Conversion of lowland forests to rubber and oil palm plantations changes nutrient leaching and nutrient retention efficiency in highly weathered soils of Sumatra, Indonesia

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Summary

Over the last two decades, Sumatra, Indonesia has experienced rapid expansion of rubber and oil palm plantations through conversion of rainforests. This is evident from the 36% decrease in forest area in this region from 1990-2010. Such rapid land-use change necessitates assessment of its environmental impacts. Forest conversion to rubber and oil palm plantations are expected to increase nutrient leaching losses and decrease nutrient retention efficiency, following the changes in soil cover, litter input, soil nutrient availability and management practices. This thesis presents two studies, which focused on the impact of forest conversion to rubber and oil palm plantations on nutrient leaching and nutrient retention efficiency, and on the difference in nutrient leaching losses between fertilized and frond-stacked areas of oil palm plantations. All studies were conducted in two landscapes of highly weathered soils that mainly differed in texture (loam and clay Acrisol soils), located in the Jambi province, Sumatra, Indonesia. Nutrient leaching losses were measured using suction cup lysimeters installed at 1.5 m soil depth and sampling frequency was bi-weekly to monthly during February to December 2013.

In the first study, nutrient leaching losses and nutrient retention efficiency in the soil were measured in four land uses: the reference land uses of lowland forest and jungle rubber (rubber trees interspersed in secondary forest), and the converted land uses of smallholder rubber and oil palm plantations. In each landscape, the first three land uses were represented by four replicate sites and the oil palm by three sites, totaling 30 sites. The results illustrated that for the reference land uses the loam Acrisol soil had higher leaching fluxes of dissolved nitrogen (N) and base cations, and lower retention efficiencies of N and base cations than the clay Acrisol soil. For the converted land uses, management practices such as fertilization and liming in oil palm plantations resulted in higher dissolved N, dissolved organic carbon (DOC), and base cations leaching fluxes, and lower N and base cation retention efficiencies in the soil than the reference land uses. On the other hand, in the unfertilized rubber plantations leaching losses of dissolved N, DOC, and base cations were lower than in the oil palm plantations. Overall, the results showed that clay content and management practices controlled nutrient leaching losses and nutrient retention efficiencies in heavily weathered Acrisol soils of these converted landscapes.

In the second study, nutrient leaching losses were measured in fertilized and frond-stacked areas of smallholder oil palm plantations in clay and loam Acrisol soils. The results exhibited higher leaching losses (i.e. N, base cations, total aluminum (Al), total manganese (Mn), total sulfur (S), and chloride (Cl)) in the fertilized area than the frond-stacked area due to pulse rates of applications of mineral fertilizers and lime. At the landscape scale, higher soil nutrient stocks and lower nutrient leaching losses in the clay Acrisol soil compared to the loam Acrisol soil both in the fertilized and frond stack areas were caused by the higher nutrient retention as a result of higher clay content.

Combining nutrient leaching losses and nutrient input (i.e. bulk precipitation and fertilizers) with ancillary studies on nutrient output through harvest export provides more comprehensive information about the changes in partial nutrient budgets of N, phosphorus (P), and base cations due to forest conversion to oil palm and rubber plantations. Fertilized oil palm plantations had the lowest annual partial budget of N, calcium (Ca) and magnesium (Mg) due to the high annual leaching losses and harvest export. However, the high negative partial budgets of N, Ca and Mg in oil palm plantations did not significantly decrease those stocks at 1-m soil depth compared to all the other land uses, except for exchangeable Mg in the loam Acrisol landscape. Even though unfertilized rubber plantations have lower leaching losses (e.g. P) than forest, harvest export caused the lower annual partial budget of P. Overall, these results from the two studies suggests for improved management practices on these highly weathered soils through synchronizing rate of application of fertilizer with plant uptake and frequency of fertilizer application.

Zusammenfassung

In den letzten zwei Jahrzehnten wurden in Sumatra (Indonesien) große Regenwaldflächen für den Anbau von Kautschuk- und Palmölplantagen zerstört. Dies zeigt sich in der Abnahme Waldfläche in dieser Region um 36% zwischen 1990-2010. Eine solch schnelle Landnutzungsänderung hat Auswirkungen auf die Umwelt: Es ist davon auszugehen, dass die Zerstörung von Regenwald und die Etablierung von Kautschuk- und Palmölplantagen aufgrund von Einflüssen auf die Bodenoberfläche, Veränderungen von Streufall, Nährstoffverfügbarkeit und Management in den Plantagen zu erhöhter Nährstoffauswaschung und einer verminderten Nährstoffretentionseffizienz führt. Diese Arbeit stellt zwei Studien vor, die sich mit den Auswirkungen der Regenwaldzerstörung - und der einhergehenden Kultivierung von Kautschuk und Ölpalmenbäumen - auf Nährstoffauswaschung und Nährstoffretentionseffizienz beschäftigt. Außerdem untersucht sie Unterschiede in der Nährstoffauswaschung zwischen gedüngten und mit Palmwedeln bedeckten Bereichen in Palmölplantagen. Beide Studien wurden in zwei Landschaften der Provinz Jambi (Sumatra, Indonesien) mit stark verwitterten Acrisol-Böden durchgeführt, die sich in der Bodenart unterscheiden (lehm- bzw. tonhaltiger Acrisol). Die Nährstoffauswaschung im Boden wurde mit Saugkerzen-Lysimetern gemessen, die in 1,5m Tiefe im Boden installiert wurden. Beprobt wurde von Februar bis Dezember 2013 zweiwöchentlich bis monatlich.

Die erste Studie beschäftigt sich mit der Nährstoffauswaschung und Nährstoffretentionseffizienz im Boden vierer verschiedener Landnutzungsarten. Dabei handelt es sich um die zwei Referenznutzungsformen Tieflandregenwald sowie Sekundärwald durchsetzt mit Kautschukbäumen, als auch um die veränderten Landnutzungsformen kleinbäuerlicher Kautschuk- und Ölpalmplantagen. Jede Landnutzung, ausgenommen der Palmölplantagen mit drei Wiederholungen, wurde durch vier Wiederholungsflächen innerhalb jeder Landschaft repräsentiert. Somit wurde die Studie auf insgesamt 30 Flächen durchgeführt. Die Ergebnisse zeigen für den lehmigen

Acrisol-Boden der Referenzflächen eine höhere Auswaschung und eine niedrigere N-Retentionseffizienz für Stickstoff (N) und basische Kationen, verglichen mit dem tonigen Acrisol-Boden bestanden. In den Palmölplantagen zeigte sich, dass Düngung und Kalkung zu erhöhter Auswaschung von gelöstem N, gelöstem organischen Kohlenstoff (DOC) und basischen Kationen führte, sowie zu einer geringeren Retentionseffizienz von N und basischen Kationen im Boden. In den ungedüngten Kautschukplantagen dagegen waren die Auswaschungsverluste von gelöstem N, DOC und basischen Kationen geringer als in den Palmölplantagen. Zusammenfassend zeigten die Ergebnisse, dass Nährstoffverluste und Nährstoffretentionseffizienz in Kautschuk- und Palmölplantagen auf stark verwitterten Acrisolen primär von Tongehalt und Management abhängen.

In der zweiten Studie wurde die Nährstoffauswaschung in den gedüngten und mit Palmwedeln bedeckten Bereichen in Palmölplantagen von Kleinbauern in lehm- bzw. tonhaltigen Acrisolen gemessen. Die Ergebnisse zeigten höhere Auswaschverluste (d.h. N, basische Kationen, Gesamt-Aluminium, Gesamt-Mangan, Gesamt-Schwefel und Chlor) in den gedüngten Bereichen als in den mit Palmwedeln bedeckten Bereichen aufgrund der Frequenz des Mineraldünger- und Kalkeinsatzes. Auf Landschaftsebene wurden die höheren Bodennährstoffvorräte und eine niedrigere Nährstoffauswaschung im Ton-Acrisol im Vergleich zum Lehm-Acrisol sowohl in den gedüngten als auch in den mit Palmwedeln bedeckten Bereichen durch die höhere Nährstoffretention (als Ergebnis höheren Tongehaltes) verursacht.

Die Kombination von Nährstoffauswaschung und Nährstoffeintrag (d.h. Gesamtniederschlag und Dünger) mit zusätzlichen Informationen über den Nährstoffaustrag durch die Ernte, geben uns umfassendere Informationen über die Veränderungen im partiellen Nährstoffhaushalt von N, Phosphor (P), und basischen Kationen bei Waldumwandlung zu Palmöl- und Kautschukplantagen. Gedüngte Palmölplantagen hatten aufgrund der hohen jährlichen Nährstoffauswaschung und des Ernteexports das niedrigste jährliche Teilbudget an N, Kalzium (Ca) und Magnesium (Mg). Dennoch verringerten die hohen negativen Teilbudgets von N, Ca und Mg in den

Palmölplantagen nicht deren Vorräte in 1m Bodentiefe verglichen mit den anderen Landnutzungsformen - außer für austauschbares Mg im Lehm-Acrisol. Obwohl ungedüngte Kautschukplantagen geringere Auswaschung zeigen als der Wald (z.B. für P), führte der Ernteexport zu einem geringeren jährlichen P-Teilbudget. Insgesamt implizieren die Ergebnisse der beiden Studien folgende verbesserte Managementverfahren für diese hochverwitterten Böden: eine Synchronisation der Düngermenge mit der Pflanzenaufnahme sowie eine Anpassung der Düngungshäufigkeit.

Chapter 1

General Introduction

1.1 Overview of deforestation in Sumatra, Indonesia

Forests can play a key role in maintaining soil fertility and nutrient balance, which is manifested by the high nutrient cycling rates and low nutrient loss by leaching measured in Indonesian forests (Dechert et al., 2004, 2005; Allen et al., 2015). However, in some regions, the role of forests in providing environmental services and supporting sustainable ecosystems has been continuously declining in the past two decades due to rapid conversion of forests to agricultural land-uses. Southeast Asia is experiencing rapid expansion of agricultural land area through rainforest conversion. According to forest resources assessments, tropical forest covered 247.3 million ha in 11 Southeast Asia countries in 1990, which was reduced to 214.1 million ha in 2010. Indeed, the deforestation rate in Southeast Asia from 1990 to 2010 was approximately 1.7 million ha per year, of which 1.2 million ha per year came from Indonesia (FAO, 2010). In Indonesia, the island of Sumatra experienced primary forest loss of approximately 36% (7.53 million ha) during this period, most of which (70%) occurred in the provinces of Riau, Jambi and South Sumatra. In total, forest-cover loss in Sumatra accounted for 31% of the total forest loss in all of Indonesia from 1990 to 2010 (Margono et al., 2012).

Most of the deforestation in Sumatra has been driven by the expansion of oil palm plantations and pulp and timber operations (Margono et al., 2012), but the rapid conversion of lowland forest to rubber (both agroforest/jungle rubber and monoculture) and other crops has also played a role (Broich et al., 2011; Villamor et al., 2014). The main actors of deforestation in Sumatra during the last two decades has consisted of large investors (i.e. large companies) and smallholders/small investors (i.e. urban-based businessman, government employees) (Holmes, 2002). According to Statistics Indonesia (2013a, 2013b), the structure of ownership in oil palm plantations consists of 51% of the area owned by smallholders and 49% of the area owned by large-scale enterprises (both state and private), whilst rubber plantations are dominated by smallholders (88%).

1.2 Soil nutrient leaching losses

Nutrient leaching losses are defined as the downward movement of dissolved nutrients below the rooting zone by percolating water (Lehman and Schroth, 2003). Nutrient leaching occurs when soil pores fill with rain water and gravity pulls the water down through the soil profile, thereby carrying away dissolved nutrients. Nutrient leaching losses can cause negative impacts such as groundwater contamination and increased operational/production costs in intensive agricultural systems (Goh and Härdter, 2003; Caliman et al., 2007).

Under natural conditions, nutrient leaching is controlled by climatic and soil factors (Kump et al., 2000; Lehman and Schroth, 2003). Climatic factors (i.e. precipitation, temperature, solar radiation, humidity and wind speed) control water usage of plants and water supply, both of which play an important role in determining soil drainage flux. Precipitation also regulates nutrient leaching losses through nutrient input from bulk precipitation (Havlin et al., 1999; Corre et al., 2010). Therefore, nutrient leaching losses are generally higher in humid climates as compared to dry climates (Havlin et al., 1999). Both precipitation and temperature also indirectly affect nutrient leaching losses, as they affect mineralization of soil organic matter (SOM) and weathering, releasing nutrients which not only can be taken up by plants but also can be lost through run off and leaching to ground water (Wright et al., 1998).

Soil characteristics affect nutrient leaching losses through 1) physical characteristics (e.g. texture, structure, soil porosity) which influence nutrient retention, water infiltration, water holding capacity, and percolation, and 2) biochemical characteristics (e.g. nutrient availability, pH, cation exchange capacity, SOM) which reflect nutrient supply and storage as influenced by weathering and mineralization (Silver et al., 2000; Lehman and Schroth, 2003; Mdemu, 2015). Soils with high nutrient retention and water holding capacity, and low water infiltration such as clay soil, generally have low nutrient leaching (Ohta et al., 1993; Lehman and Schroth, 2003). Conversely, nutrient leaching losses are usually higher in sandy-textured soils with high soil macroporosity that allow water to drain more easily (Ohta and Effendi, 1992; Silva et al., 2005). Hydrologic losses of nutrients are also controlled by weathering processes, which in part control solute concentration. Hedin et al. (2003) confirmed that the concentration of nutrients in soil solution was lower in an old soil (150000 yrs) than in a young soil (300 yrs). Furthermore, mineralization of SOM releases a large amount of nutrients to soil solution, and therefore nutrient leaching losses tend to be higher in soils with high mineralization rates than in soils with low mineralization rates. Nevertheless, soils with high mineralization rates can have low nutrient availability (i.e. NH_4^+-N) due to microbial immobilization (Allen et al., 2015), and may consequently have relatively low nutrient leaching losses. Silva et al. (2005) recorded lower NO_3^- leaching in a soil with a high C:N ratio than in a soil with a low C:N ratio, which they attributed to the higher C:N ratio leading to N immobilization and hence low net N mineralization.

In addition to climatic and soil factors, leaching losses from converted land uses can also be affected by management practices. In a study comparing soils with the same climatic conditions and soil type (Fluvic Cambisol), conversion of old-growth forest to a cacao-agroforest system (6-7 yrs-old) in Sulawesi, Indonesia, increased leaching losses of N, K, Ca, and Mg (Dechert et al., 2005). In the early period of conversion from forest to agricultural land uses, increases in nutrient leaching losses may relate to: 1) increased water percolation due to the temporarily reduced water use by vegetation, 2) increased nutrient release from decomposition and mineralization of the felled biomass (i.e. leaves), ash from burning, and dead roots, and 3) decreased nutrient uptake by plants due to the inactivation of the root systems of the former vegetation (Malmer et al., 2005). With time, nutrient leaching losses in agricultural land with no soil amendment (i.e. fertilizer) will usually decrease due to the declining store of available nutrients (Dechert et al., 2004; Kimetu et al. 2008; Ngoze et al. 2008). Conversely, soil nutrient stocks and nutrient leaching losses in agricultural land uses may increase with more intensive fertilizer application (Goh et al., 2003).

1.3 Management practices in oil palm plantations and its impact on soil fertility and nutrient losses

Oil palm (*Elaeis guineensis*) has been cultivated on approximately 11.7 million ha of land in Southeast Asia, 11 million ha of which is located in Indonesia and Malaysia (FAOSTAT, 2013). Smallholder oil palm plantations account for 40% of total oil palm area in Indonesia and Malaysia whilst the remaining 60% is owned by large-scale enterprises (both state and private) (Nagiah and Azmi, 2012). According to the roundtable for sustainable palm oil, smallholders are defined as family-based enterprises producing palm oil from less than 50 ha of land, often around 2 ha (Vermeulen and Goad, 2006). In Indonesia, there are two types of smallholders: smallholders working under the nucleus estate scheme (NES) and independent smallholders. For the NES, smallholders cultivate oil palm under the contract of state-owned or private plantation companies and receive technical assistance (i.e. land preparation, planting and maintenance) from the company whilst independent smallholders plant oil palm independently (Comte et al., 2012).

Soil management practices in smallholder plantations are generally less intensive as compared to the industrial oil palm plantations (state and private companies). Smallholders usually apply less fertilizer than oil palm companies and the dose of fertilizer may not be determined using leaf diagnosis and soil analysis due to economic considerations and/or lack of knowledge (Feintrenie et al., 2010; Comte et al., 2012). In the industrial oil palm plantations, fertilizers (inorganic and organic) are managed by block (planting areas with size 25–30 ha) and applied twice a year, by hand, on the soil surface around the palm tree or sprayed by airplane (Caliman et al., 2002; Comte et al., 2013). In both types of plantations (companies and smallholders), fronds are usually cut and deposited in frond piles along inter-rows to decompose and recycle nutrients (Comte et al., 2012).

Soil management practices (i.e. fertilization, liming, and pruning) play an important role in soil nutrient stocks and hydrologic loss of nutrients via leaching in oil palm plantations. In a mature oil palm plantation (17-25 years old) on Acrisol soil an in Sarawak, Malaysia, the higher soil NH_4^+ (measured in the top 0.15 m depth) in the fertilized and in the frond-stacked areas than in the harvest path area indicated that management practices increased soil N availability (Anuar et al., 2008). Management in oil palm plantation controls nutrient leaching losses, since the vertical movement of nutrients in the soil profile is predominantly determined by nutrient availability. In an Acrisol soil, higher leaching losses of nutrients (i.e. Ca, Mg, NH_4^+, NO_3^-) in fertilized as compared to unfertilized areas have been observed in a young (4 years old in Nigeria) and in a mature (26 years old in Malaysia) oil palm plantation (Omoti et al., 1983; Tung et al., 2009).

