.

Murniati., 2002. From *Imperata cylindrica* grasslands to productive agroforestry. Tropenbos Kalimantan Series 9. Tropenbos International, Wageningen. The Netherlands.

Nadelhoffer, K.L. and Fry, B., 1988. Controls on natural Nitrogen-15 and Carbon-13 abundances in forest soil organic matter. Soil Science Society of America Journal, 52:1633-1640.

Nierop, K.G.J., 1998. Origin of aliphatic compounds in a forest soil. Organic Geochemistry, 29: 1009-1016.

Nierop, K.G.J., Buurman, P., De Leeuw, J.W., 1999. Effect of vegetation on chemical composition of H horizons in incipient podzols as characterized by 13C-NMR and pyrolysis-GC/MS. Geoderma, 90: 111–129.

Nierop, K.G.J., Pulleman, M.M., Marinissen, J.C.Y., 2001a. Management induced organic matter differentiation in grassland and arable soil: a study using pyrolysis techniques. Soil Biology and Biochemistry, 33: 755-764.

Nierop, K.G.J., Van Lagen, B., Buurman, P., 2001b. Composition of plant tissues and soil organic matter in the first stages of a vegetation succession. Geoderma, 100: 1–24.

Nykvist, N., 1996. Regrowth of secondary vegetation after the 'Borneo Fire' of 1982-1983. Journal of Tropical Ecology, 12: 307-312.

Odeh, I.O.A., McBratney, A.B., Chittleborough, D.J., 1990. Design op optimal sample spacings for mapping soil using fuzzy-k-means and regionalized variable theory. Geoderma, 47: 93-122.

Ogle, S.M., Breidt, F.J., Eve, M.D., Paustian, K., 2003. Uncertainty in estimating land use and management impacts on soil organic Carbon storage for U.S. agricultural lands between 1982 and 1997. Global Change Biology, 9: 1521-1542.

Ohtsuka, T., 1999. Early stages of secondary succession on abandoned cropland in North-East Borneo Island. Ecological Research, 14: 281-290.

Ohta, S., Morisada, K., Tanaka, N., Kiyono, Y., Effendi, S., 2000. Are soil in degraded dipterocarp forest ecosystem deteriored? A comparison *Imperata* grasslands, degraded secondary forests, and primary forests. In: Guharja, E., Fatawi, M., Sutisna, M., Ohta, S. (Eds.), Rain Forest Ecosystem of East Kalimantan (El Nino, Drought, Fire and Human Impacts). Japan, Springer-Verlag. Ecological Studies 140, pp. 49-57.

Okimori, Y. and Matius, P., 2000. Tropical secondary forest and its succession following traditional slash-and-burn agriculture in Mencimai, East Kalimantan. In: Guhardja E, Fatawi M, Sutisna M, Mori T, Ohta S (Eds.) Rain Forest Ecosystem of East Kalimantan (El Nino, Drought, Fire and Human Impacts). Japan, Springer-Verlag. Ecological Studies 140, pp 185-197.

Oliver, G.R., Beets, P.N., Garrett, L.G., Pearce, S.H., Kimberly, M.O., Ford-Robertson, J.B., Robertson, K.A., 2004. Variation in soil Carbon in pine plantations and implications for monitoring soil Carbon stocks in relation to land-use change and forest site management in New Zealand. Forest Ecology and Management, 203: 283-295.

Osaki, M., Watanabe, T., Ishizawa, T., Nilnond, C., Nuyim, T., Shinano, T., Urayama, M., Tuah, S.J., 2003. Nutritional characteristics of the leaves of native plants growing in adverse soils of humid tropical lowlands. Plant Foods for Human Nutrition, 58: 93-115.

Palmer, C.J., Smith, W.D. and Conkling, B.L., 2002. Development of a protocol for monitoring status and trends in forest soil Carbon at a national level. Environmental Pollution, 116: 209-219.

Panda, D. K., Singh, R., Kundu, D. K., Chakraborty, H., Kumar, A., 2008. Improved estimation of soil organic Carbon storage uncertainty using first-order Taylor series approximation. Soil Science Society of America Journal, 72: 1708–1710.

Parsi, Z., Hartog, N., Gorecki, T., Poerschmann, J., 2007. Analytical pyrolysis as a tool for the characterization of natural organic matter - A comparison of different approaches. Journal of Analytical and Applied Pyrolysis, 79: 9–15.

Perobelli Ferreira, F., Vidal-Torrado, P., Buurman, P., Macias, F., Otero, X.L., Boluda, R., 2009. Pyrolysis-gas chromatography/mass spectrometry of soil organic matter extracted from a Brazilian mangrove and Spanish salt marshes. Soil Science Society of America Journal, 73: 841-851.

Plante, A.F., Conant, R.T., Stewart, C.E., Paustian, K., Six, J., 2006. Impact of soil texture on distribution of soil organic matter in physical and chemical fractions. Soil Science Society of America Journal, 70: 287-296.

Post, J., Hattermann, F.F., Krysanova, V. and Suckow, F., 2008. Parameter and input data uncertainty estimation for the assessment of long-term soil organic Carbon dynamics. Environmental Modelling and Software, 23: 125-138.

Primack, R., Supriatna, J., Indrwan, M., Kramadibrata, P., 1998. Biologi konservasi. Yayasan Obor Indonesia. Jakarta, Indonesia.

Qiming, L., Shijie, W., Hechun, P., Ziyuan, O., 2003. The variation of soil organic matter in a forest-cultivation sequence traced by stable carbon isotopes. Chinese Journal of Geochemistry, 22: 83-88.

Ralph, J. and Hatfield, R.D., 1991. Pyrolysis-GC/MS characterization of forage materials. Journal of Agricultural and Food Chemistry, 39: 1426–1437.

Raven, A.M., Van Bergen, P.F., Stott, A.W., Dudd, S.N., Evershed, R.P., 1997. Formation of long-chain ketones in archaeological pottery vessels by pyrolysis of acyl lipids. Journal of Analytical and Applied Pyrolysis, 40–41: 267–285.

Revilla, J.A.V., 1993. Preliminary study on the rate forest covers loss in Indonesia. FAO Project UTF/INS/006/INS Field Document No. 35.

Risch, A.C., Jurgensen, M.F., Frank, D.A., 2007. Effects of grazing and soil micro-climate on decomposition rates in a spatio-temporally heterogeneous grassland. Plant and Soil, 298: 191–201.

Robertson, K.M., 2006. Distributions of tree species along point bars of 10 rivers in the South-Eastern US Coastal Plain. Journal of Biogeography, 33: 121-132.

Römkens, P.F.A.M., van der Plicht, J., Hassink, J., 1998. Soil organic matter dynamics after conversion of arable land to pasture. Fractionation results and stable isotope analysis. Biology and Fertility of Soils, 28: 277-284.

Roscoe, R., Buurman, P., Velthorst, E.J., and Pereira, J.A.A., 2000. Effects of fire on soil organic matter in a 'cerrado sensu-stricto' from Southeast Brazil as revealed by changes in δ^{13}C. Geoderma, 95: 141-160.

