Effects of Forest Fire on Characteristics of Soil Ecosystem in South Sumatra, Indonesia

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Abstract

Both conversion of natural forests into human-managed land and burning of forest ecosystems are still widely practiced in Indonesia. More frequently drought related to the El Niño-Southern Oscillation (ENSO) and the irresponsible use of natural resources have also exacerbated fire problems over the last twenty years. However, informative data quantifying the effects of conversion of natural forests into human-managed ecosystem and mainly the effects of fires on the soils are still lacking, hence the effects are still poorly understood. The objectives of the present study were to study the effects of forest ecosystem practices on the characteristics of Ultisols and to investigate both the immediate effects and delayed effects of wildfire on the dynamics of chemical and biological characteristics of Ultisols in South Sumatra Province, Indonesia. This was carried out by investigating the immediate changes in the chemical and biological characteristics of the soils both under laboratory and field conditions.

The results of the preliminary study showed evident differences in soil characteristics between the natural forest and the human-managed ecosystems, and between the unburned and the burned sites. In spite of the fact that soils in the unburned sites showed a better physico-chemical potential than those in the burned sites, in general Ultisols in the study sites showed a low natural fertility potential.

Artificial heating at 100°C did not cause significant changes in soil chemical characteristics but increased the sand fraction and decreased the clay fraction of the soil. However, heating at 300 and 500 ° C, a representative of *intensive* burning, for example during forest fires, adversely affected all soil characteristics investigated. Watering to some extent showed positive effects on chemical characteristics, except on the available phosphorous.

Both forest management practices and wild fire occurrence significantly affected the dynamics of total carbon, total nitrogen, available phosphorous, pH, exchangeable bases, and cation exchange capacity of the soils. Surface soil total carbon, and total nitrogen in conservation forest, pine forest, and home garden varied significantly among periods of sampling but were generally stable. Total carbon, total nitrogen, available phosphorous, pH, and exchangeable bases significantly increased immediately after fire while Al saturation and cation exchange capacity significantly decreased. Soil nutrients still accumulated 1 yr after fire but depleted and returned to the pre-fire levels by year 2. Although fallowing aggraded the burnt sites, the nutrient levels in the burnt sites were still lower than those in the unburnt sites.

Conversion of natural forest into human-managed ecosystems significantly reduced number of spores, species richness, and Shannon-Wiener's Diversity index (*H'*) of vesicular arbuscular mycorrhizal fungal communities. Both predominant genus and species composition of VAM fungal community changed as a result of ecosystem conversion and fire occurrence. Similarly, bacterial populations decreased as the natural forest was converted into human-managed ecosystem except for the P-solubilizing bacteria in PF. The lowest population of all microbial groups was found in HG, reflecting that increases in agricultural practices had caused more pressure on microbial population. The immediate effect of wildfire on VAM fungal and bacterial community was striking but short-lived. Species richness and Shannon-Wiener Diversity index (*H'*) of VAM fungal community significantly decreased immediately after the fire but increased three months after the fire. Amongst the bacterial groups investigated, nitrifying bacteria were the most severely affected.

Although fluctuating significantly with time, bacterial populations returned to the pre-fire levels two years after the fire. The temporal improvement in soil nutrients (abiotic factors) after fire was clearly an important factor for the increases in microbial population. However, the lag time of recovery showed by *Rhizobia* also reflected the importance of biotic factors (vegetations) in determining the survival of certain groups of soil microorganisms. Considering the long-term productivity, the declines in nutrient pools in the *Acacia mangium* site might have significant implications because *Acacia mangium* plantation has about 6- to 7-yr rotation. On the other hand continuous and regular organic matter turnover, for example in the PF and HG sites, successfully maintains levels of nutrient pools mainly C and N comparable to those found in the CF sites. The high intra-periodic variation reported here suggests that care must be taken in selecting control site and suggests the necessity of repeated sampling in order to determine the long-term effects of management practices and the long-term effects of wildfires on the dynamics of microbial populations in soils.

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Chapter I

Introduction

It is widely accepted that forest resources should be managed to meet the economic, ecological, and social needs of present and future generations. Therefore forest management must respond to environmental, social, and economic issues. This requires feedback of relevant information between planning, implementation, control, and impacts of forest management. A hierarchical system of principles, criteria, and indicators can be used to organize this information in a manner useful to conceptualize, evaluate, implement sustainable management of forests (Lamnerts van Bueren and Blom, 1997; Prabhu et al., 1999).

Indonesia ranks third (behind Brazil and Zaire) in its endowment of tropical rainforests, possessing about 10% of what remains in the world of this resource. Although estimates of the area of forested land in Indonesia diverge widely, estimates from the early 1990s of actual forest cover range from 92.4 to 113 million hectares with the approximate distribution as follows: Kalimantan (32.0% of the total), Irian Jaya (29.9%), Sumatra (20.8%), Sulawesi (9.7%), Maluku (5.5%), and other (2.1%) (GOI/FAO, 1996). Kartodihardjo (1999) using the upper estimate (113 million ha) further classified the forest resources into protection forest (26.1%), conservation forest (16.9%), and production forest (57%).

Dating from the mid-1960s, commercial exploitation of forests outside of Java, Madura, and Bali has grown rapidly. In 1996, 445 logging concession were operating on 54,060,599 ha. In 1994, wood and wood products produced about US$ 5.5 billion in export revenue for Indonesia, about 15% of the total and there were approximately 700,000 people employed in the formal forest sector (Sunderlin and Resosudarmo, 1996). In the course of the development of the timber industry, there has been a marked increase in the extent and rate of disappearance of Indonesia's forest cover. Forest cover of the country has been reduced from 74% to 56% in the space of 30 to 40 years with the estimated rate of annual deforestation at between 0.6 to 1.2 million ha (Sunderlin and Resosudarmo, 1996). The most recent statistics by World Bank (1999) revealed that the average annual deforestation rate since 1986 to 1997 has actually been about 1.5 million ha, much of it caused by forest fires with Sumatra and Kalimantan being the most severely affected islands. Some 30 and 22% of forest cover in Sumatra and Kalimantan vanished during this period. Recent statistics by Bryant et al. (1997) showed that only 53 million hectares of frontier forest (relatively undisturbed forest) remained in Indonesia. If current trends continue, virtually all nonswampy lowland forests in Sumatra and Kalimantan will be destroyed by 2010 (Barber and Schweithelm, 2000; World Bank, 1999).

Further, people have been living in and around forests since the ancient time, taking what they need from the wealth of the natural resources available. The indigenous people have been farming the rain forests, felling trees to allow cultivation of crops and grazing space for animals, and moving on when the soil becomes less fertile. This is no threat to the forests, provided that it is carried out sustainably, and used areas are left to regenerate for long periods before repeating the process. However, in the last two centuries populations have expanded, requiring more and more space for housing and agriculture. As a result the long fallow period (20 years) become shorter, the land is not allowed sufficient time to recover, and intensive farming results in irreversible soil degradation. Coupled with extremely lucrative international timber trade in recent years, this has brought about unparalleled degradation of the rain forests.

Tropical rain forests have generally been regarded as ecosystems in which fire was excluded by the prevailing moist environment and the resulting unsuitable conditions of all material (fuels) for burning. The spread of frequent fires into the wet tropics is a more recent phenomenon, attributable largely to human population pressures. During the last two decades, the occurrence of fire has dramatically increased with the rapid clearing of tropical rain forests for economic development. This in turn has caused overall degradation and promoted the geographic expansion of pyrophytic grass life forms with increasing flammability and fire frequency (Mueller-Dumbois and Goldammer, 1990).

Indonesia received unaccustomed attention in the world's headlines during the latter part of 1997 as forest and land fires raged in Sumatra and Kalimantan. The fires pumped enough smoke into the air to blanket the entire region in haze, reaching as far north as southern Thailand and the Philippines, with Malaysia and Singapore being particularly affected. The fires burnt out of control again in early 1998, the second year of perhaps the worst El Niño-Southern Oscillation (ENSO)-related drought ever recorded in Indonesia. Obtaining accurate data about the spatial distribution of the 1997-98 fires, the total area burnt, and the proportion of different vegetation is difficult because of the size and wide distribution of the burns, the remoteness of many of the sites, the inability of most satellite remote-sensing devices to penetrate the thick haze while the fires are burning, and the need to verify interpretations of images. The Ministry of Forestry and Estate Crops (1998) officially estimated that in the country as a whole only 165,000 ha of designated forestlands had burnt in 1997. However, Ramon and Wall (1998) using satellite imagery made a preliminary estimate that the 1997 fires burnt 2.3 million hectares in South Sumatra alone. Furthermore a study by Goenner (1998) in East Kalimantan registered a figure of 8 to 10 million ha of lands vanishing during fire from middle of 1997 to May 1998. Similarly Liew et al. (1998) using SPOT (Syte e Pour l'observation de la Terre) satellite imagery calculated that in 1997 approximately 1.5 million ha had burnt in Sumatra and approximately 3.0 million ha in Kalimantan. Regardless of the discrepancies in the figures available, the fires were almost evenly divided between wildfires and controlled burns. The types of vegetation that burnt, in descending order of importance, were wetland vegetation being cleared to prepare rice fields, secondary brush, scrublands and herbaceous swamplands, dryland shifting agricultural plots, and grassland in coastal peat swamps. The important points are that very large areas of Sumatra and Kalimantan burnt in 1997 and 1998 and that many different types of vegetation burnt.

The heavy suffocating smoke produced by these fires caused a major health hazard, diminished agricultural production, disrupted travel and communications, and seriously disturbed wildlife over large areas. The drying of rivers and lakes as a result of drought conditions and their subsequent pollution with fire debris also had a major affect on forest inhabitants. Cities, towns and even smaller settlements throughout Sumatra and Kalimantan that were dependent on river water suffered water supply problems during this El Niño. To date there has been an attempt to value the costs of the 1997-98 fires and haze described above. BAPPENAS (1999) estimated a total loss between $8.9 billion and $9.7 billion with a mean value of $9.3 billion, as presented in Table 1.1. In fact, much of the cost of the fire damage probably cannot be estimated. Assigning a dollar value to the destruction of some of the last intact lowland forest in Sumatra, the death of a large percentage of Indonesia's remaining wild "orang-utans", or the shortened life span of medically vulnerable people made ill by the haze, is impossible. However, even this conservative and partial assessment of the monetary costs gives policymakers and the people of the region a tangible way of understanding the destruction brought by the fires and haze.

Table 1.1. The estimated economic cost of the 1997-1998 fires and haze (million U.S. dollar)

Sector	Estimated economic losses		
	Minimum	Maximum	Mean
Agriculture:			
Farm crops	2,431	2,431	2,431
Plantation crops	319	319	319
Forestry:			
Timber from natural forests (logged and unlogged)	1,461	2,165	1,813
Lost growth in natural forests	256	377	316
Timber from plantations	94	94	94
Nontimber forest products	586	586	586
Flood protection	404	404	404
Erosion and siltation	1,586	1,586	1,586
Carbon sink	1,446	1,446	1,446
Health	145	145	145
Transmigration and building and property	1	1	1
Transportation	18	18	18
Tourism	111	111	111
Firefighting costs	12	12	12
Total	8,870	9,726	9,298

Source: BAPPENAS (1999)

Apart from economic losses described above, the ecological damage was considerable. All disturbances produce impacts on ecosystems. The level and direction of impact (negative or positive) depends on ecosystem resistance and resilience, as well as on the severity of the disturbance. The variability in resource damage and response from site to site and ecosystem to ecosystem is highly dependent on burn and fire severity. Burn severity (fire severity) is a qualitative measure of the effects of fire on site resources (Hartford and Frandsen, 1992). As a physical-chemical process, fire produces a spectrum of effects that depend on interaction of energy release (intensity), duration, fuel loading and combustion type, climate, topography, soil, and area burnt (Robichaud et al., 2000).

Fire intensity is an integral part of burn severity and the terms are often incorrectly used synonymously. Intensity refers to the rate at which a fire is producing thermal energy in the fuel-climate environment (DeBano et al., 1998). Intensity is measured in terms of temperature and heat yield. Surface temperature can range from 50 to greater than 1,500 °C. Heat yields per unit area can be as little as 260 kg cal m^{-2} in short, dead grass to as high as 10,000 kg cal m^{-2} in heavy logging slash (Pyne et al., 1996). Rate of spread is an index of fire duration and can vary from 0.5 m $week^{-1}$ in smoldering peat fires to as much as 25 km hr^{-1} in catastrophic wildfires.

Fire, either wild or prescribed, may have a wide range of effects on soils. The wide range of effects is due to the inherent pre-burn variability in soil characteristics, and to fire behavior characteristics, season of burning, and prefire and postfire environmental conditions, such as timing, and duration of rainfall. However, effects of fire on some soils are poorly documented and poorly understood. Thus, generalization about effects of fire on soils is likely to be more risky and overstated than with any other managed resources (Clark, 2001).

When biomass on or above soil surface burns, a heat pulse penetrates the soil. The magnitude of the heat pulse into the soil depends on fuel loading, fuel moisture content, fuel distribution, rate of combustion, soil texture, soil moisture content, and the length of time that the heat source is present (Clark, 2001). Because fuels are not evenly distributed around a site, a mosaic of soil heating occurs. The highest soil temperatures are associated with areas of greatest fuel consumption and the areas that have the longest duration of burning. Because the pattern of soil heating varies significantly around a site, with differences in both the amount and duration of soil heating, a range of fire effects on soils can occur on burnt areas. Debano (1977) estimated that about 8% of the heat generated in California chaparral fires is absorbed and transmitted downward into the soil. In general, "lightly burnt" forest will cause maximum soil surface temperature between 100 and 200°C and the temperature 1.0 and 2.0 cm below the surface will not exceed 100 ° C (Chandler et al., 1983). In "moderately burnt" areas, surface temperatures are typically in the 300- to 400°C-range and may be between 200 and 300 ° C at 1.0-cm depth (Chandler et al., 1983). A "severely burnt" area may result in surface temperature approaching 760°C (Chandler et al., 1983).

The resulting high temperatures can alter soil properties, and kill roots, seeds and soil microbes. Quantitative data on the impacts of wildfire in the tropics on soil quality are still limited, although some information is available for other vegetation types and different regions (Campbell et al., 1995; Dunn et al., 1979; Giovannini and Lucchesi, 1997; Kang and Sajjapongse, 1980; Marion et al., 1991). Heating may cause changes in some physical properties of soils, including the loss or reduction of structure, reduction of porosity, alteration of color, increased soil erosion, the formation of water repellent layers, hydrophobicity, and alteration of soil mineralogical properties (Clark, 2001; Ghuman et al., 1991; Ketterings et al., 2000).

Several chemical changes in soils may occur as a direct result of fires. Low-intensity fires have been shown to facilitate the cycling of some nutrients (Wells et al., 1979) although short-lived (DeBano et al., 1987; Gifford, 1981). In contrast, intense fires can volatilize excessive amounts of nutrients, and destroy organic matter and CEC (Ghuman and Lal, 1991; Giovannini and Lucchesi, 1997; Wells et al., 1979). Recovery after low-intensity fire is generally rapid even though natural mechanisms such as atmospheric deposition add nutrients to a site very slowly. Increases in N-fixing organisms in many soils help offset any nitrogen losses (Metz et al., 1961; Wells, 1971). Thus, the extent of N fixation and subsequent soil N transformations occurring after fire are critical factors in evaluating the long-term effects on site quality (Harvey et al., 1976).

Effects of fire on ecosystem processes and components can be both immediate and long term. If fire removes most of the organic matter on a forested site, productivity may be significantly reduced for many years (Harvey et al., 1986). However, long-term responses of ecosystem processes to fire have not been studied in detail because such studies have been regarded costly and complex, and often inconclusive (Keane et al., 1996). Since living organisms in soils are mostly located in or near the soil surface, they are exposed to heat radiated by flaming surface fuels. As a result, alterations in their populations are simultaneously expected to occur. Although immediate decline in microbial population after burning has been shown under chaparral shrublands in southern California (Chandler et al., 1983; Dunn et al., 1985) and natural grassland in southern India (Senthilmukar et al., 1998), information pertaining the population dynamics of indigenous VAM fungi, nitrifiers, P-solubilizing bacteria, and rhizobia in relations to wildfire under tropical climate is still lacking.

The increasing fire occurrence in forested areas of South Sumatra, Indonesia and the gap in information pertaining long-term effects of wildfires on soil characteristics of the tropics prompted us to undertake an extensive research program on the effect of wild fires on the characteristics of soils in South Sumatra, Indonesia. The objectives of the current study include (1) to study the immediate modifications of soil characteristics induced by artificial heating. In order to study possible changes in soil characteristics after rain, the heated soils were then either watered and left unwatered (Chapter III), (2) to investigate the effects of forest management practices on chemical characteristics of soils, and to investigate both the immediate and leftover effects of wild fires on the dynamics of chemical characteristics (Chapter IV), and (3) to investigate the effects of forest management practices on the populations of some beneficial soil microorganisms, and to investigate both the immediate and leftover effects of wild fires on the dynamics of the populations of some beneficial microorganisms (Chapter V). In Chapter II, data on soil physico-chemical characteristics of different ecosystems with different wildfire history were noted as baseline information for further characterization in the current study.

Chapter II

Characteristics of Ultisols under Different Wildfire History in South Sumatra, Indonesia: Physico-chemical Properties

2.1. Background

In the last two decades, fire has become one of the greatest threats to Indonesian tropical rainforest. Boosted by the 1997/1998 El Niño phenomenon, uncontrolled fires have destroyed hundred thousands hectares of forests, plantations, agricultural fields, and bushes in Indonesia. Official estimates registered a figure of 263, 992 ha of various land-uses being scorched in Indonesia in 1997 alone. However, the coverage in the field might be much worse than the estimates. A recent study by Goenner (1998) reported a figure ranging from 8 to 10 million ha in East Kalimantan alone being destroyed by fire from mid 1997 to May 1998. A study by FFPCP (1997) reported a figure of 142,000 ha of land in South Sumatra destroyed by wildfires from 1993 to 1995. Bompard and Guizol (1999) reported an estimate of 786,000 ha of forest including those on peat, smallhorder rubber plantation, pulpwood, timber plantation, and bushes in South Sumatra destroyed by fire in 1997 alone.

Apart from economic losses, the ecological damage was considerable. When biomass on or above soil surface burns, a heat pulse penetrates the soil. The resulting high temperatures can alter soil properties, and kill roots, seeds and soil microbes. Quantitative data on the impacts of wildfire in the tropics on soil quality are still limited, although some information is available for other vegetation types and different regions (Campbell et al., 1995; Dunn et al., 1979; Giovannini and Lucchesi, 1997; Kang and Sajjapongse, 1980; Marion et al., 1991). Each of those studies was limited to either sudden impacts of artificial heating or impacts of prescribed burning on soil properties. On the other hand data available quantifying long-term effects of wildfire on tropical soil characteristics are still limited, hence still poorly understood.

Understanding both immediate and delayed changes in soil properties associated with fires is crucial because soil fertility may decrease and subsequent site degradation may occur if inadequate management practices are employed after fires. Previous studies indicated that the soils over which fire has passed have been intensely eroded and profoundly degraded (Giovannini and Lucchesi, 1987), which is most likely due to reduced organic matter content, polysaccharides, and water stable aggregates (Domear et al., 1979), all factors associated with soil erodibility. Therefore it is important to follow eventual successive effects of the fire on the soil ecosystem. Since 1998, we have been monitoring the natural recovery of soil of different ecosystem with varied wild fire history in South Sumatra, Indonesia. This chapter reports data on soil characteristics of different ecosystem with different wildfire history as baseline information for further characterization of dynamics of soil properties.

2.2. Materials and Methods

2.2.1. Study Site

The study area was located in Pendopo Sub-district, Muara Enim District, South Sumatra Province, about 200 km Southwest to Palembang, Indonesia, stretching from 103º30' E to 103º50' E and 3º15' N to 3º25' N (Fig. 2.1). Pendopo belongs to Type A zone with ratio of dry month to wet month <1.5 (Schmidt and Ferguson, 1951). Although there has been a seasonal shift, rainy season usually starts in September and ends in March with average annual precipitation of 2600 to 4200 mm; dry season is from April to August. Air temperature is relatively similar all year long with mean annual temperature of 25ºC.

The study sites were located at elevation of approximately 60 to 140 m above sea level with slope ranging from 3% to 16%. Based on Geological Map (Gafoer et al., 1986), the soils in Pendopo Area developed from tertiary fine sedimentary rocks. Hapludults, Hapludox and Dystropepts could be found in the study area. The soils were Yellowish-brown Podzolic, and association between Yellowish-red Podzolic and Yellowish-brown Podzolic (LPT, 1974), equivalent to Acrisols (FAO/UNESCO, 1974) or Ultisols (Soil Survey Staff, 1992).

Seven sites ranging from conservation forest (CF) to 20-year old pine plantation forest (PF), unburnt 7-year-old *Acacia mangium* plantation (AM), and *Acacia mangium* plantation burning in 1995(B95), 1997 (B97) and 1998 (B98), were selected for this study. Starting from May 1999, a home garden area (HG) was also included. In each selected site, a representative sub-plot of 50 m x 50 m was established for soil sampling purposes.

The overstory of CF was mixed-tree species. Some primary species included *Agathis loranthifolia* Salibs, *Peronema canescens* Jack, *Shorea* spp., *Altingia excelsa* Noronhae, and *Durio zibethinus* Meur. Although this area has been designated as a conservation forest, it was not free from disturbances because people living in the vicinity have been using it as a source of living. In addition, illegal logging has also been taking place, resulting in open spots. As a result shrubs and grasses could be found as understory species. Pine (*Pinus merkusii*) forest was about 20 years old at the start of this experiment (December 1998). In the 50-m x 50 m-plot, there were 56 stands of *Pinus merkusii*, with average girth at breast height (GBH) of 88 cm (54 -143 cm). The dominant understory species were ferns and *Mimosa* sp. The *Acacia mangium* plantation was

Fig. 2.1. Location of the study site in Pendopo, Muara Enim, South Sumatra Province, Indonesia.

either burnt or unburnt. The unburnt *Acacia mangium* was about 7 years old at the start of this study. There were 300 stands of *Acacia mangium* in the 50 m x 50 m-plot, with average GBH (based on 60 samples) of 44 cm (21 to 116 cm). The understory species included *Astonia* sp., *Caesalpinia pulchriina*, *Zingiber aromaticum* vail, and *Mimosa* sp. The burnt *Acacia mangium* had different burning history, namely in 1995, 1997, and 1998, respectively. Pre-wildfire vegetation was 5-year old, 6-year old, and <1-year old *Acacia mangium*, consecutively. The burnt area had sparse regrowing *Acacia mangium* mixing with shrubs and grasses, mainly *Imperata cylindrica* as understory species. The home garden has been cultivated for two years at the start of this study. It was established on a fallowed area in 1997 by a slash- and burn-method; thereafter nor burning treatment had been applied. Long-term tree species was rubber tree (*Hevea brasiliensis*) with cassava (*Manihot esculenta*), banana (*Musa* sp.), upland rice (*Oryza sativa*), bean (*Phaseolus vulgaris*), peanut (*Arachis hypoge* L.), and maize (*Zea mays* L.) in the alley.