1.4 Aims and Hypotheses

This study was conducted as part of subproject A05 "Trace gas fluxes and soil N cycling in heavily weathered soils under rainforest transformation systems" within CRC990: EEFForTs in the province of Jambi, Sumatra, Indonesia. The first study (chapter 2) focused on the impact of forest conversion to oil palm and rubber plantations on nutrient leaching losses and nutrient retention efficiency. The aim of this study was to assess: 1) how soil physical and biochemical characteristics affect nutrient leaching losses in highly weathered soils, and 2) the impact of forest conversion to oil palm and rubber plantations on leaching losses and on nitrogen and base cation retention efficiencies. The following hypotheses were tested: 1) for the reference land uses (forest and jungle rubber), clay Acrisol soil have higher nutrient retention and lower leaching fluxes compared to loam Acrisol soil, 2) oil palm plantations with management practices (i.e. fertilization and liming) will have the highest nutrient leaching losses and consequently the lowest nutrient retention whereas rubber plantations with no fertilizer input will have the lowest nutrient leaching losses.

The second study (chapter 3) focused on nutrient leaching losses in fertilized and frond-stacked areas in smallholder oil palm plantations. The aim of this study was to assess how in smallholder oil palm plantations soil management such as spreading fertilizer around the palm trees and stacking palm fronds in the palm inter-rows affects leaching losses in Acrisol soils with differing soil texture. The following hypotheses were tested in this study: 1) fertilized areas around each palm tree will have higher soil nutrient stocks and nutrient leaching losses due to the pulsed nature of nutrient addition while under frond stacks leaching will be minimal since the slow mineralization of nutrients from decomposing fronds will be taken up by roots before it is lost through leaching, 2) soils with higher clay content will have higher soil nutrient levels and lower nutrient leaching losses both in the frond stack and fertilized areas than in soils with lower clay content.

1.5 Sites

The study took place in the Bukit Duabelas National Park, Harapan Rainforest, and in the area of Sarolangun and Batanghari regency within Jambi Province, Sumatra, Indonesia (Fig. 1.1). The research was conducted in two landscapes on highly weathered soils that mainly differed in soil texture: loam Acrisol and clay Acrisol. Four land-use types were selected in each landscape: lowland forest, rubber trees interspersed in secondary forest (hereafter, jungle rubber), and smallholder plantations of monoculture rubber and oil palm (Fig. 1.1). In each landscape, the three land uses (i.e. forest, jungle rubber, and rubber plantation) was represented by four replicate sites and the oil palm plantation by three sites, totaling to 30 sites. The size of each replicate plot was 50 m x 50 m with a minimum distance of 200 m between plots. The site information is described in more detail in the following chapter.

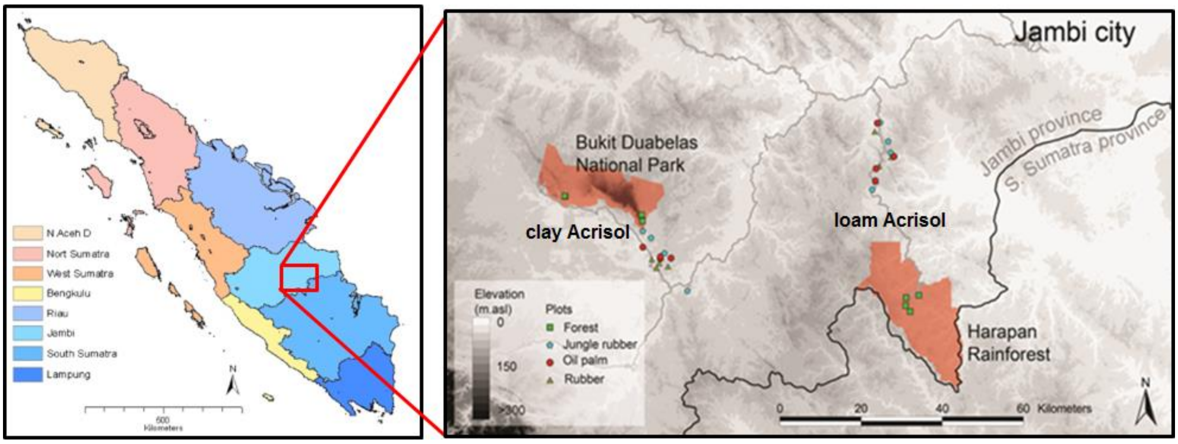

Fig. 1.1. Map of Sumatra (left) and the study sites located in four land uses (i.e. forest, jungle rubber, rubber and oil palm plantations) within the province of Jambi. Picture adapted from http://cdn.iopscience.com/images/1748-9326/7/3/034010/Full/erl428965f1_online.jpg (left) and map created by Oliver van Straaten (right).

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Chapter 2

Conversion of lowland forests to oil palm and rubber plantations impacts nutrient leaching losses and nutrient retention efficiency in highly weathered soils in Sumatra, Indonesia

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Abstract

Rapid rates of deforestation are occurring in tropical regions due to increasing global demands for palm oil and rubber. We examined the impact of forest conversion to oil palm and rubber plantations on soil nutrient leaching losses and nutrient retention efficiency. Our study was conducted in two landscapes with highly weathered soils (loam and clay Acrisol) in the province of Jambi, in Sumatra, Indonesia. Within each landscape, we investigated four land-use types: two reference land uses, lowland forest and jungle rubber (i.e. rubber interspersed in secondary forest), and two converted land uses, smallholder rubber and oil palm plantations. In each landscape, the first three land uses were represented by four replicate sites and the oil palm by three sites, totaling 30 sites. In each site, we measured leaching losses using suction cup lysimeters installed at 1.5-m soil depth. Soil water was sampled bi-weekly to monthly from February to December 2013. In the reference land uses, the clay Acrisol landscape had better soil biochemical characteristics and showed lower dissolved N and base cations leaching fluxes or, conversely, higher retention efficiency of N and base cations in the soil than the loam Acrisol landscape. Management practices in the converted land uses strongly influenced nutrient leaching losses. The fertilized oil palm plantations had higher dissolved N, organic C and base cation leaching fluxes, and lower N and base cation retention efficiencies in the soil than the reference land uses. The unfertilized rubber plantations had lower leaching fluxes of these elements than the oil palm plantations. High N fertilization in oil palm plantations of the loam Acrisol landscape had decreased soil solution pH and increased dissolved Al. Our results call for improved management practices in oil palm plantations on these highly weathered soils to minimize acidification and leaching effects on ground water quality.

2.1 Introduction

Rainforests can play an important role in maintaining ground water quality in tropical regions; however, in some regions their effectiveness may have decreased as a consequence of forest conversion to agricultural land. From 1990 to 2010, the global deforestation rate was approximately 13 million ha per year, of which 3 million ha per year occurred in South and Southeast Asia (FAO, 2010). In Indonesia, the province of Jambi (in Sumatra) experienced loss of primary forest by approximately 40% from 1990 to 2010, which accounts for 15% of the total primary forest loss in all of Sumatra island (Margono et al., 2012). The two most common land uses for converted forest in Jambi are oil palm and rubber plantations. From 2000 to 2010, the area of oil palm plantations in Jambi increased by approximately 85% whereas rubber plantations increased by 19% (Luskin et al., 2013). The expansion of rubber and oil palm plantations has had sizable benefits, by increasing the income of Jambi in general and of the smallholders in particular (Rist et al., 2010; Statistics of Jambi Province, 2012). Nevertheless, forest conversion also has negative environmental effects, including loss of soil carbon stocks (van Straaten et al., 2015), reduction in soil N availability (Allen et al., 2015) and decrease in stocks of exchangeable base cations in the soil due to leaching losses (Dechert et al., 2005).

The two major factors that influence nutrient leaching losses after forest conversion in a region with the same climatic conditions are soil texture and management practices. Soil texture affects nutrient leaching losses through its effect on soil fertility (e.g., cation exchange capacity, decomposition, and nutrient cycling) and water-holding capacity (Silver et al., 2000; Sotta et al., 2008; Allen et al., 2015). Soils with high clay contents have high cation exchange capacity and high nutrient cycling, which aid in retaining nutrients in the soil (Ohta et al., 1993; Allen et al., 2015), whereas coarse-textured soils are particularly conducive to nutrient leaching due to low nutrient retention (Lehman and Schroth, 2003). However, in heavily weathered soils such as Acrisols, which dominate the converted lowland landscapes in the province of Jambi (FAO et al., 2009), even soils with high clay contents may not contain high amounts of base cations, as the clay exchange sites are already saturated with exchangeable Al (Ohta et al., 1993; Allen et al., 2015). Finally, water percolation (which moves nutrients through the soil profile) is largely controlled by soil texture. Clay soils can hold a large amount of water against the force of gravity due to their large surface areas and dominance of small pores. Coarse-textured soils have large pores that allow water to drain easily, and consequently, the potential for leaching losses of dissolved solutes increases (Lehman and Schroth, 2003; Fujii et al., 2009). In summary, the typical characteristics of increasing clay content with depth in Acrisol soils may slow down water percolation and reduce nutrient leaching losses, leading to nutrient retention and consequently conserve soil fertility (Ohta and Effendi, 1992; Ohta et al., 1993; Silva et al., 2005).

In areas that have undergone forest conversion, soil management practices (e.g. fertilization and liming) also plays an important role in influencing nutrient leaching, since the magnitude of nutrients moving downward in the soil profile is predominantly driven by the availability of those nutrients (Dechert et al., 2004, 2005). Without fertilization, nutrient leaching losses in agricultural land will usually decrease in the years following forest conversion due to the declining store of available nutrients in the soil (Dechert et al., 2004). This would be the case in most rubber plantations as they are not - or only rarely - fertilized (Aweto, 1987). However, soils in oil palm plantations are very often supplemented with chemical fertilizer and lime applications to augment nutrient availability (Goh et al., 2003). In cases where oil palm plantations are regularly fertilized, nutrient leaching losses in older plantations may actually be higher than in younger ones, since nutrients may have accumulated in the soil over time (Omoti et al., 1983; Goh et al., 2003). As a consequence, nutrient leaching losses in agricultural land with regular fertilizer inputs are typically higher than in primary forest (Silva et al., 2005). In addition, application of fertilizer typically decreases nutrient retention efficiency in the soil-plant system due to decreases in microbial immobilization and plant uptake efficiency (Keuter et al., 2013; Hoeft et al., 2014). The low nutrient retention efficiency may also drive increases in nutrient leaching losses in agricultural land with regular fertilizer application.

Despite a growing body of information on the effects of deforestation on soil properties and processes, there is a clear lack of information on how rainforest transformation to tree cash crops, like oil palm and rubber, affects nutrient leaching and the efficiency with which nutrients are retained in the soil. Our study aimed to assess: 1) how soil physical and biochemical characteristics affect nutrient leaching losses in highly weathered soils, and 2) the impact of forest conversion to oil palm and rubber plantations on leaching losses and on nitrogen and base cation retention efficiencies. We hypothesized that: 1) for the reference land uses (forest and jungle rubber), clay Acrisol soil have higher nutrient retention and lower leaching fluxes compared to loam Acrisol soil, and 2) oil palm plantations with management practices (i.e. fertilization and liming) will have the highest nutrient leaching losses and consequently the lowest nutrient retention whereas rubber plantations with no fertilizer input will have the lowest nutrient leaching losses.

2.2 Materials and methods

2.2.1. Study sites and experimental design

The study area is located in the lowlands (35–95 m above sea level) of Jambi Province, Sumatra, Indonesia. The climate is humid tropical with a mean annual air temperature of 26.7 ± 0.1 °C and a mean annual precipitation of 2235 ± 385 mm (1991–2011; Jambi-Sultan-Thaha airport data from the Meteorological, Climatological and Geophysical Agency). The dry season is usually from May to September and the rainy season occurs from October to April. During our study period (2013), the wet season lasted slightly longer, while a drier period was detected between mid-June until end of October. During this dry season, rainfall was reduced by 35–57% compared to the wetter months during which rainfall was 333–362 mm per month.

We selected two landscapes that were both dominated by heavily weathered Acrisol soils but differed in soil texture: loam (36 ± 6% sand, 32 ± 4% silt and 32 ± 2% clay in the top 0.5 m) and clay (26 ± 6% sand, 29 ± 3% silt and 45 ± 4% clay in the top 0.5 m). This textural difference led to differences in soil fertility: forest sites in the clay Acrisol had higher base saturation, Bray-extractable P and lower Al saturation compared to those in the loam soil ($p \leq 0.01$ to 0.04; Table S1; Allen et al., 2015). The loam Acrisol landscape is in the Batanghari regency, 80 km southwest of Jambi City (01.79° S, 103.24° E and 2.19° S, 103.36° E). The forest sites in this landscape were within the Harapan Forest Reserve (administered by the Restoration Ecosystem Indonesia Harapan, PT REKI). The clay Acrisol landscape is part of the Sarolangun regency and the National Park Bukit Duabelas, 160 km southwest of Jambi City (01.94° S, 102.58° E and 02.14° S, 102.85° E). The forest sites in this landscape were within the Bukit Duabelas National Park (administered by the Ministry of Forestry, PHKA). Acrisol soils dominate the lowland area converted to plantations. They cover about half of the land area in Sumatra and about one third of Indonesia (FAO et al., 2009).

Within each soil landscape, we selected four land-use types: lowland forest, secondary forest with rubber trees (hereafter, jungle rubber), and smallholder plantations of rubber and oil palm (Table S2.2). Rubber and oil palm plantations were established on logged and/or burned forest or jungle rubber sites (Euler, 2015), and thus we consider both forest and jungle rubber as reference land uses that represent the baseline conditions with which we compared the converted smallholder plantations. Within each landscape, forest, jungle rubber and rubber were represented by four replicate sites and the oil palm by three sites, totaling 30 sites. In the clay Acrisol landscape, one landowner sold his oil palm plantation and nullified our contract for access to continue sampling; in the loam Acrisol landscape, the lysimeter for soil water sampling in one oil palm plantation was damaged by the workers. Each replicate plot was 50 m x 50 m with a minimum distance of 200 m between plots. Trees in monoculture plantations ranged from 7–17 years old, and tree species diversity, tree density, tree height and basal area were higher in the reference land uses (forest and jungle rubber) than in the converted land uses (rubber and oil palm plantations) (Table S2.2).

The oil palm and rubber plantations in both landscapes represented management practices typical for smallholders. During our study period (2013), oil palm plantations in the clay Acrisol soil were fertilized once in the rainy season (October to March) whereas those in the loam Acrisol soil were fertilized once in the rainy season and once in the dry season (April to September). Fertilization rates ranged between 48–88 kg N ha^{-1} yr^{-1} (except two smallholders who applied 138 kg N ha^{-1} yr^{-1} during our study period), 21–38 kg P ha^{-1} yr^{-1} and 40–157 kg K ha^{-1} yr^{-1} (accompanied by Cl input of 143 kg Cl ha^{-1} yr^{-1}), with the lower range in the clay Acrisol soil and the upper range in the loam Acrisol soil.

The fertilizer sources were NPK complete, urea and KCl fertilizers. One smallholder also applied lime (200 kg dolomite ($CaMg(CO_3)_2$) ha^{-1} $year^{-1}$) in the loam Acrisol soil. Prior to our study year, kieserite ($MgSO_4.H_2O$) and borate ($Na_2B_4O_2.5H_2O$) fertilizers were also used in some oil palm plantations in the loam Acrisol soil. All oil palm sites used a combination of herbicides (Gramoxone and Roundup) and manual weeding. Soil amendments were applied by hand around each palm tree at about 0.8–1.5 m distance from the palm stem. Senescing oil palm fronds were regularly cut and stacked at a distance of 4.5 m from the rows of palm trees (row spacing was about 9 m). This was done to facilitate walking and working (e.g. harvesting) in the plantations. The rubber plantations had no fertilizer or lime application but had weeding. Harvesting in oil palm plantations was done on average every two weeks, whilst latex from rubber and jungle rubber were collected every week.

The implicit assumption of our experimental design, comparing the changes in converted land uses to the reference land uses to assess effects of land-use change, is that the initial conditions were comparable prior to conversion. To test this assumption, we compared land-use independent soil characteristics (i.e. soil texture at deeper depths, ≥0.5 m) among land uses within each landscape. We did not detect significant differences in soil texture between the reference land uses and the converted plantations within a soil landscape (Table S2.1); this, together with our interviews of the smallholders about the previous land use, support our assumption of comparable soil conditions prior to land conversion such that changes in nutrient leaching can be attributed to land-use change.

2.2.2 Lysimeter installation and soil water sampling

For measuring nutrient leaching, we collected soil water samples in two subplots of 5 m x 5 m each per replicate plot (50 m x 50 m), except in oil palm plantations where we sampled only in one subplot. In each subplot, we installed a suction cup lysimeter (P80 ceramic, maximum pore size 1 µm; CeramTec AG, Marktredwitz, Germany) 1.5 m into the soil. In the oil palm sites, the lysimeters were installed 1.3–1.5 m distance from the palm stem. In all plots, the 1.5-m depth of lysimeter cup installation was well below the rooting depth. This was ascertained from the fine and course root distribution with depth (Fig. S2.1). Prior to installation, lysimeters, sample tubes and collection containers were acid-washed and rinsed with copious amounts of deionized water. Lysimeters were installed in the field 3 months prior to the first sampling to allow resettling of natural soil conditions prior to measurement. The collection containers (dark glass bottles) were placed in plastic buckets with lid and buried in the ground approximately 1.3-m distance from the lysimeters. Soil water was sampled biweekly to monthly, depending on the frequency of rainfall, from February to December 2013. Soil water was withdrawn by applying a 40 kPa vacuum on the sampling tube, which represents soil water in rapidly and slowly draining pores (Amer, 2012). The collected soil water was transferred into 100 ml plastic bottles, which were acid-washed and thoroughly rinsed with deionized water before use. Upon arrival at the laboratory, a subsample of about 20 ml from each water sample was set aside for pH measurement while the remaining water was immediately frozen. All frozen soil water samples were transported by air to the laboratory of Soil Science Tropical and Subtropical Ecosystems (SSTSE), Goettingen University, Germany, and remained frozen until analysis.