Rumpel, C., Chabbi, A., Nunan, N., Dignac, M.F., 2009. Impact of landuse change on the molecular composition of soil organic matter. Journal of Analytical and Applied Pyrolysis, 85: 431-434.

Saiz-Jimenez, C. and De Leeuw, J.W., 1986. Chemical characterization of soil organic matter factions by analytical pyrolisis-gas chromatography-mass Spectrometry. Journal of Analytical and Applied Pyrolysis, 9: 99-119.

Saiz-Jimenez, C., 1996. The chemical structure of Humic subtances: Recent advances. In: Humic subtances in terrestrial edited by A. Piccolo. Elsevier Science, pp. 1-44.

Saiz-Jimenez, C., Hermosin, B., Guggenberger, G., Zech, W., 1996. Land use effects on the composition of organic matter in soil particle size separates. III. Analytical pyrolysis. European Journal of Soil Science, 41: 61-69.

Sakalauskas, K.M., Costa, J.L., Laterra, P., Hidalgo, L., Aquirrezabal, L.A.N., 2001. Effects of burning on soil-water content and water use in *Paspalum quadrifarium* grassland. Agricultural Water Management, 50: 97-108.

Särndal, C.E. Swensson, B. Wretman, J., 1992. Model assisted survery sampling. Springer-Verlag, New York, 694 pp.

Schellekens, J., Buurman, P., Pontevedra Pombal, X., 2009. Selecting parameters for environmental interpretation of peat molecular chemistry – a pyrolysis-GC/MS study. Organic Geochemistry, 40: 678-691.

Schoning, I., Totsche, K.U., Kogel-Knabner, I., 2006. Small scale spatial variability of organic Carbon stocks in litter and solum of a forested Luvisol. Geoderma, 136: 631-642.

Schoenholtz, S.H., Van Miegroet, H., Burger, J.A., 2000. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. Forest Ecology and Management, 138: 335-356.

Schwager, S.J. and Mikhailova, E.A., 2002. Estimating variability in soil organic Carbon storage using the method of statistical differentials. Soil Science, 167: 194-200.

Scott, G.A.J., 1977. The role of fire in the creation and maintenance of savanna in the Montana of Peru. Journal of Biogeography, 4: 143-167.

Shukla, M.K., Slater, B.K., Cepuder, P., 2004. Spatial variability of soil properties and potential management classification of a chernozemic field in lower Austria. Soil Science, 169: 852-860.

Slik, J.W.F., Verburg, R.W., Kebler, P.J.A., 2002. Effects of fire and selective logging on the tree species composition of lowland dipterocarp forest in East Kalimantan, Indonesia. Biodoversity and Conservation, 11: 85-98.

Slik, J.W.F. and 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.W.F., 2006. Trees of Sungai Wain, http//www.nationaalherbarium.nl/sungaiwain/. Last accessed: 1.10.2007.

SSAC (Soil Survey Analytical Continuum)., 1996. Soil survey laboratory methods manual. Soil Survey Investigations Report 42. US Department of Agriculture, Washington.

Stein, C., 1945. A two-sample test for a linear hypothesis whose power is independent of the variance. Annals of Mathematical Statistics, 16: 243-258.

ter Braak, C.J.F. and Smilauer, P., 1998. Canoco reference manual and user's guide to Canoco for Windows: Sofware for canonical community ordination (Version 4). Centre for Biometry Wageningen (NL) and Microcomputer Power (Ithaca, NY, USA).

Toma, T., Matius, P., Hastaniah., Kiyono, Y., Watanabe, R., Okimori, Y., 2000. Dynamics of burned lowland dipterocarp forest stands in Bukit Soeharto, East Kalimantan. In: Guharja, E., Fatawi, M., Sutisna, M., Ohta, S. (Eds.), Rain Forest Ecosystem of East Kalimantan (El Nino, Drought, Fire and Human Impacts). Japan, Springer-Verlag. Ecological Studies 140, pp. 107-119.

Van Bergen, P.F., Bull, I.D., Poulton, P.R., Evershed, R.P., 1997. Organic geochemical studies of soils from the Rothamsted classical experiments: Total lipids, solvent insoluble residues and humic acids from Broadbalk Wilderness. Organic Geochemistry, 26: 117–135.

Van Bergen, P.F., Flannery, M.B., Poulton, P.R., Evershed. R.P., 1998. Organic geochemical studies of soils from Rothamsted experimental station: III. Nitrogen-containing macromolecular moieties in soil organic matter from Geescroft Wilderness. pp. 321–328. In B.A. Stankiewicz and P.F. Van Bergen (ed.) Nitrogen-containing macromolecules in the bio- and geosphere. ACS Symp. Ser. Oxford Univ. Press, New York.

Van Breemen, N. and Buurman, P., 2002. Soil formation. 2d Edition. Kluwer Academic Publishers, Dordrecht. 404 pp.

Vancampenhout, K., Wouters, K., De Vos, B., Buurman, P., Swennen, R., Deckers, J., 2009. Differences in chemical composition of soil organic matter in natural ecosystems from different climatic regions – A pyrolysis–GC/MS study. Soil Biology and Biochemistry, 41: 568-579.

VandenBygaart, A. J., Gregorich, E.G., Angers, D.A. and Stoklas, U.F., 2004. Uncertainty analysis of soil organic carbon stock change in Canadian cropland from 1991 to 2001. Global Change Biology, 10: 983-994.

VandenBygaart, A.J., Bremer, E., McConkey, B.G., Ellert, B.H., Janzen, H.H., Angers, D.A., Carter, M. R., Drury, C. F., Lafond, G. P., McKenzie. R.H., 2011. Impact of sampling depth on differences in soil Carbon stocks in Long-Term Agroecosystem experiments. Soil Science Society of America Journal, 75: 226-234.

Van der Kamp, J., Yassir, I., Buurman, P., 2009. Soil Carbon changes upon secondary succession in *Imperata* grasslands (East Kalimantan, Indonesia). Geoderma, 149: 76-83.

Van der Maarel, E., 1979. Transformation of cover-abundance values in phytosociology and its effects on community similarity. Vegetatio, 39: 97-114.

Van Noordwijk, M., Cerri, C., Woomer, P.L., Nugroho, K., Bernoux, M., 1997. Soil Carbon dynamic in the humic tropical forest zone. Geoderma, 79:187-225.

Van Putten, B., Knippers, T., Buurman, P., 2010. Concepts of classical statistical design and analysis in soil treatment experiments. Soil Science, 175: 519-529.

Veldkamp, E., 1994. Organic Carbon turnover in three tropical soils under pasture after deforestation. Soil Science Society of America Journal, 58: 175-180.

Venteris, E.R., McCarty, G.W., Ritchie, J.C., Gish, T., 2004. Influence of management history and landscape variables on soil organic Carbon and soil redistribution. Soil Science, 169: 787-795.