2.2.2. Soil Samples and Analytical Methods

Soil profiles were described in terms of color, texture, structure, consistency, and occurrence of roots. From each site, both undisturbed and disturbed soil samples were collected. The undisturbed soil cores were collected using 100cc-soil corers from the depth of 0 - 10 cm for bulk density or hydraulic conductivity measurement. Bulk density was calculated based on the 100cc-core and constant dry weight of the core following drying at 105ºC for 24 hours. Moisture content was gravimetrically calculated after drying. Saturated water permeability was measured using a permeameter (Daiki, DIK 4000). Soil hardness was measured in the field using a fall-cone-type soil penetrometer (Daito Green Ltd. Hasegawa type H-60) to the depth of 60 cm. Particles size distribution was determined by the pipette method using 6 and 30%-H_2O_2 as a decomposing agent of organic matter.

For physico-chemical characterization, soil samples were collected from each layer of pedons. Soil pH was measured in a soil to water or 1 M KCl ratio of 1 to 5 (designated as pH-H_2O and pH-KCl, respectively). Electrical conductivity (EC) was measured using the same solution as for pH-H_2O measurement. The filtrate from pH-KCl measurement was used for exchangeable Al and H analysis using the titration method. Exchangeable bases (Ca, Mg, K, and Na) were extracted with 1M-ammonium acetate (pH 7.0), followed by reciprocal shaking and centrifugation in a soil to solution ratio of 1 to 5. The amount of exchangeable bases was then measured by atomic absorption spectrophotometry (Shimadzu, AA-610S). After the extraction of exchangeable bases, the residue was washed with 20 mL of deionized water once and twice with 20 mL of 99% EtOH to remove the salt excess. The ammonium was extracted with 30 mL of 10%-NaCl solution twice, followed by reciprocal shaking for 1 h and centrifugation for 10 min at 800 G. The ammonium ion content was determined as cation exchange capacity (CEC) by steam distillation and titration method. Total Carbon (T-C) and Nitrogen (T-N) were determined by a dry combustion method with NC-Analyzer (Sumigraph NC-80). Available phosphorous (available P) was extracted with an extracting solution containing 1N NH_4F and 0.5N HCl, followed by reciprocal shaking for 1 min (Bray-1 method), colorimetrically analyzed for available P using a spectrophotometer at 660 nm. Al, Fe, and Si oxides were extracted with 0.2M ammonium oxalate solution (pH 3.0), using reciprocal shaking in the dark for 1 h in a soil to solution ratio of 1 to 25 (Mckeague and Day, 1966), then designated as Alo, Feo, and Sio. Al, Fe, and Si oxides were also extracted with citrate-bicarbonate mixed solution pH 7.3 with the addition of sodium dithionate at 75 to 80ºC for 15 min in a soil to solution ratio of 1 to 100 (Mehra and Jackson, 1960), then designated as Ald, Fed, and Si. Al, Fe, and Si were determined using a sequential plasma spectrometer (Shimadzu, ICPS-1000IV). Values of zero point of charge (ZPC) and σ_p were determined with the STPT method (Sakurai et al., 1988).

2.3. Results and Discussion

2.3.1. Soil Morphological Characteristics

All sites were located in one compound area, except CF, which was about 20 km apart from the closest site (PF). Therefore, there might not be definitive differences in climate and geological time for long-term weathering. Topographically, all pedons were located in the middle slope of hills, with slopes ranging from 6 degrees (CF, AM, B95, B97 and HG) to 8 degrees (PF and B98), respectively. The morphological characteristics of soils are given in Table 2.1. The study site had soils with deep solumn. Up to the depth of 150 cm, neither rock fragment nor parent material was found. Organic layer was 5 cm in PF, 4 cm in AM, and 2 cm in CF, and composed of plant debris of varied floor growth in CF, pine leaves and fern debris in PF, and *Acacia mangium* leaves in AM.

Darker A horizons were found in all pedons (Table 2.1). Organic matter (as will be shown in the following section) appeared to contribute significantly to the darker A horizon. Disturbances, such as land conversion and/or fire, to some extent affected soil color. Burnt areas (B95, B97, and B98) showed lighter A horizon than unburnt areas. These changes were due to loss of organic matter caused by combustion and/or subsequent erosion. Exceptionally dark brown Ap horizon in HG sites was related to regular plant debris turnover after harvest. Such condition could be maintained because slash- and-burn practice had never been applied to this home garden since it was established in 1997. Lighter colors dominated lower horizons throughout the study sites. Reddish mottling (2.5YR to 7.5YR) could be found at lower horizon in all pedons. Iron concretion was found in CF, PF, and B97 at 62 cm, 90 cm, and 90 cm, respectively.

Clay loam was a dominant texture class of surface soil except those of CF and AM which had clay and sandy clay loam,

Table 2.1. Morphological characteristics of the soils

Pedon	Hor.	Depth (cm)	Color[a]	Text.[b]	Struct.[c]	Consist.[d]	Root[e]	Bound.[f]
CF	O	-2						
	A	0-12	10YR3/4	C	3fsbk	vs/vp	m	cw
	BA	12-33	7.5YR5/6	C	3csbk	vs/vp	f	cs
	B	33-62	7.5YR6/8	C	3fabk	vs/vp	vf	gw
	BC1	62-84	7.5YR6/6	C	3fp	vs/vp	f	gw
			2.5YR3/6(ffp)					
	BC2	84-93	7.5YR5/6	C	2fp	vs/vp	vf	as
			2.5YR3/6(ffp), abundant Fe stone					
	BC3	93-150	5YR6/8	C	2fp	vs/vp	vf	
			10R4/8(mmp)					
PF	O	-5						
	A	0-11	10YR4/4	CL	3mabk	s/p	m	gw
	BA	11-29	10YR3/4	C	3mabk	s/p	f	gw
	Bt1	29-54	10YR5/6	C	4msbk	s/p	vf	gw
	Bt2	54-90	5YR6/6	C	2fp-3fabk	s/p	vf	gw
			5YR7/1(ffp)					
	BC	90-150	5YR7/3	C	2fp-3fabk	s/vp	vf	gs
			2.5YR4/6					
			10R4/6 (Fe concretion, few, platty)					
			Cutan was found in BA to Bt2 horizons (pressure surface)					
AM	O	-4						
	A	0-12	10YR5/6	SCL	3msbk	s/p	m	as
	Bw	12-57	10YR5/8	CL	3msbk	s/vp	f	as
	Bt	57-74	10YR5/8	C	3msbk	s/p	vf	cs
			7.5YR6/4					
			10YR6/6(mmd)					
	BC	74-150	7.5YR6/4		Sfp	vs/vp	vf	
			5YR5/6(cmp)	C				
B95	A	0-21	10YR5/8	CL	3fsbk	ss/vp	m	cs
	BA	21-30	10YR6/6	CL	3msbk	s/vp	c	cs
	Bw1	30-65	10YR5/4	C	3msbk	s/p	f	gw
			7.5YR5/6(mmf)					
	Bw2	65-80	10YR6/6	C	3msbk	s/p	f	cs
			5YR5/8(mfd)					
	BC1	80-110	10YR5/6, many Fe stone	C	3fabk	s/p	f	gs
			5YR5/8(mfd)					
	BC2	110-150	10YR7/8		3fabk	vs/vp	vf	
			5YR5/8(mmp)	C				

a) Lower row of each horizon indicates mottle color, followed by abundance, size, and contrast in the parenthesis. Abundance: f, few; m, many. Size: f, fine; m, medium. Contrast: f, faint; d, distinct; p, prominent.
b) Abbreviation used for soil texture: C, clay, CL, clay loam.
c) Abbreviation used for soil structure. Grade: 2, weak; 3, moderate. Size: f, fine; m, medium; c, coarse. Shape: sbk, sub-angular blocky; abk, angular blocky; p, prismatic.
d) Abbreviation used for consistence. Stickiness: ss, slightly sticky; s, sticky; vs, very sticky. Plasticity: p, plastic; vp, very plastic.
e) Abbreviation used for root abundance. vf, very few; f, few; m, many.
f) Abbreviation used for boundary. as, abrupt smooth; aw, abrupt wavy; cs, clear smooth; cw, clear wavy; gs, gradual smooth; gw, gradual wavy.

Table 2.1. (cont.)

Pedon	Hor.	Depth (cm)	Color[a]	Text.[b]	Struct.[c]	Consist.[d]	Root[e]	Bound.[f]
B97	A	0-15	10YR5/5	CL	3fsbk	s/vp	m	aw
	BA	15-35	10YR6/6	C	3msbk	vs/vp	c	gs
	Bt1	35-55	10YR7/4 10YR5/8(mff)	C	3msbk	vs/vp	f	gs
	Bt2	55-90	10YR6/6 7.5YR5/8(mff)	C	4-3fsbk	vs/vp	f	cs
	BC1	90-110	10YR6/6 5YR4/6(cfd)	C	3fsbk	vs/vp	vf	gs
	BC2	110-150	7.5YR7/2 5YR5/8(mfp)	C	2fp-2fsbk	vs/vp		
B98	A	0-11	10YR5/4	CL	3fsbk	s/p	m	cw
	BA	11-40	7.5YR6/6	C	3fsbk	s/p	f	gs
	BW	40-70	7.5YR7/4 7.5YR5/6(aff)	C	3fabk	s/p	f	gs
	Bt	70-104	7.5YR7/4 5YR5/6(mfd)	C	3fabk	vs/vp	vf	gs
	BC	104-114	10YR7/2 2.5YR5/8(cfp)	C	2fp	vs/vp	vf	
HG	Ap	0-17	10YR3/6	CL	2fsbk	vs/vp	f	aw
	Bt1	17-47	7.5YR5/6	C	2mabk	vs/vp	f	cw
	Bt2	47-105	7.5YR4/6 5YR5/8(ffp)	C	2fsbk	vs/p	vf	cs
	BC	105-150	7.5YR6/2 5YR4/6(afc)	C	2fp-2fsbk	vs/p	vf	

a) Lower row of each horizon indicates mottle color, followed by abundance, size, and contrast in the parenthesis. Abundance: f, few; m, many. Size: f, fine; m, medium. Contrast: f, faint; d, distinct; p, prominent.
b) Abbreviation used for soil texture: C, clay, CL, clay loam.
c) Abbreviation used for soil structure. Grade: 2, weak; 3, moderate. Size: f, fine; m, medium; c, coarse. Shape: sbk, sub-angular blocky; abk, angular blocky; p, prismatic.
d) Abbreviation used for consistence. Stickiness: ss, slightly sticky; s, sticky; vs, very sticky. Plasticity: p, plastic; vp, very plastic.
e) Abbreviation used for root abundance. vf, very few; f, few; m, many.
f) Abbreviation used for boundary. as, abrupt smooth; aw, abrupt wavy; cs, clear smooth; cw, clear wavy; gs, gradual smooth; gw, gradual wavy.

respectively (Table 2.1). At lower horizons, clay texture was found in all pedons. Also, blocky structures either moderately developed or strongly developed angular blocky and sub-angular blocky were found across the study sites. However weakly developed prismatic structure was also found in lower horizon of CF, PF, AM, B97, and B98, respectively. In accordance with soil texture, soil consistence ranged from sticky to very sticky and plastic to very plastic except in surface soil of B95, which was slightly sticky (Table 2.1). The slightly sticky consistence could be attributed to clay loam texture of this layer (Table 2.1) due to high content of sand as confirmed by the result of soil particle analysis (Table 2.2). With depth, both the stickiness and plasticity increased mainly due to the increasing clay content. Fine roots concentrated in A horizon and decreased with depth.

2.3.2. Soil Physical Properties

2.3.2.1. Soil Bulk Density

Soil bulk density varied according to land-use system and fire history (Table 2.2). In the unburnt sites, the lowest bulk density (0.99 g cm^{-3}) was observed in the CF soil, followed by PF soil (1.10 g cm^{-3}) and AM soil (1.15 g cm^{-3}). In the burnt sites the bulk densities were in the order of 1.01 g cm^{-3}, 1.10 g cm^{-3}, and 1.13 g cm^{-3} for B95, B97, and B98, respectively.

When the natural forest was cleared for certain purposes or underwent disturbances, such as fire, soil bulk density also increased. These increases were due to soil-structure deterioration during the operation of heavy machinery for land clearing and logging. Similar results on Nigerian and Australian soils had also been reported by Ghuman et al. (1991) and Huang et al. (1996), respectively. A study by Matarangan and Kobayashi (1999) showed that increases in soil bulk density of Indonesian soil (Riau and East Kalimantan) was to some extent affected by the frequency of tractor traffics. A significant increase was observed after the first and the second passage, followed by a slight increase after the third to the fifth passage.

Table 2.2. Physical characteristics of soils in the study sites

Site	Bulk density (g cm^{-3})	Moisture content (%)	Saturated water permeability (cm s^{-1})	Three-phase distribution (%)		
				Air	Liquid	Solid
CF	0.99	41.9	9.38×10^{-4}	33.5	44.2	22.3
PF	1.10	36.9	2.65×10^{-4}	6.6	66.6	26.8
AM	1.15	26.7	8.16×10^{-5}	9.6	55.7	34.7
B95	1.01	32.3	2.93×10^{-4}	4.3	58.8	36.9
B97	1.10	32.5	9.61×10^{-5}	2.9	49.9	47.2
B98	1.13	24.6	1.87×10^{-7}	0.0	43.3	56.7
HG	NA	NA	1.85×10^{-6}	8.2	50.9	40.9

NA, not available

Current study did not detect significant difference in soil bulk density among sites. However, in the unburnt sites (CF, PF and AM), the length of vegetation coverage (tree species) appeared to be influential. This was shown by decreasing bulk density of PF and AM soils, approaching the value of forested soil. The turnover of organic matter in these sites seemed to be an important contributing factor. As shown previously, an accumulation of plant debris could be found both on the surface of PF and AM sites (Table 2.1). The importance of organic matter turnover into degraded soils in soil bulk density recovery had been demonstrated both in young temperate-zone soils and in the old, highly weathered soils of the tropics (Nye and Greenland, 1964; Sanchez et al., 1985).

Soil bulk density of CF (0.99 g cm^{-3}) was somewhat higher than those reported earlier (e.g. Fisher, 1995; Huang et al., 1996). Heavy texture (reflected by high clay content) seemed to be the main cause of higher BD in the current study than those appeared in the literatures. In addition, the anthropogenic disturbances, such as the exploration for rattan, resin of *Shorea* sp. and firewoods, and illegal logging, might also have contributed. With time such activities increased as the populations increased, causing more pressure to the forest. However, the values were still lower than those of other sites mainly because those activities were carried out manually whereas in other sites (PF and AM) it was carried out mechanically.

In the burnt sites, soil bulk density also varied according to the length of post-fire period (Table 2.2). The highest bulk density (1.13 g cm^{-3}) was detected in B98. This increase was resulted by decreasing organic volume of the soil due to combustion during fire or to subsequent erosion; whereas the contribution of materials with higher density (mineral) increased. In the current experiment, increasing solid phase was also detected as the natural ecosystem (CF) was converted into plantation (PF and AM) or agricultural land (HG). The effect was even stronger when the sites burnt, as in the B95, B97, and B98. However, with time, the value decreased to 1.10 g cm^{-3} in B97 and 1.01 g cm^{-3} in B95, and approaching the value of CF soil. Interestingly, time required for recovery appeared to be shorter compared with the unburnt sites. When plant biomass (fuel) on soil surface burns, a heat pulse penetrated the soil to certain depth. Increases in soil temperature above normal, beyond the tolerable degree, killed below-ground biomass including roots. Wild fires occurring in the study sites had killed most if not all vegetations. Then the dead roots became a good source of organic matter turnover into the soil. Because of the effect of soil heating, the dead roots decomposed easily, contributing to the earlier replenishment of soil fertility.

Moisture contents were markedly different among the study sites (Table 2.2). In the unburnt sites, the highest moisture content (42%) was recorded in CF soil, followed by PF soil (37%), AM soil (27%), respectively. Naturally, forest soils especially under mature natural forests, in comparison with cultivated soils, generally had high infiltration, as shown by faster water permeability in Table 2.2 (9.38×10^{-4} cm s^{-1}), and relatively low run-off and erosion potential. The water holding capacity was also ensured due to high organic matter content (as will be shown in the subsequent section). In contrast, the evaporation rate was low due to dense canopy closure. As disturbance, for example land clearing and wild fires as the case in current study, occurred, compaction that might adversely affect soil structure and caused changes to soil hydraulic conductivity would result in changes in water distribution. This case might have occurred in PF and AM soils. After land clearing usually the area was not directly planted. When the rain came, the soils easily got saturated and erosion started to occur, exposing subsoil to the surface. The interventions of wildfire caused further degradation of soil quality. Therefore, in the current study the moisture contents of soils in the planted forest (PF and AM) were lower than that of CF soil. In addition, the vertical movement of water in the soil profile, as shown by water permeability was also decreased (Table 2.2).

In the burnt sites, the lowest moisture content (25%) was observed in B98 soil; there was no difference between B97 and B95 soils, which had water content of 32% (Table 2.2). Mechanisms described earlier might have significant contribution to moisture status of these two sites. In addition, open space resulted from wildfire supported the chances of high evaporation to occur. These results confirmed the importance of soil coverage in conserving water in soil because it could reduce evaporation. In addition, accumulation of plant debris on the surface also had significant contribution.

2.3.2.2. Soil Hardness

Soil hardness measurement using a cone-type penetrometer provides valuable information to predict physical hazard for plant growth without causing significant disturbance to the plant stands (Hasegawa et al., 1984; Sakurai et al., 1995). In this study, soil hardness was measured along transect (upper, middle, and lower slopes) and within different times (September 1999/dry season, December 1999/rainy season and March 2000/beginning of dry season) to study the spatial and seasonal variability of soil hardness. However, in CF data for December 1999 were not available because we could not approach this site due to muddy road. Each point of observation was about 15 m apart. Representative vertical distributions of soil hardness in the study area are given in Fig. 2.2.

Although the penetrometer could very well penetrate the 60-cm depth, indicating the absence of gravel or rock layer, the total count at each site and each point was varied. The absence of gravel layer was also confirmed by the observation made on the soil profile (Table 2.1). The highest count of penetrometer (125 counts) was observed in the lower slope of AM, whereas the lowest (36 counts) was observed in the middle slope of B97. The total count in other sites ranged between those two figures.

At the first 10-cm depth, soil hardness showed a similar pattern across space and season. However, the burnt areas had higher counts in the first 10-cm layer than the unburnt areas (Fig. 2.2). These results were in agreement with the results on the bulk density, which was higher in the burnt areas than in the unburnt areas (Table 2.2). The lower counts in HG compared with B98 was due to regular soil tillage carried out in this area prior to crop planting.

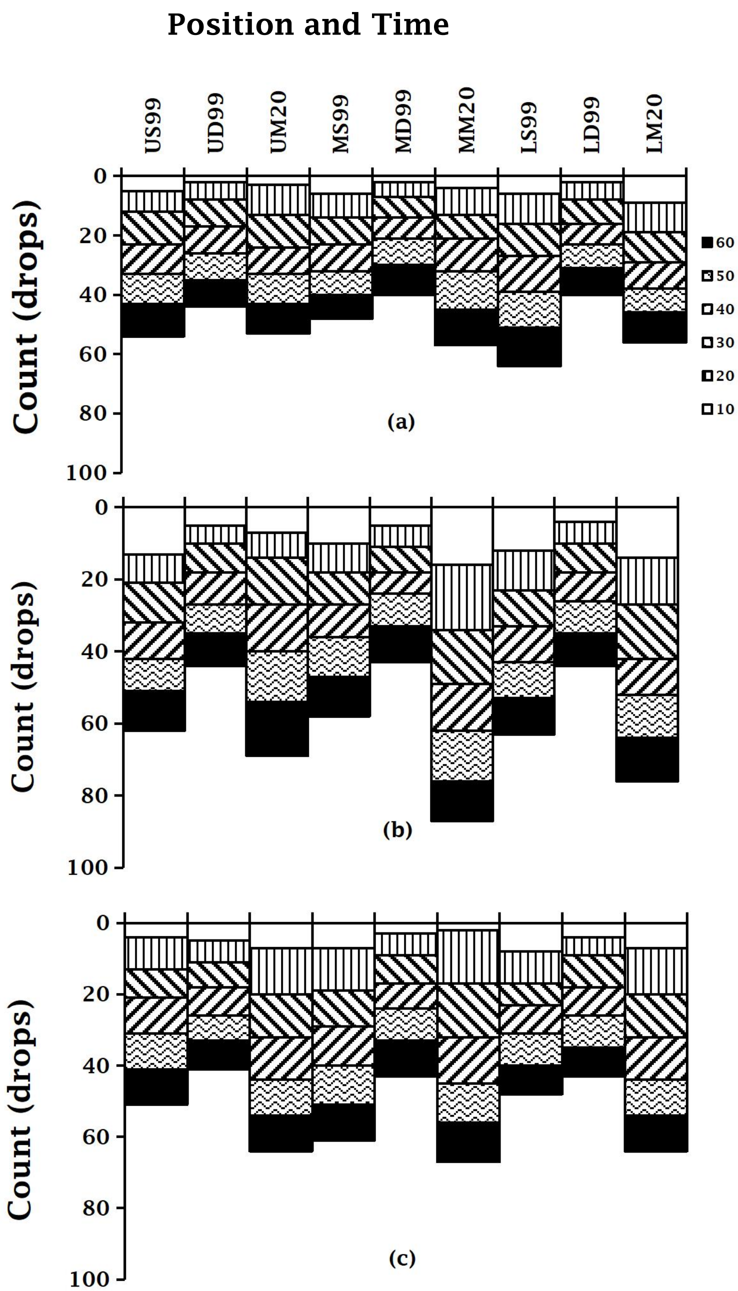

Fig. 2.2. Penetrometer Count

Increasing depth tended to be followed by increasing total count of penetrometer across the study site. Because soil hardness could reflect the presence of clayey materials in lower layers of the pedon, the increasing soil hardness with depth might be attributed to increasing clay content (Table 2.4). However, in current study there was no enough information to support this argument because soil penetrability was calculated with 10-cm increment of soil depth, whereas clay content was measured in each soil horizon, which had different depth. Nevertheless, regression on three-point observation, for example in the AM Area, showed a significant correlation ($r^2 = 0.99$, $P<0.001$) of soil hardness and clay content.