The total dissolved N (TDN), NH_4^+, NO_3^- and Cl concentrations were measured using continuous flow injection colorimetry (SEAL Analytical AA3, SEAL Analytical GmbH, Norderstedt, Germany). Total dissolved N was determined by ultraviolet-persulfate digestion followed by hydrazine sulfate reduction (Autoanalyzer Method G-157-96); NH_4^+ was analyzed by salicylate and dicloro isocyanuric acid reaction (Autoanalyzer Method G-

102-93) and NO_3^- by cadmium reduction method with NH_4Cl buffer (Autoanalyzer Method G-254-02); Cl was determined with an ion strength adjustor reagent that is pumped through an ion selective chloride electrode with an integrated reference electrode (Auto analyzer Method G-329-05). Dissolved organic N (DON) is the difference between TDN and mineral N (NH_4^+ + NO_3^-). Dissolved organic C (DOC) was determined using a Total Organic Carbon Analyzer (TOC-Vwp, Shimadzu Europa GmbH, Duisburg, Germany). DOC was analyzed by pre-treating the samples with H_3PO_4 solution (to remove inorganic C) followed by UV-enhanced persulfate oxidation of organic C to CO_2, and determined by an infrared detector. Base cations (Na, K, Ca, Mg), total Al, total Fe, total Mn, total S, total P, and total Si in soil water were analyzed using inductively coupled plasma-atomic emission spectrometer (iCAP 6300 Duo View ICP Spectrometer, Thermo Fischer Scientific GmbH, Dreieich, Germany). Method detection limits for each element were: 6 µg NH_4^+-N l^{-1}, 5 µg NO_3^--N l^{-1}, 2 µg TDN l^{-1}, 4 µg DOC l^{-1}, 30 µg Na l^{-1}, 50 µg K l^{-1}, 3 µg Ca l^{-1}, 3 µg Mg l^{-1}, 2 µg total Al l^{-1}, 3 µg total Fe l^{-1}, 2 µg total Mn l^{-1}, 10 µg P l^{-1}, 10 µg total S l^{-1}, 1 µg total Si l^{-1} and 30 µg Cl l^{-1}. For concentrations below these detection limits, we assigned a value of zero. Partial cation-anion charge balance of the major solutes (i.e. concentrations >0.03 mg l^{-1}) in soil solution was conducted by expressing solute concentrations into μmol_c l^{-1} (molar concentration multiplied by the equivalent charge of each solute). Contributions of organic acids ($RCOO^-$) and bicarbonate (HCO_3^-) were not measured, but were calculated together with S (having very low concentration) from the difference of cations minus anions. Charge contributions of total Al were assumed to be 3^+, respectively; other solutes (total Fe, Mn and P that had very low concentrations and thus minimal charge contribution) as well as total dissolved Si (commonly in a form of monosilicic acid ($H_4SiO_4^0$) that has no net charge) were excluded (Hedin et al., 2003).

2.2.3 Soil water modelling and calculation of nutrient leaching fluxes

Daily drainage water fluxes were estimated using the soil water module of the Expert-N model (Priesack, 2005). This model was used successfully in our earlier work on nutrient leaching losses from conversion of montane forest to agricultural land uses in Sulawesi, Indonesia (Dechert et al., 2005). The model was parameterized with the conditions in our sites (i.e. climate, vegetation, and soil data). The climate data consisted of daily minimum, maximum and average air temperature, daily average relative humidity, daily average wind speed, daily total solar radiation, and daily total precipitation. For the loam Acrisol landscape, the climate data were taken from a climatological station at the Harapan Forest Reserve approximately 10–20 km from our plots. For the clay Acrisol landscape, the climate data were taken from climatological stations at the villages of Sarolangun and Lubuk Kepayang, approximately 20 km and 10 km, respectively, from our plots. The vegetation data consist of leaf area index (LAI in m^2 m^{-2}) and fine root mass distribution. The LAI in the loam soil landscape was 5.8 for forest, 4.8 for jungle rubber, 3.5 for rubber, and 3.9 for oil palm, whereas the LAI of forest, jungle rubber, rubber and oil palm in the clay soil was 6.2, 4.5, 2.8 and 3.1, respectively (Rembold et al., unpublished data). Our measured fine root biomass distribution (Fig. S2.1) was used to partition root water uptake at various depths and assumed that water uptake by evapotranspiration mainly occurred in the upper 1.5-m depth. Further data input of soil characteristics included soil bulk density, texture (Table S2.1) and the water retention curve. The soil water retention curve was determined using the pressure plate procedure; we took 250-cm^3 intact soil cores from one soil pit per land use and landscape at depths of 0.05, 0.2, 0.4, 0.75 and 1.25 m and water contents were measured at pressure heads of 0, 100, 330 and 15000 hPa.

Calculation of daily drainage water fluxes follows the equation of the water balance:

$$\Delta W + D = P - R - ET \text{ and } ET = I + E + T$$

in which ΔW = change in soil water storage, D = drainage water below rooting zone, P = precipitation, R = runoff, and ET = evapotranspiration, which is equal to the sum of three terms: I = interception of water by plant foliage, assumed to evaporate, E = evaporation from soil, and T = transpiration by plants following water uptake. The Expert-N model calculates actual evapo-transpiration using the Penman-Monteith method, actual runoff based on the sites' slope, and vertical water movement using Richards equation, of which the parameterization of the hydraulic functions was based on the measured soil texture and water retention curve using standard equations (Mualem, 1976; Van Genuchten, 1980).

To validate the output of the Expert-N model, we compared the modelled soil matrix potential with the measured matrix potential. Soil matrix potential was measured biweekly to monthly from February to December 2013, using tensiometers (P80 ceramic, maximum pore size 1 µm; CeramTec AG, Marktredwitz, Germany), which were installed at 0.3 m and 0.6 m depths in two replicate plots per land use and landscape. The modelled and measured soil matrix potential were strongly correlated (Pearson correlation coefficients of 0.79 to 0.98, p = 0.000–0.007; Fig. S2.2). Predicted daily drainage water fluxes at a depth of 1.5 m were summed to get the biweekly or monthly drainage fluxes. Nutrient leaching fluxes from each replicate plot were calculated by multiplying the average element concentrations from two lysimeters per plot (except for oil palm sites, which had one lysimeter per plot) on each sampling period with the total biweekly or monthly drainage water flux at 1.5 m.

2.2.4 Nutrient retention efficiency

To evaluate the efficiency with which nutrients were retained in soil, we calculated for each replicate plot the N and base cation retention efficiency as: 1 – (nutrient leaching loss/soil available nutrient), an index that is shown to be sensitive for evaluating effects of management practices on nutrient retention in the soil (Hoeft et al., 2014). This calculation does not include harvest export and thus we emphasize that this index of nutrient retention entails the fraction of nutrient retained in the soil in relation to the index of available nutrients in the soil. For N retention efficiency, N loss was TDN leaching flux and soil available N used gross N mineralization rate as an index, with both terms expressed in mg N m^{-2} d^{-1}. For base cation retention efficiency, base cation leaching flux was the sum of K, Na, Mg and Ca in units of mol_{charge} ha^{-1} yr^{-1} and soil available base cations are the sum of these exchangeable cations in units of mol_{charge} ha^{-1}. Gross N mineralization in the top 5-cm depth and exchangeable bases in the top 10-cm depth were measured on the same plots of our present study in 2013 and reported earlier by Allen et al. (2015). Retention efficiency of P in the soil was not reported because total P leaching flux was very low (see result section).

2.2.5 Supporting parameter: nutrient input through bulk precipitation

In each landscape, we installed two rain samplers in an open area at 1.5 m above ground level. Rain samplers consisted of 1 l high-density polyethylene bottles with lids attached to funnels that were covered with a 0.5-mm sieve to prevent insects, twigs or leaves from entering, and were placed inside polyvinyl chloride tubes (to shield from sunlight and prevent algae from growing). These rain samplers were washed with acid and rinsed with deionized water immediately after each collection. Rain was sampled during the same sampling period as the soil water. Each rain sample was immediately filtered through prewashed (with deionized water) filter paper (4 µm nominal pore size) into 100 ml plastic bottles and stored frozen for transport to SSTSE. The element analyses were the same as those described for soil water. The biweekly or monthly measurement of element concentrations in rain water was weighted with the rainfall volume during the two-week or 1-month collection period to get volume-weighted concentrations. The annual element input from bulk precipitation was calculated by multiplying the volume-weighted average concentration in a year with the annual rainfall in each landscape.

2.2.6 Statistical analysis

Tests for normality (Shapiro-Wilk's test) and homogeneity of variance (Levene's test) were conducted for each variable across landscapes or across land-use types prior to tests of differences between landscapes for each land use or differences among land-use types within each landscape. Logarithmic or square-root transformation was used for variables that showed non-normal distribution or heterogeneity of variance. We used linear mixed effects (LME) model (Crawley, 2009) to assess: 1) differences between landscapes for the reference land uses (hypothesis 1), and 2) differences among land-use types within each landscape (hypothesis 2). For element concentrations, the LME model had landscape or land-use type as the fixed effect with spatial replication (plot) and time (biweekly or monthly sampling period of element concentrations) as random effects. For the annual leaching fluxes (which were the sum of the bi-weekly or monthly sampling), the LME model had landscape or land-use type as the fixed effect with only spatial replication (plot) as a random effect. We extended the LME model to include: either 1) a variance function that allows different variances of the fixed effect, 2) a first-order temporal autoregressive process that assumes that correlation between measurement periods decreases with increasing time difference, or both if these improved the relative goodness of the model fit based on the Akaike information criterion. Fixed effect was considered significant based on analysis of variance at $p \leq 0.05$, and differences between landscapes or land-use types were assessed using Fisher's least significant difference test $p \leq 0.05$. For a few specified parameters, we also considered marginal significance at $p \leq 0.09$, because our experimental design encompassed the inherent spatial variability in our study area. Pearson correlation analysis was conducted to assess the relationships among cation and anion charge concentrations in soil solution for each land use within each landscape, using the monthly average ($n = 12$ within one year of measurement) of the four replicate plots per land use. Finally, Spearman's rank correlation test was conducted to assess the relationships between annual nutrient leaching fluxes and soil biochemical characteristics across landscapes, separately for the reference land uses and the converted land uses ($n = 16$). All statistical analyses were conducted using R 3.0.2 (R Development Core Team, 2013).

2.3 Results

2.3.1 Water balance and nutrient input from bulk precipitation

The trends (since statistical comparison was not possible) of the simulated water balance showed that evapotranspiration (ET) and runoff were higher in the clay than loam Acrisol soils, whilst cumulative water drainage showed the reverse trend (Table 2.1). Transpiration was the largest component of total ET in the reference land uses in both landscapes (74% of total ET for forest and 67% of total ET for jungle rubber). Within each landscape, rubber and oil palm plantations had lower ET and higher cumulative water drainage and runoff than the reference land uses (Table 2.1).

Table 2.1. The simulated water balance for 2013 in four different land uses (forest, jungle rubber, rubber plantations, oil palm plantations) within two soil landscapes (loam and clay Acrisols) in Jambi, Sumatra, Indonesia.

Water balance components (mm yr^{-1})	Forest	Jungle rubber	Rubber plantations	Oil palm plantations
	loam Acrisol landscape (precipitation: 3418 mm yr^{-1})			
Evapotranspiration	1384	1224	1077	1027
Transpiration	1033	815	594	437
Evaporation	155	213	287	408
Interception	196	196	196	182
Water drainage	1483	1487	1544	1614
Runoff	545	704	800	761
	clay Acrisol landscape (precipitation: 3475 mm yr^{-1})			
Evapotranspiration	1622	1271	1114	1071
Transpiration	1284	861	402	446
Evaporation	157	242	548	459
Interception	181	168	164	166
Water drainage	1117	1268	1280	1311
Runoff	722	932	1070	1087

Mean volume-weighted element concentrations of collected rain water (Table 2.2) between landscapes were also not tested statistically since we only had n = 2. In the clay Acrisol soil, the volume-weighted concentration of DOC tended to increase during the dry season (May-October: 9.28 ± 0.58 mg l^{-1}) compared to the wet season (November-April: 6.80 ± 1.51 mg l^{-1}) whereas in the loam Acrisol soil, they were similar (May-October: 6.70 ± 1.80 mg l^{-1}; November-April: 6.74 ± 0.66 mg l^{-1}). Most other element concentrations were similar between the two landscapes. Averaged across landscapes, annual input from bulk precipitation was dominated by DOC (58% of total element deposition rate), followed by Na (14%), Cl (12%), total dissolved N (3%), Ca (3%), K (2%), and total S (2%) (Table 2.2). We also detected small amounts of Mg, total Al, total Fe, total Mn, total P and total Si from bulk precipitation. Average chlorinity ratios across landscapes were: 1.13 ± 0.05 for Na:Cl, 0.05 ± 0.01 for Mg:Cl, 0.20 ± 0.02 for Ca:Cl and 0.13 ± 0.04 for K:Cl.

Table 2.2. Mean (± SE, n = 2) volume-weighted element concentrations and annual inputs from bulk precipitation from February to December 2013 within two soil landscapes (loam and clay Acrisol) in Jambi, Sumatra, Indonesia.

Elements	Volume-weighted concentration (mg l^{-1})		Annual input (kg ha^{-1} yr^{-1})	
	loam Acrisol	clay Acrisol	loam Acrisol	clay Acrisol
Ammonium (NH_4^+-N)	0.17 (0.02)	0.20 (0.02)	5.8 (0.6)	6.9 (0.7)
Nitrate (NO_3^--N)	0.04 (0.02)	0.07 (0.01)	1.3 (0.6)	2.6 (0.4)
Dissolved organic nitrogen (N)	0.17 (0.01)	0.20 (0.04)	5.8 (0.2)	7.0 (1.4)
Total dissolved nitrogen (N)	0.38 (0.00)	0.47 (0.07)	12.9 (0.1)	16.4 (2.6)
Dissolved organic carbon (C)	8.15 (0.19)	7.44 (0.07)	278.4 (6.6)	258.6 (2.5)
Sodium (Na)	1.84 (0.04)	1.90 (0.18)	63.0 (1.3)	66.1 (6.3)
Potassium (K)	0.16 (0.04)	0.28 (0.14)	5.5 (1.5)	9.6 (4.9)
Calcium (Ca)	0.32 (0.02)	0.36 (0.07)	10.9 (0.8)	12.4 (2.4)
Magnesium (Mg)	0.07 (0.01)	0.09 (0.01)	2.4 (0.5)	3.0 (0.4)
Total aluminum (Al)	0.02 (0.01)	0.01 (0.00)	0.5 (0.3)	0.4 (0.1)
Total iron (Fe)	0.01 (0.00)	0.01 (0.00)	0.4 (0.1)	0.3 (0.1)
Total manganese (Mn)	0.001 (0.00)	0.001 (0.00)	0.03 (0.0)	0.04 (0.0)
Total phosphorus (P)	0.01 (0.00)	0.02 (0.00)	0.4 (0.1)	0.8 (0.1)
Total sulfur (S)	0.26 (0.00)	0.30 (0.03)	9.0 (0.1)	10.4 (1.0)
Total silica (Si)	0.02 (0.01)	0.03 (0.01)	0.6 (0.2)	0.9 (0.3)
Chloride (Cl)	1.79 (0.25)	1.54 (0.30)	61.1 (8.4)	53.4 (10.6)

2.3.2 Leaching losses and nutrient retention efficiencies in the reference land uses – forest and jungle rubber

Differences in soil characteristics between the two landscapes were more pronounced in jungle rubber than forest. In the forest sites, exchangeable Na and Bray-extractable P were lower in the loam than clay Acrisol soils (all $p \leq 0.05$, Table S2.1). In the jungle rubber sites, soil organic C (SOC), total N, and exchangeable K, Na, Ca and Mg were lower in the loam than clay Acrisol soils (all $p \leq 0.05$, except $p \leq 0.09$ for Ca; Table S2.1). Averaged exchangeable Al saturation was 78-80% (with 11-16% exchangeable base saturation) and 61-71% (with 23% base saturation) in the loam and clay Acrisol soils, respectively (Table S2.1).

Differences in nutrient concentrations in soil solution at 1.5-m depth between the two landscapes were stronger in forest than jungle rubber (Table 2.3). In the jungle rubber sites, NO_3^--N was higher ($p \leq 0.05$) and total Si was lower ($p \leq 0.09$) in the loam than clay Acrisol soils (Table 2.3). In the forest sites, NH_4^+-N, DON, Na, Mg, total Al, total Fe and Cl were higher (all $p \leq 0.05$, except $p \leq 0.09$ for NH_4^+-N, DON, total Fe and Cl) in the loam than clay Acrisol soils (Table 2.3). The partial charge balance of cations and anions in soil solution showed that forests in the loam Acrisol soil had higher ($p = 0.01$) total ionic charges (274 ± 19 μmol_{charge} l^{-1}) than forests in the clay Acrisol soil (203 ± 20 μmol_{charge} l^{-1}) (Fig. 2.1). Element concentrations in soil solutions of the forests, particularly in the loam Acrisol soil that had high leaching fluxes, exhibited a strong positive correlations between solute cations (NH_4^+-N, Ca, Mg and Al) and anions (DOC, DON and Cl) (Table S2.3). For the jungle rubber, the total ionic charges were comparable between the loam (199 ± 31 μmol_{charge} l^{-1}) and clay (207 ± 24 μmol_{charge} l^{-1}) Acrisol soils (Fig. 2.1) and there were also strong correlations between solute cations (NH_4^+-N, K, Ca, Mg and Al) and anions (DOC, DON, Cl and NO_3^--N) in both landscapes (Table S2.3 and S2.4).