Vitorello, V.A., Cerri, C.C., Andreux, F., Feller, C., Victoria, R.L., 1989. Organic matter and natural carbon-13 distribution in forest and cultivated Oxisol. Soil Science Society of America Journal, 53: 773-778.

Watanabe, T. and Osaki, M., 2001. Influence of Aluminum and Phosphorus on growth and xylem sap composition in *Melastoma malabathricum* L. Plant and Soil, 237: 63-70.

Wattel-Koekkoek, E.J.W., 2002. Clay-associated organic matter in kaolonitic and smectitic soils. PhD Thesis, Wageningen University, Wageningen.

Webster, R. and Oliver, M.A., 1992. Sampling adequacy to estimate variograms of soil properties. Journal of Soil Science, 43: 177-192.

Whitehouse, A. E. and Mulyana, A.A.S., 2004. Coal fires in Indonesia. International Journal of Coal Geology, 59: 91-97.

Wilcke, W. and Lilienfein, J., 2004. Soil Carbon-13 natural abundance under native and managed vegetation in Brazil. . Soil Science Society of America Journal, 68: 827-832.

Woomer, P. L., Palm, C.A, Alegre, J., Castilla, C., Cordeiro, D.G., Hairiah, K., Kotto-Same, J., Moukam, A., Riese, A., Rodrigues, V., van Noordwijk, M., 2000. Slash-and-burn effects on carbon stocks in the humid tropics. In: Lal, R., Kimble, J.M., Stewart, B. A. (Eds.), Global Climate Change and Tropical Ecosystems. CRC Press, Boca Raton, pp. 99-115.

World Bank, 2000. The World Bank forest strategy: Striking the right balance. World Bank Operations Evaluation Department. Washington DC.

World Bank, 2003. East Asia and Pacific forest law enforcement and governance (FLEG). World Bank, Jakarta, Indonesia.

Yamashita, N., Ohta, S., Hardjono, A., 2008. Soil changes induced by *Acacia mangium* plantation establishment: Comparison with secondary forest and *Imperata cylindrica* grassland soils in South Sumatra, Indonesia. Forest Ecology and Management, 254: 362-370.

Yasreby, J., Saffari, M., Fathi, H., Karimian, N., Emadi, M., Baghernejad, M., 2008. Spatial variability of soil fertility properties for precision agriculture in Southern Iran. Journal of Applied Sciences, 8: 1642-1650.

Yassir, I. and Omon, M., 2006. Hubungan potensi cendawan mikoriza arbuskula (CMA) dengan sifat-sifat tanah pada lahan kritis di Samboja, Kalimantan Timur (The relationship between potency arbuscular mycorrhizal fungi (AMF) and soil properties in marginal land Samboja, East Kalimantan). Jurnal Penelitian Hutan Tanaman. Pusat Penelitian dan Pengembangan Hutan Tanaman, 3:107-115.

Yassir, I., Van der Kamp, J., Buurman, P., 2010. Secondary succession after fire in *Imperata* grasslands of East Kalimantan, Indonesia. Agriculture, Ecosystems and Environment, 137: 172-182.

Yonekura, Y., Ohta, S., Kiyono, Y., Aksa, D., Morisada, K., Tanaka, N., Kanzaki, M., 2010. Changes in soil carbon stock after deforestation and subsequent establishment of “*Imperata*” grassland in the Asian humid tropic. Plant and Soil, 329: 495–507.

Zang, C. and McGrath, D., 2004. Geostatistical and GIS analyses on soil organic Carbon concentrations in grassland of southeastern Ireland from two different periods. Geoderma, 119: 261-275.

Summary

Imperata grassland is a common vegetation type in Kalimantan (Indonesia), and other parts of South-East Asia. It indicates a high degree of degradation of the vegetation, and mostly occurs after slashing and burning of primary forest. Through secondary succession *Imperata* grassland is converted into new secondary forest and much of the original biodiversity is restored.

The overall objective of the thesis was to study the regeneration of *Imperata* grasslands in East Kalimantan, and to measure the effects of regeneration on soil properties, with emphasis on the organic fraction. The research strategy was to compare plots of different regeneration stages, characterized by the last year that the vegetation was burned.

The crucial research questions for the thesis were:

a) How does the vegetation community develop after fire?
b) How is the quantitative development of carbon contents and stocks under vegetation succession?
c) How fast is decay and build-up of SOM fractions contributed by C3 and C4 vegetation? Does carbon dynamics throw a light on differences in carbon stocks?
d) Is there a relation between carbon stocks, carbon dynamics, and soil organic matter chemistry?
e) Using geo-referenced information, how accurate are carbon stock estimates for larger areas under *Imperata* grassland? Which estimation procedure is optimal?

In chapter 2, an early succession in *Imperata* grasslands was studied, both to examine how the species community develops after fire and to explore the relation between community structure and environmental gradients. We analysed 291 plots representing five different stages of succession: plots that last burned 3, 4, and 9 years before sampling, respectively, a secondary forest (approximately 15 years old) and a primary forest. The percentage of shrubs and young trees increased from *Imperata* grassland to secondary forest, whereas *Imperata* decreased. In our case study, soil texture is an important factor in the vegetation succession. On sandy soils, there was a strong increase with time of bracken fern (*Pteridium aquilinum* L.), while the number of other species was lower. This slows down the development towards secondary forest. Canonical Correspondence Analysis (CCA) of the environmental gradient and vegetation showed that pH, bulk density, sand and clay were the factors related to the distribution of species. The rapid secondary succession indicates that *Imperata* grasslands are not a final and stable stage of land degradation, but, when not maintained by frequent fires and human disturbances, regenerate spontaneously to secondary forest.

In chapter 3, using the same regeneration stages but including a larger number of plots than in Chapter 2, changes in soil carbon stocks were studied with the dual purpose of testing differences between stocks and to find out whether there is a trend of carbon stocks and other soil properties with the succession phases. We analysed 366 plots representing the five different stages of succession mentioned before. Soil samples were taken of the A-, AB- and B-horizon, except under the primary forest where AB- horizons were lacking. Results showed that during regeneration not only vegetation composition but also soil properties change. The effects of regeneration on soil were strongest in the A-horizon, where an increase in carbon stock, N stock, and C/N ratio, and a decrease in bulk density and pH were observed. Soil carbon stocks increased upon natural regeneration from grassland to secondary forest, and highest carbon stocks were found in the later regeneration phases and lowest under primary forest. Contrary to the situation in other areas, the lowest soil carbon stocks were found under primary forest. The cause of this discrepancy was not completely clear, but lower carbon stocks under primary forests in East Kalimantan may be due to extremely low fertility, combined with shallow soils and low root mass in the topsoil. Root density – as observed in the field – is much higher under the grass vegetation.