Spatial variability of soil hardness seemed to be season dependent. During dry season (September 1999), the highest count was obtained in the lower part of all sites except CF and HG where the lower slopes showed the least count. In rainy season, the total counts decreased markedly but did not show a great spatial variability (31 to 44), except in B97 where a total count of 69 was recorded. By the end of rainy season (March 2000), the total counts increased but did not show similar pattern across the study site. Based on these results, it was concluded that soil hardness was not a limiting factor for plant growth. It was also confirmed by the presence of roots up to the depth of 150 cm (Table 2.1).

2.3.3. Soil Chemical Properties

The chemical properties of soils in the study sites are given in Table 2.3. Soil acidity ranged from very acid to slightly acid. Release and hydrolysis of Al under strong leaching condition produced acidity in these soils. At low pH, Al is present in the exchange complex and diffuses into soil solution where it may lower the pH and cause toxicity. Higher pH mainly in the upper layer of PF soil (pH=5.73) was most likely due to much higher content of Ca in this soil than in any other soils (Table 2.3). However, only at CF Al saturation was slightly lower compared with other plantation forest and HG. This might be due to disturbances by human management. PF might be affected by the translocation of Al from the surface layer, dissolved by the organic matter accumulated on the surface. The deeper layer of HG might also be affected greatly by the agricultural practices, which enabled water percolation by the cultivation at the surface.

Soils in the burnt sites in general had higher pH than those in the unburnt sites. The ash from plant materials was high in basic cations such as calcium, potassium, and magnesium. Hence, the liming effect of these cations tended to raise the pH. Somewhat higher pH in the HG soil was due to liming applied prior to crop planting. Unfortunately, no definitive rate of liming could be recorded. However, based on the results on the Al saturation (Table 2.3), which was still high, it could be

Table 2.3

assumed that the amount of liming material used was still below the recommended rate. Hence only might be the active acidity neutralized.

In the unburnt site, total C and N contents were markedly different among sites. Permanently covered sites (CF) and the sites free from fire disturbance (PF) had higher C and N contents than the other sites. The highest C content in the surface layer was found in PF soil (6.81%); whereas the lowest was in AM soil (2.08%). As shown previously, the PF site had the thickest organic layer (5 cm), comprising of decomposed, partly decomposed, and fresh pine leaves and fern debris. On the other hand although the thickness of organic layer of AM soil (4 cm) was not different from that of PF soil, the C content was the lowest. Observation in the field showed that organic layer in AM soil mostly consisted of undecomposed *Acacia mangium* leaves, which were bulky but not densely packed.

In the burnt sites, the most recently burnt area (B98) had the highest total C and N in the surface layer (Table 2.3). At 2 yr after fire (B97), however, both total C and N slightly decreased and increased again at 4 yr (B95), approaching the value of CF soil (Table 2.3). These results suggested that in term of total C and N, 4 yr after passage of the fire, the surface soils regained their original organic C and N levels. An extra input from the decomposing dead roots was probably responsible for their recovery.

Vertical distribution of C and N shared similar pattern across the sites. As the soil depth increased, both C and N contents decreased. In accordance with lateral and vertical variability of C and N contents, C/N ratio also varied according to sites but shared similar vertical variability. Toward deeper layers, C/N ratio dropped significantly (r^2=0.72, P<0.001). This occurred because the higher consumption of N than C by microbes at the surface layer.

The amount of exchangeable bases, except Na was much higher in the surface layer than in the subsurface layer. In particular Ca content in surface soil of PF, B98 and HG was much higher than in any other sites (Table 2.3). Several mechanisms could be responsible for the high solubility of Ca in these three sites. In B98, it was possibly due to wild fire occurring prior to sampling. Heating at low temperature up to 100°C had been reported to increase NH_4-extractable Ca, which then decreased at temperature >200 ° C (Kang and Sajjapongse, 1980; Kitur and Frye, 1983).

In addition, the input from the ash might also have contributed significantly. The high exchangeable Ca in HG soil was mainly due to liming to eliminate Al toxicity. The commonly used lime material was $CaMg(CO_3)_2$. Therefore this material could also be expected as a source of both Ca and Mg. However, the liming failed to minimize the solubility of Al in the soil of HG, shown by high Al saturation (30%) in this soil (Table 2.3). The failure was most probably related to insufficient amount of liming materials used. In PF, a different mechanism might have taken place. Pine has been suggested as a species that acts a cation pump (Alban, 1982) that takes up high level of cations from deeper layer and accumulates those cations in the upper layer. The high availability of Ca in the surface layer of PF and AM-b98 caused the dominance of this cation on the exchange complex. This was reflected by the high base saturation in the surface soil of these two sites. On the hand the solubility of Al was restrained, shown by the low Al saturation (Table 2.3).

Cation exchange capacity (CEC) did not show a consistent pattern in all sites (Table 2.1). In general the soils in the study area had low CEC, irrespective of the high clay content. CEC value less than 16 cmol(+) kg^{-1} soil or 10 cmol(+) kg^{-1} soil is one of the criteria to differentiate the low activity clay prevailing in the tropics. Thus, most of the soils in this region can be classified as soils with low activity clay. The CEC/Clay ratio of soil varied in the surface soil but not in the deeper layer. Because the soil across the study site had high clay content, the CEC/Clay ratio was also low except in AM soil (Table 2.3). It suggested that the main source of CEC was the clay. However, because the clay mineral was dominated by kaolinite, the CEC remained low.

Available P was low across the study sites (Table 2.3). However, available P in the surface layer was consistently higher than in the subsurface layer. In the subsurface layer of some soils the available P was even too low to be detected. The low P availability was significantly correlated (r^2=0.53, P=0.002) with high Al solubility in these soils. As solubility of Al was high, P formed Al-P complexes, which were not soluble. As generally acknowledged, the only source of P supply in nature is organic matter decomposition. In current study we also found a significant relationship (r^2=0.67, P<0.001) between the available P and the total C content. Because the total C content decreased with increasing depth, the availability of P also declined with depth.

The results of oxide extraction are given in Table 2.4. Crystalline Al, Si, and Fe (Ald, Fed, and Sid) contents of all soils were higher than amorphous Al, Si, and Fe oxides (Alo, Feo, and Sio). In particular, Fed contents in all soils were significantly higher than that of Feo. Higher values of Ald and Fed indicated the relative accumulation of oxidized Al and Fe associated with strong weathering. The activity ratio of Al (Alo/Ald) ranged from 0.17 to 0.76, while that of Fe (Feo/Fed) ranged from 0.02 to 0.17 (Table 2.4). When compared throughout the study area, permanently covered sites (for example CF and PF) showed higher value of Alo, resulting in higher Alo/Ald, than any other areas. These results were in agreement with those previously reported for Thai soils under para rubber plantation (Sakurai et al., 1996). They further argued that the continuous supply of organic matter might have prevented weathering and leaching of the amorphous forms of these elements. This argument was also in accordance with findings in the current study, which showed that both CF and PF had the highest C among soils examined (Table 2.3).

Soils across the study sites had very low ZPC values, mostly below 4.00 (Table 2.4). The soils under natural vegetation usually showed a lower ZPC value at the surface layer than the subsurface layer due to

Table 2.4.

higher negative charge generated by organic matter at the surface (Sakurai et al., 1989). This was true for most of the soils in the current study. However, PF and AM-b98 showed higher ZPC value at the surface layer than the subsurface layer. This phenomenon could be attributed to the higher amorphous oxides contents (Alo and Feo) and exchangeable Ca and Mg at the surface layer. These characteristics had been known to make ZPC value higher (Sakurai et al., 1989; Sakurai et al., 1996). The increase in Alo and Feo contents was brought by an accumulation of organic matter at the surface layer. Higher Ca and Mg content at the surface layer would be brought as the addition of the ash for B98 and as an accumulation of undecomposed organic matter for PF. Excluding the surface layer of all sites, ZPC values of subsurface layers were in a narrow pH range across the study area, suggesting the weathering status of the soils would be similar. At the deepest layers in each profile, ZPC values were slightly higher than the overlying layers. This could be ascribed to the accumulation of oxides along with the accumulation of clayey materials, as indicated by Sakurai et al. (1989 and 1996).

The values of σ_p ranged from very low to very high (0.33 to 10.00). Only at the surface layers of PF and HG, this value exceeded 6.0. Toward lower horizon, the σ_p decreased. Similar results had also been reported for Thai soils (Sakurai et al., 1989). Since the exchange site of permanent negative charge is mostly occupied by Al when Al content is high, the amounts of basic cations retained by the permanent negative charge are low. Therefore, base saturation is low but Al saturation is high in the

Ultisols. When evaluating the ZPC value by the STPT method (Sakurai et al., 1988), Al dissolution during determination of ZPC did not proceed greatly (Sakurai et al., 1990). Since σ_p is defined as a remaining charge at ZPC, the magnitude of σ_p was closely correlated with the amount of exchangeable bases. Thus the value of σ_p can be an estimate of acting permanent negative charge in the field condition (Sakurai et al., 1988). Hirai et al. (1991) showed a significant correlation between σ_p value and the sum of cations. In addition, the sum of cations (Ca+Mg+K+Na) at the surface layers often exceeded the magnitude of σ_p value (Tables 2.3 and 2.4), suggesting that the cations were existing as free ions or loosely complexed with the organic matter.

Chapter III

Effect of Heating and Rewetting on Properties of Ultisols from South Sumatra Province, Indonesia

3.1. Background

Vegetation fires during prolonged dry seasons in South Sumatra Province of Indonesia have been recorded for almost a hundred years. However, it is only in the last twenty years have they become a regular phenomenon and caused major economic and environmental damage not only within the country but also to its neighbors and to the global climate. In addition, farmers have also been practicing shifting cultivations known as *ladang* since the ancient time. *Ladang* is the felling and burning of the forest followed by the planting of upland rice with other crops for one or two years. In such system, no chemical fertilizers are used. When the crop productivity declines, the fields are abandoned to forest re-growth. The long fallow (20 years) enables the fertility to regenerate that would be available for the next cycle. With increasing population, the fallow periods are shortened, and grasses, mainly *Imperata cylindrica*, which are more prone to fire, invade the areas. Thus, fire pressure has grown even more intense. However, the study on the effects of heating on soil characteristics is still limited although fire has assumed a prominent role in Indonesian forest.

During actual fire the resulting temperature and the duration of the thermal effects in the upper soil might be variable. Severe or high intensity burning is typically found in areas of high fuel concentration, such as fallen logs and slash piles, and may have created surface temperature of 500 ° C (Chandler et al., 1983; Wells et al., 1979). Giovannini and Lucchesi (1997) in an artificial burning experiment detected a temperature above 500°C, 57°C, and 18°C at the depth of 0, 2.5, and 5 cm, respectively. In forest fires and chaparral fires, a temperature of 850°C at the soil surface and slightly lower temperature at upper soil had also been reported (DeBano et al., 1979). Although Giovaninni et al (1988) regarded duration of heating less important because the soil responded according to discrete model to heating, the actual fact in the field frequently showed that fire could persist for hours or days. Kang and Sajjapongse (1980) registered a temperature of 40 to 80°C, which could persist for several hours at the upper soil layer.

Heating caused by fire has been known to have important effects on soil chemical properties and could have beneficial and detrimental effects on soil fertility and plant growth. An increase in soil temperature above normal has been known to alter soil properties, kill roots, seeds, and soil microbes (Hungerfold et al., 1991). These effects depend on the amount of combustible fuel per unit area, the humidity of the fuel and the soil, the type of soil, the temperature level, and the length of heating.

Heating has been reported to decrease organic matter content and CEC (Kitur and Frye, 1983; Giovannini et al., 1990; Giovannini and Lucchesi, 1997; Nishita and Haug, 1972; Sertsu and Sanchez, 1978). Nishita and Haug (1972), Sertsu and Sanchez (1978), and Kitur and Frye (1983) reported a decrease in Ca, Mg, and Na of heated soils. Potassium was reported to either increase (Nishita and Haug, 1972) or decrease (Sertsu and Sanchez, 1978; Kitur and Frye, 1983). Although there has been research regarding natural and prescribed fires in other vegetation types and in different regions as described above, we were unaware, however, of studies that have measured changes in properties of burnt tropical forest soils, which is immediately receiving water treatment.

Since 1998, a study on the changes and possible natural recovery of soils under different ecosystems with varied wildfire history in South Sumatra, Indonesia had been conducted. However, at field scale it was difficult to distinct the possible causes of changes in soil properties because the effects of heat and input of ash were concomitant. To overcome these difficulties, soil samples were subjected to artificial heating under controlled condition. To study possible changes in soil characteristics after rain in the` field, the heated soils were then subjected to watering treatment. In the current chapter, the immediate modifications of soil properties induced by heating and watering are described.

3.2. Materials and Methods

Soil material used in this study was from A horizon of an undisturbed forest soil in Pendopo, South Sumatra, Indonesia. A description of the climate, vegetation, and soil type in the study area was already given in Chapter II. The sampling site can be considered as a representative undisturbed site because the area was under a conservation forest. Bulk samples of the soil were collected and air-dried, and passed through a 2-mm sieve. Selected soil properties are given in Table 3.1.

Table 3.1. Initial properties of the soil used in the experiment

Soil properties	Indonesia Ultisols
pH-H_2O	4.36
pH-KCl	3.61
EC (mS m^{-1})	7.65
T-C (g kg^{-1})	31.21
T-N (g kg^{-1})	2.37
Exch. Ca (cmol(+) kg^{-1})	0.56
Exch. Mg (cmol(+) kg^{-1})	0.30
Exch. K(cmol(+) kg^{-1})	0.27
Exch. Na (cmol(+) kg^{-1})	0.02
CEC (cmol(+) kg $soil^{-1}$)	27.56
Exch. Al (cmol(+) kg^{-1})	3.57
Exch. H (cmol(+) kg^{-1})	0.42
Available P (mg kg^{-1})	12.32
Sand (%)	22.1
Silt (%)	8.1
Clay (%)	69.8
Texture	Clay
Clay mineralogy	Kaolinitic

Soil samples weighing 200 g were placed in shallow porcelain crucibles and heated at 100, 300, and 500 °C in an oven or a muffle furnace for 4, 8, and 12 hours. The thickness of the soil in the porcelain crucibles was about 10 mm. Each treatment consisted of three replications arranged in a completely randomized design (CRD). After cooling, the samples were divided into two compartments. One was stored in plastic bags and tied closely until analyzed; while the other was wetted with deionized water to field capacity and equilibrated in 250-mL plastic cups for one week.

Moist soil colors were determined with a Munsell soil color chart. Particles size distribution was determined by the pipette method using 6 and 30%-H_2O_2 as a decomposing agent of organic matter. Clay minerals were identified by X-ray diffraction after saturation of the clay with K and Mg. Soil pH, exchangeable bases (Ca, Mg, K, and Na), exchangeable Al, total carbon (T-C) and nitrogen (T-N), available P, and CEC of the soils were determined. The analytical methods employed were the same as those described in Chapter II.

All data collected were statistically analyzed with two objectives. The first objective was to compare the effect of the three different temperatures and the three different heating times on the characteristics of heated soils. The Costat V. 2.0 (Cohort) was used to perform a two-way analysis of variance and where significant differences were found at $P<0.05$, LSDs were calculated for mean separation. The second objective was to discriminate the effect of watering treatment on soil characteristics. The paired *t*-test was used to test whether differences between unwatering and watering treatment were statistically significant at $P<0.05$.

3.3. Results and Discussion

Because the soil in the porcelain crucibles was about 10 mm thick, all results obtained were representative of similar thickness of soils in the field. The summary of ANOVA is given in Table 3.2. Main effect of temperature was significant ($P<0.05$) on Fed and highly significant ($P<0.001$) on other parameters. The effect of heating time was significant on T-C, exchangeable Mg, K, Na and Fed, and highly significant on T-N. The interaction between temperature and heating time was significant on T-C and exchangeable Mg and highly significant on T-N and exchangeable K. The main effects of each treatment and the effects of watering on chemical characteristics of the soil are given in Tables 3.4 to 3.6.

3.3.1. Soil Color

Although heating the soil at 100°C did not significantly change the color of the dark Ultisols, it decreased the chroma of soil material by two units (Table 3.3). Visually, the soils heated at 100°C were slightly darker than the unheated soil. The darkening of the soils was due to the

Table 3.2. Statistical significance of the effect of heating temperature and time on the soil properties

Parameters	Treatments		
	Temperature (T)	Time (D)	T x D
pH-H_2O	***	NS	NS
EC	***	NS	NS
T- C	***	*	*
T-N	***	***	***
Available P	***	NS	NS
Exchangeable cations :			
Al	***	NS	NS
H	***	NS	NS
Ca	***	NS	NS
Mg	***	*	*
K	***	*	**
Na	***	*	NS
CEC	***	NS	NS
Amorphous oxides :			
Alo	***	NS	NS
Feo	***	NS	NS
Sio	***	NS	NS
Crystalline oxides :			
Ald	***	NS	NS
Fed	*	*	NS
Sid	***	NS	NS

*, **, *** indicate significance at P<0.05, 0.01, 0.001, respectively.
NS indicates not significant at P<0.05.

Table 3.3. Effect of heating on soil colors

Temperature (°C)	Color
30	10YR3/4 (dark brown)
100	10YR3/3 (dark brown)
300	5YR3/2 (dark reddish brown)
500	10R4/6 (red)

charring of organic matter (Boyer and Dell, 1980; Ulery and Graham, 1993). This mechanism seemed to operate in the current experiment. Heating the soil at 100°C did not burn the organic C of the soil but it was enough to char the organic C, resulting in darker color to the soil. This is also confirmed by lack of changes in C content of soils heated at 100°C relative to unheated soils (Table 3.5).

The effect of heating was much more prominent at 300 and 500°C at which, the color of soil material changed to reddish and red, respectively. The reddening can be attributed to the removal of organic C, the oxidation of Fe compounds, and the dehydroxylation of Fe compounds (Sertsu and Sanchez, 1978). The changes in Fe compounds, such as Fe oxides and significant removal of organic C were also observed in current study, as will be given in the following section. In addition, the thermal conversion of goethite (yellow) to maghemite (reddish) and hematite (red), especially if organic matter is present, which contributes to an O_2-deficient environmenthas also been reported to be responsible for the reddening color (Schwertmann and Fechter, 1984; Ketterings et al., 2000). Maghemite and hematite are highly effective pigmenting agents, imparting reddish brown and red colors, respectively, even when present in very small quantities (Schwertmann and Taylor, 1989). Scheffer et al. (1958) and Torrent et al. (1983) stated that only 1 to 1.7% of hematite could give a soil a red color.

3.3.2. Particle Size, Mineralogy and Oxide

The soil used in the current study was initially high in clay (69.3%) but low in sand (22.1%) prior to heating. Exposing the soil to 100°C had little effect on the particle-size distribution and did not change the textural class of the soil (Fig. 3.1a). At 300°C, the laterization process occurred, causing a significant increase of sand (from 22.1% to 27.2%) but a significant decrease of clay (from 69.3% to 51.5%), which then resulted in coarser textural class. These changes were caused by the fusion of clay particles into sand-sized particles as a result of thermal modification of the iron and aluminosilicate (Betremieux et al., 1960), and were in a good agreement with those obtained in a laboratory study by Sertsu and Sanchez (1978), and a field study in Chapter II.

The length of heating above-normal temperatures appeared to be crucial to soils. Although exposing the soil to high temperature for 4 hr had little effect on silt fraction seemingly, this length seemed to be critical time for changes in clay and sand fractions. During the first four hours, clay fraction decreased by 35% (from 69.3% to 44.9%), while sand fraction increased by 115% (from 22.1% to 47.6%) (Fig. 3.1b). A further increase in heating time caused further decreases in clay fraction, increases in silt fraction, but did not alter the sand fraction. These results implied that the fusion of clay particles into sand-sized particles mostly occurred during the first 4 hr of continuous exposure to elevated temperatures. Therefore, in addition to heat pulse penetration produced by burning, the length of a fire persisting in the field is also a crucial factor determining the total effect of fire on physical properties of soils.

Mineralogically, kaolinite was a predominant (94%) clay mineral found in the soil, while the others existed in much smaller proportion

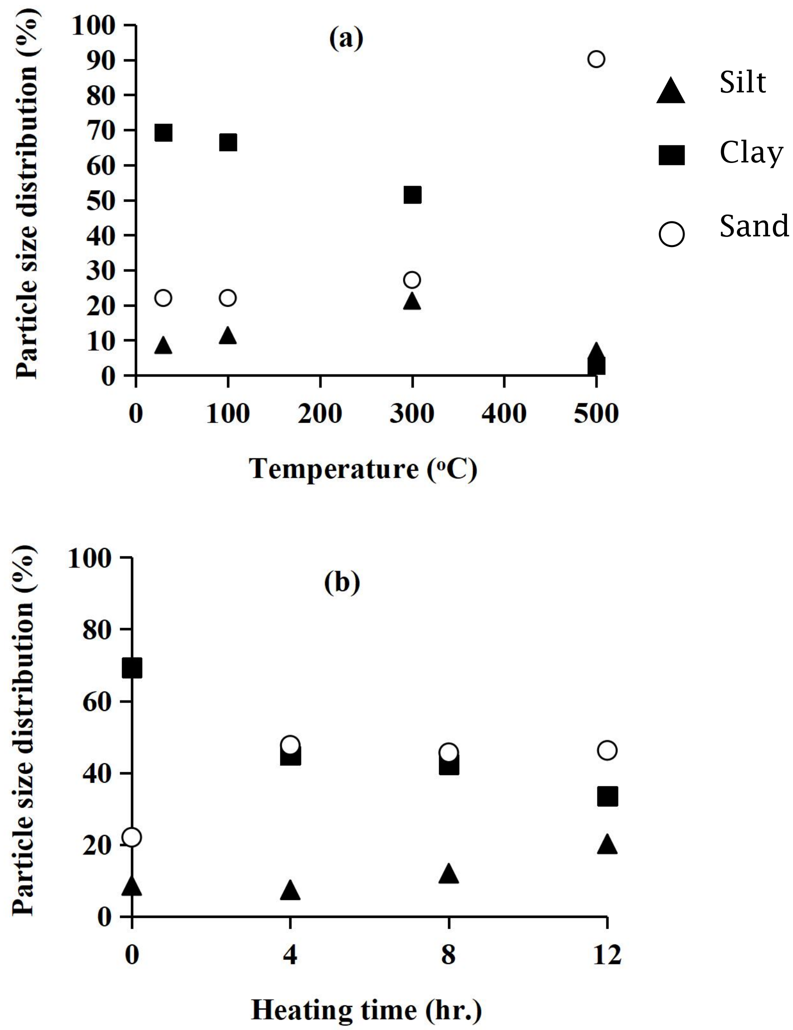

Fig. 3.1. Effect of temperature (a) and heating time (b) on particle distribution of soils.

prior to heating treatment. The weakening of diagnostic XRD peaks for kaolinite was observable at 100 °C, gradually decreased at 300 °C, and completely disappeared at 500 ° C. Unfortunately changes in other minerals, especially those exist in much lower proportion, could not be further detected with XRD technique. However, our results showed that soil heating at low temperature (100°C) was enough to cause significant changes in soil mineralogy, which then might have significant effect on other characteristics of soils.