Annual leaching fluxes of NH_4^+-N, DON, Na, Ca, Mg, total Al, total Si and Cl in the forest sites were higher (all $p \leq 0.05$ except $p \leq 0.09$ for NH_4^+-N and DON) in the loam than clay Acrisol soils, whereas in the jungle rubber sites, only annual NO_3^--N leaching flux was higher ($p \leq 0.05$) in the loam than clay Acrisol soils (Table 2.4). We correlated the annual nutrient leaching fluxes with potential soil controlling factors across both landscapes; there are no other significant correlations observed except those that are presented here. Annual leaching fluxes of negatively charged solutes, DON and NO_3^--N, were correlated with indicators of soil exchangeable cations: base saturation, effective cation exchange capacity (ECEC) or exchangeable Al in the reference land uses across landscapes (*Spearman's* ρ = -0.51 - -0.61, $n = 16$, $p \leq 0.05$). On the other hand, annual leaching flux of positively charged NH_4^+-N was negatively correlated with SOC (*Spearman's* ρ = -0.53, $n = 16$, $p = 0.04$). The higher leaching fluxes in the loam than clay Acrisol soils were mirrored by decreased N and base cation retention efficiency (Table 2.5). N and base cation retention efficiency in the soils of these reference land uses were also positively correlated with base saturation, ECEC and SOC across landscapes (*Spearman's* ρ = 0.52–0.70, $n = 16$, $p \leq 0.04$). These soil biochemical properties (base saturation, exchangeable Al, ECEC and SOC) were also positively correlated with clay contents across landscapes (*Spearman's* ρ = 0.55–0.59, $n = 12$ sites with analysis of clay content, $p \leq 0.05$).

2.3.3 Leaching losses and nutrient retention efficiency in unfertilized rubber plantations

In the loam Acrisol landscape, rubber plantations had lower NO_3^--N and DOC concentrations in soil solution than both forest and jungle rubber ($p \leq 0.09$) and lower DON, Na, Ca, total P, total S and Cl concentrations than forest (all $p \leq 0.05$, except $p \leq 0.09$ for total P and total S) (Table 2.3). The low concentrations of organic (DON and DOC) and inorganic (Na, Ca and Cl) elements resulted in the lower ($p < 0.01$) total ionic charges in soil solutions of rubber plantations (200 ± 21 μmol_{charge} l^{-1}) as compared to forest (Fig. 2.1). There were strong positive correlations between dissolved positive ions (Ca, Mg and total Al) and Cl as well as weaker correlations between dissolved positive ions (Na, Ca and Mg) and negatively charged DOC (Table S2.3). Finally, unfertilized rubber plantations had lower annual total P leaching than forest ($p \leq 0.09$) and lower annual DOC leaching than jungle rubber ($p \leq 0.05$) (Table 2.4). N and base cation retention efficiency in soils of unfertilized rubber plantations were comparable with the reference land uses (Table 2.5).

In the clay Acrisol landscape, rubber plantations had 30% lower DOC ($p = 0.07$) and 20% lower Na ($p \leq 0.01$) concentrations in soil solution than jungle rubber; also total S and total Si were 30% lower (all $p \leq 0.09$) compared to forest (Table 2.3). The total solute ionic charges in rubber plantations (189 ± 23 $\mu mol_{charge}\ l^{-1}$) were comparable to both reference land uses (Fig. 2.1). As was the case with rubber plantations in the loam Acrisol landscape, we detected strong positive correlations of dissolved positive ions (Ca, Mg and Al) with negative ions (Cl), and strong positive correlations of dissolved monovalent bases (Na and K) with negatively charged DOC (Table S2.4). Only annual DOC leaching fluxes in rubber plantations were lower than in jungle rubber ($p \leq 0.05$) (Table 2.4). These unfertilized rubber plantations had similar N and base cation retention effeciency as compared to the reference land uses (Table 2.5).

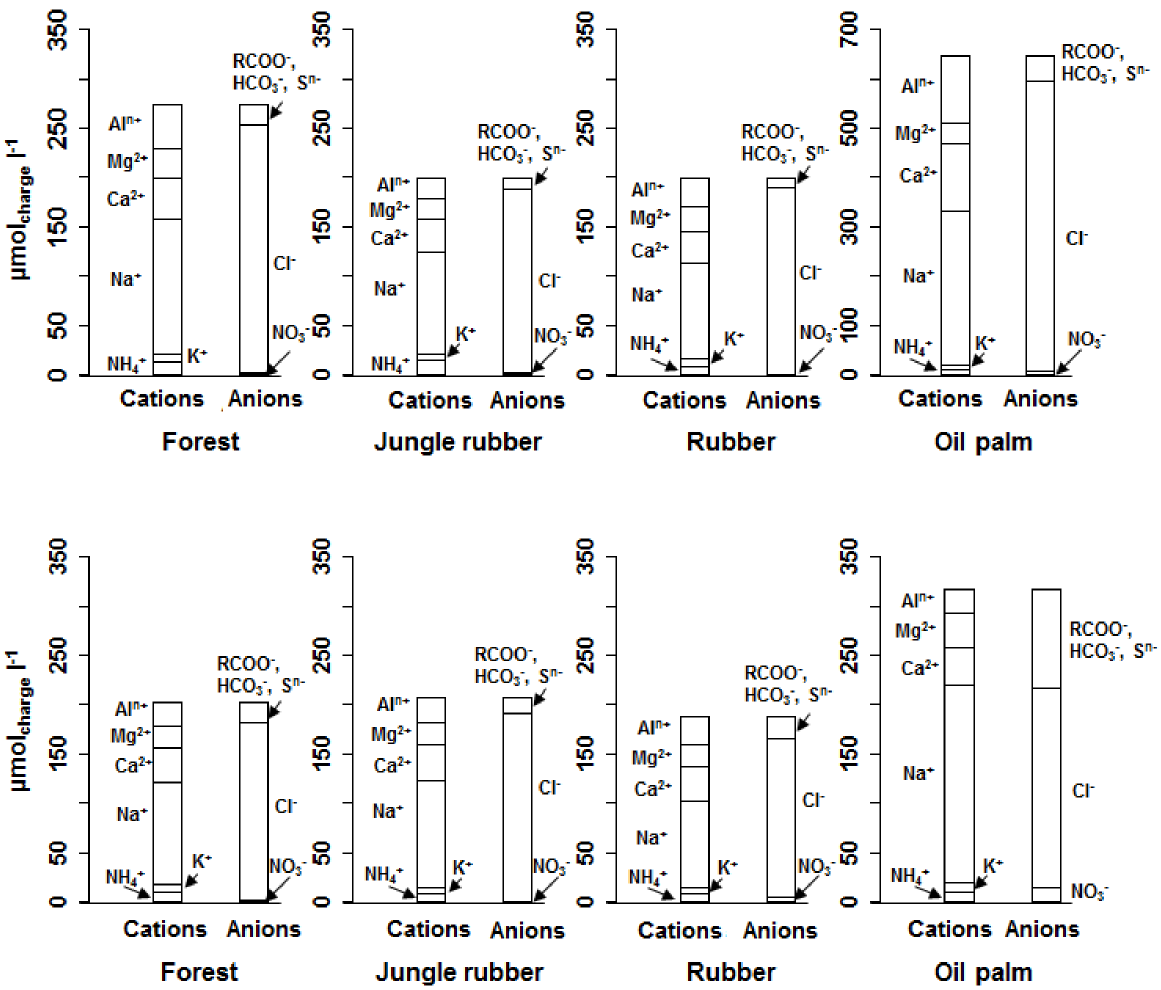

Fig. 2.1. Partial cation-anion charge balance of the major solutes (solutes with concentrations >0.03 mg l^{-1}) in soil water at a depth of 1.5 m in different land uses (forest, jungle rubber, rubber plantations and oil palm plantations) within two soil landscapes (loam and clay Acrisols) in Jambi, Sumatra, Indonesia.

2.3.4 Leaching losses and nutrient retention efficiencies in fertilized oil palm plantations

In the loam Acrisol soil, application of dolomite and K-containing fertilizers during our study year (2013) and application of kieserite and borate in previous years to these oil palm plantations (see section 2.2.1) led to two to three times higher saturation of exchangeable bases ($p = 0.06$) and four times higher exchangeable Na in the soil ($p \leq 0.01$) than in forest and jungle rubber (Table S2.1). Application of dolomite also increased the Ca concentration in the soil solution up to four months after application, with a monthly rate increment of 26% (2.64 mg Ca l^{-1} before application and 3.08, 3.80, 5.17, and 6.61 mg Ca l^{-1} in the following four consecutive months). Despite dolomite application and temporal increase in Ca concentrations in the soil solution, stocks of exchangeable Ca in the soil were not significantly different between oil palm and the reference land uses due to the high variability among oil palm sites (as indicated by the large standard errors; Table S2.1).

Oil palm plantations in the loam Acrisol landscape lower soil solution pH and higher concentrations of NO_3^--N, DOC, Na, Ca, Mg, total Al and Cl (all $p \leq 0.05$, except $p \leq 0.09$ for pH, NO_3^--N and DOC) in soil solution compared to forest and jungle rubber (Table 2.3). We observed negative correlations of NO_3^--N and total Al concentrations with soil solution pH (r = -0.57 – -0.76, $p \leq 0.05$, $n = 12$) and positive correlations between NO_3^--N and total Al concentration ($p = 0.03$; Table S2.3). The total ionic charge concentrations in soil solutions of oil palm plantations (648 ± 306 μmol_{charge} l^{-1}) were higher ($p < 0.01$) than the reference land uses (Fig. 2.1). As opposed to the other land uses, we did not detect correlations of cations with Cl in soil solutions of oil palm plantations. Instead, we found that base cations (K, Ca and Mg) were positively correlated with total S concentration in the soil solution (all $p \leq 0.05$; Table S2.3). In oil palm plantations, annual leaching fluxes of Na, Ca, Mg, total Al and Cl were higher (all $p \leq 0.05$, except $p \leq 0.09$ for Mg) than any other land uses, DOC were higher ($p = 0.04$) than in forest, and NH_4^+-N, NO_3^--N, total P and total S were higher than in unfertilized rubber plantations (all $p \leq 0.05$, except p = 0.08 for total P; Table 2.4). Consequently, N and base cation retention efficiency decreased in oil palm plantations (all $p \leq 0.01$; Table 2.5).

In the clay Acrisol landscape, oil palm plantations had lower DON ($p \leq 0.09$) and higher Na concentrations ($p = 0.05$) in soil solution than both reference land uses, as well as higher DOC and total Si concentrations (all $p \leq 0.09$) as compared to forest (Table 2.3). The total ionic charges of soil water in oil palm plantations (317 ± 83 $\mu mol_{charge}\ l^{-1}$) were higher ($p < 0.01$) than in both reference land uses (Fig. 2.1). In this landscape, we observed strong correlations between base cations (Na, K, Ca and Mg) and anions (Cl, total S and DOC) (Table S2.4). Although we did not find significant differences in base cation retention efficiency among land uses (Table 2.5), annual Na and Mg leaching fluxes were higher in the oil palm than forest and jungle rubber (all $p \leq 0.05$) as well as higher annual Ca leaching than forest ($p = 0.03$) (Table 2.4).

Lastly, we related the annual nutrient leaching fluxes in these converted land uses (smallholder oil palm and rubber plantations) to potential soil controlling factors across landscapes. Annual N leaching fluxes (NH_4^+-N and DON) were negatively correlated with clay content (*Spearman's* ρ = -0.54 – -0.73, n = 12 sites analyzed for clay content, $p \leq 0.05$). Base cation retention efficiency in the soil was positively correlated with ECEC and SOC (*Spearman's* ρ = 0.66 - 0.87, $n \leq 14$, $p \leq 0.01$), which in turn were positive correlated with clay content (*Spearman's* ρ = 0.87 - 0.90, n = 12 sites analyzed for clay content, $p \leq 0.05$). There were no other significant correlations observed.

Table 2.3. Nutrient concentrations in soil solution from a depth of 1.5 m in different land uses (forest, jungle rubber, rubber plantations, oil palm plantations) within two soil landscapes (loam and clay Acrisols) in Jambi, Sumatra, Indonesia.

Elements	loam Acrisol landscape				clay Acrisol landscape			
	Forest	Jungle rubber	Rubber	Oil palm	Forest	Jungle rubber	Rubber	Oil palm
pH	[a]4.26 (0.03)$_{a\dagger}$	4.33 (0.09)$_{a\dagger}$	4.37 (0.04)$_{a\dagger}$	4.11 (0.11)$_{b\dagger\ B\dagger}$	4.35 (0.11)	4.38 (0.10)	4.37 (0.04)	4.56 (0.14)$_{A\dagger}$
Ammonium (mg NH_4^+-N l^{-1})	0.22 (0.03)$_{A\dagger}$	0.27 (0.12)	0.15 (0.00)	0.17 (0.01)	0.18 (0.02)$_{B\dagger}$	0.15 (0.00)	0.14 (0.00)	0.15 (0.01)
Nitrate (mg NO_3^--N l^{-1})	0.12 (0.05)$_{b\dagger}$	0.09 (0.04)$_{b\dagger\ A}$	0.02 (0.00)$_{c\dagger\ B\dagger}$	0.32 (0.15)$_{a\dagger}$	0.08 (0.04)	0.02 (0.00)$_{B}$	0.24 (0.15)$_{A\dagger}$	0.90 (0.88)
Dissolved organic N (mg N l^{-1})	0.17 (0.03)$_{a\ A\dagger}$	0.08 (0.02)$_{b}$	0.08 (0.02)$_{b}$	0.11 (0.03)$_{ab\ A}$	0.07 (0.02)$_{a\dagger\ B\dagger}$	0.09 (0.01)$_{a}$	0.05 (0.01)$_{ab}$	0.04 (0.01)$_{b\ B}$
Total dissolved N (mg N l^{-1})	0.51 (0.06)$_{ab\dagger\ A\dagger}$	0.44 (0.13)$_{b\dagger\ A\dagger}$	0.25 (0.02)$_{c\dagger}$	0.60 (0.18)$_{a\dagger}$	0.34 (0.04)$_{B\dagger}$	0.24 (0.02)$_{B\dagger}$	0.43 (0.14)	1.09 (0.89)
Dissolved organic C (mg C l^{-1})	3.69 (0.28)$_{b\dagger}$	3.98 (0.49)$_{b\dagger}$	3.15 (0.17)$_{c\dagger}$	4.19 (0.10)$_{a\dagger}$	3.31 (0.45)$_{b\dagger}$	4.04 (0.28)$_{a\dagger}$	2.87 (0.07)$_{b\dagger}$	4.79 (0.88)$_{a\dagger}$
Sodium (mg Na l^{-1})	3.16 (0.10)$_{b\ A}$	2.37 (0.23)$_{c}$	2.24 (0.18)$_{c}$	7.20 (3.88)$_{a}$	2.36 (0.18)$_{bc\ B}$	2.49 (0.14)$_{b}$	2.02 (0.08)$_{c}$	4.63 (1.20)$_{a}$
Potassium (mg K l^{-1})	0.35 (0.03)	0.25 (0.08)	0.27 (0.09)	0.39 (0.14)	0.31 (0.04)	0.26 (0.05)	0.26 (0.04)	0.38 (0.05)
Calcium (mg Ca l^{-1})	0.83 (0.04)$_{b}$	0.68 (0.10)$_{c}$	0.66 (0.06)$_{c}$	2.74 (0.91)$_{a\ A\dagger}$	0.72 (0.07)	0.73 (0.04)	0.69 (0.08)	0.77 (0.17)$_{B\dagger}$
Magnesium (mg Mg l^{-1})	0.35 (0.03)$_{b\ A}$	0.25 (0.03)$_{c}$	0.30 (0.07)$_{b}$	0.49 (0.11)$_{a\ A\dagger}$	0.27 (0.03)$_{B}$	0.27 (0.03)	0.27 (0.03)	0.43 (0.10)$_{B\dagger}$
Total aluminum (mg Al l^{-1})	0.41 (0.07)$_{b\ A}$	0.18 (0.04)$_{c}$	0.26 (0.02)$_{b}$	1.24 (0.71)$_{a\ A\dagger}$	0.21 (0.03)$_{B}$	0.22 (0.10)	0.27 (0.05)	0.21 (0.11)$_{B\dagger}$
Total iron (mg Fe l^{-1})	0.19 (0.15)$_{A\dagger}$	0.02 (0.00)	0.02 (0.01)	0.02 (0.00)$_{A}$	0.02 (0.00)$_{B\dagger}$	0.03 (0.01)	0.02 (0.00)	0.01 (0.00)$_{B}$
Total manganese (mg Mn l^{-1})	0.02 (0.01)	0.01 (0.01)	0.01 (0.00)	0.01 (0.01)	0.01 (0.00)	0.01 (0.00)	0.01 (0.00)	0.08 (0.06)
Total phosphorus (mg P l^{-1})	0.01 (0.00)$_{a\dagger}$	0.00 (0.00)$_{b\dagger}$	0.00 (0.00)$_{c\dagger\ B\dagger}$	0.00 (0.00)$_{ab\dagger}$	0.01 (0.00)	0.00 (0.00)	0.00 (0.00)$_{A\dagger}$	0.00 (0.00)
Total sulfur (mg S l^{-1})	0.16 (0.02)$_{a\dagger}$	0.14 (0.04)$_{ab\dagger}$	0.10 (0.01)$_{b\dagger}$	0.14 (0.01)$_{ab\dagger}$	0.15 (0.02)$_{a\dagger}$	0.11 (0.00)$_{b\dagger}$	0.11 (0.00)$_{b\dagger}$	0.13 (0.01)$_{ab\dagger}$
Total silica (mg Si l^{-1})	0.53 (0.07)	0.33 (0.12)$_{B\dagger}$	0.22 (0.07)	0.31 (0.13)$_{B\dagger}$	0.37 (0.05)$_{b\dagger}$	0.60 (0.10)$_{ab\dagger\ A\dagger}$	0.26 (0.02)$_{c\dagger}$	1.03 (0.39)$_{a\dagger\ A\dagger}$
Chloride (mg Cl l^{-1})	8.91 (0.83)$_{b\ A\dagger}$	6.61 (0.76)$_{c}$	6.70 (0.64)$_{c}$	20.99 (2.72)$_{a\ A}$	6.39 (0.57)$_{B\dagger}$	6.76 (0.87)	5.73 (0.83)	7.19 (2.10)$_{B}$

[a] Means (± SE, n = 4, except for oil palm n = 3) followed by different lowercase letters indicate significant differences among land uses for each landscape and different uppercase letters indicate significant differences between landscapes for each land use (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