In Chapter 4, in order to better understand the effect of vegetation succession in *Imperata* grassland on soil carbon dynamics and stocks, stable C isotopes were used. We analysed twenty plots representing the 3-years, 9-years, secondary forest and primary forest to determine the proportion of C in soil derived from C3- and C4-vegetation (the latter representing the *Imperata* vegetation). Soil samples were taken of the A-, AB- and B-horizons, except under the primary forest where AB- horizons were lacking. Isotopic signatures of all soil horizons of the regeneration stages differ significantly from those of the primary forest. The A horizon under the 3-years Imperata plot still contains 23% forest (C3) carbon, and this fraction increases to 51% in the B-horizon. In the 9-years plot and in the secondary forest, the C3 carbon on the A horizon increased to 51% and 96%, respectively. The absolute increases in C3 carbon were significant in the 9-years and secondary forest plots. The absolute amounts of C4 carbon in the A-horizon decreases rapidly from the 3-years to the 9-years plot, but the decreases in the AB-horizon were not significant. The decay of C4-C in the topsoil amounted to approximately 4.9 g per kg soil in approximately 6 years, or a decay rate of C4-C of 0.8 g kg^{-1} per year. In the primary forest, the in A-horizon contained 14.3 g kg^{-1} C3-C, of which only 3.6 g kg^{-1} remained in the 3-years plot. This is a loss of 10.7 g kg^{-1} in approximately 25 years, or a decay rate of 0.4 g kg^{-1} per year. Results indicate that the strong decay C4-C in topsoil might be due to rather high carbohydrate contents in both of the litter and soil organic matter.

In Chapter 5, the chemical composition of the SOM in the succession was studied by pyrolysis-gas chromatography/mass spectrometry (PG-MS). The aim of this research was to find out whether (1) the chemical composition would change in the various succession stages, and (2) whether changes in carbon stocks were logically connected to changes in SOM chemical composition. We analysed 46 litter and soil samples from the 3-years, 9-years, secondary forest and primary forest plots and also included soils under *Acacia mangium* (9 years). Chemical composition of litters was clearly different for the various vegetations. There was a large difference between litter composition and (extractable) SOM composition, which was largely due to a rapid decay of lignin in the A-horizon. Both litter and SOM under *Imperata* contained larger amounts of carbohydrates and fewer lignin moieties, aliphatics and N-compounds than those under secondary and primary forest. Nevertheless, SOM was less decayed under *Imperata* grassland. SOM decomposition was most advanced under forest, as indicated by a higher contribution of microbial matter. The results indicate that decomposition efficiency is related to SOM chemical composition, but more to abundance of N-compounds than to that of potentially recalcitrant compounds.

In Chapter 6, using geo-referenced information, we tested the accuracy of carbon stock estimates for a larger area under *Imperata* grassland, and determined which estimation procedures are optimal. The dataset contains 18 transects with a length varying from 200 to 1000 meters. The number of observations per transect varied from 6 to 24 and distance between observations varied from 2 to 150 meters. We used 250 plots that had 3, 4 and 9 years of regeneration after fire. We combined these three regeneration phases because an earlier study did not indicate that their soil carbon stocks differed significantly. The various purposes of this research were (i) to study the spatial variability of soil carbon stock under *Imperata* grassland from geo-referenced information at various scales; (ii) to estimate the total carbon stock and to quantify the quality of the estimator used, and (iii) to formulate recommendations for efficient sampling in similar areas. Results showed that spatial variability was modelled following an isotropic stationary model with Spherical and Exponential variogram functions. Range and sill were estimated at 100 m and 82.29 ton^2 ha^{-2}, respectively. The nugget: sill ratio was about 24%, implying a rather strong spatial dependence. We applied three types of estimators, namely a 'naive average procedure' (NAP), a 'spatial average procedure' (SAP) and a 'spatial optimal procedure' (SOP). Results showed that estimates of total carbon stock (in ton ha^{-1}) following NAP (which erroneously ignores the spatial dependence), resulted in a considerably too narrow 95%-confidence interval (37.52, 39.75), whereas the outcomes using SAP and SOP were (36.54, 40.73) and (37.14, 40.78), respectively for the Spherical model, and (36.63, 40.64) and (37.07, 40.64), respectively for the Exponential model. The results also showed that when total carbon stock estimation is the main goal, random sampling is optimal, whereas 'wide design' sampling can be preferred for good reasons.

In conclusion, results showed that during regeneration of *Imperata* grasslands, both vegetation composition and soil properties, including chemistry of SOM change. Soil carbon stocks are larger under *Imperata* grasslands than under primary forest, and increase further upon natural regeneration of grassland to secondary forest. Highest carbon stocks are found in the later regeneration phases. The fact that soil carbon stocks under primary forest are lower than under *Imperata* and its regeneration phases may have important consequences for carbon sequestration projects in this area, because (soil) carbon may be lost in the later stages of forest regeneration. This research supports the statement by Mori et al. (2000) that the hypothetical rapid decrease in soil carbon as a result of forest loss and its slow recovery under secondary vegetation are misconceptions.

Samenvatting

Imperata graslanden behoren tot een algemeen voorkomend vegetatietype in Kalimantan (Indonesia) en andere delen van Zuid Oost Azie. Deze vegetatie is typisch voor gedegradeerde terreinen en is vaak het resultaat van zwerflandbouw en bosbranden. Middels natuurlijke regeneratie ontwikkelen zij zich weer tot (secundair) bos waarbij veel van de biodiversiteit terugkeert.

Het doel van deze thesis was om het herstel van *Imperata* graslanden in Oost-Kalimantan te bestuderen en om de effecten van deze regeneratie op bodemkwaliteiten te meten met speciale nadruk op de organische fractie. De onderzoeksstrategie was erop gericht om locaties met verschillende ontwikkelingsstadia te vergelijken, uitgaand van het jaar waarin de vegetatie het laatst verbrandde.

De belangrijkste onderzoeksvragen voor deze thesis waren:

a) Hoe ontwikkelt zich de soortencompositie na brand?

b) Hoe ontwikkelt zich de koolstof hoeveelheid en opslag tijdens de bosregeneratie?

c) Hoe snel is afbraak en opbouw van de SOM fracties bijgedragen door C3 en C4 planten? Leidt de koolstofdynamiek tot meer inzicht in de verschillen in koolstofopslag?

d) Is er een relatie tussen koolstofvoorraden, koolstofdynamiek, en de chemie van de organische stof in de bodem?

e) Hoe nauwkeurig is de schatting van de koolstofvoorraad in de bodem onder *Imperata* grasland voor grotere oppervlakken bij gebruik van geostatistische informatie? Welke schattingsprocedure is optimaal?