Heating significantly affected Al, Fe and Si oxides of the soils but the effect of heating time was significant only on Fed oxides (Table 3.4). Exposing the soils to 100 ° C did not cause marked changes in oxides status of the soils. Given the fact that the content of oxides has been known closely related with organic matter content (Sakurai et al., 1996), the release of Alo, Feo, and Sio did not occur at 100°C because at this temperature, the organic C was charring, instead of being combusted. It was confirmed by lowering chroma, resulting in darker soils. In addition, total C of soils heated at 100°C was not different from the unheated soils, as will be shown in the following section. Similarly, no changes in Ald, Fed, and Sid could be observed at 100°C because at this temperature, the oxidation of oxide compounds, the removal of organic C, and the dehydroxylation of oxide compounds as described by Sertsu and Sanchez (1978) did not occur yet. However, increasing temperature to 300 and 500°C significantly increased the amount of oxides of Al, Fe, and Si of soils. Combustion of organic C released the Alo, Feo, and Sio of the soil. In addition, increasing temperature also caused thermal modification of iron and aluminosilicate, releasing of Ald, Fed, and Sid. The decomposition of poorly crystalline aluminosilicates and amorphous Si and Al from kaolin during burning has been found to contribute to the formation of sand-sized aggregates in severely burnt soils (Ulery and Graham, 1993).

In the current experiment, consistent effects of watering on oxides of the heated soils were not detected (Table 3.4). Intense heating has been shown to induce water repellency in soils (DeBano et al., 1976; Giovannini and Lucchesi, 1983). The effect of watering on oxides was significant only on Alo of soils heated at 500°C, Sio of soils heated at 100°C and 500°C and on Sid of soils heated at 100°C and 300°C (Table 3.4). The losses of organic matter and thermal modification of the iron and aluminosilicate seemed to be the main cause.

Exposing the soil to high temperatures for 4 hr increased levels of soil oxides (Table 3.4). It confirmed that the changes in oxides might have occurred during the first 4 hr of heating.

3.3.3. Soil Total Carbon and Nitrogen

Resulting temperature and heating time are two important factors determining the effects of fire on soil characteristics (Kang and Sajjapongse, 1980). Artificial heating in the current experiment showed that temperature and heating time significantly affected both T-C and T-N. There was no change of T-C at 100°C as compared with unheated soil but at 300 and 500°C a significant decrease occurred (Table 3.5). The lack of changes in T-C content of the soil at 100°C was likely related to

Table 3.4.

Table 3.5.

the initial stage of combustion operating at this temperature. As mentioned by Sandberg et al. (1979) and Pyne et al. (1996), at pre-ignition stage of a fire, both dehydration and pyrolysis occurred, at which moisture in the soil was driven out of the soil, and to some extent could cause decomposition of long-chain organic molecules (Shafizadeh, 1984). Pyrolysis causes formation of char, which is a carbonaceous residue left in the soil that is neither intact organic compound nor pure C. In the current study we found a shift in color toward darker color of soils exposed to 100 ° C (Table 3.2), which is usually related with charring organic matter. Organic C in the soil is usually destructively distilled between 200 and 300 ° C (Hungerfold et al., 1991), and almost totally combusted at 500 ° C (Hosking, 1938). Current results showed that heating at 300 and 500°C resulted in losses of 78% and 99% of the soil T-C, respectively. These figures were in a good agreement with those reported by Hosking (1938) and DeBano et al. (1998).

The temperature and length of heating combination is a crucial factor. Although heating the soil at 100°C did not change T-C and T-N content of the soil, increasing heating temperature from 100°C to 500°C was followed by significant decrease in T-C and T-N of soil (Fig. 3.2). The results of current experiment indicated that the temperature of 300 ° C appeared to be crucial. At this temperature, a significant decrease of T-C and T-N was already observable 4 hours after exposure (Fig. 3.2). At 500°C, almost total elimination of T-C and T-N occurred regardless of the

Fig. 3.2. Effect of watering on total C and N

duration (Fig. 3.2) because at this temperature, most if not all of organic matter, which was the main reserve of C and N, was already combusted.

When treated with water, the T-C of soil exposed to 100°C within different times did not change. Although soils exposed to 300°C and received water treatment tended to have higher T-C content than those exposed to the same temperature but did not receive water treatment, significant difference was detected only on soils exposed to 300°C for 8 hr. However, soils treated 500°C and received water treatment had T-C content of 5 to 7.5 times higher after receiving water treatment and statistically different from those did not receive water treatment (Fig. 3.2). On the other hand, although watering increased T-N of soil at all temperature levels, the difference was not enough to be statistically different from unwatered soils (Fig. 3.2). The source of both C and N turnover in current experiment was not apparent. However, possible sources included dissolved C and N in water and microbial cells.

3.3.4. Soil pH, EC, Aluminum, and Hydrogen Levels

Heating significantly affected soil pH, EC, and solubility of exchangeable Al and H of the soil. When the soil was heated at 100°C, there was a slight decrease of soil pH as compared with the unheated soil, followed by a significant increase at 300°C and a significant decrease at 500°C (Table 3.5). The cause of decrease in pH at 100°C was most likely due the increasing solubility of H ions. Similar results had also been reported by Kitur and Frye (1983) who argued that the decreases in pH might be due to organic acids being released more rapidly than base elements at this temperature. In a field experiment Blank et al. (1994) also found elevated amount of organic acids immediately after wildfire in the surface 5 cm of under-shrub soils. As described earlier, during dehydration and pyrolysis at the preignition stage of a fire, decomposition of long-chain organic molecules also occurred (Shafizadeh, 1984; Blank et al., 1994). In addition, the lowering of buffer capacity due to combustion of organic C and the exposure of new surfaces were other possible mechanisms contributing to the lowering pH (Giovannini and Lucchesi, 1997; Nishita and Haug, 1972). Johnson (1916) suggested that the lowering pH could also be attributed to an increase in CO_2 with heating, but it seems unlikely that H_2CO_3 could persist to an appreciable extent at 100°C. On the other hand, the increasing pH at higher temperatures might be attributed to significant losses or precipitation of Al and H ions due to liming effect of heating (Table 3.5), the formation of oxides of several elements (Nishita and Haug, 1972), and release of NH_3 (Kitur and Frye, 1983).

Watering increased the pH of the soils heated at each temperature (Table 3.5). This increase was in accordance with decreases in soluble Al and H in the heated soils receiving water treatment (Table 3.5). Watering had caused precipitation of Al and H in the heated soils, most likely mediated by carbonates. In contrast, the solubility of exchangeable bases increased as will be given in the following section.

Electrical conductivity, taken as a measure of soluble salts, was significantly affected ($p<0.001$) by the temperature but not by the heating time (Table 3.3). The EC of soil heated at 100°C doubled as compared with the unheated soil, then decreased significantly above this temperature (Table 3.5). The large increase in the EC at 100°C might have been due to the increase in Ca and Na of the soil at this temperature. When regressed on the exchangeable bases, the EC was significantly correlated ($p<0.001$) with the exchangeable Ca, Mg and Na but not with the exchangeable K. On the other hand the following decrease at higher temperature would be due to significant decreases of all exchangeable bases (Table 3.6)

When the heated soils were treated with water, the EC of soil heated at 100ºC significantly decreased but the EC of soils heated at 300ºC in contrast significantly increased (Table 3.5). The cause of decreased EC in soil heated at 100°C was not apparent because all exchangeable bases but Ca significantly increased after watering (Table 3.6). However, an increase in water-soluble SO_4^{2-} at 110 and 200°C as described by Kitur and Frye (1983) might have occurred in the current experiment. On the other hand, significant increases in EC of heated soil and receiving water treatment were in accordance with the increasing all exchangeable bases observed in the current study.

3.3.5. Exchangeable Bases and CEC

Heating significantly affected exchangeable bases of forest soil (Table 3.6). Exchangeable Ca, K, and Na increased as the soil was heated up to 300°C. Such increases had also been indicated in previous reports (Nishita and Haug, 1972; Kitur and Frye, 1983; Giovannini et al., 1990; Ketterings et al., 2000) but none reported the possible mechanisms responsible for such increases. However Kim et al. (1999) related the increase with the accretion of ash from burning of surface organic matter. Although soils used in the current experiment were visually free from plant residue, the initial T-C content was 31 g kg^{-1} (Table 3.5). The decomposition of this C source due to heating might have contributed to the increases in exchangeable bases.

Exchangeable bases significantly decreased at 500 ° C (Table 3.6). Such a decrease may have been due to the formation of insoluble oxides and carbonate of these elements, as suggested by Sertsu and Sanchez (1978). In addition to losses of organic C, the decrease was also related with decreases in clay fraction of the soils. The aggregation of fine particle into sand-sized particles caused a decrease in the surface area in

Table 3.6.

contact with the NH_4-OAc. Therefore the exchangeable bases remained trapped inside the aggregates and not be extracted.

Exchangeable Mg behaved differently from the other bases. This cation significantly decreased with increasing temperature. It suggested that the formation on insoluble oxides and carbonates of this element as suggested by Sertzu and Sanchez (1978) occurred at lower temperature than that for Ca, K, and Na.

When the heated soils were watered, exchangeable bases significantly increased at all temperature levels, compared with the unwatered soils (Table 3.6). As the water was added, the oxides and mainly carbonate of the exchangeable bases (Sertsu and Sanchez, 1978) might have been dissolved and released into soil solution. Although the exchangeable bases except Ca were significantly affected by the heating time of a soil to different temperatures (Table 3.3), the differences within the times were not statistically different. If the mechanisms suggested by Sertsu and Sanchez (1978) above occurred, the results obtained in Table 3.6 implied that most of oxides and carbonates of the exchangeable bases formed in the first 4 hours of heating.

Cation exchange capacity significantly decreased with increasing temperature but not with increasing time (Table 3.6). The combustion of organic matter was contributing to the decreased CEC following heating in the current experiment. The data showed that CEC was significantly correlated ($p<0.001$) with T-C content (CEC= 16.337 + 0.3359T-C). Due to the combustion of C, most if not all of the exchange sites had been disrupted, leaving the clay as the only possible source of CEC. However, the clay fraction of the soils also dropped significantly at these temperatures (Fig. 3.1). The aggregation of finer particles into sand-sized particles at higher temperature with the consequent decrease of the reactive surface area (Giovannini and Lucchesi, 1997), and dehydration of the mineral crystal lattice and the resulting breakdown of the lattice (Giovannini et al., 1990) had caused further reduction of the CEC at higher temperatures.

In the current study, a significant relationship between CEC and amorphous oxides represented as Alo ($P<0.001$), Feo ($P<0.005$), and Sio ($P<0.005$) and crystalline oxides represented as Ald ($P<0.005$) and Sid ($P<0.005$) was also found. The increases in these oxides due to heating were followed by decreases in CEC of the soils. Heating has enhanced the release of these elements through the disruption of mineralogical properties of the soils. However, because the CEC of Al and Fe oxides is low, the increases in their content adversely decrease the CEC of the soil.

3.3.6. Available P

The available phosphorus significantly decreased with increasing temperature (Table 3.6). This result was in contradictory with those reported in the previous studies (Giovannini et al., 1990; Sertsu and Sanchez, 1978). Giovannini et al. (1990) reported an increase in available P from 5 to 92 mg P kg^{-1} with increasing temperature from 25 to 460°C, while Sertsu and Sanchez (1978) reported an increase in available P from below to above the critical level of 10 mg P kg^{-1}. They attributed this increase to the decomposition of organic matter on the soil surface. It suggested that the increase in P they observed did not present the actual changes in available P in soil. In addition, they did not consider the losses of P through nonparticulate transfer, which could account for 60%, when organic matter is totally combusted (Raison et al., 1985). In the current experiment, the soil used was free from plant residue and the available P significantly decreased from 12.32 to 1.69 mg P kg^{-1} with increasing temperature from no heating treatment to 500°C. Therefore, the only possible source of P was T-C pool in the soil. However, instead of gaining available P decreased as temperatures increased. Although the losses of P were negligible at 100°C as compared with pre-heating level, about 62 and 86% of P lost at 300 and 500°C, consecutively (Table 6). These losses were corresponding with 78 and 99% loss of organic C at the same temperatures as compared with the pre-heating level (Table 3.5). Statistically the reduction in available P was significantly correlated with the loss of T-C from the heated soils (Available P=2.0625 + 0.3162T-C, $p<0.001$).

Heating time did not significantly affect available P of the soils (Table 3.3). However, exposing the soils to high temperature continuously for 4 hr caused 52% reduction of available P but further increase in the heating time did not cause any changes in P status of the soil (Table 3.6). These results suggested that the losses of P from the soil in relation to heating occurred during the first 4 hr. It once again confirmed that significant changes in soil characteristics due to elevating temperatures occurred within 4 hr.

Water treatment decreased available P of heated soil (Table 3.6). Under acidic condition, as found in the current study, P is immobilized in insoluble compounds. Williams et al. (1958) and Williams (1960) found that the P sorption capacity of acid Scottish soils was closely related to the Alo. In the current experiment, the oxides of Fe, Al, and Si significantly increased as the soils were heated. The decreases in available P were significantly correlated with increases in Alo ($P<0.001$), Feo ($P<0.001$), and Sio ($P<0.001$) and Ald ($P<0.001$) and Sdi ($P<0.001$). With the addition of water, the chemical reactions involved in the formation of insoluble complex of Al- and Fe-P were insured, as shown by lower available P in soils receiving water treatment than those not receiving water treatment (Table 3.6).

3.4. Practical Implication

Darkened soils that resulted from low- and medium-severity burns are preferred for agricultural productivity by farmers in Sumatra, Indonesia, because they were associated with higher crop yield and crop establishment. On the other hand, areas of reddened topsoil were classified as undesirable due to their perceived low fertility status and poor water-holding capacity (Ketterings, 1999). From the results obtained it is clear that heating the soils at 100 °C for a short period, which could be regarded as *light* burning usually practiced by farmers in the field, did not cause significant changes in soil chemical characteristics. However, cautions must be given because increases in sand fraction but decreases in clay fraction started occur at this temperature. At field scale such increase would have more important implication because it can increase soil erodibility. In contrast, heating at 300 and 500°C, a representative of *intensive* burning for longer periods, for example during forest fires, adversely affected soil characteristics. In the current experiment it was found that changes in all characteristics occurred during the first four hours.

Watering to some extent showed positive effects on chemical characteristics, except available P of soils. On the other hand nutrient losses through leaching and/or erosion could concomitantly occur due to direct expose of the soils to precipitation and increases in erodibility. In the field, burning and watering of the surface soils occur repeatedly. In other words, intensive oxidation and reduction occur alternatively, which cause more weathering and degradation of soils. To protect land and biological resources in the tropics, prevention of forest fire is one of the most important practices.

Chapter IV

Characteristics of Ultisols under Different Wildfire History in South Sumatra, Indonesia: Dynamics of Chemical Properties

4.1. Background

Tropical rain forests have been regarded as ecosystems in which natural fire is excluded by the prevailing moist environment and the resulting unsuitable conditions of all materials (fuels) for burning. However, recent changes in climatic conditions, which contribute to changes in fuel characteristics and flammability of rain forests have favored the occurrence of natural and anthropogenic fires. Furthermore, pressures on tropical lands are also rapidly increasing, causing overall degradation, and conversion of rainforest to pyrophytic life forms with increased flammability and fire frequency (Mueller-Dumbois and Goldammer 1990).

Forest fires during the El Niño-Southern Oscillation (ENSO)-related prolonged dry season in South Sumatra, Indonesia have been recorded for almost a hundred years. More frequent droughts and the irresponsible use of land and natural resources have greatly exacerbated the fire problems over the last twenty years (Bompard and Guizol 1999). Widespread logging using flawed technique, large-scale land clearance by agro-industrial companies, land clearance for major transmigration schemes, and land acquisition by companies and government with little consideration for the rights of local people are contributing factors to the increased fire occurrence in South Sumatra Province, Indonesia (Gouyon 1999).

Removal of forest by fire can result in immediate and drastic degradative changes in soil properties. The risks of fire on forest soils include (i) nutrient losses in smoke, ash, runoff and erosion (Ewel et al. 1981), (ii) a short-term improved availability of nutrients (Ghuman and Lal 1991), (iii) increased soil erosion due to decreased soil stability (Ghuman et al. 1991), and (iv) changes in soil physical and mineralogical properties (Ketterings et al. 2000). While the soil of a forest ecosystem is one of important components in sustainable nutrient cycling, much less work has been carried out to assess the leftover effects of fire. Therefore, little information is known about the long-term effects of fire on forest soil properties. The loss of nutrients from forest sites by the removal of trees appears to be particularly important. Measuring the recovery of ecosystem nutrient capital to pre-disturbance levels provides an assessment of biogeochemical resilience and may also facilitate cross-biome comparisons of ecosystem recovery rate (Zarin and Johnson 1995). The objectives of this study were: (i) to investigate the effect of forest management practices on total carbon and nitrogen, available phosphorus, and exchangeable bases; and (ii) to investigate both the immediate and the leftover effects of forest fires on the total carbon and nitrogen, available phosphorus, and exchangeable bases of Ultisols in South Sumatra Province, Indonesia.

4.2. Materials and Methods

4.2.1. Study Site

A detailed description of the study sites has been given in Chapter II. In order to study the immediate effect of wildfire on soil characteristics, a most recently burnt *Acacia mangium* plantation (B00) and an unburnt *Acacia mangium* plantation adjacent to the burnt plantation were also sampled.

4.2.2. Soil Samples and Analytical Methods

Soil samples were collected every 3 months from December 1998 through December 2000 from 0- to 10-cm depth at approximately adjacent points in three replicates per site. Each replicate consisted of four sub-samples, which were combined to get one composited sample for each replicate. To investigate the immediate effect of wildfire, an unburnt area adjacent the recently burnt site (B00) was also sampled as a control. Soil pH(H_2O), exchangeable bases (Ca, Mg, K, and Na), Al, total carbon (T-C) and nitrogen (T-N), available P, and CEC of the soils were determined. The analytical methods employed here were the same as those described previously in Chapter II.

All data collected were statistically analyzed to discriminate the effect of ecosystem types and the immediate effect of fire on soil characteristics. Analysis of variance (ANOVA) was carried out to compare the effect of ecosystem types and the effect of years of fire on soil characteristics. The Costat V. 2.0 (Cohort) was used to perform a one-way analysis of variance and where significant differences were found at $P<0.05$, LSDs were calculated for mean separation. The paired *t*-test was used to test whether differences in nutrients between pre-fire and after fire were statistically significant at $P<0.05$.

4.3. Results and Discussion

4.3.1. Natural Forest vs. Human-managed Ecosystem

In order to follow the dynamics of characteristics of soils under different ecosystems during 2 yr, T-C, T-N, available P, exchangeable bases, and CEC of surface soil were plotted against time of sampling as given in Figs. 4.1 and 4.2. Despite the intra-period variability, certain

Fig. 4.1.

Fig. 4.2.

patterns in the reported soil characteristics can be summarized as follows.

Although total C and N measurements were limited to total pools in the soil, the results indicated that the ecosystem studied was resilient with respect to C and N (Figs. 4.1a and b). Although differences in vegetative composition among these three sites might have resulted in differences in net primary production and nutrient sequestration in biomass, response of surface soil C and N for the forested sites (CF and PF) and home garden (HG) during 2-yr monitoring showed similar pattern. A significant trend toward increasing or decreasing was not shown statistically (Figs. 4.1a and b). Forest or forest fallow, for example in CF and PF sites, have very tight nutrient cycles. The decomposing litter layers are intensely perfused with fine roots and mycorrhizal hyphae, so nutrients are captured as they are mineralized, reducing leaching losses to near zero. The results in the HG sites to some extent contradicted with those reported by Tanaka et al. (1997), who found a decrease in organic C with increasing cultivation period for Thai soils where there was no organic input except from crop residue and sometime fire invaded from the surroundings or farmers burnt the sites for weed control. Whereas T-C and T-N in soil of HG in the current study could be maintained because organic debris after harvest or litterfall has been regularly incorporated into the soil, and burning has not been performed since the area was established in 1997.

The intra-periodic variability observed in T-C and T-N resulted from the differences in vegetation types, which may have resulted in differences in net primary production, nutrient sequestration in biomass, and turnover of soil C and N. The critical role of N fixation in primary succession sequence has been reported for a wide range of ecosystems (Knoepp and Swank 1997).