Table 2.4. Annual (2013) nutrient leaching fluxes measured at a depth of 1.5 m, in different land uses (forest, jungle rubber, rubber plantations, oil palm plantations) within two soil landscapes (loam and clay Acrisols) in Jambi, Sumatra, Indonesia

Elements	loam Acrisol landscape				clay Acrisol landscape			
	Forest	Jungle rubber	Rubber	Oil palm	Forest	Jungle rubber	Rubber	Oil palm
Ammonium ($kg\ NH_4^+$-N $ha^{-1}\ yr^{-1}$)	[a]2.7 (0.4)$_{ab\ A†}$	4.9 (2.9)$_{ab}$	2.0 (0.1)$_{b\ A}$	3.2 (0.1)$_{a\ A}$	1.7 (0.2)$_{B†}$	1.8 (0.1)	1.7 (0.1)$_{B}$	2.0 (0.2)$_{B}$
Nitrate (kg NO_3^--N $ha^{-1}\ yr^{-1}$)	1.5 (0.6)$_{ab}$	1.3 (0.6)$_{ab\ A}$	0.2 (0.02)$_{b}$	5.9 (2.9)$_{a}$	1.0 (0.7)	0.2 (0.0)$_{B}$	3.3 (2.0)	11.4 (11.2)
Dissolved organic N (kg N $ha^{-1}\ yr^{-1}$)	1.8 (0.4)$_{A†}$	1.5 (0.5)	1.3 (0.4)	2.2 (0.6)$_{A}$	0.7 (0.2)$_{B†}$	1.0 (0.2)	0.8 (0.1)	0.5 (0.0)$_{B}$
Total dissolved N (kg N $ha^{-1}\ yr^{-1}$)	6.0 (0.8)$_{ab†\ A†}$	7.7 (3.3)$_{ab†}$	3.5 (0.4)$_{b†}$	11.3 (3.1)$_{a†}$	3.4 (0.8)$_{B†}$	3.0 (0.2)	5.7 (2.0)	14.0 (11.2)
Dissolved organic C (kg C $ha^{-1}\ yr^{-1}$)	41.7 (5.1)$_{bc}$	62.1 (14.6)$_{ab}$	38.6 (2.1)$_{c}$	72.9 (2.5)$_{a}$	33.6 (3.7)$_{c}$	53.8 (7.4)$_{ab}$	36.2 (1.7)$_{bc}$	62.5 (13.9)$_{a}$
Sodium (kg Na $ha^{-1}\ yr^{-1}$)	37.7 (3.6)$_{b\ A}$	37.0 (7.6)$_{b}$	31.0 (2.7)$_{b\ A†}$	130.6 (75.8)$_{a}$	24.9 (3.8)$_{b\ B}$	32.3 (3.2)$_{b}$	25.2 (0.8)$_{b\ B†}$	62.6 (17.7)$_{a}$
Potassium (kg K $ha^{-1}\ yr^{-1}$)	4.3 (0.57)	4.3 (1.9)	4.1 (1.4)	7.0 (2.2)	3.1 (0.4)	3.4 (0.9)	3.2 (0.5)	4.9 (0.6)
Calcium (kg Ca $ha^{-1}\ yr^{-1}$)	10.0 (0.7)$_{b\ A}$	11.6 (3.3)$_{b}$	9.4 (0.9)$_{b}$	46.4 (13.4)$_{a\ A†}$	6.8 (0.6)$_{b\ B}$	9.0 (0.4)$_{ab}$	8.4 (0.9)$_{ab}$	10.2 (2.3)$_{a\ B†}$
Magnesium (kg Mg $ha^{-1}\ yr^{-1}$)	4.1 (0.3)$_{b†\ A}$	4.4 (1.2)$_{b†}$	4.4 (0.9)$_{b†}$	8.8 (2.1)$_{a†}$	2.5 (0.2)$_{b\ B}$	3.3 (0.5)$_{b}$	3.2 (0.3)$_{b}$	5.7 (1.4)$_{a}$
Total aluminum (kg Al $ha^{-1}\ yr^{-1}$)	4.4 (0.7)$_{b\ A}$	3.1 (0.8)$_{b}$	4.0 (0.2)$_{b}$	23.2 (13.4)$_{a\ A†}$	1.8 (0.2)$_{B}$	2.3 (0.9)	3.5 (0.7)	2.7 (1.2)$_{B†}$
Total iron (kg Fe $ha^{-1}\ yr^{-1}$)	1.7 (1.3)	0.2 (0.1)	0.3 (0.1)	0.4 (0.1)$_{A}$	0.2 (0.01)	0.3 (0.1)	0.2 (0.02)	0.1 (0.0)$_{B}$
Total manganese (kg Mn $ha^{-1}\ yr^{-1}$)	0.2 (0.1)	0.3 (0.2)	0.1 (0.1)	0.3 (0.1)	0.1 (0.0)	0.2 (0.1)	0.1 (0.0)	0.9 (0.7)
Total phosphorus (kg P $ha^{-1}\ yr^{-1}$)	0.1 (0.0)$_{a†}$	0.1 (0.0)$_{ab†}$	0.0 (0.0)$_{b†\ B}$	0.1 (0.0)$_{a†}$	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)$_{A}$	0.1 (0.0)
Total sulfur (kg S $ha^{-1}\ yr^{-1}$)	2.1 (0.3)$_{ab}$	2.3 (1.0)$_{ab}$	1.3 (0.1)$_{b}$	2.4 (0.2)$_{a\ A}$	1.6 (0.3)	1.5 (0.1)	1.4 (0.1)	1.7 (0.2)$_{B}$
Total silica (kg Si $ha^{-1}\ yr^{-1}$)	7.4 (1.8)$_{A†}$	6.2 (2.6)	3.6 (1.3)	4.3 (1.1)	3.3 (0.5)$_{b\ B†}$	6.8 (1.2)$_{ab}$	3.0 (0.3)$_{b}$	12.9 (5.8)$_{a}$
Chloride (kg Cl $ha^{-1}\ yr^{-1}$)	105.2 (9.0)$_{b\ A}$	114.8 (24.4)$_{b}$	90.6 (5.6)$_{b}$	380.2 (67.3)$_{a\ A}$	60.0 (3.2)$_{B}$	82.5 (11.3)	69.3 (9.7)	97.7 (30.0)$_{B}$

[a] Means (± SE, n = 4, except for oil palm n = 3) followed by different lower case letters indicate significant differences among land uses for each landscape and different upper case letters indicate significant differences between landscapes for each land use (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

Table 2.5. Nitrogen and base cation retention efficiency from different land uses in two soil landscapes of Jambi, Sumatra, Indonesia.

Characteristic	Forest	Jungle rubber	Rubber plantation	Oil palm plantation
	loam Acrisol landscape			
N retention efficiency ($mg\ N\ m^{-2}\ d^{-1}$/ $mg\ N\ m^{-2}\ d^{-1}$)	[a]0.69 (0.04)$_{a\ B}$	0.55 (0.16)$_{ab\ B†}$	0.84 (0.01)$_{a}$	0.33 (0.16)$_{b}$
Base cation retention efficiency ($mol_{charge}\ ha^{-1}\ yr^{-1}$/ $mol_{charge}\ ha^{-1}$)	0.45 (0.09)$_{a\ B}$	0.59 (0.09)$_{a\ B†}$	0.70 (0.08)$_{a}$	0.07 (0.07)$_{b\ B}$
	clay Acrisol landscape			
N retention efficiency ($mg\ N\ m^{-2}\ d^{-1}$/ $mg\ N\ m^{-2}\ d^{-1}$)	0.91 (0.03)$_{A}$	0.91 (0.02)$_{A†}$	0.74 (0.08)	0.72 (0.18)
Base cation retention efficiency ($mol_{charge}\ ha^{-1}\ yr^{-1}$/ $mol_{charge}\ ha^{-1}$)	0.81 (0.08)$_{A}$	0.85 (0.08)$_{A†}$	0.84 (0.02)	0.86 (0.04)$_{A}$

[a] Means (± SE, n = 4, except for oil palm *n* = 3) followed by different lower case letters indicate significant differences among land uses for each landscape and different upper case letters indicate significant differences between landscapes for each land use (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$)

2.4 Discussion

2.4.1. Water balance and nutrient input from bulk precipitation

Our estimated ET (Table 2.1) was comparable to the ET from a tropical rainforest in central Kalimantan, Indonesia (1217–1519 mm yr^{-1}; Suryatmojo et al., 2013). Also, our estimates of daily ET in the oil palm plantations (2.4 ± 0.1 and 2.2 ± 0.1 mm d^{-1} in the loam and clay Acrisol soils, respectively) were similar to those reported by Niu et al. (2015) (2.6 ± 0.7 mm d^{-1}) for the same oil palm sites with the micro-meteorological data measured on site. The lower ET in the forest sites of the loam Acrisol soil compared to the clay Acrisol soil was primarily due to the lower transpiration from plants (Table 2.1), which was probably related to the lower LAI in the loam Acrisol soils. Similarly, the lower LAI in rubber and oil palm plantations as compared to forests likely led to the lower ET (Table 2.1), as leaf area is closely related to transpiration and plant water use (Granier et al., 1996; Santiago et al., 2000), a resulting in higher water drainage in the converted land uses (Table 2.1).

The high DOC contents in precipitation from our study area reflected the high biomass burning (Coelho et al., 2008) and dusts common in these landscapes with active land-use conversion. The Na:Cl and K:Cl ratios in bulk precipitation at our sites were higher than those in seawater (Na:Cl = 0.56 and K:Cl = 0.02; Schlesinger, 1997), whereas the Mg:Cl and Ca:Cl ratios were comparable to those in seawater (Mg:Cl = 0.07 and Ca:Cl = 0.02; Schlesinger, 1997). The high Na:Cl and K:Cl ratios, accompanied by high concentrations of elements associated with organic molecules (TDN, DOC and total S), in precipitation at our studied landscapes are common for areas influenced by biomass burning and terrigenous dust from agriculture (Eklund et al., 1997; Balasubramanian et al., 1999), which are common features in our study region.

2.4.2 Leaching losses and nutrient retention efficiency and in the reference land uses – forest and jungle rubber

Heavily weathered soils, like Acrisols, have relatively little internal input from rock-derived nutrients through weathering (Markewitz et al., 2001; Hedin et al., 2003) and thus nutrient leaching fluxes are largely influenced by internal input from decomposition/mineralization of organic matter, external input from atmospheric deposition (including sources like biomass burning), nutrient retention processes in the soil and water balance. The higher soil nutrient stocks (i.e. SOC, total N, extractable P and exchangeable bases) in the clay Acrisol soil (Table S2.1) were reflected by lower nutrient leaching losses (e.g. total N, Na, Ca, Mg and total Al) than in the loam Acrisol soil (Table 2.4). We attribute this to the high nutrient and water holding capacity of clay soil (Ohta and Effendi, 1992; Ohta et al., 1993) which, in turn, facilitate high plant productivity and efficient cycling of nutrient between vegetation and soil (Silver et al., 2000). This is evident from the higher ECEC (Table S2.1) and water-filled pore space of these reference sites in the clay than loam Acrisol soils (Hassler et al., 2015). In addition, clay content also affects the soil pore size distribution, with higher proportion of small pores in clay soils compared to loamy soils, which slows water percolation and thus also contributes to lower nutrient leaching losses (Ohta and Effendi, 1992; Silva et al., 2005).

The generally low total ionic charge concentrations in soil solutions of our forest sites are typical for highly weathered soils (Hedin et al., 2003), for which weathering of primary minerals must be already low (Markewitz et al., 2001). For Na and K, which ratios to Cl suggest large influence from biomass burning and dust, their inputs from bulk deposition (Table 2.2) were at most three times larger than their annual leaching losses (Table 2.4), suggesting the importance of atmospheric deposition (common in our study region with land clearing and biomass burning) as external sources of these elements. Additionally, internal supply of nutrients from decomposition of organic matter was possibly the reason for our observed high correlations between inorganic cations (Ca, Mg and Al) and organic anions (DOC and DON) (Table S2.3 and S2.4).

Nitrogen (NH_4^+-N, NO_3^--N and DON) leaching losses are influenced by N availability (e.g. measured as gross N mineralization) and N retention processes (e.g. microbial N immobilization) in the soil (Corre et al., 2010). In our reference land uses, the clay Acrisol had higher gross N mineralization and NH_4^+ immobilization than the loam Acrisol (Allen et al., 2015), corroborating the lower N leaching losses (Table 2.4) and higher N retention efficiency in the clay than loam Acrisol soils (Table 2.5). Across landscapes, the negative correlation of annual DON and NO_3^--N leaching losses with soil base saturation, ECEC and exchangeable Al suggested a link between N leaching and the buffering capacity of the soils. Acrisol soils are characterized by low pH and low base saturation (Table S2.1) and these correlations observed in our sites suggest that the buffering reactions for DON and NO_3^--N losses included not only the soil exchangeable bases but also the Al buffering range (through Al solubilization at pH 3-5; Van Breemen et al., 1983). Similarly, the negative correlation of annual NH_4^+-N leaching losses with SOC suggested both biotic and abiotic mechanisms of increased NH_4^+ retention with increasing SOC (i.e. clay Acrisol; Table S2.1). In the same reference sites, microbial biomass and microbial NH_4^+ immobilization are higher in clay Acrisol with high SOC than in loam Acrisol with low SOC (Allen et al., 2015). Also, abiotic NH_4^+ immobilization via physical condensation with organic compounds and clay fixation (Davidson et al., 1991) could be higher in clay Acrisol than in loam Acrisol (Table S2.1; Allen et al., 2015). All these mechanisms contributed to our observation of positive correlations between N and base cation retention efficiency with base saturation, ECEC and SOC (see section 2.3.2).

2.4.3 Leaching losses and nutrient retention efficiency in unfertilized rubber plantations

In converted land uses with the same soil types and climate, age of land use and management practices are important factors that influence soil nutrient levels and leaching losses (e.g. Dechert et al., 2005; Corre et al., 2006; Ngoze et al., 2008). In our loam Acrisol landscape, the smallholder rubber plantations were already 14–17 years old (Table S2.2) without external nutrient input from fertilization. In this landscape, input of organic material from aboveground litterfall is lower than in forest and jungle rubber (Kotowska et al., 2015) and together with harvest export these might have resulted in lower replenishment of soil nutrients than in the reference land uses. This was reflected in the lower total ionic charges in soil solutions of rubber plantations compared to forest (Fig. 2.1). Such reduction in total ionic charges was not statistically different from the reference land uses in soil solutions of younger rubber plantations (7–8 years old, except one site that was 16 years; Table S2.2) in the clay Acrisol landscape (Fig. 2.1), possibly because the legacy effect of ashes (from burning of the original vegetation) on leaching losses (Markewitz et al., 2001) was still evident during the relatively early years.

Nonetheless, the ultimate results after years of agricultural production without soil amendments are decreases in soil nutrient levels and cycling (e.g. soil N availability, Corre et al., 2006; Davidson et al., 2007; Allen et al., 2015; P availability, Ngoze et al., 2008). This was evident in the lower annual P leaching in rubber plantations compared to forest in the loam Acrisol soil (Table 2.4) that already had low levels of extractable P (Table S2.1). In these unfertilized rubber plantations, extractable P decreases not only in the top 10 cm (Allen et al., 2015) but also down to a 2-m depth when compared to forest (Allen, 2015). Similarly, the decrease in annual DOC leaching flux in rubber plantations compared to jungle rubber in both landscapes was due to a reduced amount of C in this land use, as shown by its decreases in microbial C (Allen et al., 2015), litterfall and root production (Kotowska et al., 2015) and SOC stocks (van Straaten et al., 2015). Interestingly, our observation of reduced NO_3^--N leaching losses (i.e. rubber plantations in the loam Acrisol soil; Tables 2.3 and 2.4) were also mirrored with reduced soil extractable NO_3^- in rubber plantations compared to forest in both landscapes (Allen et al., 2015), which was attributed to monoterpenes produced by rubber trees (Wang et al., 2007). Monoterpenes serve as a C source that increases microbial activity and reduces NO_3^- level in soil (White, 1991), possibly through increases in dissimilatory NO_3^- reduction to NH_4^+ and NO_3^- immobilization (Allen et al., 2015), resulting in the reduced NO_3^- leaching in these rubber plantations. Altogether, the high N and base cation retention efficiency in the soils of these unfertilized rubber plantations were because of low leaching from decreased levels of these nutrients (i.e. gross N mineralization and base saturation were lower in rubber than either oil palm or forest; Allen et al., 2015; Allen, 2015).

2.4.4 Leaching losses and nutrient retention efficiency in fertilized oil palm plantations

Given that land-uses within the same landscape had similar nutrient input from bulk precipitation, the most important factor contributing to the increase in nutrient leaching in oil palm plantations (i.e. higher total ionic charge concentrations; Fig. 2.1) was the external input of nutrients from fertilization. This difference was evident in both landscapes between oil palm plantations and all other land uses (i.e. reference land uses and unfertilized rubber plantations). Although harvest export is high from these oil palm plantations (i.e. harvested fruit bunches are 60% and 50% of the total annual net primary production in the loam and clay Acrisol soils, respectively; Kotowska et al., 2015), depletion of soil nutrients, as was detected in the unfertilized rubber plantations (e.g. soil available N and extractable P; Allen et al., 2015), was abated through fertilization. The drawback, however, was increases in nutrient leaching. We attributed the more pronounced increases in solute concentrations (Table 2.3), total ionic charge concentrations (Fig. 2.1) and annual leaching fluxes (Table 2.4) in the loam than the clay Acrisol landscapes to the higher fertilization rate (see section 2.2.1) and lower nutrient and water holding capacity (i.e. lower ECEC, Table S2.1; higher drainage flux, Table 2.1). Clay content influences the nutrient and water-holding capacity of soil which, in turn, regulates the cycling of nutrients between vegetation and soil (e.g. Silver et al., 2000). Therefore, altogether, the main factors influencing leaching losses from these smallholder oil palm plantations were fertilization rate and clay content.