In hoofdstuk 2 werd de vroege regeneratie in *Imperat*a graslanden bestudeerd, zowel om te kijken hoe de soortensamenstelling zich na branden ontwikkelde alswel om de relatie tussen de soortenopbouw en milieugradiënten te onderzoeken. Wij onderzochten 291 locaties die vijf verschillende ontwikkelingsstadia van regeneratie vertegenwoordigden: locaties die 3, 4 en 9 jaar geleden verbrandden, een secundair bos van ongeveer 15 jaar oud en een primair regenbos. Het percentage struiken en jonge bomen neemt toe van *Imperata* grasland naar secundair bos terwijl het gras afneemt. In deze studie bleek de bodemtextuur een belangrijke factor te zijn in de regeneratie van de vegetatie. Op zandige bodems werd een sterke correlatie gevonden met adelaarsvaren (*Pteridium aquilium* L.), terwijl het aantal andere soorten lager was. Dit ging samen met een langzamere ontwikkeling tot secundair bos. Canonical correspondence analysis (CCA) van de milieugradiënten en vegetatie liet zien dat pH, bodemdichtheid, en bodemtextuur (met name zand en klei) de belangrijkste factoren waren met betrekking tot de aanwezige soorten. De snelle secundaire successie laat zien dat Imperata graslanden niet een stabiel eindstadium van landdegradatie zijn, maar dat, indien niet in stand gehouden door frequente branden en menselijke verstoringen, deze spontaan tot secundair bos regenereren.

Met gebruik van dezelfde regeneratiefasen, maar met een groter aantal locaties, werden in hoofdstuk 3 de veranderingen in bodem organische stof gehalte en voorraad bestudeerd. Het doel was vast te stellen of regeneratie leidt tot verschillen in koolstofvoorraad en tegelijk te onderzoeken of er een verband bestaat tussen hoeveelheid koolstof en andere bodemeigenschappen enerzijds met de regeneratie fases anderzijds. We analyseerden 366 locaties die de vijf hierboven genoemde regeneratiefasen vertegenwoordigen. Er werden bodemmonsters genomen van de

A, AB en B bodemhorizonten, behalve onder het primaire bos waar een AB horizont afwezig was. De resultaten tonen dat gedurende de regeneratie zowel de vegetatie als de bodemkarakteristieken veranderden. De effecten van regeneratie op de bodem waren het sterkst in de A horizont, waar een toename van koolstof en de C/N verhouding en een vermindering van de bodemdichtheid en zuurgraad werden waargenomen. Met de regeneratie van grasland naar secundair bos nam de hoeveelheid koolstof in de bodem toe waarbij de hoogste voorraden werden gevonden in oud secundair bos en de laagste onder primair bos. Dit is in tegenstelling tot de situatie in andere regio's. De oorzaak van dit verschil was niet geheel duidelijk, maar de lagere bodemkoolstof onder primair bos is mogelijk gerelateerd aan de extreem lage bodemvruchtbaarheid, gecombineerd met ondiepe bodems en lage wortelmassa in de bovengrond. Worteldichtheden – zoals waargenomen in het veld – zijn veel hoger onder grasvegetaties.

In hoofdstuk 4 werden stabiele C isotopen gebruikt om een beter inzicht te verkrijgen in de dynamiek van bodemkoolstof onder *Imperata* graslanden. We analyseerden monsters van 20 locaties die regeneratie stadia van 3 en 9 jaar oud, secundair bos en primair bos vertegenwoordigden. Van deze monsters bepaalden we de proportie C in de bodem afkomstig van C3- en C4-vegetatie, waarbij de laatste groep het *Imperata* gras vertegenwoordigt, en de eerste de overige planten. Monsters werden verzameld van de A, AB en B horizonten behalve onder primair bos waar de AB horizont afwezig was. De isotoop signatuur van alle bodemhorizonten onder de regeneratiefasen waren significant verschillend van die onder primair bos. De A horizont van de drie-jaar *Imperata* locatie had 23% C3 koolstof tegenover 51% in de B horizont. De 9-jaar locatie en het secundaire bos hadden in de A horizont respectievelijk 51 en 96% C3 koolstof. De absolute toename in C3 koolstof was significant in de 9-jaar en secundair bos locaties. De absolute waarden van C4 koolstof in de A horizont nam af van de 3-jaar naar de 9-jaar locaties, maar dit verschil was niet significant in de AB horizont. De afbraak van C4 koolstof in de bovengrond bedroeg ongeveer 4.9 g per kg grond over ongeveer 6 jaar. Dit vertegenwoordigt een afbraak van 0.8 g koolstof per jaar per kg grond. In het primaire bos bevatte de organische bodemlaag 14.3 g C3 koolstof per kg grond tegenover slechts 3.6 g per kg in de 3-jaar oude locatie. Dit betekent een afbraak van 10.7 g C per kg in ongeveer 25 jaar, oftewel een afbraak van 0.4 g koolstof per kg grond per jaar. De resultaten tonen een sterke afbraak van C4 koolstof in de bovengrond, wat mogelijk gerelateerd is aan het de hoge carbohydraat gehalte in zowel het strooisel als de bodem organische stof van deze vegetatie.

In hoofdstuk 5 werd de chemische samenstelling van de (extraheerbare) organische fractie in de bodem bestudeerd middels pyrolysis gaschromatografie/massa spectrometrie. De bedoeling van dit onderzoek was om te bekijken of (1) de chemische samenstelling varieerde per regeneratiestadium en (2) of veranderingen in koolstofvoorraad gerelateerd waren aan de veranderingen in chemische samenstelling van de organische stof. We analyseerden 46 strooiselmonsters en bodemmonsters van de regeneratiestadia 3-jaar, 9-jaar, secundair bos en primair bos en namen ook monsters van een 9 jaar oude *Acacia mangium* plantage mee. De chemische compositie van de verschillende strooisels was zeer verschillend per type locatie. Er was een groot verschil tussen strooiselsamenstelling en (extraheerbare) organische stof samenstelling, wat voornamelijk terug te voeren was tot de snelle afbraak van lignine in de A horizont.

Zowel strooisel en bodem organische stof in *Imperata* locaties bevatte grotere hoeveelheden carbohydraten en minder lignine fragmenten, alifaten en N-verbindingen dan onder secundair en primair bos. Desalniettemin was de organische stof in de *Imperata* locaties minder afgebroken. De afbraak was het verst voortgeschreden onder primair bos, waar minder plantaardige bestanddelen en meer microbieel materiaal aanwezig was. De resultaten suggereren dat de afbraak-efficiëntie samenhangt met de chemische samenstelling van de bodem organische stof, maar meer met het gehalte aan N-verbindingen dan dat van potentieel moeilijk afbreekbare verbindingen.