The CEC showed significant ($p<0.01$) fluctuations during the 2-yr-period of the study, ranging from 11.15 to 31.30 cmol(+) kg^{-1} soil in CF, 14.61 to 38.83 cmol(+) kg^{-1} soil in PF, and 10.96 to 28.18 cmol(+) kg^{-1} soil in HG, respectively (Fig. 4.2c). Given the fact that the permanent negative charges in soil remain unchanged and the clay fraction is primarily dominated by kaolinite, the wide variability in the CEC reflects the contribution of variable negative charges generated by deprotonation of hydroxide groups on organic matter. This is supported by the fact that the CEC values were higher than the sum of exchangeable cations (Ca, Mg, K, Na, and Al). The significant intra-periodic variability in the T-C and the pH as shown in Figs. 4.1a and 4.2a supported this possibility. There was an indication that the fluctuation in the CEC was significantly correlated ($p<0.05$) with the fluctuation in the organic C except in the PF site (Fig. 4.3). Lack of significant correlation in the PF site indicates that the high content of exchangeable bases mainly Ca and Mg in the surface soil of PF masked the role of organic matter in determining the CEC of the soil in this site. The dynamics of soil acidity and exchangeable bases in these three sites differs considerably. Both pH and exchangeable

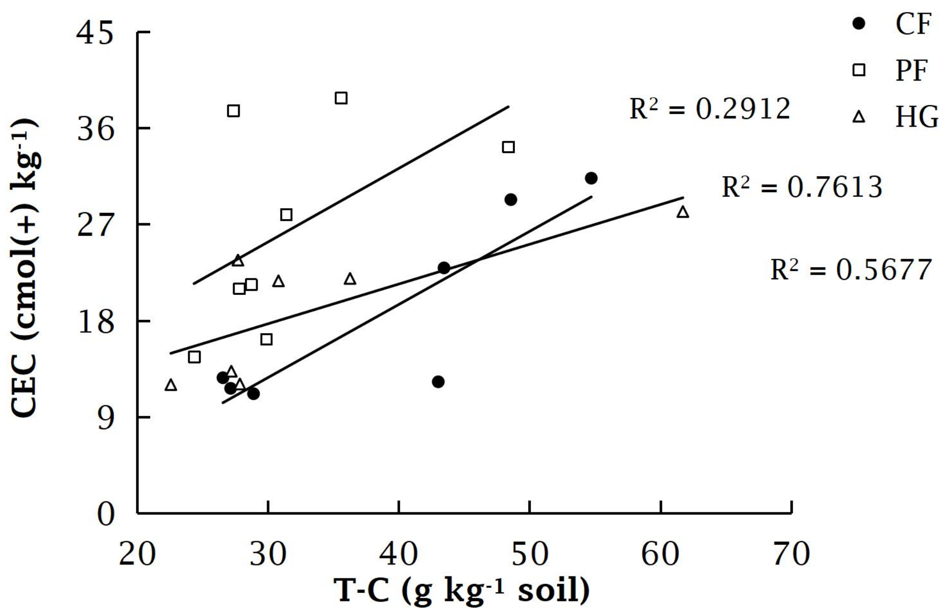

Fig. 4.3. Relationship between CEC and total C of surface soils (0 – 10 cm) from the natural forest and human-managed ecosystem.

bases in CF did not show significant fluctuation (Figs. 4.2a and b). In PF, the pH was still aggrading due to accumulation of Ca and Mg (Figs. 4.2a and b). On the other hand, the pH in HG fluctuated according to planting season, increasing during the growing season (usually September of the year to March of the following year) and decreasing after harvest (May to June of the year) (Fig. 4.2a). The increases in the soil pH occurred due to liming carried out prior to crop planting, usually at the end of August to the beginning of September of the year. The important objective of liming Ultisols is to eliminate the Al toxicity that impairs root development and consequently reduces nutrient and water uptake by the plants. Therefore the usual liming recommendations aim at applying as much as is needed to eliminate the Al toxicity in the surface layers. Hence the amount depends on the solubility of Al in the soils. The commonly used recommendation for Indonesian Ultisols is one time equivalent to exchangeable Al. Because the surface soil of the HG site contained 3.2 cmol(+) kg^{-1} of Al in September 1999, the amount of liming material needed was 3.2 x 10^3 kg $CaCO_3$ ha^{-1}. As the farmer could not afford large investments, the amount used was only about 25 kg $CaCO_3$ ha^{-1}, which was still far below the recommended rate for agricultural purposes. Although the Al saturation in this site remained high (around 30%) an increase in soil pH in the subsequent period (September 1999 through March 2000) could still be observed (Fig. 4.2a), most likely due to the neutralization of the active acidity of the soil. After harvesting the effects of liming on the soil pH diminished, as shown by the decreased pH in June 2000 (Fig. 4.2 a). This decrease coincided with the decreases in the exchangeable Ca and Mg in the same period (Fig. 4.2b). Absorption by plants and vertical leaching of these two cations are considered to be responsible for such depletion. It has been shown in Chapter II that subsoil of HG had reasonably high content of Ca, Mg, and K.

In spite of the considerable fluctuation, available P in all sites was still increasing with time (Fig. 4.1c). As generally acknowledged, the main source of P supply is organic matter decomposition. In addition, the dissolution of inorganic P (Al-P and Fe-P) by mycorrhizal fungi and P-solubilizing bacteria P might also be important in the study sites. This was supported by the abundant inoculum potentials of indigenous vesicular arbuscular mycorrhizal fungi and P-solubilizing bacteria in the study sites, as will be given in Chapter V.

Forest productivity is directly related to soil organic matter content because decomposition of organic matter provides significant amounts of nutrients for plant uptake. Soil organic matter content can change due to normal site process with time such as succession and biomass accumulation or due to anthropogenic manipulations such as species conversion or harvesting. Because vegetation and soils are inextricably linked in forest ecosystem, forest management practices directly affect vegetation and may also change soil organic matter and nutrient pools in the soil. Considerable fluctuations in all soil characteristics across the study sites reflect the significant spatial and temporal dynamics of soil nutrients despite the similarity in soil parent material and in general climatic patterns. Differences in vegetation types might have resulted in differences in net primary production and nutrient sequestration in biomass. Similarly management practices, i.e. crop planting and harvesting in HG, might have caused seasonal effect on fine root production and turnover of soil nutrients.

4.3.2. Immediate Dynamics of Soil Properties after Fire

Because the current study was carried out under wildfire conditions, temperatures during fire could not be measured. To evaluate the immediate effects of wildfire on soil properties, a comparison was made on the results in the most recently burnt site (B00) and the adjacent unburnt site. Fire consumed most of 9-yr old *Acacia* trees and all litter layer. As a result, ash accumulation ranging from 1 to 2 cm thick and soil cracks could be found on the soil surface immediately after the wildfire. Chemical analysis of fire-exposed mineral horizon (0- to 10-cm depth) immediately after fire found significant increases ($p<0.05$) in T-C, T-N, available P, pH, and exchangeable bases. In contrast Al saturation and CEC significantly ($p<0.05$) depleted (Table 4.1).

Table 4.1. Pools and changes in chemical properties of surface soils (0 - 10 cm) immediately after fire occurrence

Variable	Before fire	After fire
T-C (g kg^{-1})	21.3 a	22.6 b
T-N (g kg^{-1})	2.0 a	2.2 b
Available P (mg P kg^{-1})	14.3 a	19.8 b
pH	4.9 a	5.4 b
Al saturation (%)	40.4 a	17.7 b
Ex. bases (cmol(+) kg^{-1}):		
Ca	1.9 a	9.0 b
Mg	1.8 a	3.4 b
K	0.3 a	0.3 a
CEC (cmol(+) kg^{-1})	16.9 a	12.0 b

Means within the same rows followed by the same letters are not significantly different at $p<0.05$ (based on *paired-t test*). Values are means for $n=3$.

The increases in T-C from 21.3 to 22.6 g kg^{-1} (6%) and T-N from 2.0 to 2.2 g kg^{-1} (10%) immediately after fire were related mainly with the increases in charring materials produced during pyrolitic stage of burning. Similar results had been reported both in the laboratory (Chapter III) and in the field studies (Gimeno-García et al. 2000). As described in Chapter III, T-C and T-N slightly increased at 100ºC, destructively distilled at 300ºC, and almost totally combusted at 500ºC. In a field experiment using 20 and 40 t biomass ha^{-1}, Gimeno-García et al. (2000) found and increase of 24% and 5% of T-C and T-N of surface soils (0 to 5 cm) and recorded surface temperatures between 200 and 400ºC, respectively. On the other hand, a small decrease (2%) in organic matter and T-N was found in soils from the plots suffering a high intensity fire (400 and 600ºC). Therefore the current results implied that the temperatures during fire were likely still lower than the temperatures at which T-C and T-N combustion could occur. Fire intensity and temperatures are related to the amount and condition of fuel. Severe or high-intensity burns are typically found in areas of high fuel concentration, such as fallen logs and slash piles, and may have created surface temperatures of 500°C or more (Wells et al. 1979; Chandler et al. 1983). The concentrated fuel may burn slowly, over several days or weeks, baking the underlying soil at high temperatures and resulting in a reddened surface soil (Ulery and Graham 1993). In the current study, the conditions described by Wells et al. (1979) and Chandler et al. (1983) were not found because the field was a 9-yr old *Acacia mangium* stands with sparse floor growth underneath. Although organic debris (about 2 cm thick) existed, it burnt quickly during fire due to the dry climatic condition when the fire occurred. In addition reddening of surface soil as described by Ulery and Graham (1993) did not occur in the current study. Therefore, it is concluded that the increases in T-C and T-N immediately after fire in the current study were related mainly with the increases in charring materials produced during pyrolitic stage of burning. Charcoal produced during burning might have been also included at sampling and estimated as soil materials at analysis.

Soil moisture content is also important in determining the effect of fire on soil properties as water can increase both the heat capacity and conductivity of soil (DeBano et al. 1998). Soil moisture content at 0- to 10-cm depth in the study site was about 28% prior to fire and dropped to about 20% after fire. The moist condition of soils prior to fire allowed the soils to absorb large amounts of heat without a subsequent increase in temperature until water was vaporized and to conduct heat more rapidly due to the increased thermal conductivity. However, severe effects of fire on soil drying hence on other chemical properties might have occurred in the first 1- to 2-cm depth where soil cracking was found. Soil cracking found in the first 1- to 2-cm depth of the soil surface indicated that this depth had suffered dehydration and the temperature during fire might have reached 220ºC (Giovannini and Lucchesi 1993; Raison 1979). Therefore it was most likely that the depletions in T-C and T-N occurred in the first 1 to 2 cm of soil surface but the 10-cm sampling carried out in the current study did not permit the detection of these potential changes. In addition, some researchers considered that litterfall and other organic material are added as charcoal during burning and estimated as soil material.

Significant increases in the soil pH and exchangeable bases were attributed to the accumulation of ash from burning of plant biomass, plant litters on soil surface, and to some extent charring of organic C pools in the soil, as has been shown in the previous studies (Ketterings et al. 2000; Tanaka et al. 1997). Since the increase in pH was accompanied by a considerable decrease in soluble Al and concomitant increase in exchangeable bases, it suggested that the alkalinity properties induced by ash accretion precipitated soluble Al into $Al(OH)_3$, followed by liberation of exchange sites and absorption of inorganic bases from the ash.

Similarly plant-available P increased by about 39% after fire (from 5.55 mg P kg^{-1} soil before fire to 19.8 mg P kg^{-1} soil after fire) (Table 4.1). Current results contradicted the results obtained in the artificial heating experiment (Chapter III) but they were in agreement with those reported by Giardina and Rhoades (2001) and Kang and Sajjapongse (1980). In the previous laboratory experiment (Chapter III), forest soil free from plant materials had been used, whereas in the current study the site was an *Acacia mangium* plantation with plant biomass aboveground. Therefore some possible sources of the increases in available P can be explained as follows. Thermal decomposition of organic materials on the soil surface had also brought about decomposition of organic P into available P. Several studies have reported increases in plant-available P and reductions in organic pools of soil P following fire (Giardina et al. 2000). Fire-related increases in the available P found in the current study might have been caused by an input of P from consumed litter layer, *Acacia* trees, and organic matter in the first 1- to 2-cm soil surface. In addition, this increase could also occur through ash included at sampling. Although chemical analysis of ash was not carried out in the current study, a study by DeBano and Klopatek (1988) on Pinyon-Juniper soils in Arizona found that litter ash of pinyon and juniper contained 378 and 422 mg P kg^{-1}, or approximately 26 and 29% of the total P remaining, respectively. Furthermore the increases in soil pH due to the alkalinity of ash would also release P from Al-P complex. Alternatively the increases in available P could also be attributed to the heat-induced death of soil microbial population and the release of microbial P (DeBano and Klopatek 1988; Serrasolsa dan Khanna 1995). Even it is so, thermal decomposition is required because Bray method can measure only inorganic P of soils. Given the fact that soil microbes are concentrating in the surface soil and this part was severely affected by fire (as shown by soil cracking), thermal decomposition of microbial pools also occurred because microbial pools have been shown to have threshold temperature of less than 100°C (Dunn et al. 1985; Klopatek et al. 1988).

In spite of significant increases in exchangeable bases, cation exchange capacity (CEC) in contrast significantly ($p<0.05$) decreased after fire (Table 4.1). Two important sources of CEC are organic matter and clay. However, in the current study, decreases in CEC could not be attributed to the decreases in T-C because instead of decreasing, T-C markedly increased immediately after fire. Because the increases in T-C in the current study was caused by the increases in charring materials, these materials, which were neither intact organic compound nor pure C, chemically did not contribute significantly to soil CEC. However, it is probable that from a long-term viewpoint, the charring materials have important roles in predetermining many of the chemical, physical, and biological properties of soils, especially after the burnt sites experience wet and dry cycles and these materials decompose. Alternatively, the possible source of CEC was clay. Fire might have caused fusion of clay into sand-sized particle as a result of thermal modification of the iron and aluminosilicate (Betremieux et al., 1960; Sertsu and Sanchez, 1978). The result of the laboratory study showed that this fusion occurred at 100 °C (Chapter III). The aggregation of finer particles into sand-sized particles at higher temperature with the consequent decrease of the reactive surface area and dehydration of the mineral crystal lattice and the resulting breakdown of the lattice had caused further reduction of the CEC.

4.3.3. Leftover Effect of Fire on Soil Properties

The leftover effects of fire include the effects of fire on the short-term and long-term dynamics of soil properties. The short-term dynamics evaluates the nutrient budget of the B98 site during the first two years, which is given by the differences between nutrient contents of the B98 site before and those obtained one month, one year and two years after fire, and the results are given in Table 4.2. The leftover effect of fire on T-C and T-N observed in the current study appeared to be more

Table 4.2. Nutrient budget of surface soils (0 - 10 cm) from *Acacia mangium* plantation during the first two years after fire

Variable	Before fire	After 1 mo	After 1 yr	After 2 yr
T- C (g kg^{-1})	19.7 a	24.9 b	28.3 c	20.4 d
T-N (g kg^{-1})	1.7 a	3.2 b	2.3 c	1.8 a
Available P (mg P kg^{-1})	6.9 a	3.4 b	5.4 c	12.6 d
pH	4.9 a	5.2 b	5.5 c	5.3 b
Al saturation (%)	30.0 a	28.6 a	14.3 b	29.9 a
Exch. Bases (cmol(+) kg^{-1}):				
Ca	2.6 a	3.9 b	3.6 c	4.8 d
Mg	1.1 a	1.3 a	1.9 b	1.9 b
K	0.3 a	0.3 a	0.3 a	0.3 a
CEC (cmol(+) kg^{-1})	10.2 a	10.9 b	20.3 c	11.6 d

Means within the same rows followed by the same letters are not significantly different at $P<0.05$ (based on *Tukey's t- test*). Values are means for $n = 3$.

prominent than the immediate effect. T-C and T-N one month after fire was significantly ($p<0.05$) higher than before fire (Table 4.2). In addition to charring materials, the elevating temperature during fire might have killed most if not all of fine roots in the first 10-cm depth of the soils. Because the site had received sufficient amount of precipitation and experienced wet- and dry-cycle, decomposition of organic matter derived from these two sources might have been responsible for these significant increases in T-C and T-N observed one month after fire.

Similarly, pH, exchangeable bases, and CEC were also still significantly ($p<0.05$) aggrading one month after fire (Table 4.2), indicating the residual effect of ash accretion on these two properties. On the other hand the source of acidity was slightly suppressed as shown by the decreasing Al saturation from 30% before fire to 28.6% one month after fire (Table 4.2). Concomitantly exchangeable bases were dissolved and released into soil solution. In Chapter III, it was also shown that watering on heated Ultisols increased exchangeable bases of the heated soils. The combination of significant increases in exchangeable bases and in organic matter was thought to be responsible for the significant increases in CEC one month after fire (Table 4.2).

Available P significantly ($p<0.05$) decreased one month after fire leaving only about 50% of the pre-fire level of available P (Table 4.2). This finding confirmed that the increases in available P immediately after fire (Table 4.1) were in fact short- lived under field conditions. Although soil pH has a strong control on the availability of P in acid soils, a significant decrease observed in the current study could not be attributed to soil acidity because soil pH also increased by 0.3 unit (Table 4.2). Exchangeable Al also showed a slight decrease one month after fire (Table 4.2). Therefore, we assumed that the demanding regrowing vegetations for P as pointed out by Vitousek and Matson (1984) had absorbed most of P from the soils. Furthermore, because the site has experienced wet and dry cycles during the period between fire occurrence and sampling, a proportion of the available P might have equilibrated with the lesser available P fraction in soil, as pointed out by Giardina and Rhoades (2001). This mechanism was possible since Al saturation one month after fire was not different from the prefire level (Table 4.2).

The significant increases in T-C, available P, and pH still continued until year 1 after fire (Table 4.2). The increases in T-C and available P indicated accumulation process through mineralization organic matter and through solubilization of inorganic P by P-solubilizing bacteria and/or vesicular arbuscular mycorrhizal fungi. Similarly pH also increased, indicating that the residual effect of ash accumulation on soil acidity was still evident one year after fire. In contrast, Al solubility was suppressed due to the alkalinity effect of ash. However a significant depletion of N was observed two years after fire, indicating that consumption of organic N by soil microbial population was faster than accumulation due to active mineralization. Results on nitrifiers (as will be given in a Chapter V) shows that the population NH_4^+- oxidizers was high between 6 to 9 months after fire and significantly decreases afterward. Similarly the population of NO_2^--oxidizers was high between 9 to 18 months after fire and significantly depleted afterward. The high populations of these two groups of bacteria indicated that the depletion of T-N most likely occurred as a result of active nitrification process.

A significant ($p<0.05$) depletion in exchangeable bases one year after fire was observed only for Ca. On the contrary, exchangeable Mg significantly ($p<0.05$) increased and no changes were observed in exchangeable K (Table 4.2). The combination of plant uptake and leaching might have been the major mechanisms contributing to the decreases in exchangeable Ca in the burnt sites. Tanaka et al. (1997) in Thailand also found rapid depletion of Ca under continuous cropping because Ca occurred only in the exchangeable forms and could not be supplied from weathering of soil mineral. On the other hand, the significant increases in exchangeable Mg and no changes in exchangeable K reflected the resilience of these two cations in soils, presumably because they could be supplied along with mineral weathering after heating.

By year 2 after fire, the sites were fully occupied by secondary growth, including the regrowing *Acacia mangium*. At this stage, rates of biomass and nutrient accumulation approach a maximum placing a great demand on soil reserves. By this time, mineralization of organic matter pools resulted in significant ($p<0.05$) decreases in T-C and T-N but still comparable to their original levels. Concomitantly P and exchangeable Ca availability significantly increased as compared to their levels one year after fire (Table 4.2). Although a deterioration of soil organic matter-related properties for fields under shifting cultivation in Thailand had been reported by Funakawa et al. (1997), current results showed that available P was still high and other parameters, such as T-C, T-N, exchangeable bases, and CEC, were still comparable to their original levels, most likely because the burnts site were fallowed after the fire.

To facilitate the investigation on the long-term dynamics of soil properties in the burnt sites, a comparison was made between the results obtained from the unburnt *A. mangium* plantation (AM) and those burnt in 1995 (b95 and 1997 (B97). Table 4.3 presents the effect of year of fire occurrence on the soil characteristics of each site and the comparison of the soil characteristics of unburnt and burnt *A. mangium* plantations. The accumulation of nutrients under the control unburnt *A. mangium* plantation (AM) in fact occurred in all nutrients. However, the effect of year was significant only on the T-C and available P, which were significantly higher at year 2 than that at year 1. Similarly, under *A. mangium* burnt in 1997 and 1995, a significant effect was found only on available, which was significantly higher at year 2 than at year 1.

Table 4.3. Chemical characteristics of surface soils (0 - 10 cm) from the unburnt *Acacia mangium* (AM), and *Acacia mangium* burnt in 1995 (B95) and 1997 (B97)

Year	T-C	T-N	Avail. P	pH	Exchangeable bases				CEC
					Al	Ca	Mg	K	
	---- g kg^{-1} ----		mg P kg^{-1}		---------------cmol(+) kg^{-1}----------------				
AM	22.6 a	2.0 a	11.9 a	4.9 a	3.9 a	1.4 a	1.7 a	0.2 a	17.7 a
P > F	*	NS	**	NS	NS	NS	NS	NS	NS
Year 1	20.7 (4.6)	1.9 (0.3)	5.8 (2.8)	5.0 (0.1)	3.5 (1.9)	1.3 (0.3)	1.6 (0.6)	0.2 (0.05)	16.6 (5.5)
Year 2	24.4 (3.4)	2.1 (0.0)	18.1 (4.6)	4.8 (0.1)	4.4 (0.4)	1.5 (0.4)	1.8 (0.1)	0.3 (0.04)	18.9 (5.5)
B97	23.8 a	1.9 a	7.5 ab	5.0 a	4.2 a	1.0 a	1.3 b	0.2 a	14.9 ab
P > F	NS	NS	**	NS	NS	NS	NS	NS	NS
Year 1	23.6 (3.9)	1.9 (0.3)	4.6 (1.1)	5.1 (0.1)	4.6 (1.4)	0.6 (0.1)	0.9 (0.2)	0.2 (0.02)	14.3 (4.4)
Year 2	23.9 (1.2)	2.1 (0.3)	10.4 (2.3)	4.9 (0.1)	3.8 (0.2)	1.5 (0.1)	1.5 (0.6)	0.3 (0.06)	15.6 (5.9)
B95	22.9 a	1.7 a	5.5 b	5.0 a	3.3 a	1.6a	1.1 b	0.3 a	13.3 b
P > F	NS	NS	**	NS	NS	NS	NS	NS	NS
Year 1	22.0 (4.6)	1.7 (0.2)	2.6 (1.2)	5.1 (0.1)	3.1 (0.4)	1.1 (0.3)	0.9 (0.3)	0.3 (0.1)	13.9 (2.5)
Year 2	23.7 (1.7)	1.7 (0.1)	7.5 (1.3)	5.0 (0.2)	3.4 (0.5)	2.0 (1.0)	1.3 (0.4)	0.3 (0.1)	12.9 (4.4)

Means with standard error in parentheses; *n=8*; For the comparison among sites, *n = 6*; * and ** means that ANOVA is significant at P<0.05 and at P<0.01; Means within the same columns followed by the same letters are not significantly different at P<0.05 (based on *Tukey's t-test*).

The *A. mangium* stands have been shown to experience experience mortality as early as year 2 after planting (Fisher, 1995). Because *A. mangium* stands in the current study were already 7 years old at the start of the study, a large amount of organic matter, particularly fine- and medium-sized roots, may have been returned to the soil.