In the loam Acrisol landscape, increased NO_3^- concentrations and acidity in soil solution of oil palm plantations compared to the other land uses (Table 2.3) were likely due to nitrification of added N fertilizer. This suggests that the pulse N application had exceeded the N demand of plant and microbial biomass at the time of application (e.g. Corre et al., 2010). In these oil palm sites, elevated NO_3^- and NH_4^+ concentrations in the soil were observed up to six weeks following fertilization application (Hassler et al., unpublished data), and both compounds are susceptible to leaching losses. Other studies in Indonesia and Malaysia have also reported increase in soil acidity due to N fertilization in oil palm plantations (Anuar et al., 2008; Comte et al., 2013). Since our loam Acrisol soil had initially low acid-buffering capacity (i.e. low base saturation in reference land uses; Table S2.1), nitrification-induced acidity may have enhanced the Al acid-buffering reaction and thus led to the increases in soluble Al (Table 2.3) and its correlation with NO_3^- concentrations in soil solution (Table S2.3). Even though the effect of ashes from biomass burning can linger years after conversion (e.g. van Straaten et al., 2015) and these smallholder oil palm plantations experienced occasional liming (i.e. in at least one site during our study year), the soil pH (Table S2.1) was still within the Al buffering range (pH 3-5; Van Breemen et al., 1983). Concurrent with increased soluble Al we also observed increased dissolved base cations (Table 2.3); the correlation of the latter with total S in the loam Acrisol landscape (Table S2.3) reflected the application of base cation- and sulfur-containing fertilizer (i.e. dolomite and kieserite) at these sites (see section 2.2.1). In the clay Acrisol landscape, where fertilizer applications were lower than in the loam Acrisol, dissolved cations correlated with the typical dominant anions (Cl, DOC and total S; Table S2.4). In sum, pulse N fertilization in oil palm plantations in these soils with low acid-buffering capacity caused acidic soil water and elevated Al concentrations, which may have caused decreases in mychorrhizal colonization of fine roots and the increases in distorted root tips found in these oil palm plantations compared to the reference land uses (Sahner et al., 2015).

High net primary production in these oil palm plantations (Kotowska et al., 2015), despite the highly weathered soils, was clearly sustained due to fertilizer application. However, increased leaching losses from these fertilized oil palm plantations compared to the original land uses showed decreased in nutrient retention efficiency (Table 2.5) and implied deleterious effects on ground water quality. Increased annual leaching fluxes of Na, Ca and Mg (Table 2.4) were primarily due to the application of borate fertilizer and lime. These values were within the range of those reported for fertilized oil palm plantations on Acrisol soil in Nigeria (26 kg Ca ha^{-1} and 6.5 kg Mg ha^{-1} during six-month measurement period; Omoti et al., 1983). The annual NH_4^+-N and NO_3^--N leaching fluxes in our oil palm sites were also comparable with those from fertilized oil palm plantation on Acrisol soil in Malaysia (3-6 kg N ha^{-1} during five-month measurement period; Tung et al., 2009). These annual NH_4^+-N + NO_3^--N leaching losses from oil palm sites in the loam Acrisol landscape (Table 2.4) were 10.4% of the typically applied rate of N fertilizer (88 kg N ha^{-1}; see section 2.2.1). Low annual P leaching is a common feature in highly weathered soils like Acrisols, as P tends to sorb onto Al and Fe hydrous oxides (McDowell et al., 2001). However, despite the generally low annual P leaching fluxes across land uses (Table 2.4), the higher annual total P leaching losses in the fertilized oil palm plantations than in unfertilized rubber plantations in the loam Acrisol landscape (Table 2.4) was presumably due to the relatively higher rate of inorganic P fertilization than in the clay Acrisol landscape. Moreover, the increased annual DOC fluxes in these fertilized oil palm plantations (Table 2.4) together with the large harvest export and decrease in litterfall input compared to forests (Kotowska et al., 2015) provided strong support for the observed decreases in soil organic C stocks in these oil palm plantations (van Straaten et al., 2015). Finally, the reduced N and base cation retention efficiency in the fertilized oil palm plantation occurred only in the loam Acrisol landscape (Table 2.5), which had a higher fertilization rate (see section 2.2.1) and lower nutrient and water holding capacity (i.e. lower ECEC, Table S2.1; higher drainage flux, Table 2.1) than the clay Acrisol landscape. Across these converted land uses, correlations of annual N leaching fluxes (NH_4^+-N and DON) and base cation retention efficiency with ECEC and SOC, which in turn were correlated with clay content, indicated the influence of clay content on nutrient adsorption and water holding capacity in these highly weathered soils to enhance cycling or retention of nutrients within the system.

In conclusion, this study highlighted the importance of soil texture in these highly weathered Acrisol landscapes. Higher clay content supported better soil biochemical characteristics, which supported efficient cycling of nutrients in the reference land uses and resulted in low nutrient leaching losses or conversely high nutrient retention efficiency in the soil. This supported our first hypothesis. Management practices in converted land uses strongly influenced nutrient leaching and retention efficiency in the soils. The unfertilized rubber plantations had lower nutrient leaching losses than the fertilized oil palm plantations. However, the high N and base cation retention efficiency in the unfertilized rubber plantations had a different meaning from that of the reference land uses since the low leaching fluxes from the rubber plantations clearly reflected the decreased levels of these nutrients in the soil (Allen et al., 2015), possibly due to harvest export and less input from litterfall and root production (Kotowska et al., 2015). This has implications for sustainability of yield from such smallholder rubber plantations as well as for the duration until such land use must be further converted. On the contrary, the fertilized oil palm plantations showed increased leaching of mineral N, accompanied by increased dissolved Al and acidity of soil solution, increased leaching of DOC, total P and total S compared to the rubber plantations and higher base cation leaching fluxes than any of the land-use types. This supported our second hypothesis. The high net primary production from these oil palm plantations (Kotowska et al., 2015), despite the highly weathered soils with low initial soil fertility, was aided by fertilizer input. Sustainability of palm oil yield must take into account the long-term effect of pulse N application on soil acidity, dependency on liming input that requires additional capital by smallholders, and the impact of increased nutrient leaching on ground water quality. In general, these smallholder oil palm plantations had lower fertilization rates compared to the large-scale plantations in our study region (e.g. a large-scale plantation in the loam Acrisol landscape (PTPN VI) has fertilization rates of 175-40-175 kg N-P-K ha^{-1} yr^{-1}), implying that the effects of increased leaching losses from oil palm plantations on ground water quality may be substantially higher than what we see from these smallholder plantations. Thus, our results suggest the need to improve fertilization practices on these highly weathered soils, synchronizing time and rate of application with plant uptake (e.g. by basing fertilizer rate to only compensate harvest export) and enhancing microbial immobilization (e.g. by maintaining the microbial biomass as in the original land use; Allen et al., 2015).

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Supplementary information

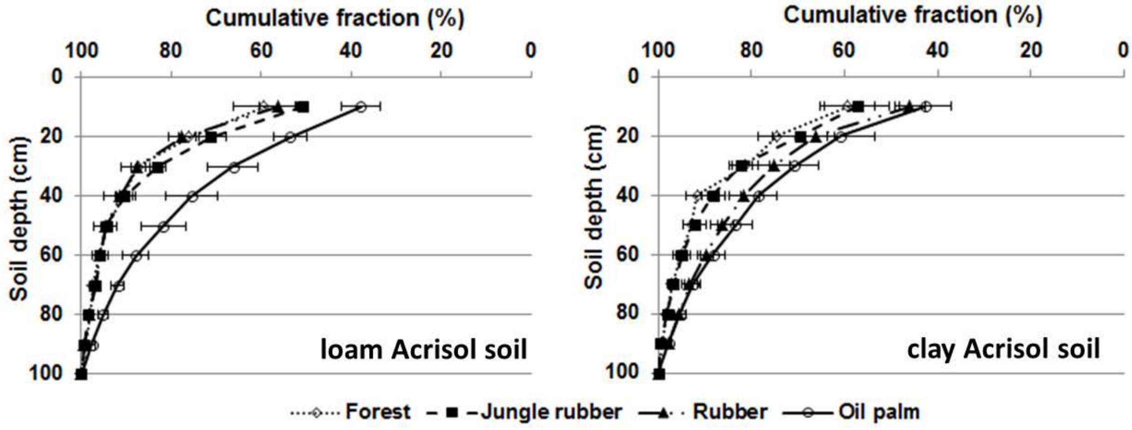

Fig. S2.1. Fine root biomass distribution to a depth of 1-m in different land uses (forest, jungle rubber, rubber plantations and oil palm plantations) within two soil landscapes (loam and clay Acrisol), in Jambi, Sumatra, Indonesia.

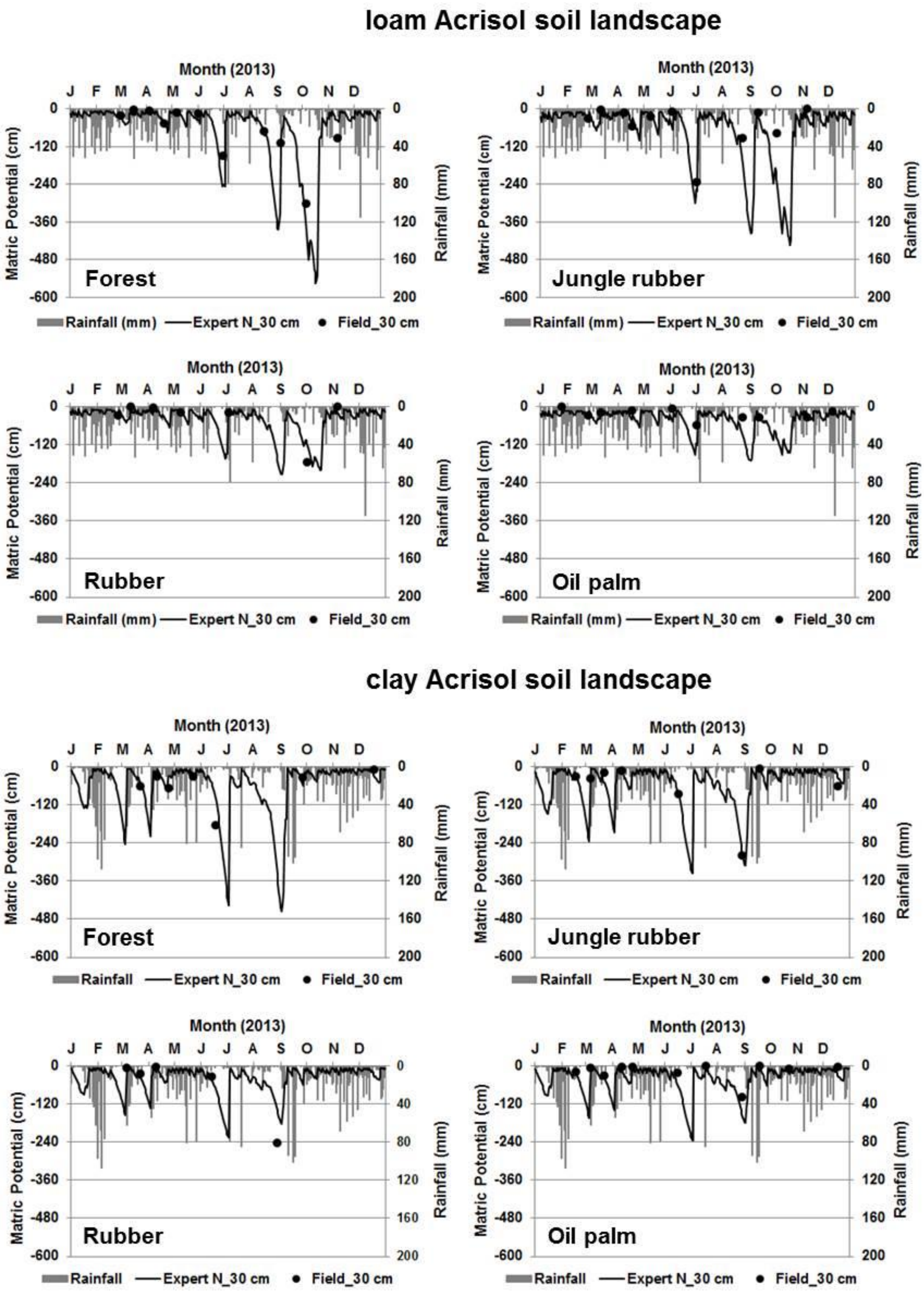

Fig. S2.2. Validation between Expert N-modelled and field-measured matric potential at 0.3-m depth in different land uses at two soil landscapes, Jambi, Sumatra, Indonesia.

Table S2.1. Soil characteristics[a] in the top 0.1 m of soil (except for clay content, which is given in depth intervals), from different land uses in two soil landscapes of Jambi, Sumatra, Indonesia.

Characteristic	Forest	Jungle rubber	Rubber plantation	Oil palm plantation
	loam Acrisol landscape			
Bulk density (g cm^{-3})	1.0 (0.0)$_{ab}$	0.9 (0.0)$_{b\ A}$	1.1 (0.1)$_{a}$	1.1 (0.1)$_{a\ A}$
pH (1:4 H_2O)	4.3 (0.0)$_{b†}$	4.3 (0.0)$_{b†\ B}$	4.5 (0.1)$_{ab†}$	4.5 (0.1)$_{a†}$
Soil organic C (kg C m^{-2})	2.6 (0.2)	2.7 (0.3)$_{B}$	2.0 (0.3)	1.8 (0.2)$_{B}$
Total N (g N m^{-2})	182.9 (10.8)	186.1 (11.0)$_{B}$	172.6 (23.8)	145.0 (13.5)$_{B}$
C:N ratio	14.3 (0.2)$_{a}$	13.7 (0.8)$_{a}$	11.7 (0.7)$_{b\ B}$	12.5 (0.5)$_{ab}$
Effective cation exchange capacity (mmol$_{charge}$ kg^{-1})	44.8 (5.0)	40.6 (7.6)$_{B}$	46.0 (5.4)	39.5 (7.9)$_{B}$
Base saturation (%)	10.6 (0.5)$_{b†\ B}$	16.0 (2.2)$_{ab†}$	21.1 (7.5)$_{ab†}$	27.9 (5.4)$_{a†}$
Potassium (g K m^{-2})	3.3 (0.3)	2.6 (0.2)$_{B}$	3.4 (0.8)	2.1 (0.8)$_{B†}$
Sodium (g Na m^{-2})	0.5 (0.1)$_{c\ B}$	1.5 (0.2)$_{b\ B}$	1.4 (0.1)$_{b}$	3.9 (1.1)$_{a}$
Calcium (g Ca m^{-2})	5.5 (2.0)	6.9 (0.8)$_{B†}$	14.5 (7.1)	18.5 (7.4)$_{B†}$
Magnesium (g Mg m^{-2})	1.8 (0.1)	2.0 (0.3)$_{B}$	3.4 (1.4)	1.7 (0.9)
Aluminum (g Al m^{-2})	33.1 (3.5)	29.6 (6.6)$_{B}$	30.7 (4.3)	23.5 (2.7)$_{B}$
Iron (g Fe m^{-2})	0.8 (0.1)$_{a\ B}$	0.3 (0.0)$_{bc\ B}$	0.3 (0.1)$_{c\ B}$	0.5 (0.0)$_{ab}$
Manganese (g Mn m^{-2})	0.3 (0.1)	0.4 (0.2)$_{B}$	0.8 (0.3)	0.5 (0.2)$_{B}$
Bray-extractable phosphorus (g P m^{-2})	0.5 (0.1)$_{B}$	0.7 (0.1)	0.5 (0.1)	0.8 (0.1)$_{B†}$
Clay at 0-0.5 m (%, weighted for the top 50 cm)	26.0 (2.6)	30.6 (4.6)	37.3 (10.2)	33.4 (2.2)$_{B}$
Clay at 0.5-1.0 m (%)	28.7 (4.8)	38.8 (9.0)	45.1 (11.3)	41.0 (3.1)$_{B}$
Clay at 1.0-1.5 m (%)	33.3 (7.6)	42.4 (9.9)	46.1 (9.9)	43.3 (2.8)$_{B}$
Clay at 1.5-2.0 m (%)	37.3 (8.7)	44.5 (10.0)	43.4 (6.5)	47.6 (4.5)
	clay Acrisol landscape			
Bulk density (g cm^{-3})	1.0 (0.1)	0.8 (0.1)$_{B}$	0.9 (0.1)	0.9 (0.1)$_{B}$
pH (1:4 H_2O)	4.2 (0.0)$_{b}$	4.5 (0.0)$_{a\ A}$	4.5 (0.1)$_{a}$	4.4 (0.0)$_{a}$
Soil organic C (kg C m^{-2})	3.3 (0.5)	4.3 (0.4)$_{A}$	2.8 (0.4)	3.5 (0.2)$_{A}$
Total N (g N m^{-2})	263.4 (67.1)	331.4 (34.1)$_{A}$	198.9 (32.5)	260.2 (22.6)$_{A}$
C:N ratio	13.1 (1.3)	13.0 (0.3)	14.3 (0.6)$_{A}$	13.5 (0.2)
Effective cation exchange capacity (mmol$_{charge}$ kg^{-1})	94.3 (40.8)	124.5 (25.5)$_{A}$	71.3 (22.3)	78.1 (8.4)$_{A}$
Base saturation (%)	22.9 (5.6)$_{A}$	23.2 (5.8)	20.1 (2.6)	37.5 (7.1)
Potassium (g K m^{-2})	9.4 (3.9)	9.6 (2.6)$_{A}$	4.2 (1.1)	4.8 (0.9)$_{A†}$
Sodium (g Na m^{-2})	3.6 (0.8)$_{A}$	4.2 (0.2)$_{A}$	3.7 (1.3)	1.9 (1.3)
Calcium (g Ca m^{-2})	32.3(21.2)	33.3 (10.9)$_{A†}$	14.7 (2.8)	59.1 (19.5)$_{A†}$

Magnesium (g Mg m^{-2})	7.3 (3.9)	12.0 (4.1)$_{A}$	4.0 (0.9)	3.5 (0.8)
Aluminum (g Al m^{-2})	50.9 (22.7)	76.6 (15.6)$_{A}$	47.2 (17.6)	34.4 (2.0)$_{A}$
Iron (g Fe m^{-2})	3.7 (1.1)$_{a\ A}$	3.0 (0.4)$_{a\ A}$	2.3 (0.6)$_{a\ A}$	0.7 (0.3)$_{b}$
Manganese (g Mn m^{-2})	4.5 (3.1)	2.5 (0.7)$_{A}$	1.5 (0.4)	3.4 (1.3)$_{A}$
Bray-extractable phosphorus (g P m^{-2})	1.4 (0.1)$_{ab\ A}$	0.8 (0.1)$_{bc}$	0.4 (0.04)$_{c}$	4.7 (1.5)$_{a\ A\dagger}$
Clay at 0-0.5 m (%, depth-weighted average)	31.5 (5.4)	47.2 (12.4)	42.4 (3.1)	59.7 (5.2)$_{A}$
Clay at 0.5-1.0 m (%)	34.9 (9.0)$_{b\dagger}$	51.4 (12.6)$_{ab}$	36.8 (8.0)$_{b}$	69.7 (4.8)$_{a\ A}$
Clay at 1.0-1.5 m (%)	39.0 (13.0)	62.8 (12.6)	40.8 (10.3)	62.8 (3.7)$_{A}$
Clay at 1.5-2.0 m (%)	41.3 (11.2)	46.6 (16.2)	36.5 (10.8)	63.3 (6.1)

[a] Allen et al. (2015).