In hoofdstuk 6, met gebruikmaking van coördinaat gerelateerde data, hebben we getest hoe nauwkeurig de schatting van (bodem)koolstofvoorraden onder *Imperata* graslanden kan zijn, en bepaald welke procedure het beste werkt. De dataset bevat 18 transecten met lengtes die variëren van 200 tot 1000 meter. Het aantal waarnemingen per transect varieerde van 6 tot 24, en de afstand tussen de waarnemingspunten van 2 tot 150 meter. Er werden 250 locaties van 3, 4 en 9 jaar regeneratie na brand geanalyseerd. Omdat eerder onderzoek had uitgewezen dat de koolstofvoorraden onder deze drie regeneratiestadia niet significant verschillen, werden deze tezamen genomen. Het doel was om de ruimtelijke variatie van bodem organische stof onder *Imperata* graslanden te bestuderen gebaseerd op de verschillende geografische locaties met verschillende afstanden ertussen. Verder wilden we nagaan welke methode om de totale koolstofvoorraad te bepalen het best was en aanbevelingen geven voor optimale bemonsteringsmethoden in andere vergelijkbare gebieden. De resultaten laten zien dat de ruimtelijke variatie goed beschreven werd door een stationair model met Sferische en Exponentiële variogram functies. *Range* en *sill* werden geraamd op 100 m en 82.29 ton^2 ha^{-2}, respectievelijk. De *nugget/sill* verhouding was ongeveer 24%, hetgeen een behoorlijk sterke ruimtelijke afhankelijkheid aanduidt. We pasten drie soorten van schatters toe, namelijk, een 'naïeve gemiddelde procedure' (NAP), een 'ruimtelijke gemiddelde procedure' (SAP) en een 'ruimtelijke optimale procedure’ (SOP). Resultaten toonden aan dat schattingen van de totale voorraad koolstof (in ton ha^{-1}) volgende NAP (die ten onrechte de ruimtelijke afhankelijkheid negeert), resulteerde in een aanzienlijk te klein 95%-betrouwbaarheidsinterval (37.52, 39.75), terwijl de uitkomsten met behulp van SAP en SOP waren (36.54, 40.73) en (37.14, 40.78), respectievelijk voor het sferische model, en (36.63, 40.64) en (37.07, 40.64), respectievelijk voor het exponentiële model. De resultaten toonden ook aan dat wanneer de schatting van de totale koolstofvoorraad het belangrijkste doel is, een aselecte steekproef optimaal is, terwijl de 'brede design' monstername om goede redenen de voorkeur kan hebben.

Samenvattend laten de resultaten van het onderzoek zien dat gedurende de regeneratie van *Imperata* graslanden zowel de samenstelling van de vegetatie en bodemeigenschappen, waaronder bodemchemie, veranderen. Koolstofvoorraden in de bodem onder *Imperata* grasland zijn groter dan die onder primair bos en nemen toe met het aantal jaren van regeneratie. Het feit dat koolstofvoorraden lager zijn onder oerwoud dan in de regeneratie vegetatietypes kan belangrijke gevolgen hebben voor koolstofsequestratie projecten, omdat blijkbaar (bodem)koolstof verloren gaat in ouder bos. Dit onderzoek ondersteunt daarmee de stelling van Mori et al. (2000) dat het veronderstelde verlies van bodemkoolstof door bosverlies en langzame wederopbouw in regenererende vegetatie op een vergissing berust.

Annex 1.

Species of BOS Samboja Lestari (SL) and Sungai Wain (SW)

No	Species	Family	Location
1	*Acacia mangium**	Leguminosae-Mimosaceae	SL
2	*Acalypha sp.*	Euphorbiaceae	SL
3	*Actinodaphne diadenum*	Lauraceae	SL
4	*Agelaea borneensis*	Connaraceae	SL
5	*Alocasia* sp.	Araceae	SW
6	*Alphitonia excelsa*	Rhamnaceae	SL
7	*Alpinia galanga*	Zingiberaceae	SW
8	*Alpinia* sp.*	Zingiberaceae	SW/SL
9	*Alseodaphne elmeri*	Lauraceae	SW
10	*Amyema* sp.	Loranthaceae	SL
11	*Aporosa frutescens*	Euphorbiaceae	SL
12	*Aporosa nitida*	Euphorbiaceae	SW
13	*Aporosa* sp.	Euphorbiaceae	SL
14	*Aporosa subcaudata*	Euphorbiaceae	SW
15	*Archidendron clypearia*	Leguminosae/ Mimosaceae	SL
16	*Archidendron jiringa*	Leguminosae/ Mimosaceae	SL
17	*Archidendron* sp.	Leguminosae/ Mimosaceae	SL
18	*Ardisia korthalsiana*	Myrsinaceae	SW
19	*Ardisia* sp.	Myrsinaceae	SL
20	*Artabotrys* sp.	Annonaceae	SL
21	*Artocarpus integer*	Moraceae	SL
22	*Artocarpus lanceifolius*	Moraceae	SL
23	*Artocarpus nitidus*	Moraceae	SW
24	*Artocarpus rigidus*	Moraceae	SL
25	*Artocarpus* sp.	Moraceae	SL
26	*Baccaurea* sp.	Euphorbiaceae	SW
27	*Bambusa* sp.	Gramineae	SL
28	*Barringtonia macrostachya*	Lecythidaceae	SL
29	*Barringtonia* sp.	Lecythidaceae	SL

No	Species	Family	Location
30	*Bauhinia excelsa*	Leguminosae/ Mimosaceae	SW/SL
31	*Bauhinia purpurea*	Leguminosae/ Mimosaceae	SL
32	*Begonia* sp.	Begoniaceae	SW
33	*Blechnum* sp.*	Blechnaceae	SL
34	*Borrassodendron borneensis*	Palmae	SW
35	*Bridelia glauca**	Euphorbiaceae	SL
36	*Calamus* sp.	Palmae	SW/SL
37	*Callicarpa longifolia**	Verbenaceae	SL
38	*Calophyllum nodosum*	Guttiferae	SW
39	*Calophyllum* sp.	Guttiferae	SW
40	*Carex* sp.	Cyperaceae	SL
41	*Cayratia* sp.	Vitaceae	SW
42	*Clerodendrum adenophysum*	Verbenaceae	SL
43	*Clerodendrum* sp.	Verbenaceae	SL
44	*Clidemia hirta**	Melastomataceae	SL
45	*Cnestis platantha*	Connaraceae	SW/SL
46	*Combretum* sp.	Combretaceae	SW
47	*Connarus semidecandrus*	Connaraceae	SW/SL
48	*Costus* sp.	Zingiberaceae	SL
49	*Cotylelobium melanoxylum*	Dipterocarpaceae	SW
50	*Cratoxylum formosum**	Hypericaceae	SL
51	*Cratoxylum arborescens*	Hypericaceae	SL
52	*Curculigo* sp.	Amaryllidaceae	SL
53	*Cynanchum* sp.	Asclepiadaceae	SL
54	*Cyperus* sp.	Cyperaceae	SL
55	*Cyrtandra* sp.	Gesneriaceae	SW
56	*Dacryodes rostrata*	Burseraceae	SW/SL
57	*Dacryodes rugosa*	Burseraceae	SW
58	*Daemonorops* sp.	Palmae	SW
59	*Dalbergia* sp.	Leguminosae-Pap	SL
60	*Dehaasia* sp.	Lauraceae	SL
61	*Desmos cochinchinensis*	Annonaceae	SL