In the soils of the study sites, P tightly bound by Al and Fe. However, in spite of the decrease in pH and the increase in exchangeable Al, available P in year 2 was significantly higher than that in year 1 (Table 4.3). At this stage ay specific cause cannot be confirmed but it is hypothesized that decomposition of organic P and the activity of P-solubilizing bacteria and mycorrhizae may have released significant amount of P into the soil.

Both unburnt and burnt *Acacia mangium* plantations were still aggrading and accumulating soil nutrients (Table 4.3). The increase in organic matter is crucial in disturbed ecosystems because decomposition of organic matter provides significant amounts of nutrients for plant uptake. Potential sources for nutrient accumulation observed in the current study include nutrient cycles through plant biomass and N_2 fixation. A study by Polglase and Attiwill (1992) in Australia showed thatthe return of N and P in *Acacia* leaves was in the range of 1880 to 1970 mg m^{-2} y^{-1} and 54.6 to 77.4 mg m^{-2} y^{-1} at the age of 5 and 9 years, respectively. *Acacia mangium* plantation in the current study was 9 years old. Because both B95 and b97 were left fallow after fire, there has been enough time since they burnt in 1995 and 1997 for litterfall to accumulate and to be recycled.

The introduction of or the invasion by N_2 fixers to a disturbed ecosystem has been shown to increase N availability and to enhance site fertility. Adam and Atwill (1984) in a study in Australia estimated that N_2 fixation by *Acacia* was as high as 2,000 mg m^{-2} yr^{-1} for at least the first five years of growth. Although quantitative data on the N_2 fixation are not measured, the abundant rhizobium in all sites (data will be given in Chapter V) indicates the similar potentials as reported by other authors (Adam and Atwill 1984; Vitousek and Walker 1989). Similarly in B95 and B97, the regrowing *Acacia mangium* and the sprouting *Mimosa* spp. seemed to play similar important role in N accumulation in these sites.

Although atmospheric deposition has been shown to be a potential source for nutrient accumulation in disturbed ecosystems (Zarin and Johnson 1995; McDowell and Asbury 1994), data to support this possibility are not available in the current experiment. However, McDowell and Ashbury (1994) in the Rio Icacos watershed, Puerto Rico, had estimated atmospheric input onfN, P, K, Ca, and Mg at 2.4, 0.3, 4.2, 15.0, and 9.5 kg ha^{-1} yr^{-1}, respectively.

When a comparison is made among sites, available P and CEC in the B95 site were significantly lower than those in the control AM but they were not different from those in the B97 site. Exchangeable Mg was significantly lower in the B95 site than that in the control AM and B97 sites. In general soils under the burnt sites showed poorer quality, indicating that the leftover effects of fire on the soil characteristics were still evident five years after the fire.

The tree plantations, particularly those with close canopies, obviously alter the microclimate within the stand. Measurement of soil temperature at 10-cm depth in the AM and B97 sites did not show significant variations among sites and within sites. They were ranging from 25 to 26.2°C at the AM site and from 25.2 to 26.2°C at the B97 during wet season and from 25.9 to 26.3°C at the AM site and from 25.6 to 25.9°C at the B97 site during dry season. Due their similarity, soil temperature did not appear to cause a strong control on the variability of nutrients through their effect on decomposition. However, soil moisture was significantly higher in wet season than that in dry season (Table 4.4). The increased soil moisture in the near surface soil may favor biotic activity in this zone. These results emphasize the importance of extensive temporal replication of sampling and the importance of reference sites that normalize factors influencing nutrient cycle.

Table 4.4. Moisture content of surface soils (0 – 10 cm) from the unburnt and burnt *Acacia mangium* plantations

Site	Wet season	Dry season
	---------------- %	-----------------
AM	40.2 a	27.5 b
B97	35.2 a	27.6 b
B95	34.6 a	23.9 b

Means within the same rows followed by the same letters are not significantly different at $p<0.05$ (based on *paired t-test*).

4.4. Summary

Forest management practices significantly affect the dynamics of T-C, T-N, available P, pH, exchangeable bases, and CEC. Both T-C and T-N under CF, PF, and HG show considerable intra-periodic variation (Figs. 4.1a and b). The available P also fluctuates significantly (Fig. 4.1c). The pH and the exchangeable bases under CF are lower but show less fluctuation than those under PF and HG (Figs. 4.2a and b). Similarly, the CEC values significantly vary with time (Fig. 2c) and significantly correlate with the fluctuations in T-C except in the PF site (Fig. 3) where much higher contents of exchangeable Ca and Mg are found.

Fire has significant direct and indirect effects on nutrient dynamics in the soils. The significant increases in nutrients level immediately after fire (Table 4.1) indicate the positive effects of fire on nutrient status. However, these advantageous effects are short-lived. The results show that although the accumulation of nutrients in burnt site is still evident one year after fire (Table 4.2), nutrients levels start to deplete and return to the pre-fire levels by year two (Table 4.2). The combination of plant uptake and leaching is thought to be responsible for the depletion. The depletion also indicates that the rate of plant uptake and leaching is faster than that of deposition.

Fallowing helps the burnt sites aggrading (Table 4.3). In spite of the considerable intra-periodic variability and the lack of significant effect of year except on T-C in the unburnt AM and available P in all sites, both the unburnt and the burnt sites are aggrading. However, the nutrient levels in the burnt sites are still lower than those in the unburnt sites. Current findings provide evidence that the leftover effects of fire on soil nutrient levels are still evident five years after fire. Therefore prevention of forest fire is a crucial action in order to maintain sustainability of ecosystem in the tropics. In addition, considering the long-term productivity, the declines in nutrient pools in the AM site may have significant implications because *Acacia mangium* plantation has about 6- to 7-yr rotation. It means that site disturbance will cycle every 6 to 7 years as the trees are clear-cut and the sites are prepared for the next cycle. On the other hand continuous and regular organic matter turnover, for example in the PF and HG sites, successfully maintains levels of nutrient pools mainly C and N comparable to those found in the CF sites.

Chapter V

Characteristics of Ultisols under different Wildfire History in South Sumatra, Indonesia: Dynamics of Microbial Population

5.1. Background

In recent years, there has been increasing evidence that microbial communities of soil have an important role in the maintenance of ecosystem sustainability and in the reconstruction of degraded lands. Although microorganisms make only 1 to 8% of the organic matter, they influence ecosystem productivity by acting as a catalyst for nutrient biotransformation through the process of mineralization and immobilization (Roder et al., 1988). Amongst the important microorganisms are those involved in the cycle of P and N, which are usually limiting plant growth in the tropical soils.

Two important microbial groups in P cycle include P-solubilizing bacteria (PSB) and Vesicular arbuscular mycorrhizal (VAM) fungi. PSB are responsible for the solubilization of Al-P and Fe-P, two inorganic complexes of P commonly found in the tropical soils. A study by Senthilkumar et al. (1998) on grassland soils in southern India found a significantly higher population of PSB on the burnt site than that on the unburnt site. The major microbiological means, by which insoluble P-compounds are mobilized, is by the production of organic acids.

VAM fungi have been found to be essential components of sustainable soil-plant systems. Benefits derived by plants from VAM symbiosis include (i) increased uptake of P (Bolan, 1991), N (Barea et al., 1991), and micronutrients (Bürkert and Robson, 1994), (ii) enhanced water absorption (George et al., 1992), and (iii) improved plant health by providing protection against some pathogens (Dehne, 1982). VAM fungi have also been shown to hasten the rate of succession and restoration of vegetations in disturbed ecosystems and contribute to the composition of plant community (Francis and Read, 1994). Although much of basic knowledge of VAM ecology comes from studies conducted on disturbed ecosystems (Jasper et al., 1988), much less information is available on the recovery of VAM fungal community on burnt sites (Klopatek et al., 1988).

Prolonged periods of fire exclusion commonly result in development of forests with N-limiting conditions. The majority of soil N is found in organic forms and accumulates either in the organic horizon or mixed with the uppermost mineral horizon. Nitrifying bacteria (e.g. *Nitrosomonas* and *Nitrobacter*) have the ability to bring about important changes in soil N affecting its retention and movement. The importance of the nitrifying microorganisms rest to a great degree on their capacity to produce nitrate that is the major N source assimilated by higher plants.

Although the increases in the availability of NH_4-N and NO_3-N after prescribed burning have been reported for shifting cultivation (Tanaka et al., 2001), the effects of fire on the dynamics of nitrifying bacteria have not been thoroughly studied.

Keenan et al. (1995) stated that only a small portion of organic N undergoes microbial decomposition, the majority of which is rapidly immobilized by soil microbes, leaving as little as 0.01% of total N for plant uptake. Bormann et al. (1977) suggested that about 68% of the N added annually to a temperate deciduous forest was estimated from biological N_2 fixation. However, data from tropical forests and managed forest are still lacking. Although biological N_2 fixation can be carried out both by free-living microorganisms (e.g. *Azotobacter* and *Beijerinckia*) and by symbiotic microorganisms (e.g. *Rhizoium*), perhaps the greatest potential for better exploiting N_2-fixing organisms in forestry is the use of symbiotic N_2-fixing plants. Trees, shrubs, and herbaceous plants capable of symbiosis with *Rhizobium* can often sustain high rate of productivity themselves while increasing soil N available to associated plants (Dawson, 1983).

Since living organisms in soils are mostly located in or near the soil surface, they are exposed to heat radiated by flaming surface fuels. As a result, alterations in their populations are simultaneously expected to occur. Immediate decline in microbial population after burning has been shown under chaparral shrublands in southern California (Dunn et al., 1985) and natural grassland in southern India (Senthilmukar et al., 1998). However, information pertaining the population dynamics of indigenous VAM fungi, nitrifiers, P-solubilizing bacteria, and rhizobia in relations to wildfire under tropical climate is still lacking. Current study highlighted the effects of forest management practices as well as the immediate and the long-term effects of wildfire on the survival of indigenous VAM fungi, nitrifiers, P-solubilizing bacteria, and rhizobia in Ultisols of South Sumatra, Indonesia for two consecutive years.

5.2. Materials and Methods

5.2.1. Site Description and Soil Sampling

Detailed descriptions of the study site and soil sampling procedures were already given in Chapter II and Chapter IV. All samples were immediately stored in a cooler box with ice for transportation to laboratory for storage. Prior to storage, the soils were divided into two compartments. One was used for VAM spore counting; the other was used for the enumeration of soil bacteria. Soil samples for spore counting were immediately air-dried at room temperature to avoid germination of the spores while those for the enumeration of soil bacteria were immediately transferred to a cooling chamber (4°C).

5.2.2. Vesicular Arbuscular Mycorrhizal Fungi

Spores were isolated from 50 g of soil by wet sieving (Gerdemann and Nicolson, 1963). Fine sieves (38 and 106 μm opening) were used to collect the small spores, and the coarse materials remaining on the top sieves (250 and 1000 μm opening) were also checked for sporocarps and large spores. Under a stereomicroscope (Olympus SZ60), spores were counted, and typical and clean spores were separated into groups according to general morphological similarities. Permanent slides of all typical spores were prepared by mounting in polyvinyl alcohol-lactic acid-glycerol (PVLG) and Melzer's reagent (Koske and Tessier. 1983). On each slide, two sets of permanent slide were prepared with the same amount of spore (10 to 15 spores). On one side, the spores were intact under the cover slip, whereas on the other side, the spores were cracked open under the cover slip to allow for observation of spore wall and the inner wall characteristics. Spores were identified to species according to classical morphological analysis under a stereomicroscope and their frequencies were recorded. Important taxonomic characteristics (Schenk and Pérez, 1988) included color, size range of spores, number and type of spore wall layer and their reaction to PVLG and Melzer's reagent, and when present; morphology of hyphal attachment.

Species richness, Shannon-Wiener index of diversity, evenness, and Simpson's index of dominance were calculated (see Table 5.1 for formulae). Analysis of variance (ANOVA) was performed according to

GLM procedure in Cohort V. 2.2 to examine the effect of changes in land use and fire history on ecological measures of diversity of VAM fungal communities. Significant differences between means of each of ecosystem types and fire history were determined using Tukey's mean separation in Cohort V. 2.2.

Table 5.1. Diversity measures used to describe communities of VAM fungi

Species richness (*S*)	The number of species in a specified study site
Shanoon-Wiever index of diversity(*H'*)	$H' = -\Sigma p_i \ln p_i$
Evenness (*E*)	$E = H'/\ln S$
Simpson index of dominance (*D*)	$D = \Sigma p_i^2$

p_i is the proportion of individuals found in the *i*-th species, estimated as n_i/N, where n_i is the number of individuals in the *i*-th species and the *N* is the total number of individuals (Begon et al., 1996).

5.2.3. Enumeration of Soil Bacteria

Three indigenous groups of bacteria investigated included nitrifying bacteria, P-solubilizing bacteria, and *rhizobium*. Nitrifying bacteria were enumerated using the most probable number (MPN) method with 5-tube procedure (Alexander, 1982) in a growth medium as described by Schmidt and Besler (1982). The soil suspension (1 mL) of each dilution level (10^{-2} to 10^{-7}) was aseptically transferred into each of 5 test tubes containing 4 mL of either sterilized NH_4^+- or sterilized NO_2^--media for the enumeration of NH_4^+- and NO_2^- oxidizers, respectively. The inoculated MPN tubes were incubated at 30ºC in the dark. Initial observation was made after 3 weeks using a spot test. The NH_4^+-medium was checked for the presence of NO_2^-, and the NO_2^- -medium for NO_3^-. For the NO_2^--oxidizers, the NO_2^--medium was checked for the disappearance of NO_2^- using NO_3^- spot test (Schmidt and Besler, 1982). One drop of the NH_4^+-medium was aseptically transferred to a spot plate depression; added with one drop of diazotizing reagent and coupling reagent, respectively. If the NH_4-medium gave a strong reaction (dark red) compared with the uninoculated control, the tube was scored positive for NH_4^+-oxidizers. To guard against a false-negative score, NO_3^- spot test reagent was also added. Tubes containing NO_2^--medium was scored positive for NO_2^--oxidizers if the test indicated that this substrate had decreased or disappeared. Control NH_4^+-medium developed a light pink reaction to the reagents. Control NO_2^--medium did not change perceptibly. Tubes scored negative were reincubated with weekly monitoring for a period of six weeks until there was no change in the number of positive tubes for consecutive weeks. The MPN, based on the positive tubes, was calculated with the MPN table (Alexander, 1982).

The enumeration of rhizobia and P-solubilizaing bacteria (PSB) was carried out using pour-plate method (Wollum II, 1982) in yeast extract mannitol agar-congo red (YEMA-CR) medium and Pikovkaya's medium, respectively. A 1-mL of soil suspension (10^{-2} to 10^{-7}) was aseptically transferred into each of five sterile petridishes. Subsequently, a 15-mL of sterilized YEMA-CR and Pikovkaya's medium was poured into the petridishes. Immediately after pouring, the petridishes were swirled to thoroughly mix the medium and the soil suspension. After the medium solidified, the inoculated petriplates were inverted, wrapped in paper, and incubated at 30ºC. Colony counting was carried out 4 days after incubation. The colonies of *rhizobium* were light pink, circular and raised on the agar surface. *Rhizobia* growing on the media were grouped into one of the following categories: fast-growing with colonies 2 to 4 mm in diameter in 3-5 days; and slow-growing with colonies smaller than 1 mm in 5 days (Jordan, 1984; Barnet and Catt, 1991). The colonies of PSB were easily recognized because they produced clear zones around the colony. The petridishes yielding 30 to 300 colony forming units (cfu) were selected.

Data on nitrifying bacteria, P-solubilizing bacteria, and rhizobium were average of five test tubes and five petriplates per replicate of soil samples, respectively. Analysis of variance (ANOVA) was performed according to GLM procedure in Cohort V. 2.2 to examine the effect of changes in land use and fire history on microbial populations. Significant differences were determined using Tukey's mean separation in Cohort V. 2.2.

5.3. Results and Discussion

5.3.1. Effect of Landuse on VAM Fungi

Most spores extracted (above 80%) possessed recognizable morphological characteristics, which permitted their identification. The abundance of spores, species richness, and diversity of VAM fungai varied significantly ($p<0.001$), depending on the landuse. Conversion of natural forest (CF) into human-managed ecosystems (PF and HG) significantly reduced the number of spores, species richness, and Shannon-Wiener's diversity indices of VAM fungal communities in the study site (Table 5.2). These results indicated that the selective pressure caused losses of some species of indigenous VAM fungi. On the other hand, the evenness and Simpson's index of dominance in the CF, PF, and HG were similar (Table 5.2), reflecting the general resemblance of the proportional abundances of spore of different species among these three communities.

The removals of photosynthetic tissues from plants (grazing) have been reported to decrease spore densities (Bethlenfalvay et al, 1985). In the current study, severe disturbance on the pine forest occurred mainly during land preparation although the planted pine grew with little human interference in subsequent years. On the HG site, the grazing due to tillage, harvest, and weeding occurred regularly, especially at every

Table 5.2. Effect of landuse on the VAM fungal community

Ecological parameter	Site		
	CF	PF	HG
Total spores (per g of soil)	40 a	36 b	24 c
Species richness	13 a	9 b	10.7 c
Shannon and Wiener diversity index (*H'*)	2.4 a	1.9 b	1.9 b
Evenness (*E*)	0.9 a	0.8 a	0.8 a
Simpson's index (*D*)	0.1 a	0.2 a	0.2 a

Values in the row followed by the same letter are not significantly different (based on Tukey's t-test at p=0.05). Values are means for $n = 3$.

growing season. Such activities can cause more pressure on the sporulation by VAM fungi. Therefore, the number of spore in the HG site was significantly lower than those in the other sites (Table 5.2). Because VAM fungi are ecologically obligate symbionts and depend their C needs for growth and sporulation on their host, the removal of photosynthetic tissue limits the availability and allocation of C to the roots.

Several workers have also noted that diversity of VAM fungal communities tended to decrease when natural ecosystems are converted to agroecosystems, and diversity decreases as the intensity of agricultural inputs increases (Schenck et al., 1989; Sieverding, 1990). Two possible explanations could account for this pattern. Firstly, fungal diversity is related to diversity of plant communities (Rabatin and Stinner, 1989). Natural ecosystems contain a great variety of plant species, while human-managed ecosystems consist of very few plant species, generally a single crop per field with occasional weeds. Natural forest in the current experiment (CF) consisted of a great variety of tree species with a great variety of floor growths. On the other hand, human-managed plantations (PF, AM and HG) involved single tree species, pine with occasional weeds growing on the floor. Second, cultural practices might have exerted strong selective pressure on VAM fungal communities. The shift from mixed species prior to conversion into tree plantation of monospecies might have changed predominat microsymbionts from VAM fungi into ectomycorrhizal fungi, while VAM fungal species most tolerant of these stresses persisted and proliferated. However this hypothesis still needs further investigation.

Although species richness in HG was significantly higher than that in PF, the Shannon-Wiener's index of diversity (*H'*) was similar (Table 5.2). Since VAM fungi are obligatory plant symbionts, it is not surprising that cropping system influence their population in soil. VAM fungi are typically considered generalists. However evidence of specificity between plants and VAM fungi is growing (Johnson and Pfleger, 1992). Because certain crops can preferentially select for certain VAM fungal species, cropping sequences may influence the species composition of VAM fungal communities. In the current study, crops planted in HG included cassava (*Manihot esculenta*), banana (*Musa* spp.), upland rice (*Oryza sativa*), bean (*Phaseolus vulgaris*), peanut (*Arachis hypogea* L.), and maize (*Zea mays* L.). These crops were not cultivated in monoculture but in a mix of two or three of those species in one growing season. Therefore, the species of VAM fungi in the HG site were more diverse than those in the PF site although they were still less diverse than those in the CF site.

The HG site could be regarded as a low-input agriculture since no chemical fertilizer or pesticide has been used. The only input was liming material to eliminate Al toxicity. Furthermore plant materials after harvest were also regularly returned and incorporated into the soil. Saif (1986) showed that returning plant residues to soil increased mycorrhizal infection and spore populations in tropical forage system. Apparently VAM hyphae can respond to microsite accumulation of organic matter by proliferating into the microsites. These microsites of organic debris may be a necessary step in the active proliferation of external hyphae and, concomitantly, in the whole life cycle of the VAM fungal community.

5.3.2. Species Composition of VAM Fungi

Land management practices may affect mycorrhizae both quantitatively and qualitatively. A study typically examining the influence of land management and environmental stresses on total number of spores is an enlightening first step. Thus, when considering cultural stresses on VAM fungi, assessing the species composition of VAM fungal communities may sometimes be more important than assessing spore densities and root colonization (Johnson and Pfleger, 1992).

To unveil community members that are undetected in the initial extraction of spores from field soils, trap cultures have been recommended and widely used (Morton et al., 1995). However, trap cultures tended to encourage preferential sporulation by a few species

(Bever et al., 1996). Therefore in the current study, a repeated soil sampling technique during two consecutive years was used to unveil the species composition of VAM community in the study sites. With this technique, some species that were not detected in the previous sampling periods could be revealed without causing any preferential sporulation of certain species of VAM fungi. Therefore the species recorded were an accurate reflection of the richness and a good representation of the VAM fungal community in the study sites.

Morphological analysis on spore characteristics resulted in five genera of family Endogenaceae, i.e. *Glomus*, *Gigaspora*, *Acaulospora*, *Sclerocystis*, and *Scutellospora*. Those five genera could be found across the study site, probably because the study sites were located in one compound area with similar type of landuse prior to conversion. The occurrence of each genus in the study sites is summarized in Fig. 5.1 and Table 5.3. *Glomus* was the most frequently found genus except in the most recently burnt site (B00), which was dominated by *Acaulospora*. A unit of land should not be viewed as a landscape containing plants linked by a single mycelial network. Rather it consists of several vegetation units, and multiple fungal species with different levels of inoculum densities and mycorrhizal types. This system is also dynamic, the members of associated mycorrhizal fungi change as a patch of vegetation changes through time. Disturbance, such as conversion or alteration of natural vegetation into human-managed ecosystem, and also fire,

FIGURE 5.1.