[b] Means (± SE, n = 4, except for clay content n = 3) followed by different lower case letters indicate significant differences among land uses for each landscape and different upper case letters indicate significant differences between landscapes for each land use (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

Table S2.2. Mean (± SE, n = 4) tree density, diameter at breast height (DBH), basal area, height, the most common species of trees with DBH ≥ 0.10 m and cumulative fine root mass in the top 1 m depth in different land uses at two soil landscapes, Jambi, Sumatra, Indonesia.

Characteristics	Forest	Jungle rubber	Rubber	Oil palm
	loam Acrisol landscape			
Plantation age (years)	not determined (ND)	ND	14–17	12–16
Tree density (trees ha^{-1})[a]	658 (26)	525 (60)	440 (81)	140 (4)
DBH (cm)[a]	21.0 (0.5)	16.8 (0.5)	17.8 (1.2)	not applicable (NA)
Basal area (m^2 ha^{-1})[a]	30.7 (1.0)	16.6 (0.4)	12.2 (1.6)	NA
Tree height (m)[a]	20.0 (0.6)	14.0 (0.2)	13.4 (0.5)	4.9 (0.6)
Cumulative fine root biomass in the top 1 m of soil (g m^{-2})	290.2 (82.6) $_{ab†}$	143.9 (33.0) $_{b}$	188.2 (37.6) $_{b}$	356.8 (49.9) $_{a}$
Most common tree species[b]	*Aporosa spp., Burseraceae spp., Dipterocarpaceae spp., Fabaceae spp., Gironniera spp., Myrtaceae spp., Plaquium spp., Porterandia sp., Shorea spp.*	*Alstonia spp., Artocarpus spp., Fabaceae sp., Hevea sp., Macaranga spp., Porterandia sp., Sloetia sp.*	*Hevea brasiliensis*	*Elaeis guineensis*
	clay Acrisol landscape			
Plantation age (years)	ND	ND	7–16	9–13
Tree density (trees ha^{-1})[a]	471 (31)	685 (72)	497 (15)	134 (6)
DBH (cm)[a]	23.0 (0.4)	17.3 (0.6)	15.2 (0.7)	NA
Basal area[1] (m^2 ha^{-1})[a]	29.4 (1.7)	21.1 (1.4)	10.0 (1.4)	NA
Tree height (m)[a]	17.0 (0.5)	15.2 (0.3)	13.4 (0.1)	4.0 (0.3)
Cumulative fine root biomass in the top 1 m of	140.4 (33.0) $_{c}$	402.2 (65.9) $_{b}$	309.6 (16.0) $_{bc}$	630.1 (86.2) $_{a}$

soil (g m^{-2})				
Most common tree species[b]	*Archidendron sp., Baccaurea spp., Ochanostachys sp.*	*Artocarpus spp., Endospermum sp., Hevea sp., Macaranga spp.*	*Hevea brasiliensis*	*Elaeis guineensis*

[a] Kotowska et al., 2015.

[b] Rembold et al. (unpublished data), based on trees found in five subplots (5 m x 5 m) of each replicate plot (50 m x 50 m) which had ≥ 20 individuals, except Fabaceae spp. which had ≤ 20 individuals.

Table S2.3. Pearson correlations among element concentrations (mg l^{-1}) in soil solution (1.5-m depth) from four land uses in a loam Acrisol soil landscape, Jambi, Sumatra, Indonesia. Correlations were carried out using monthly averages of four replicate plots, except oil palm plantation using average of three replicate plots, per land use (*n* = 12).

Elements	NO_3^--N	DON	DOC	Na	K	Ca	Mg	Total Al	Total S	Cl
	Forest									
NH_4^+-N	0.22	0.79**	0.48	0.23	0.64*	0.67*	0.65*	0.58*	0.30	0.58*
NO_3^--N		-0.24	-0.12	-0.09	0.35	-0.26	-0.25	-0.45	0.63*	-0.47
DON			0.77**	0.36	0.43	0.80**	0.77**	0.84**	-0.17	0.86**
DOC				0.36	0.45	0.72**	0.71**	0.73**	-0.02	0.68*
Na					0.58*	0.53†	0.46	0.34	0.23	0.45
K						0.51†	0.45	0.29	0.71**	0.33
Ca							0.99**	0.94**	0.00	0.92**
Mg								0.95**	-0.03	0.92**
Total Al									-0.28	0.95**
Total S										-0.23
	Jungle rubber									
NH_4^+-N	0.32	0.80**	0.73**	0.35	0.77**	0.53†	0.67*	0.55*	0.17	0.79**
NO_3^--N		0.28	0.35	0.17	0.20	0.65*	0.62*	0.61*	-0.11	0.65*
DON			0.77**	0.72**	0.85**	0.72**	0.79**	0.30	0.60*	0.68*
DOC				0.63*	0.76**	0.51†	0.53†	0.13	0.57*	0.49†
Na					0.80**	0.58*	0.55*	-0.18	0.93**	0.29
K						0.65*	0.70**	0.12	0.65*	0.60*
Ca							0.97**	0.56*	0.32	0.84**
Mg								0.65*	0.27	0.93**
Total Al									-0.47	0.85**
Total S										-0.02
	Rubber plantation									
NH_4^+-N	0.10	-0.12	0.31	0.61*	-0.05	0.17	-0.07	-0.41	0.65*	-0.18
NO_3^--N		-0.32	-0.25	0.25	-0.48	0.42	0.15	-0.09	0.26	0.31
DON			0.53†	0.04	0.65*	0.37	0.65	0.67*	-0.28	0.39
DOC				0.50†	0.46	0.51†	0.50†	0.29	0.30	0.34
Na					0.17	0.46	0.08	-0.34	0.85**	0.00
K						0.24	0.55*	0.54†	-0.15	0.38
Ca							0.81**	0.40	0.27	0.72**
Mg								0.84**	-0.26	0.92**
Total Al									-0.7**	0.83**
Total S										-0.35

Elements	NO_3^--N	DON	DOC	Na	K	Ca	Mg	Total Al	Total S	Cl
	Oil palm plantation									
NH_4^+-N	0.54†	-0.28	-0.12	0.00	0.50	0.15	0.37	0.46	0.22	0.46
NO_3^--N		0.08	-0.12	0.14	-0.02	-0.49	0.00	0.63*	-0.38	0.10
DON			-0.18	-0.57*	-0.12	0.16	0.31	0.50	-0.06	0.08
DOC				-0.22	0.08	0.02	0.29	-0.17	0.40	-0.47
Na					-0.12	-0.45	-0.45	-0.37	-0.38	0.22
K						0.58*	0.43	-0.17	0.58*	0.27
Ca							0.48	-0.19	0.79**	0.45
Mg								0.40	0.72**	0.41
Total Al									-0.16	0.27
Total S										0.30

*, ** - significant at $p \leq 0.05$ and $p \leq 0.01$, respectively; † - marginally significant at $p \leq 0.09$; element that had concentrations < 0.03 mg l^{-1} (total Fe, total Mn, and total P) and total Si (that has no net charge and did not show correlation with other elements) were excluded.

Table S2.4. Pearson correlations among element concentrations (mg l^{-1}) in soil solution (1.5 m depth of soil) from four land uses in a clay Acrisol soil landscape, Jambi, Sumatra, Indonesia. Correlations were carried out using monthly averages of four replicate plots, except oil palm plantation using average of three replicate plots, per land use (*n* = 12).

Elements	NO_3^--N	DON	DOC	Na	K	Ca	Mg	Total Al	Total S	Cl
	Forest									
NH_4^+-N	-0.48	0.10	0.81**	0.63*	0.23	0.51†	0.28	-0.11	-0.27	0.09
NO_3^--N		-0.39	-0.48	-0.24	-0.18	-0.05	-0.03	0.36	0.12	0.37
DON			0.57*	0.32	0.53†	0.17	0.20	-0.28	0.25	-0.20
DOC				0.66*	0.41	0.48	0.31	-0.25	-0.15	-0.06
Na					0.69*	0.52†	0.54†	-0.22	-0.24	-0.10
K						0.74**	0.88**	0.22	-0.17	0.26
Ca							0.93**	0.54†	-0.29	0.70**
Mg								0.52†	-0.34	0.59*
Total Al									-0.15	0.94**
Total S										-0.10
	Jungle rubber									
NH_4^+-N	0.01	0.23	0.36	0.35	0.35	0.29	0.29	0.16	0.31	0.18
NO_3^--N		0.55*	0.32	0.30	0.49†	0.51†	0.50†	0.35	0.13	0.42
DON			0.58*	0.19	0.69**	0.50†	0.63*	0.70**	-0.22	0.49†
DOC				-0.24	0.11	-0.14	-0.05	0.29	0.06	-0.20
Na					0.68**	0.84**	0.73**	0.01	0.52†	0.66*
K						0.87**	0.93**	0.63*	0.09	0.84**
Ca							0.97**	0.50†	0.09	0.95**
Mg								0.66*	-0.04	0.97**
Total Al									-0.62*	0.68*
Total S										-0.18
	Rubber plantation									
NH_4^+-N	0.22	-0.20	0.81**	0.85**	0.47	0.19	0.10	-0.20	0.52†	-0.06
NO_3^--N		-0.18	-0.07	-0.16	-0.44	-0.68*	-0.60*	-0.38	0.05	-0.63*
DON			0.21	-0.29	0.41	0.40	0.55*	0.65*	-0.57*	0.48
DOC				0.79**	0.71**	0.54†	0.45	0.20	0.43	0.30
Na					0.61*	0.38	0.21	-0.15	0.65*	0.07
K						0.67*	0.66*	0.46	0.08	0.64*
Ca							0.93**	0.73**	-0.16	0.83**
Mg								0.88**	-0.39	0.93**
Total Al									-0.58*	0.89**
Total S										-0.40

Elements	NO_3^--N	DON	DOC	Na	K	Ca	Mg	Total Al	Total S	Cl
	Oil palm plantation									
NH_4^+-N	0.08	0.02	0.15	0.39	0.37	0.16	0.06	0.06	0.46	-0.01
NO_3^--N		-0.09	-0.18	0.03	0.46	0.51†	-0.01	0.19	0.33	-0.49
DON			0.49	0.70*	0.69*	0.67*	0.42	0.45	0.54†	0.63*
DOC				0.52†	0.66*	0.56†	0.50	0.56†	0.25	0.70*
Na					0.61*	0.61*	0.29	0.21	0.75**	0.55†
K						0.85**	0.74**	0.78**	0.52†	0.59*
Ca							0.81**	0.74**	0.69*	0.64*
Mg								0.95**	0.26	0.74**
Total Al									0.15	0.75**
Total S										0.26

*, ** - significant at $p \leq 0.05$ and $p \leq 0.01$, respectively; † - marginally significant at $p \leq 0.09$; element that had concentrations < 0.03 mg l^{-1} (total Fe, total Mn, and total P) and total Si (that has no net charge and did not show correlation with other elements) were excluded.

Chapter 3

Leaching losses differ between fertilized and frond-stacked areas of oil palm plantations in Sumatra, Indonesia

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Abstract

In oil palm plantations converted from tropical forests, it is not known whether and how management practices affect nutrient leaching losses. Our study aimed to quantify nutrient leaching losses from fertilized areas and areas covered by frond stacks in smallholder oil palm plantations. In Jambi Province, Indonesia, we selected two landscapes with highly weathered Acrisol soils that differed in texture: loam and clay. In each landscape, we investigated four sites in the loam Acrisol and three sites in the clay Acrisol. Using suction cup lysimeters installed under frond stacks and within fertilized area at 1.5-m soil depth, we sampled soil water at bi-weekly to monthly interval from February to December 2013 to measure leaching losses. In the loam Acrisol landscape with high fertilization rates and recent lime application, leaching fluxes of dissolved N, base cations, S, Mn, Al and Cl were higher in the fertilized than the frond-stacked areas. Dissolved organic carbon (DOC) leaching losses in the fertilized area were 70% higher than the frond-stacked area, probably as a result of high DOC production and/or low DOC retention. In the clay Acrisol landscape with low fertilization rates, trends were the same but differences were only significant for dissolved K leaching. Differences in dissolved organic N leaching losses between loam and clay Acrisol soils were related to clay contents. Lower total ionic concentrations in soil solution of the fertilized area in the clay Acrisol soil compared to the loam Acrisol soil were due to the higher clay contents and lower fertilization rates of the former than the latter. The present fertilizer application practice of smallholders (i.e. pulse rate of fertilization around oil palm trees) results in soil water acidification and increased Al concentrations, which may potentially affect ground water quality.

3.1 Introduction

In the last two decades, annual world palm oil consumption has strongly increased, driving the global expansion of oil palm plantations. The majority of new oil palm plantation development is occurring in Southeast Asia (between 10^0N and 10^0S of the equator) due to the crop's particular climatic requirements, with the two largest cultivation areas located in Indonesia and Malaysia (Fairhust and Härdter, 2003). In Indonesia, the province of Jambi, Sumatra, experienced oil palm expansion of approximately 85% from 2000 to 2010, conversely the primary forest area in Jambi decreased by approximately 17% during this period (Margono et al., 2012; Luskin et al., 2013). According to Statistics Indonesia (2013), 61% of the oil palm plantation area in the province of Jambi is owned by smallholders whilst 39% is owned by large-scale enterprises (both state and private). Whereas the economy benefits from the oil palm expansion by providing jobs and income to many people are considerable (Rist et al., 2010; Statistics of Jambi Province, 2012), it also strongly reduces the quality of ecosystem functions provided such as strong decreases in soil carbon stocks (van Straaten et al., 2015), changes in nitrogen cycling (Allen et al., 2015) and trace gas fluxes (Hassler et al., 2015).

Oil palm plantations require large quantities of nutrients to support vegetative growth and fruit production (Goh and Härdter, 2003). The nutrient demand of nitrogen (N), phosphorus (P), potassium (K) and magnesium (Mg) per hectare per year increases up to five times from 3- to 15-year old oil palm plantations (Ng et al., 1999). On the other hand, the majority of land conversion for plantations in Sumatra has occurred on heavily weathered soils such as Acrisol soils (Red-yellow Podzolic soil in the Indonesian classification system), which are characterized by low soil fertility (Tan, 2008; FAO et al., 2009). Since low soil fertility inhibits oil palm production, copious amounts of fertilizer are typically applied in oil palm plantations to maintain or increase nutrient availability especially on soils with inherently low fertility such as Acrisol soils.

In most oil palm plantations, nutrient levels in soils are increased or replenished by application of chemical fertilizer around the palm tree base or by stacking pruned fronds in palm inter-rows. Recycling of oil palm residue is less commonly practiced, but if it is done, this is also concentrated around the palm trees (Comte et al., 2012). These management practices cause considerable spatial variation in nutrient input which is also likely to affect nutrient leaching losses, since the rate of nutrient leaching is typically determined by the availability of nutrients in soil profile (Dechert et al., 2005). Soil nutrient stocks are usually higher in fertilized areas than in unfertilized areas, as a direct result of the spatial placement of chemical fertilizer (Anuar et al., 2008). The spatial variability in nutrient input is also likely to increase nutrient concentrations of draining water from fertilized compared to unfertilized areas (Tung et al., 2009). In the frond-stacked area, decomposition of fronds can replenish substantial quantities of nutrients back into the soil via mineralization processes since fronds contain considerable amount of nutrients (e.g. 10 ton of frond dry matter contained 82 kg N, 7 kg P and 102 kg K; Kee and Chew, 1997). The temporal release of nutrients (i.e. N, P, K, Calcium (Ca), and Mg) from the pruned fronds typically follows an exponential decrease with larger nutrient release in the beginning followed by slower release during later stages of decomposition (Moradi et al., 2014). Nevertheless, the availability of nutrients in soils under the frond stack has been shown to be lower than in the area with inorganic fertilizer application, since chemical fertilizer dissolve quickly, causing a peak in nutrient concentrations in the soil solution, which is absent when nutrients are mineralized in decomposing fronds (Anuar et al., 2008; Banabas et al., 2008).

Apart from management practices, nutrient leaching losses are also strongly affected by soil texture through its effect on soil fertility (Silver et al., 2000; Silva et al., 2005). In our study area with highly weathered Acrisol soils, it has been shown that soils with higher clay content exhibited larger soil-N cycling pools and rates (Allen et al., 2015). Additionally, soils with high clay contents have more negative charge and consequently retain more cations than soils with low clay contents (Ohta et al., 1993). As a result the increasing clay content with soil depth which is typical for Acrisol soils may prevent nutrient leaching.