No	Species	Family	Location
62	*Desmos* sp.	Annonaceae	SW
63	*Dicranopteris linearis**	Gleicheniaceae	SL
64	*Dillenia reticulate*	Dilleniaceae	SL
65	*Dillenia* sp.	Dilleniaceae	SL
66	*Dioscorea* sp.	Dioscoreaceae	SL
67	*Diospyros borneensis*	Ebenaceae	SW
68	*Diospyros buxifolia*	Ebenaceae	SW
69	*Diospyros confertiflora*	Ebenaceae	SL
70	*Diospyros pendula*	Ebenaceae	SL
71	*Diospyros elliptifolia*	Ebenaceae	SW
72	*Diospyros* sp.	Ebenaceae	SL/SW
73	*Dipterocarpus cornutus*	Dipterocarpaceae	SW
74	*Dischidia* sp.	Asclepiadaceae	SL
75	*Dissochaeta* sp.	Melastomataceae	SL
76	*Dracaena* sp.	Liliaceae	SL
77	*Dripetis kikir*	Euphorbiaceae	SW
78	*Durio acutifolius*	Bombacaceae	SW
79	*Elaeocarpus beccarii*	Elaeocarpaceae	SL
80	*Elaeocarpus stipularis*	Elaeocarpaceae	SW
81	*Elaeocarpus* sp.	Elaeocarpaceae	SL
82	*Embelia javanica*	Myrsinaceae	SL
83	*Embelia* sp.	Myrsinaceae	SL
84	*Emilia sonchifolia*	Asteraceae	SL
85	*Emilia* sp.	Asteraceae	SL
86	*Enkleia malaccensis*	Thymelaeaceae	SW
87	*Erycibe* sp.	Convolvulaceae	SW/SL
88	*Etlingera* sp.	Zingiberaceae	SL
89	*Eugenia heteroclada*	Myrtaceae	SL
90	*Eugenia* sp.*	Myrtaceae	SL
91	*Eupatorium inulaefolium**	Asteraceae	SL
92	*Euphorbia* sp.	Euphorbiaceae	SL
93	*Eurya* sp.	Theaceae	SW

No	Species	Family	Location
94	*Eurycoma longifolia*	Simaroubaceae	SW/SL
95	*Eusideroxylon zwageri*	Lauraceae	SW
96	*Fagraea racemosa**	Loganiaceae	SL
97	*Ficus aurata**	Moraceae	S L
98	*Ficus deltoidea*	Moraceae	SL
99	*Ficus grossularioides*	Moraceae	SL
100	*Ficus obscura*	Moraceae	SL
101	*Ficus variegate*	Moraceae	SL
102	*Ficus* sp.*	Moraceae	SW/SL
103	*Fissistigma manubriatum*	Annonaceae	SW
104	*Flagellaria indica*	Flagellariaceae	SL
105	*Fordia splendidissima**	Leguminosae-Pap	SW/SL
106	*Freycinetia* sp.	Pandanaceae	SW
107	*Friesodielsia borneensis*	Annonaceae	SW
108	*Garcinia parvifolia*	Guttiferae	SW
109	*Garcinia* sp.	Guttiferae	SL/SW
110	*Gironniera nervosa*	Ulmaceae	SW
111	*Glochidion sericeum**	Euphorbiaceae	SL
112	*Gmelina* sp.	Verbenaceae	SL
113	*Gordonia* sp.	Theaceae	SW
114	*Geunsia pentandra*	Verbenaceae	SL
115	*Guioa* sp.	Sapindaceae	SL
116	*Gymnacrathera farquhariana*	Myristicaceae	SW
117	*Hedyotis* sp.*	Rubiaceae	SW/SL
118	*Heliciopsis* sp.	Proteaceae	SW
119	*Homalanthus populneus*	Euphorbiaceae	SL
120	*Hypobatrum* sp.	Rubiaceae	SW/SL
121	*Ilex cymosa*	Aquifoliaceae	SL
122	*Imperata cylindrica**	Gramineae	SL
123	*Ixora* sp.	Rubiaceae	SW
124	*Knema latericia*	Myristicaceae	SW
125	*Knema pallens*	Myristicaceae	SW

No	Species	Family	Location
126	*Knema purpuracea*	Myristicaceae	SW
127	*Knema* sp.	Myristicaceae	SW
128	*Koompassia malaccensis*	Leguminosea/ Caes	SL
129	*Korthalsia* sp.	Palmae	SW/SL
130	*Lantana camara**	Verbenaceae	SL
131	*Lasianthus* sp.	Rubiaceae	SL
132	*Leea indica*	Leeaceae	SL
133	*Lepisanthes alata*	Sapindaceae	SL
134	*Lepisanthes amoena*	Sapindaceae	SL
135	*Licania splendens*	Chrysobalanaceae	SW/SL
136	*Licuala* sp.	Palmae	SW
137	*Lindsaea* sp.	Lindsaeaceae	SL
138	*Lithocarpus coopertus*	Fagaceae	SW
139	*Litsea elliptica*	Lauraceae	SL
140	*Litsea firma*	Lauraceae	SW/SL
141	*Litsea angulata.*	Lauraceae	SW
142	*Litsea* sp.	Lauraceae	SW
143	*Lophopetalum javanicum*	Celastraceae	SW
144	*Lycopodium cernum**	Lycopodiaceae	SW/SL
145	*Lygodium circinatum**	Schizaeaceae	SL
146	*Lygodium flexuosum**	Schizaeaceae	SL
147	*Lygodium microphyllum**	Schizaeaceae	SL
148	*Macaranga gigantea*	Euphorbiaceae	SL
149	*Macaranga lowii*	Euphorbiaceae	SW
150	*Macaranga beccariana*	Euphorbiaceae	SL
151	*Macaranga trichocarpa*	Euphorbiaceae	SL
152	*Macaranga hypoleuca*	Euphorbiaceae	SL
153	*Macaranga* sp.*	Euphorbiaceae	SW/SL
154	*Madhuca kingiana*	Sapotaceae	SW
155	*Madhuca pallida*	Sapotaceae	SW
156	*Madhuca sericea*	Sapotaceae	SW
157	*Maesa ramentacea*	Myrsinaceae	SW/SL
158	*Maesa* sp.*	Myrsinaceae	SL