Table 5.3. Species composition and percentage of identified species of VAM fungal community in the study sites

Species	Site							
	CF	PF	AM	B95	B97	B98	HG	B00
Glomus caledonium (Nicol. & Gerd.) Trappe & Gerdemann	O	X	O	O	O	O	X	O
G. etunicatum Becker and Gerdemann	X	O	X	X	X	X	O	X
G. fuegianum (Spegazzini) Trappe & Gerdemann	O	O	X	X	X	X	X	X
G. geosforum (Nicolson & Gerdemann) Ealker	X	O	X	X	X	O	X	X
G. globiferum Koske & Walker	X	X	X	O	X	O	X	X
G. manihotis Howeler, Sierverding & Schenck	X	X	X	X	X	X	O	X
G. mosseae (Nicol & Gerd.) Gerdemann & Trappe	O	O	O	O	O	O	O	O
G. reticulatum Bhattachaerjee & Mukerji	X	X	X	X	O	O	X	O
G. turtuosum Schenck & Smith	X	X	O	X	X	O	X	X
G. versiforme (Karsten) Berch	O	O	O	O	O	O	O	O
Acaulospora appendicula Spain, Sieverding & Schenck	X	O	X	X	O	O	X	O
A. bireticulata Rothwell & Trappe	O	O	O	O	O	O	O	O
A. foveata Trappe & Janos	X	X	O	O	O	X	O	X
A. laevis Gerdemann & Trappe	O	O	O	X	O	X	O	X
A. rugosa Morton	O	X	X	X	X	X	X	X
A. tuberculata Janos & Trappe	O	O	O	O	O	O	O	O
Gigaspora albida Schenck & Smith	O	O	X	O	X	X	X	X
Gi. decipiens Hall & Abbott	X	X	X	O	O	X	X	X
Gi. gigantea (Nicol. & Gerd.) Gerdemann & Trappe	O	O	O	O	O	O	O	O
Sclerocystis clavispora Trappe	O	O	O	O	O	O	O	X
S. microcarpus Iqbal & Bushra	O	O	O	O	O	O	O	O
S. pachycaulis Wu & Chen	X	X	X	O	X	X	X	X
S. rubiformis Gerdemann & Trappe	X	O	X	X	X	X	O	O
S. sinuosa Gerdemann & Bakshi	X	X	X	O	X	X	O	X
Scutellospora nigra (Redhead) Walker & Sanders	O	O	O	O	O	O	O	O
Level of identification	86	89	89	87	83	88	81	88

O = found; X = not found; CF = conservation forest; PF = pine forest; AM = unburnt *Acacia mangium* plantation; B95, B97, B98, and B00 = *Acacia mangium* burnt in 1995, 1997, 1998, and 2000; HG = home garden.

generally results in not only shifts in plant species composition but also types of mycorrhizal association and fungal species. As the *Acacia mangium* plantation was burnt, the predominat genus of indigenous VAM fungi changed from *Glomus* to *Acaulospora* (Fig. 5.1). As the post-fire period proceeded, i.e. from B00 to B98, B97 and B95, the predominant genus changed and returned to the pre-fire condition (AM) (Fig. 5.1).

Different habitats containing different plant and fungal species are well recognized. Twenty-five species of VAM fungi were found across the study area (Table 5.3). Only one species belonged to the genus *Scutellospora* Walker & Sanders; 10 species to the genus *Glomus* (Tul & Tul) Gerdemann & Trappe; 3 species to the genus *Gigaspora* Gerdemann &Trappe emend. Walker & Sanders; 6 six species to the genus *Acaulospora* Gerdemann &Trappe emend. Berch; and 5 species to the genus *Sclerocystis* Berk. & Broome. In spite of the fact that the whole five genera could be found across the study sites, conversion of natural ecosystem into human-managed ecosystem and interference of wild fire also had changed species composition of VAM fungi. Some species, for example *A. rugosa* Morton totally disappeared, most probably due to loss of specific host plant and unfavorable condition after conversion. The second factor appeared to be more plausible because VAM fungi have been shown to form mycorrhizae with diverse host plants, typically crossing between plant families, orders, and even classes. Harley and Smith (1983) introduced the term "ecological specificity" to emphasize that mycorrhizal associations induced in the laboratory experiments may not occur under field conditions and that competition in the field may increase expression of specificity. Because fungal diversity is related to diversity of plant communities, the colonization of soil by VAM fungi following conversion of natural ecosystem coincided with the introduction of exotic plants and weeds. The new, dominant host plant (mono species) was found to exert selective pressure on VAM fungal population.

On the other hand, some species, for example *G. etunicatum* Becker and Gerdemann in PF; *G. globiferum* Koske and Walter in B95 and B98, *G. manihotis* Howeler, Sieverding & Schenck in HG, *A. foveata* Trappe & Janos in AM, B95, B97 and HG; *G. decipiens* Hall & Abbott in B95 and B97, *S. pachycaulis* Wu & Chen in B95, *S. rubiformis* Gerdemann & Trappe in PF and HG; and *S. sinuosa* Gerdemann & Bakshi in B95 and HG, which did not exist in the CF site, could be found. The existence of those species in human-managed ecosystem was likely introduced through the introduction of new vegetation. Because the seedlings were propagated in the nursery before they were transferred to the plantations, they were hosts for the exotic species of VAM fungi. Since the conditions in the field were favorable, those exotic species of VAM fungal communities proliferated and persisted in the soils.

Similarly some other species, e.g. *G. mosseae* (Nicol. & Gerd.) Gerdemann & Trappe, *A. bireticulata* Rothwell & Trappe, *A. tuberculata* Janos & Trappe, *G. gigantea* (Nicol. & Gerd.) Gerdemann & Trappe, *S. clavispora* Trappe, *S. microcarpus* Iqbal & Bushra, and *Sc. nigra* (Redhead)

Walker & Sanders, could be found across the study sites, reflecting their ability to persist in spite of ecological stresses due to changes in their habitats and to passage of fire. Identifying which VAM fungi are tolerant to ecological stresses and determining their effects on individual crop or tree are the challenge that must be met before those fungi can be fully understood and managed as components of agroecosystem and/or agroforestry.

The taxonomic diversity found in the current study cannot necessarily be linked to the functional role of these fungi in their habitats. Even though some species occupied different ecosystems, the functional redundancy could not be predicted. Similarly, based on the results of the current study, functional aspects of different VAM fungal community in the same niche could not be predicted either. Future tests comparing symbiotic efficacy of these fungal communities should indicate whether the uniformity is also reflected in the capabilities of these fungi to promote plant growth in each ecosystem, or if there is functional diversity among these communities both within the same and different ecosystems. However, the association of plant-VAM fungi found in the study site may partly explain the establishment of vegetation in spite of the low natural fertility level of soils and further degradation of soil fertility due to fire occurrence in the study site.

5.3.3. Effect of Fire on VAM Fungi

Results obtained from the most recently burnt *Acacia mangium* plantation (B00) and those from plantation burnt in December 1998 (B98) enabled us to trace both immediate and delayed effects of fire on VAM fungal community, respectively. In the B00 site, the fire passage significantly ($p<0.01$) affected all ecological parameters except Simpson's index of dominance (D). Number of spores, species richness, and Shannon-Wiener index of diversity decreased significantly immediately after fire and increased significantly three months after fire but still lower than pre-fire levels (Table 5.4).

Table 5.4. Immediate and delayed effect of fire on VAM fungal community

Fire history	Ecological parameter				
	Total spore (per g soil)	Species richness	Shannon-Wiener Diversity index (*H'*)	Evenness (*E*)	Simpson's index (*D*)
Before fire	21.67 a	7.67 a	2.01 a	0.97 a	0.14 a
After fire	11.67 c	6.67 b	1.86 c	0.96 a	0.17 a
3 months	15.67 b	8.33 a	1.91 b	0.91 b	0.15 a

Values in the same column followed by the same letter are not statistically different (based on Tukey's t-test at p=0.05). Values are means for $n = 3$.

Raison (1979) stated that heating soil for 10 min at 70°C killed soil biota, while temperatures above 127°C would nearly sterilize soil. Although soil temperature during the fire was not measured in the current experiment, the changes in physical and chemical characteristics of soil after fire and the existence of soil cracking (Chapter IV) indicated the effects of heating. Because heat penetration decreased with depth as shown by DeBano et al. (1979) and Giovannini and Lucjesi (1997), the increasing temperature due to fire might have killed spores found in the first few centimeter of soil layer. The spores located in the deeper part could escape the heating effect and were responsible for the increases in total number of spores three months after fire, as shown for ectomycorrhizal fungus community in ponderosa pine (*Pinus ponderosa* Laws) in Sierra Nevada, California (Stendell et al., 1999).

In the current study, only one species of VAM fungi, i.e. *S. clavispora* Trappe disappeared from the site due to the fire passage. A slight increase in the Simpson index of dominance (Table 5.4) reflected a shift of dominance of individual species of VAM fungi. This increase was resulted by the higher occurrence of spores of *A. bireticulata* Rothwell & Trappe than any other species in this site immediately after fire. However, the increase in the frequencies of *A. bireticulata* Rothwell & Trappe found was not high enough to result in significant shift in the proportional abundance of all species of VAM fungi immediately after fire. Similarly the evenness after fire remained the same as before fire (Table 5.4). These results confirmed the natural potentials of VAM fungal inoculum in spite of the fire passage. The suggestion that the rate of succession or restoration of disturbed areas may be hastened by inoculation (Reeves et al., 1979) may not be appropriate to be applied in the study site because naturally all sites have` potential inoculum of

VAM fungi. However, it does not mean that mycorrhizal fungi facilitate the skipping of succesional stages. Rather, the importance of mycorrhizal fungi more likely lies in preventing the stagnation of community development. For example, immediately after the first rain, the recently burnt site was instantaneously invaded by herbs, shrubs (including legumes, e.g. *Mimosa* spp.), and grasses, which were presumably mycotrophic.

Three months after fire, there were significant increases in ecological parameters of VAM fungal communities in the recently burnt site except Simpson's index of dominance, which was similar to the pre-fire level (Table 5.4). Although the species richness increased only by one unit, the actual composition of the species changed. Three species, i.e. *G. versiforme* (Karsten) Berch, *A. bireticulata* Rothwell & Trappe, and *A. tuberculata* Janos & Trappe) found immediately after fire could not be revealed three months after fire. As mentioned before, repeated sampling technique had been very helpful tools to get the full reflection of VAM fungal communities without changing their niches. Therefore given the fact that the findings in the recently burnt site were based on only two sampling times, to conclude that these three species had totally disappeared from the site is tenuous. Franke-Snyder et al. (2001) in their study also found *G. fasciculatum* once in several years of research in a rotation of maize, soybeans, small grains and leguminous cover crops. On the other hand four species, i.e. *G. reticulatum* Bhattacharjee & Mukerji, *G. gigantea* (Nicol. & Gerd.) Gerdemann & Trappe, *A. appendicula*

Spain, Sieverding & Schenck, and *S. rubiformis* Gerdemann & Trappe, which could not be revealed immediately after fire, could be detected three months after fire. These two findings reflected the dynamics of VAM fungi in relation to changes in the above ground vegetation. Because mycorrhizal fungi are obligatory symbionts, the deleterious anthropogenic pressures on vegetations and the destructions of roots due to burning could have significant negative effects on the development of the fungi in the soil. On the other hand the growth of roots through the soil after fire could have positive effects. Therefore these dynamics indicate that biotic and abiotic factors interact to induce a fungus to sporulate and proliferate in the soils. In addition, the possibility of succession of fungi colonizing roots after fire disturbance highlight the potential for considering VAM fungi in terms of their capacity to remain infective.

Similar results were obtained for the *Acacia mangium* plantation burnt in 1998 (B98). All ecological parameters of VAM fungal community except species richness were significantly ($p<0.01$) affected by period after fire. Dynamics of these characteristics are given in Fig. 5.2. Total spores significantly fluctuated over time. The highest numbers of spore were 57 and 54 spores detected 6 and 18 months after fire, which were significantly larger than those found in any other periods (Fig. 5.2a). These two periods (6 and 18 months after fire) were coincided with dry season while the other two periods (12 and 24 months after fire) were

FIGURE 5.2.

rainy season. Because the VAM mycelia associated with roots sporulate abundantly just before the host dies, most likely that the mortality of the host plants due to seasonal cycle had facilitated the sporulation.

Although species richness was not significantly affected by the post-fire period, the species composition changed, reflecting the loss of some species from and the invasion of new species of VAM fungi into the burnt site. Five species, i.e. *G. mosseae* (Nicol. & Gerd.) Gerdemann & Trappe, *G. versiforme* (Karsten) Berch, *A. bireticulata* Rothwell & Trappe, *A. foveata* Trappe & Janos, and *S. microcarpus* Iqbal & Bushra existing before the fire could not be revealed 6 months after the fire. On the other hand, five other species, i.e. *G. globiferum* Koske & Walker, *G. reticulatum* Bhattacharjee & Mukerji, *A. appendicula* Spain, Sieverding & Schenck, *G. gigantea* (Nicol. & Gerd.) Gerdemann & Trappe, and *S. nigra* (Redhead) Walker & Sanders were found. Similarly 12 months afterward, some species disappeared but some others could be revealed.

Shannon-Wiener diversity index (*H'*) also fluctuated significantly over time (Fig. 5.2b). This index decreased significantly 6 and 12 months after fire and increased again 18 and 24 months after fire (Fig. 5.2b). However, this significant variability was not reflected in the species richness because the number of the VAM fungal species did not change over time (Fig. 5.2b). Instead, this variability was induced more by the significant changes in species composition rather than by species enrichment in each period. The only new species that could consistently be revealed after the fire was *G. turtuosum* Schenck & Smith.

Prior to the fire, the index of dominance (*D*) was 0.18, reflecting absence of dominance of certain species of VAM fungi. Six and twelve months after the fire this value significantly increased to 0.23 and 0.27 (Fig. 5.2b), respectively, reflecting a shift of dominance of individual species of VAM fungi. On the other hand the evenness index (*E*) significantly decreased from 0.89 before fire to 0.77 and 0.66 six and twelve months after fire (Fig. 5.2c), reflecting the significant changes in proportional abundance of spores of different species between these two periods. The increases in the *D* and the decreases in the *E* indices were resulted by higher occurrence of *S. clavispora* Trappe (32%), *G. caledonium* (Nicol. & Gerd.) Trappe & Gerdemann (23%), and *G. reticulatum* Bhattacharjee & Mukerji (21%) six months after fire. The domimant species twelve months after fire included *S. clavispora* Trappe (41%), *S. microcarpus* Iqbal & Bushra (24%), and *G. mosseae* (Nicol. & Gerd.) Gerdemann & Trappe (20%). From 18 months afterward, the *D* value significantly decreased and returned to pre-fire level (Fig. 5.2b).

Based on the results reported above it was clear that the deleterious effects of fire on the VAM fungal community were short-lived. All ecological parameters investigated returned to their original levels 18 months after the fire passage. This was also true for the sites that burnt in 1995 and 1997. These two sites showed similar levels of species richness, Shannon-Wiener diversity index (*H'*), evenness (*E*), and Simpson's index of dominance (*D*) to the unburnt *Acacia mangium* plantation (Table 5.5). It seems that fallowing the sites after the fire helped the recovery of the sites. The more diverse invading vegetations following fire were likely important, confirming the important of both biotic and abiotic factors in predetermining the direction of succession.

Table 5.5. Long-term effect of wildfire on the VAM fungal community

Fire history	Ecological parameter			
	Species richness	Shannon-Wiener Diversity index (*H*')	Evenness (*E*)	Simpson's index (*D*)
AM	8.14 a	1.66 a	0.79 a	0.25 a
B95	9.43 a	1.65 a	0.76 a	0.27 a
B97	9.29 a	1.73 a	0.79 a	0.21 a

Values in the same column followed by the same letter are not statistically different (based on Tukey's t-test at p=0.05). Values are means for $n = 3$.

5.3.4. Effect of Landuse on Bacterial Population

In the current study the effects of three different land use systems on two groups of soil bacteria involved in nitrogen cycle and one group involved in P cycle were studied. The results showed that ecosystem types had a significant effect ($p<0.001$) on the bacterial population in soils of the study sites.

Conversion of natural forest into human-managed ecosystem decreased in bacterial populations except P-solubilizing bacteria in PF, which were significantly higher than those in the natural forest (Table 5.6). The lowest population of all microbial groups was found in HG, reflecting that increases in agricultural practices had caused more pressure on microbial populations.

Table 5.6. Microbial population in the natural and human-managed ecosystems

Site	NH_4^+-oxidizer	NO_2^--oxidizer	PSB	Rhizobium
	---- MPN g^{-1} soil, log x ----		--- cfu g^{-1} soil, log x ---	
CF	2.96 a	3.42 a	5.44 b	5.76 a
PF	2.98 a	3.33 a	5.83 a	5.50 b
HG	2.17 b	2.88 b	5.20 c	5.36 c

Values in the same column followed by the same letter are not statistically different (based on Tuke's t-test at p=0.05). Values are means for $n = 3$.

Both NH_4^+- and NO_3^--oxidizers appeared to have completed their growth within 6 weeks. The pH of growth media decreased significantly from 7.1 at the start of the incubation to 3.0 to 5.5 for the NH_4^+-medium and to 5.0 to 5.5 for NO_2^--medium at the end of the incubation period. Although conversion of natural forest into Pine plantation did not cause significant changes in the population of NH_4^+-oxidizers, the conversion into home garden decreased the population of NH_4^+-oxidizers significantly (Table 5.6). In general, populations (MPN) obtained in the current study were lower than those reported for a wide range of soils in temperate regions in the temperate regions (Both et al., 1990). The discrepancy of MPN between the results of the current study and those of Both et al., (1990) can be explained as follows. Provided that aerobic condition exists, the most regulating factor for nitrification is substrate availability. Both et al., (1990) in their study used grassland soils with high nitrite concentration. Although the availability of NH_4^+ was not measured in the current study, mineralization of organic N in the study sites presumably occurred, as reflected by low C/N ratio (11 to 15) of soils across the study sites.

Differences in climate may be also an important factor. In the current study, bacterial population fluctuated significantly according to period of measurement (Fig. 5.3), and population of both groups of nitrifying bacteria in wet season was higher than that in dry season. Two environmental variables that could exert a strong control on nitrifying bacteria include soil temperature and moisture content. Because of their slow growth rate and relatively inefficient metabolism, nitrifying bacteria are sensitive to temperature. The rate of nitrification has been confirmed to be very slow below 5ºC and above 40ºC (Alexander, 1977). In current study soil temperature at 10-cm depth ranged from 25.0 to 25.9ºC during wet season and from 26.1 to 28.7ºC during dry season. Although these ranges of soil temperature were within the range which did not exert significant influence, the results in the current study showed that under field conditions even small decreases and increases in soil temperature could have large effects on the population of nitrifying bacteria.

Because moisture affects the aeration regime of soil, the moisture status of microbial habitat could have marked influence on the population of nitrifrying bacteria. Since amonification and nitrification are carried out by autototrphic bacteria, water logging limits the diffusion of O_2, and nitrification is suppressed. On the other hand, at water shortage condition, bacterial proliferation is not retarded by the supply of O_2 but rather by insufficiency of water. A study by Tietema et

FIGURE 5.3.

al. (1992) found a nearly correlation between nitrate production and moisture content when organic matter from organic layer of acid forest soil was incubated at water content ranging from 10 to 29%. In the current study moisture content in all sites was consistently lower (26.2 to 39.2%) during dry season than wet season (41.1 to 44.5%). Therefore although the range of moisture content was still much higher than those used by Tietema et al. (1992), these ranges were enough to affect population of nitrifying bacteria in the field during the dry season. Because the water content itself seemed to be rather high, most of the water could not be considered as available source, judging from the high clay content of the soils.

The incorporation of gaseous N_2 is a major input of nitrogen to forest ecosystem. Some 68% of the nitrogen added annually to a temperate deciduous forest was estimated to be from N_2 fixation (Bormann et al., 1977), both through symbiotic and non-symbiotic process. However, data on N_2 fixation from tropical forest ecosystems are still noticeably lacking (Paul, 1978). In the current study, the potential of indigenous population of *Rhizobium* in different ecosystem in the tropics was investigated. Changes in ecosystem type significantly ($p<0.001$) affected population of indigenous *Rhizobium* and conversion of natural forest into pine plantation and agricultural land significantly ($p<0.05$) reduced the population of indigenous *Rhizobium* (Table 5.6).

Although the indigenous *Rhizobia* were abundant in the natural forest ecosystem (Table 5.6), leguminous shrubs growing on the floor were rarely nodulated. Similar results have also been previously reported for undisturbed primary forests in Central Amazonia in South America (Sylvester-Bradely et al., 1980). The lack of nodulation observed in the current study was due either to nitrogen equilibrium of the forest ecosystem or to insufficient supply of phosphorus. It has been shown in Chapter II that soil under natural forest in the current study had the highest potential total N content. Given the fact that this soil also had narrow C to N ratio (14), this N pools were presumably mineralizing, as was supported by the occurrence of nitrifying bacteria. It has been widely known that high availability of inorganic nitrogen inhibits biological nitrogen fixation (BNF) system. Suppressive effect of inorganic N on BNF has been shown in *Leucaena leucocephala* (Sanginga et al., 1987), *Alnus* spp. and *Myrica* spp. (Stewart and Bond, 1961), and *Acacia* spp. (Hansen and Pate, 1987). However, to relate the lack of nodulation in understorey legumes to high level of inorganic N availability in the current study is tenuous because the NH_4^+- and NO_3^--N released during mineralization was directly absorbed by plants or used by other microorganisms for their assimilation. Therefore it is assumed that phosphorus, not nitrogen, was the soil nutrient most limiting to nodulation on the leguminous shrubs found on the floor of the forested site. This seems to be reasonable because available P level in the CF site is low (14.81 mg P kg^{-1} of soil). However, this hypothesis still deserves further confirmation.

Significantly higher population of indigenous *Rhizobia* in the CF site than any other sites might also be due to high variability in leguminous understorey growth and to the presence of both fast- and slow-growing isolates. Because *Rhizobium* has been shown to be host specific, the existence of various legumes were enable these communities to proliferate and infect the appropriate host plants so they can persist in the soil. Some leguminous trees and shrubs appear to nodulate only with fast-growing strains of *Rhizobium*, some only with the slow-growing *Bradyrhizobium* and some with both *Rhizobium* and *Bradyrhizobium* (Danso et al., 1992).

It seems that the losses of some host plants due to conversion of natural forest into human-managed ecosystem (PF and HG) had a strong control on the significant reduction in population of the indigenous *Rhizoium*. Therefore only those adapted to the new ecosystem could persist in the soil. Such ecological selection occurred because only one species of leguminous shrub, i.e. *Mimosa* sp. grew in the PF site and two species of leguminous crops, i.e. bean (*Phaseolus vulgaris*) and peanut (*Arachis hypogea* L.) were planted in the HG sites. Similar to those found in the CF site, the isolates obtained in the PF sites also consisted of both fast- and slow-growing *Rhizobium* although those originating from the PF sites took shorter to form colonies than strains from the CF site (5 to 6 days to reach 1-mm colony size compared with 7 to 8 days for slow-growing isolates from the CF site). On the other hand all isolates originating from the HG site had colony size more than 1 mm in 4 days.

While *Phaseolus vulgaris* has been known to form a mutualistic symbiosis with *Rhizobium phaseoli*, and *Arachis hypogea* with *Rhizobium xinjiangensis*, respectively, it is likely that the slow-growing strains found in the CF sites belong to cowpea *Rhizobia*, which are nonspecific in nodulation. Current results confirm that there are real differences in the composition of populations of root-nodule bacteria both within the same sites with various host plants and between different sites with different host plants. The apparent differences in population and probably strains between sites reflect the selection pressure induced by changes in host plants due to ecosystem conversion. However, the taxonomy of these bacteria still requires further study before definitive generic allocations can be made.

In accordance with its adverse effects on the populations of bacteria involved in N cycling, the changes in ecosystem type also significantly ($p<0.001$) affected the population of P-solubilizing bacteria (PSB). However, a significant decrease in their population was observed only when the natural forest was converted into home garden. On the other hand the population of PSB in the PF site was significantly higher than that in the CF site (Table 5.6).

Soil microorganisms are important agents in nutrient cycle and energy flow in ecosystems and are sensitive to environmental changes. The specific environmental conditions, which select for the surviving *Rhizobia* in the study sites, have still to be defined. However, the conversion of natural forest (CF) into human-managed ecosystems (PF, AM, and HG) has been shown to cause significant reduction in moisture content, organic C and available P (Chapter II). In addition sparse plant cover and consequent limited shading in the HG site produced the highest soil temperature at 10-cm depth and the lowest moisture content. The averages of soil temperature in the PF, AM, and HG sites were 25, 26.3, and 27.8ºC during dry season and 24.9, 25.7, and 25.7ºC during wet season for, respectively.

It has been shown in Chapter II that after 22, 8, and 4 years of Pine (PF), Acacia (AM), and home garden (HG) cultivations, total C had declined by 16, 29, and 12%, respectively as compared with the undisturbed natural forest (CF). Results of regression and correlation analysis indicated that the decreases in total C showed small but significant relationship with decreases in the populations of P-solubilizing bacteria ($r=0.30$, $p<0.01$), *Rhizobium* ($r=0.65$, $p<0.01$), and NH_4^+-oxidizers ($r=0.22$, $p<0.05$) but not with NO_2^--oxidizers. These results imply that from a long-term viewpoint, the losses of passive pools of soil organic matter due to conversion of natural ecosystems into human-managed ecosystems may threaten the sustainability of nutrient supply in the systems. Although organic matter turnover in all human-managed ecosystem occurred, e.g. through litter fall in the PF site and through the incorporation in the HG site, the rate of losses not only through rapid decomposition but also and more importantly through erosion might be still higher. If this is the case, the biomass and activity of soil microbes in the study site will continue to decrease because of limited C supply.

It is not only the roots of the trees that affect the microbial community in soil, but also different mycorrhizal species and their exudates have been found to change soil bacterial communities (Olson and Wallancer. 1998). As has been described in the earlier section that HG had the lowest number of spores, which was a reflection of potential inoculum in the field. Therefore, higher VAM fungal population in the other three sites (CF, PF, and AM) might have given stronger effect on the bacterial communities in these three sites.

The three sites studied were located in the same soil and climatic zone, yet bacterial population was different between them. Hence it appears that both spatial variations in soil characteristics and differences in plant community play a large role in controlling the bacterial population in this study. Other investigators, however, have found that soil factors play a minor role in determining the diversity of root-associated bacteria (Germida et al., 1998). While our results indicate that bacterial population is affected both by plants and soil factors. It confirms that the relative influence of soil and plant factors on bacterial population may be dependent upon the plants species investigated. If the plant species plays a dominating role in controlling the ecology of the root-associated microbial community, practices that affect plant diversity may also alter plant-associated bacterial diversity.

5.3.5. Immediate Effect of Fire on Bacterial Population

It has been shown that the sudden effects of artificial heating (Chapter III) and the effects of wild fire on chemical characteristics (Chapter IV) were evident. Because soil microorganisms are sensitive to environmental changes, they respond directly to variations in their environment. An immediate and significant reduction in soil microbial population after fire has been well acknowledged (Senthilkumar et al., 1998). Fire affects soil organisms directly by killing or injuring the organisms, and indirectly by its effects on plant succession, soil organic matter transformations, and microclimate.

The population of bacterial communities significantly decreased immediately after fire (Table 5.7). Nitrifying bacteria were the most severely affected, with 96 and 56% reduction for NH_4^+- and NO_2^--oxidizers, respectively, relative to their population before fire. Similarly both PSB and *Rhizobium* lost about 37% of their initial population after the passage of the wildfire.

Living organisms including microbes are mostly located in the surface soil and, therefore, are directly exposed to heat radiated by flaming surface fuels and vegetations, and smoldering of litter and duff layer. Heat directly affects survival of soil microbial population because cellular components such as proteins, membrane lipids, and nucleic acids are unstable at 60 to 70ºC. As a result, the thickness and the extent

Table 5.7. Immediate and short-term effect of fire on microbial populations

Time	Microbial group			
	NH_4^+-oxidizer	NO_2^--oxidizer	PSB	Rhizobium
	---- MPN g^{-1} soil, log x ----		------- cfu g^{-1} soil ------	
Before fire	0.88 b	0.86 b	5.16 a	5.13 a
After fire	0.04 c	0.38 c	3.27 b	3.26 b
3 months after fire	3.62 a	2.45 a	5.15 a	4.62 a

Values in the same column followed by the same letter are not statistically different (based on Tuke's t-test at p=0.05). Values are means for $n = 3$.

of combustion in the duff layer are important factors affecting the survival of living organisms. Prior to the fire passage, the most recently burnt site (B00) was 8-yr old of *Acacia mangium* plantation, which had about 4 cm of litter layer above the mineral soil. Because wildfire occurring in this site had combusted this litter layer, most likely that microbes in the litter layer were killed outright, whilst those in the deeper part of the soil survived. Heating soil for 10 min at 70ºC kills non-spore-forming fungi, protozoa, and some bacteria, while temperature above 127ºC would nearly sterilize soil (Raison, 1979). Therefore, even low-severe fires can kill or injure organisms in soil, especially those near the surface. Soil temperatures during fire were not measured in the current study, however, significant reduction in microbial population observed was an indication of elevating temperature effect above the limit at which soil microbes could resist. Because soil samples were collected from 0- to 10-cm depth, significant decreases in microbial population observed in current study reflected that heat penetration might have reached the 10-cm depth. In addition, it was also found that wildfire in the study site occurred for more than 5 hr, allowing more heat to penetrate the deeper part of the soil.

The striking effect of heating on soil microbial population observed in the current study was also likely due to moist condition of the soil prior to fire. Physiologically active populations in moist soils are often more sensitive to temperature increases than dormant populations in dry soil. Results of a study in a chaparral ecosystem by Dunn et al. (1985) showed that survival of NO_2^--oxidizers was reduced to 1% at a temperature of about 70ºC if the soil was moist (20% water). In the current study a decrease of about 96% and 56% of the initial population of NH_4^+-oxidizers and NO_2^--oxidizers, and 37% of PSB and *Rhizobium* immediately after fire were observed (Table 5.7). Soil moisture content at 0- to 10-cm depth (measured in the adjacent site) was about 28% prior to fire and dropped to about 20% after the fire. It has been known that water increases both the heat capacity and the heat conductivity of soil. The increased heat capacity allows soil to absorb large amounts of heat without a subsequent increase in temperature until water is vaporized. Furthermore because of the increased thermal conductivity, moist soils also conduct heat more rapidly.

Several workers have reported rapid increase in microbial population after the first postfire rainfall, and after a few months their population may increase above, even more active than their original levels (Pritchett and Fisher, 1987). In the present study, bacterial population significantly increased three months after the fire passage (Table 5.7). The highest increase was observed in the nitrifying bacteria, with an increase of about 8,950 and 1,053% for NH_4^+- and NO_2^--oxidizers, respectively from their population immediately after the fire. PSB and Rhizoium also increased by 57 and 44%, respectively as compared to their levels immediately after the fire.

Survival of soil microbes subsequent to fire is affected by a variety of edaphic and biotic factors. The correlations between microbial populations with certain edaphic factors are given in Table 5.8. The increases in microbial population three months after fire are significantly ($p<0.05$ and $p<0.01$) correlated with the increases in moisture content, pH, available P, and exchangeable bases but not with total C and N. In addition potential predators and competing microorganisms might have been significantly reduced after fire. Reduced predation would influence the recovery of beneficial microbes observed in the current study.

Table 5.8. Correlation between certain soil characteristics and microbial population in the B00 site three months after fire

Microbial group	Soil Characteristic								
	Moist.	pH	C	N	P	Ca	Mg	K	Na
NH_4^+-oxidizer	0.88*	0.99**	0.74	0.75	0.97**	1.00**	0.98**	0.91**	0.41
NO_2^--oxidizer	0.87*	0.99**	0.74	0.76	0.99**	1.00**	0.98**	0.90**	0.40
PSB	0.88*	0.98**	0.74	0.76	0.99**	0.99**	0.98**	0.91**	0.44
Rhizobium	0.70*	0.97**	0.56	0.83	0.92**	0.94**	0.98**	0.94**	0.34

* and ** denote significant correlation at p<0.05 and p<0.01, respectively.

5.3.6. Left-over Effects of Fire on Bacterial Population

Although a rapid recovery of microbial population after burning has been shown above and by others both under laboratory and under field conditions (Serrasolsa and Khanna, 1995; Tanaka et al., 2001), the left-over effects of burning on the dynamics of soil microbes are still poorly understood. The results obtained from the B98 sites allowed us to study the left-over effects of wildfire on the dynamics of microbial population during two years. The populations of indigenous nitrifying bacteria, P-solubilizing bacteria, and rhizobium were significantly ($p<0.01$) varied over time (Fig. 5.4). In spite of their intra-periodic variability, the dynamics of each group of the investigated soil microbes showed different patterns of survival and can be summarized as follows.

1) The NH_4^+-oxidizers and NO_2^--oxidizers significantly increased in the first 9 and 15 months after the fire but thereafter the numbers of survivors fell off significantly, and returned to its original level 24 months after fire (Figs. 5.4a and b). In a study in shifting cultivation in Thailand, Tanaka et al. (2001) reported a significant increase in NH_4-N following burning, followed by a significant decrease in the rainy season. The increases in NH_4-N after burning were mainly derived from the thermal decomposition of biomass during burning instead of by microbial decomposition. This was also supported by the lack of changes in NO_3-N after burning reported in Tanaka et al. (2001), indicating the absence of NO_2-oxidizers. In the current study the

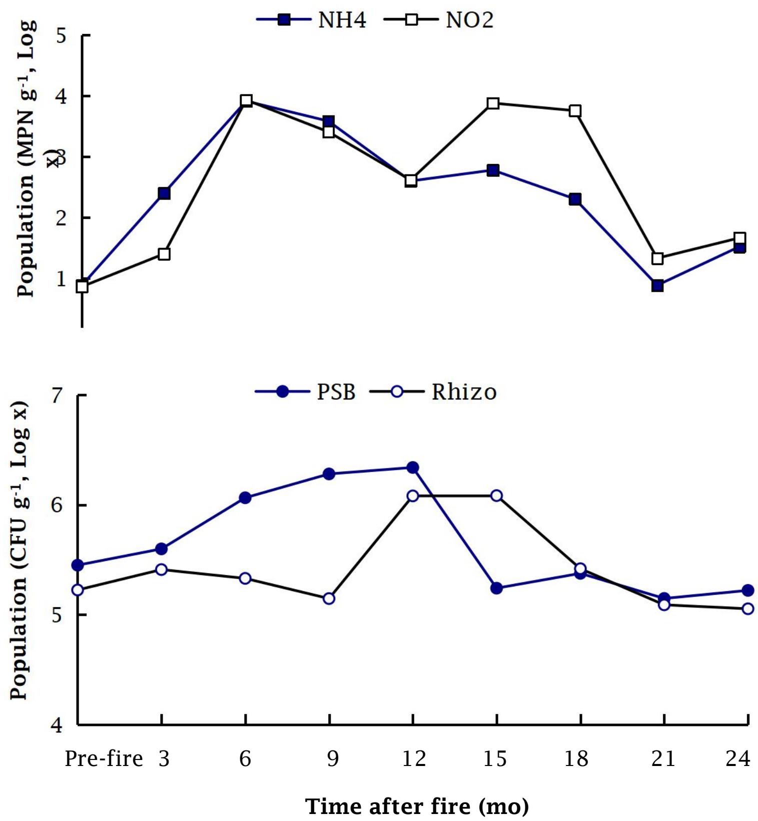

Fig. 5.4. Dynamics of bacterial population in the B98 site.

increases in the numbers of the nitrifying bacteria were initially observed 3 months after fire, coincided with the rainy season. Because the site had received enough precipitation and had experienced wet and dry cycles, the increases in the populations of the nitrifying bacteria reflected the increases in the nitrification process due to the availability of NH_4 and NO_2 ions from the burnt plant biomass. The increases in the number of NH_4-oxidizers were significantly correlated ($p<0.05$) with soil T-C, T-N, and pH. This suggested that soil reaction caused by ash input had created a favorable condition for decomposers to mineralize organic C and N, resulting enough amount of NH_4-N for nitrification.

The positive effects of fire on the nitrifiers population were found to be temporary, the population of the nitrifiers decreased as the elapsed time after the fire proceeded. Although the decreases in the population of nitrifiers coincided with the decreases in soil nutrients with time, no significant relationship between these two parameters could be obtained. Given suitable conditions, the limiting factor for nitrification is the supply of NH_4 ions. There was an indication that the increases in T-C and T-N were inversely followed by the decreases in population of the nitrifiers. This indicates that the growth of nitrifiers in the study site might have been limited by the availability of NH_4 ions. Although decomposition occurred as shown by a narrow C to N ratio of the soils (Chapter II), accumulation of organic C might have caused immobilization of inorganic N by decomposers, limiting the availability of NH_4^+ ions for nitrification. In addition, a part of the NH_4^+ ions released during organic N mineralization was absorbed by the regrowing plants, making NH_4^+ ions unavailable for use by the nitrifiers. It has been shown that nitrifiers have low energy efficiency. For Nitrosomonas, the free energy efficiency ranges from 5 to 14% while the efficiency of Nitrobacter has been calculated to be 5 to 10% (Alexander, 1977).

2) PSB significantly increased in the first 12 months after fire, then significantly decreased in the next 3 months, but thereafter the numbers did not changed and returned to its original level (Fig. 5.4c).

The initial increases in PSB were significantly correlated ($p<0.05$) with the increases in soil T-C, T-N, and pH. However, decreases in their population in the second 12 months after fire were not correlated with the decreases in the nutrient level of the soil. The lack of such effects was caused by the narrow range of the PSB populations found from 15 to 24 months after the fire (5.1 to 5.4 CFU g^{-1}). In addition it also indicates that the survival of PSB was also dependent on other factors, more important in the field than nutrient level, such as soil temperature as has been described in the previous section.

3) Rhizobium showed three phases of survival; no significant changes during the first 9 months, followed by a significant increase in the next 6 months but thereafter the numbers gradually fell-off and returned to the pre-fire level at the end of the second year (Fig. 5.4d). The fact that *Rhizobium* did not change during the first 9 months reflected their slow recovery after heat injury. This lag time for recovery was caused by the fact that *Rhizobia* are symbiotic microorganisms and reflected that their recovery was also affected by the existence of a host plant. Because the wildfire had burnt all vegetations in the study site, the survival of *Rhizobia* after fire relied on the invading leguminous plants, including *Mimosa* spp. and *Acacia mangium*.

Based on the results above it is clear that the positive effects of fire on the bacterial population are short-lived. The temporary improvement in soil nutrients (abiotic factors) after the fire was clearly an important factor for the increases in microbial population. Provided that the nitrifiers and PSB are free-living microorganisms in soils, their rapid recovery reflected their quick response to the favorable conditions induced by the fire. However, the lag time of recovery showed by *Rhizobia* also reflected the importance of biotic factors (vegetations) in determining the survival of certain groups of soil microorganisms, at least as shown by *Rhizobia* in the current study. It also applies to the depletions of bacterial populations one year after fire. The fact that the depletion in their populations was not significantly related with soil nutrient status indicated that the survival of soil microbes after fire was dependent not only on the abiotic factors but also on biotic factors or on the combination of the two.

5.4. Summary

Comparison of similar soil types with different management showed that the conversion of natural ecosystem into human-managed ecosystems caused a selective pressure on the microbial populations. The results indicated that sporulation, species richness, and Shannon and Wiener's diversity indices of VAM fungal communities significantly decreased as the natural forest was converted into planted forest or into agricultural land.

The immediate effect of wildfire on VAM fungal and bacterial community was striking but short-lived. Species richness and Shannon-Wiener Diversity index (*H'*) of VAM fungal community significantly decreased immediately after the fire but significantly increased three months after the fire. Three species, i.e. *G. versiforme* (Karsten) Berch, *A. bireticulata* Rothwell & Trappe, and *A. tuberculata* Janos & Trappe) found immediately after fire could not be revealed three months after fire. On the other hand four species, i.e. *G. reticulatum* Bhattacharjee & Mukerji, *G. gigantea* (Nicol. & Gerd.) Gerdemann & Trappe, *A. appendicula* Spain, Sieverding & Schenck, and *S. rubiformis* Gerdemann & Trappe were revealed. Given the fact that the findings were based on only two sampling times, to conclude that that these three species had totally disappeared from the site is tenuous. Instead these two findings reflected the dynamics of VAM fungi in relation to changes in the above ground vegetation. Based on the short- and long-term effects obtained it seems that the critical period for VAM fungal community development is between 3 to 6 months after fire. Fallowing appeared to help the biological recovery of the burnt sites.

Similar results were obtained for the population of the indigenous nitrifiers, P-solubilizing bacteria, and *Rhizobia*. In general conversion of natural forest into human-managed ecosystem decreased bacterial populations except P-solubilizing bacteria in PF. The lowest population of all microbial groups was found in HG, reflecting that increases in agricultural practices had caused more pressure on microbial populations.

Amongst the bacterial groups investigated, nitrifying bacteria were the most severely affected, with 96 and 56% reduction for NH_4^+- and NO_2^--

oxidizers, respectively relative to their population before fire. The decreases in the populations of PSB and *Rhizobium* were 37% of their initial population. However, bacterial populations significantly increased three months after the fire passage with the highest increase (8,950% and 1,053%) being observed for the NH_4^+- and NO_2^--oxidizers, respectively, followed by 57% and 44% for PSB and *Rhizobia*, respectively. The recovery of bacterial population three months after fire were significantly ($p<0.05$ and $p<0.01$) correlated with the increases in moisture content, pH, available P, and exchangeable bases but not with total C and N.

The assessment on the long-term effects of wildfire on the bacterial populations showed that the bacterial populations returned to the pre-fire levels two years after the fire occurrence. Based on the results obtained it is clear that the temporary improvement in soil nutrients (abiotic factors) after fire was clearly important for the increases in microbial populations. However, the lag time of recovery showed by *Rhizobia* also reflected the importance of biotic factors (vegetations) in determining the survival of certain groups of soil microorganisms. The repeated sampling technique employed in the current study was found to be very helpful tools to get the full reflection of microbial community without changing their natural niches in the field.

Chapter VI

Summary and Conclusion

The overall objective of the present study was to investigate both the immediate and delayed effects of wildfire on the characteristics of Ultisols in the tropics. This was carried out by investigating the changes in the chemical and biological characteristics of the soils both under laboratory and under field conditions.

The results of the preliminary study presented in Chapter II showed evident differences in soil characteristics between the natural forest and the human-managed ecosystems, and between the unburnt and the burnt sites. Based on the results obtained it was found that soils in the unburnt sites showed a better physico-chemical potential than those in the burnt sites. However, in general soils in the study sites showed a low natural fertility potential.

Artificial heating at 100 °C for a short period did not change soil chemical characteristics significantly but heating at 300 and 500 °C, a representative of *intensive* burning, for example during forest fires, adversely affected soil characteristics (Chapter III). Although watering to some extent showed positive effects on chemical characteristics, at field scale, care must be taken because heavy precipitation could cause nutrient leaching due to erosion.

The dynamics of T-C, T-N, available P, pH, exchangeable bases, and CEC was significantly affected by forest management practices and by wild fire occurrence (Chapter IV). The significant but short-lived increases in nutrients level immediately after fire indicated the positive effects of fire on available nutrients. Nutrients were still accumulating one year after the fire, then depleting after two years. Although fallowing appeared to help the burnt sites aggrading, the leftover effects of fire on soil nutrients were still evident five years after the fire. It indicated that nutrients losses during and after the fire could not be recovered at least during the first five years. Based on these results it is clear that the long-term effects of fire are deleterious to nutrient levels of the soils.

Conversion of natural forest into a human-managed ecosystem reduced number of spores, species richness, and Shannon-Wiener's diversity index, and changed genus and species composition of VAM fungal communities (Chapter V). Similarly, bacterial population decreased as the natural forest was converted into a human-managed ecosystem except for the P-solubilizing bacteria in the pine forest. The lowest population of microbial groups was found in the home garden, reflecting the increases in the intensity of human impact on microbial population.

Although decreasing immediately after the fire, species richness and Shannon-Wiener Diversity index (*H'*) of VAM fungal community significantly increased three months after the fire. Fire occurrence also changed the species composition of VAM fungi. Bacterial populations significantly increased three months after the fire with the highest increase (8,950% and 1,053%) being observed for the NH_4^+- and NO_2^--oxidizers, followed by 57% and 44% for PSB and *Rhizobia*, respectively. The significant correlation between bacterial population and increases in soil nutrients three months after the fire reflected the importance of edaphic or abiotic factors in determining the survival of soil microbes after fire disturbance. However, the lag time of recovery to 12 months after the fire shown by *Rhizobia* also reflected the importance of biotic factors, for example vegetations, in determining the survival of soil microorganisms.

Site productivity can be maintained if anthropogenic disturbances that result in nutrient loss are balanced with the natural rates of nutrient inputs. Given the evident long-term effects of wildfire on soil chemical properties, which are still observable five years after the fire occurrence, coupled with increasing human pressures, it is likely that declines in long-term site productivity will occur. Although the deleterious effects of fire on the biological properties of the soils are short-lived, the long-term unfavorable chemical conditions following fire may have negative effects on the biological diversity of soils. Therefore, to protect land and biological resources and to maintain the sustainability of ecosystems in the tropics, prevention of forest fire is one of the most important practices to be employed.

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