No	Species	Family	Location
159	*Mallotus paniculatus*	Euphorbiaceae	SL
160	*Mapania* sp.	Cyperaceae	SW
161	*Melanochyla fulvinervis*	Anacardiaceae	SW
162	*Melastoma malabathricum**	Melastomataceae	SL
163	*Melicope* sp.	Rutaceae	SL
164	*Meliosma* sp.	Sabiaceae	SL
165	*Microcos cinnnamomifolia*	Tiliaceae	SW
166	*Microcos* sp.	Tiliaceae	SW
167	*Mikania cordata**	Asteraceae	SL
168	*Mikania* sp.	Asteraceae	SL
169	*Mimosa invisa*	Leguminosea/ Mimosaceae	SL
170	*Mitrella kentii*	Annonaceae	SL
171	*Monocarpia kalimantanesis*	Annonaceae	SL
172	*Mussaenda* sp.	Rubiaceae	SL
173	*Nauclea subdita*	Rubiaceae	SL
174	*Neo-uvaria acuminatissima*	Annonaceae	SW
175	Nephrolepis biserrata*	Neprolepidaceae	SL
176	Nephrolepis *sp*	Neprolepidaceae	SW
177	*Nephenthes sp.*	Nepenthaceae	SL
178	*Oldenlandia* sp.	Rubiaceae	SL
179	*Omphalea bracteata*	Euphorbiaceae	SW/SL
180	*Paederia verticellata*	Rubiaceae	SL
181	*Pandanus* sp.	Pandanaceae	SW
182	*Panicum* sp.*	Poaceae	SL
183	*Paspalum* sp.	Poaceae	SL
184	*Passiflora foetida*	Passifloraceae	SL
185	*Pavetta* sp.	Rubiaceae	SL
186	*Pentace* sp.	Tiliaceae	SW
187	*Photos* sp.	Araceae	SW
188	*Piper aduncum**	Piperaceae	SL
189	*Piper* sp.	Piperaceae	SW
190	*Pipturus* sp.	Urticaceae	SW

No	Species	Family	Location
191	*Plyllanthus emblica*	Euphorbiaceae	SL
192	*Polyalthia lateriflora*	Annonaceae	SW
193	*Polyalthia sumatrana*	Annonaceae	SW
194	*Polyalthia* sp.	Annonaceae	SL
195	*Praravinia parviflora*	Rubiaceae	SW
196	*Praravinia* sp.	Rubiaceae	SL
197	*Psychotria* sp.*	Rubiaceae	SW/SL
198	*Pteridium aquilinum**	Hypolepidaceae	SL
199	*Pternandra azurea**	Melastomataceae	SW/SL
200	*Pternandra rostrata*	Melastomataceae	SW
201	*Reinwortiodendron humile*	Meliaceae	SW
202	*Rinorea* sp.	Violaceae	SL
203	*Roureopsis acutipetala*	Connaraceae	SL
204	*Saccharum spontaneum**	Poaceae	SL
205	*Salacia* sp.	Celastraceae	SW/SL
206	*Santiria graffithii*	Burseraceae	SW
207	*Schima wallichii*	Theaceae	SL
208	*Schizaea digitata*	Schizaeaceae	SL
209	*Schyzostachyum* sp.	Poaceae	SL
210	*Scleria* sp.*	Cyperaecae	SL
211	*Shorea laevis*	Dipterocarpaceae	SW
212	*Shorea ovalis*	Dipterocarpaceae	SW
213	*Shorea parvistipulata*	Dipterocarpaceae	SW
214	*Shorea* sp.	Dipterocarpaceae	SW
215	*Solanum* sp.	Solananceae	SL
216	*Spatolobus* sp.*	Leguminosae-Pap	SL
217	*Stachyprynium* sp.	Marantaceae	SW
218	*Stenochlaeana palustris**	Blechnaceae	SL
219	*Stephania corymbosa**	Menispermaceae	SL
220	*Strychnos* sp.	Loganiaceae	SW
221	*Styrax* sp.	Styracaceae	SL
222	*Symplocos* sp.	Symplocaceae	SL

No	Species	Family	Location
223	_Syzygium caudatilimbum_	Myrtaceae	SW
224	_Syzygium lineatum*_	Myrtaceae	SL
225	_Syzygium napiforme_	Myrtaceae	SL
226	_Syzygium_ napiforme	Myrtaceae	SW
227	_Syzygium hirtum_	Myrtaceae	SW
228	_Syzygium tawahense_	Myrtaceae	SW
229	_Syzygyum_ sp.	Myrtaceae	SW
230	_Tarenna_ sp.	Rubiaceae	SL
231	_Teijsmanniodendron coriaceum_	Verbenaceae	SW
232	_Tetracera akara_	Dilleniaceae	SL
233	_Tetracera_ sp.	Dilleniaceae	SW/SL
234	_Tetrastigma_ sp.	Vitaceae	SW
235	_Timonius lasianthoides_	Rubiaceae	SL
236	_Timonius_ sp.	Rubiaceae	SL
237	_Trema tomentosa_	Ulmaceae	SL
238	_Trigonostemon_ sp.	Euphorbiaceae	SW
239	_Uncaria cordata_	Rubiaceae	SL
240	_Urena lobata_	Malvaceae	SL
241	_Urophyllum_ sp.	Rubiaceae	SW
242	_Uvaria lobbiana_	Annonaceae	SL
243	_Uvaria_ ovalifolia	Annonaceae	SW
244	_Uvaria_ sp.	Annonaceae	SW
245	_Vatica umbonata_	Dipterocarpaceae	SW
246	_Vernonia arborea*_	Asteraceae	SL
247	_Vitex pinnata*_	Verbenaceae	SL
248	_Willughbeia_ sp.	Apocynaceae	SW
249	_Xanthophyllum affine_	Polygalaceae	SW/SL
250	_Xanthophyllum_ sp.	Polygalaceae	SL
251	_Xylopia ferruginea_	Annonaceae	SW
252	_Xylopia malayana_	Annonacaea	SW

*) _species included in canonical correspondence analysis (CCA)_

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Wageningen, 4 September 2012

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Short Biography

Ishak Yassir was born on the 22nd of May 1973, in Balikpapan, East Kalimantan-Indonesia. He completed his high school education in his hometown. He studied forestry at the Faculty of Forestry, Mulawarman University in Samarinda, Indonesia. He obtained his diploma with forest planning as his major in 1997. From 1997 to 2000, he was head of Forest Planning at the Planting Division of PT. Sumalindo Lestari Jaya, Muara Karangan-East Kalimantan, Indonesia. Since 2000, he has been working as staff member in the Forestry Research Development Agency (FORDA), Ministry of Forestry of Indonesia at Wanariset Samboja, East Kalimantan. In 2003, he continued his study at Bogor Agriculture University, and he obtained his degree of Master Science with forest ecology as his major in 2005. In May 2006 he started his PhD thesis at Wageningen University, the Netherlands. Since 2000 until now, he has been also working and supporting in the Orangutan conservation project in Kalimantan, Indonesia. After completing his PhD in 2012, he is going to return to Indonesia and continue with his career at the Ministry of Forestry.

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- Basic and advanced statistic
- The art of Modelling

Other PhD Courses

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- Co-organizing symposium on Mined Land Rehabilitation: Mainstreaming Landscape Restoration in Indonesia, 6-7 March 2012, Balikpapan, Indonesia
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Oral Presentations

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- Applying an ecological approach to create new forest from *Imperata* grassland in Samboja-East Kalimantan . Restoring Forests for Communities, Biodiversity, and Ecosystem Services, 12 September, 2011, Bogor, Indonesia
- Secondary succession after fire in *Imperata* grasslands of East Kalimantan, Indonesia, 19th World Congress of Soil Science , 1-6 August , 2010, Brisbane , Australia

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