To our knowledge, in oil palm no studies of nutrient leaching have been conducted that account for the spatial and temporal variability caused by frond stack area and fertilizer application. Our research aimed to assess how in smallholder oil palm plantations soil management such as spreading fertilizer around the palm trees and stacking palm fronds in the palm inter-rows affects leaching losses in Acrisol soils with differing soil texture. We hypothesized that: 1) fertilized area around each palm tree will have higher soil nutrient stocks and nutrient leaching losses due to the pulsed nature of nutrient addition while under frond stacks leaching will be minimal since the slow mineralization of nutrients from decomposing fronds will be taken up by roots before it is lost through leaching, and 2) soils with higher clay content will have higher soil nutrient levels and lower nutrient leaching losses both in the frond-stacked and fertilized areas than in soils with lower clay content.

3.2 Materials and methods

3.2.1 Study sites and experimental design

The study was conducted in smallholder oil palm plantations that were located at elevations between 35-84 m above sea level and had slopes ≤ 8 % in Jambi Province, Sumatra, Indonesia. The climate is humid tropical with a mean annual air temperature of 26.7 ± 0.1 °C and a mean annual precipitation of 2235 ± 385 mm (1991-2011; Jambi-Sultan-Thaha airport data from the Meteorological, Climatological and Geophysical Agency). During our study period (2013), the dry season lasted from mid-June until end of October, when rainfall was reduced by 35-57 % compared to the wetter months during which rainfall was between 333 and 362 mm per month.

We selected eight smallholder oil palm plantations that were established after logging, clearing and burning of either forest or secondary forest enriched with rubber trees (Euler, 2015). The oil palm plantations were selected in two different landscapes (four smallholders in each landscape) that mainly differed in soil texture: loam Acrisol soils and clay Acrisol soils. In the loam Acrisol landscape, the particle size distribution in the top 0.5 m depth of oil palm sites consisted of 38.0 ± 11.5% of sand, 28.6 ± 9.9% of silt and 33.4 ± 2.2% of clay whilst in the clay Acrisol landscape the soil contained 10.3 ± 2.4% sand, 30.0 ± 5.6% of silt and 59.7 ± 5.2% clay in the top 0.5 m depth. At each plantation we established a plot of 50 m x 50 m with a minimum distance of 200 m between plots. Annual nutrient deposition from bulk precipitation was comparable between the loam and the clay Acrisol landscape (Table 2.2; pg. 28).

For the loam Acrisol landscape, the smallholder oil palm plantations were all located in the Bajubang and Muara Bulian districts within the Batanghari regency, 80 km southwest of Jambi City (01°55'40" S, 103°15'33" E). The plantations in this landscape ranged from 12-16 years old, and had a tree density of 140 ± 5 tree ha^{-1}, tree height 4.9 ± 0.6 m and cumulative fine root biomass in the top 0.5 m of soil of 6.6 ± 0.9 Mg ha^{-1} (Kotowska et al. 2015). In the clay Acrisol landscape, the plantations were located in the Air Hitam district within the Sarolangun regency, 160 km southwest of Jambi City (02°0'57" S, 102°45'12" E). Oil palm trees in the clay landscape ranged from 9 to 13 years old with tree densities of 134 ± 6 tree ha^{-1}, tree heights of 4.0 ± 0.3 m and cumulative fine root biomass in the top 0.5 m of soil 3.7 ± 0.5 Mg ha^{-1} (Kotowska et al., 2015).

Since all oil palm sites in both landscapes were owned and managed by smallholders, there was considerable variation in management practices employed. During our study period

(2013), fertilization were applied once in the rainy season (October to March) in the clay Acrisol soil whereas those in the loam Acrisol soil were applied once in the rainy season and once in the dry season (April to September). Fertilization rates ranged between 300-550 kg NPK-fertilizer ha^{-1} $year^{-1}$ (equivalent to 48-88 kg N ha^{-1} $year^{-1}$, 21-38 kg P ha^{-1} $year^{-1}$ and 40-73 kg K ha^{-1} $year^{-1}$), with the lower range in the clay Acrisol soil and the upper range in the loam Acrisol soil. Additionally, three of the smallholders applied 157 kg K-KCl ha^{-1} $year^{-1}$ and 143 kg Cl-KCl ha^{-1} $year^{-1}$; two of the smallholders applied 138 kg urea-N ha^{-1} $year^{-1}$. One smallholder also applied lime (200 kg dolomite ($CaMg(CO_3)_2$) ha^{-1} $year^{-1}$) in the loam Acrisol soil. Prior to our study year, kieserite ($MgSO_4.H_2O$) and borate ($Na_2B_4O_2.5H_2O$) fertilizers were also used in some oil palm plantations in the loam Acrisol soil. Although the type, rate and timing of fertilizer application varied among smallholders, the spatial location of fertilizer application in all sites was similar: fertilizers and lime were spread by hand around each palm tree at about 0.8-1.5 m distance from the palm stem. For weed control, smallholders used herbicide (i.e. Gramoxone and Roundup) at 2.5 to 5 liters ha^{-1} $year^{-1}$ and/or manual weeding, twice a year. In all plantations, senescing oil palm fronds were regularly cut and stacked at a distance of 4-5 m from the rows of palm trees (row spacing was about 9 m). This was done to facilitate walking and working (e.g. harvesting) in the plantations. The annual input of pruned fronds was 5.5 ± 0.2 Mg ha^{-1} on the loam Acrisol soil and 7.2 ± 1.0 Mg ha^{-1} on the clay Acrisol soil (Kotowska et al., 2015). Harvesting was done on average every two weeks with a mean total dry yield of 19.9 ± 1.3 Mg ha^{-1} $year^{-1}$ in the loam Acrisol and 14.7 ± 1.6 Mg ha^{-1} $year^{-1}$ in the clay Acrisol (Kotowska et al., 2015).

3.2.2 Lysimeter installation and soil water sampling and laboratory analysis

We measured nutrient leaching losses by collecting soil water samples at 1.5 m depth (below the rooting depth) from each replicate plot in the fertilized and frond-stacked areas. Soil water was sampled using suction cup lysimeters (P80 ceramic, maximum pore size 1 μm;

CeramTec AG, Marktredwitz, Germany) that were connected to collection containers (dark glass bottles). All equipment (i.e. lysimeters, sample tubes and collection containers) were acid-washed and rinsed with copious amounts of deionized water before installation. We installed lysimeters in the field three months before the first sampling to allow resettling of natural soil conditions prior to measurement. The collection containers were placed in plastic buckets with lids and buried in the ground approximately 1.3-m from the lysimeters. In the fertilized areas, the lysimeters were installed 1.3 to 1.5 m from the palm stem, whilst in the frond-stacked areas, lysimeters were installed under the frond stack (4-5 m from the palm tree).

In the clay Acrisol landscape, soil water samples were collected on three replicate plots only, because one of the land-owners sold his plantation and nullified our contract for access to continue sampling; an alternative plot was established in the middle of sample collection which did not give enough time to collect water samples since acclimatization should be done for three months prior to first water sampling. In the loam Acrisol landscape, the lysimeter for soil water sampling in one fertilized area was damaged by the workers. Soil water was sampled biweekly to monthly from February to December 2013 by applying a 40 kPa vacuum in the collection containers, thus collecting water from rapidly and slowly draining soil pores. Soil water samples were transferred from the collection container into 100 ml plastic bottles, which were acid-washed and thoroughly rinsed with deionized water prior to sampling time. After arrival in the laboratory of Jambi University, we used a 20 ml subsample for pH measurement while the remaining water was immediately frozen and transported by air to the laboratory of Soil Science Tropical and Subtropical Ecosystems (SSTSE), Goettingen University, Germany, for measuring element concentrations.

We analyzed soil water samples for macro and micro element concentrations. Total dissolved nitrogen (TDN), ammonium (NH_4^+), nitrate (NO_3^-) and chloride (Cl^-)

concentrations were measured using flow injection colorimetry (SEAL Analytical AA3, SEAL Analytical GmbH, Norderstedt, Germany). We calculated dissolved organic nitrogen (DON) as: [DON] = [TDN] – [NH_4^+-N] – [NO_3^--N]. Dissolved organic carbon (DOC) was determined using a Total Organic Carbon Analyzer (TOC-Vwp, Shimadzu Europa GmbH, Duisburg, Germany). Sodium (Na), K, Ca, Mg, total aluminum (Al), total iron (Fe), total manganese (Mn), total sulfur (S), total P, and total silica (Si) in soil water were measured using inductively coupled plasma-atomic emission spectrometer (iCAP 6300 Duo View ICP Spectrometer, Thermo Fischer Scientific GmbH, Dreieich, Germany). Method detection limits for each element were: 6 µg NH_4^+-N l^{-1}, 5 µg NO_3^--N l^{-1}, 2 µg TDN l^{-1}, 4 µg DOC l^{-1}, 30 µg Na l^{-1}, 50 µg K l^{-1}, 3 µg Ca l^{-1}, 3 µg Mg l^{-1}, 2 µg total Al l^{-1}, 3 µg total Fe l^{-1}, 2 µg total Mn l^{-1}, 10 µg total P l^{-1}, 10 µg total S l^{-1}, 1 µg total Si l^{-1} and 30 µg Cl l^{-1}. For concentrations below these detection limits, we assigned a value of zero.

We made partial cations-anions charge balances (Fig. 3.1) of the major solutes (i.e. concentrations > 0.03 mg l^{-1}) in soil solution by expressing solute concentrations into μmol_{charge} l^{-1} (molar concentration multiplied by the equivalent charge of each solute). Contributions of organic acids ($RCOO^-$) and bicarbonate (HCO_3^-) were not measured, but were calculated together with S (having very low concentration) from the difference of cations minus anions. Charge contributions of total Al were assumed to be 3^+. Elements that had concentrations < 0.03 mg l^{-1} (total Fe, Mn and P) and total Si (commonly in a form of monosilicic acid ($H_4SiO_4^0$) that has no net charge) were excluded (Hedin et al., 2003).

3.2.3 Soil characteristics

To evaluate the effect of management practices (i.e. fertilizer and frond stack) on soil nutrient stocks and its relation to nutrient leaching losses, we measured soil characteristics data for the top 0.1 m soil depth (Allen et al., 2015). Soil samples were collected from the area where fronds were piled in the inter-rows; from the fertilized area and from inter-row where no fronds were stacked. The sampling distance of the frond-stacked area from the planted rows of palm trees was 3.5 m, while fertilizer was never placed more than 1.5 m from the palm tree. For the fertilized area, the soil samples were taken at 1.4 ± 0.1 m from the palm tree. The soil samples from the inter-row without frond were taken at approximately > 2.0 m from the tree base. All soil samples were air-dried and sieved (2 mm sieve) in the laboratory of Jambi University before transport by air to the laboratory of SSTSE, Goettingen University, Germany, for measuring soil biochemical characteristics (i.e. pH, organic C, total N, effective cation exchange capacity (ECEC), exchangeable Ca, Mg, K, Na, Al, Fe, Mn, and extractable P). Soil sampling and analysis are described in detail by Allen et al. (2015).

3.2.4 Calculation of element leaching fluxes and water balance

Nutrient leaching fluxes were calculated by multiplying the element concentrations from lysimeters in each sampling period with the total biweekly or monthly drainage water fluxes calculated for 1.5 m depth. For the biweekly or monthly drainage fluxes, we used the cumulative predicted daily drainage water fluxes at depth of 1.5 m. We estimated daily drainage water fluxes using the soil water module of the Expert-N model (Priesack, 2005). This model was used successfully in our earlier work on nutrient leaching losses from conversion of montane forest to agricultural land uses in Sulawesi, Indonesia (Dechert et al., 2005). Calculation of daily drainage water fluxes follows the equation of the water balance:

$$\Delta W + D = P - R - ET \text{ and } ET = I + E + T$$

in which ΔW = change in soil water storage, D = drainage water below rooting zone, P = precipitation, R = runoff, and ET = total evapotranspiration, which is equal to the sum of three terms: I = interception of water by plant foliage, assumed to evaporate, E = evaporation from soil, and T = transpiration by plants following water uptake.

The Expert-N model calculates actual evapotranspiration using the Penman-Monteith method, with aerodynamic and canopy conductance adjusted to the site conditions. Estimation of actual runoff based on the sites' slope, whilst vertical water movement using Richards equation, of which the parameterization of the hydraulic functions was based on the measured water retention curve using standard equations (Mualem, 1976; Van Genuchten, 1980). The data input for water balance simulation consisted of daily climate (i.e. air temperature, average relative humidity, average wind speed, total solar radiation, and total precipitation), vegetation (i.e. leaf area index (LAI) and fine root biomass distribution), and soil characteristics (i.e. soil texture, water retention curve, and bulk density). For the loam Acrisol landscape, the climate data were taken from a climatological station at the Harapan Forest Reserve approximately 10-20 km from our plots. For the clay Acrisol landscape, we used the climate data from climatological stations at the villages of Sarolangun and Lubuk Kepayang, approximately 20 km and 10 km, respectively, from our plots. The LAI were taken from field measurements (Rembold et al. unpublished data), while the fine root biomass and soil characteristics were taken from our measurements.

The output of the Expert-N model was validated by comparing the modelled soil matrix potential with the field measurement of matrix potential. We measured soil matrix potential biweekly to monthly from February to December 2013, using tensiometers (P80 ceramic, maximum pore size 1 μm; CeramTec AG, Marktredwitz, Germany), which were installed at 0.3 m and 0.6 m depths in two smallholder oil palm plantation per landscape. We accepted the output of the Expert-N model if the modelled and measured soil matrix potential at each depth were comparable (paired t-test; all $p \geq 0.21$, n = 10 for 30 cm depth and n = 9 for 60 cm depth) and had strong correlations (Pearson correlation coefficients of 0.79 to 0.96, all $p \leq 0.01$). The resulting annual water balance in the loam Acrisol landscape showed 3418 mm of precipitation, 1384 mm of evapotranspiration, 545 mm of runoff, and 1483 mm of drainage fluxes. In the clay Acrisol landscape, the annual water balance consisted of 3475 mm of precipitation, 1622 mm of evapotranspiration, 722 mm of runoff, and 1117 mm of water drainage (Table 2.1; pg. 27).

3.2.5 Statistical analysis

Prior to statistical analyses, we tested all data for normality (Shapiro-Wilk's test) and homogeneity of variance (Levene's test) across sampling area (i.e. fertilized and frond-stacked areas) in each soil landscapes and across soil landscapes in each sampling area. Logarithmic or square-root transformation was used for data that were non-normally distributed or heterogeneity of variance. We tested differences in soil nutrient stocks for 0.1 m depth, in element concentration in soil solution at 1.5 m depth, and in annual nutrient leaching fluxes at 1.5 m depth: 1) between fertilized and frond-stacked areas for each soil landscape (hypothesis 1), and 2) between loam and clay Acrisol soils for each sampling area (fertilized and frond-stacked; hypothesis 2) using linear mixed effects (LME) models (Crawley, 2009). For element concentrations, the LME model had sampling area or landscape as the fixed effect with spatial replication (plot) and time (biweekly or monthly sampling period of element concentrations) as random effects. For the annual leaching fluxes (which were the sum of the bi-weekly or monthly sampling), the LME model had sampling area or landscape as the fixed effect with only spatial replication (plot) as a random effect. We extended the LME model to include: either 1) a variance function that allows different variances of the fixed effect, 2) a first-order temporal autoregressive process that assumes that correlation between measurement periods decreases with increasing time difference, or both if these improved the relative goodness of the model fit based on the Akaike information criterion. Fixed effect were considered significant based on analysis of variance at $p \leq 0.05$, and differences between sampling site or landscape were assessed using Fisher's least significant difference test $p \leq 0.05$. For a few specified parameters, we also considered marginal significance at $p \leq 0.09$, because our experimental design encompassed the inherent spatial variability in our study area. We tested the relationships among cation and anion charge concentrations in soil solution across one year (i.e. $n = 12$) for each sampling area within each landscape with Pearson correlation analysis; the data taken from the average of four replicate plots per land use and landscape (Table S3.1). Additionally, Spearman's rank correlation tests were conducted to assess relationships between annual nutrient leaching fluxes and soil biochemical characteristics across sampling area (fertilized and frond-stacked) for loam and clay Acrisol soil landscape ($n = 7$ and 6, respectively) and across landscapes within each sampling area ($n = 7$). All statistical analyses were conducted using R 3.0.2 (R Development Core Team, 2013).

3.3 Results

3.3.1 Difference between frond stack area, fertilized area and inter-row area within each landscape

In the loam Acrisol landscape, the fertilized area in smallholder oil palm plantation had higher soil pH and soil nutrient stocks (i.e. exchangeable Ca, extractable P and base saturation) as compared to the frond-stacked and inter-row areas (all $p \leq 0.05$, except $p \leq 0.09$ for exchangeable Ca; Table 3.1). We did not detect any difference between the frond-stacked and inter-row areas. Furthermore, the soil solution under the fertilized area in the loam landscape had a lower pH and higher concentrations of DOC, Na, Ca, Mg, total Al, total Mn and Cl (all $p \leq 0.05$, except $p \leq 0.09$ for pH and Na) compared to the frond-stacked area (Table 3.2). Management practices (i.e. fertilization and liming) also resulted in increased annual leaching fluxes of N-NH_4^+, TDN, Na, Ca, Mg, total Al, total Mn, total S and Cl (all $p \leq 0.05$, except $p \leq 0.09$ for Na, total Mn, and total S) in the fertilized area compared to the frond-stacked area within the loam landscape (Table 3.3). Also annual leaching fluxes of

DOC were higher ($p \leq 0.00$) in the fertilized area than in the frond-stacked area (Table 3.3). Finally, we observed negative correlations ($p \leq 0.05$) of soil solution pH with total Al (in fertilized and frond-stacked areas) and total Mn (in frond-stacked area) concentrations in soil solution within the loam Acrisol landscape (Table S3.1) and positive correlation of annual leaching fluxes (i.e. NH_4^+-N, DOC, Na, total S) with soil pH (all $p \leq 0.05$, r = 0.89 to 0.99) and Na leaching fluxes with base saturation ($p = 0.03$, r = 0.89) across sampling areas in the loam Acrisol landscape.

In the clay Acrisol landscape, the top soil in the fertilized area had lower exchangeable Al and higher C:N ratio, ECEC, base saturation, exchangeable Ca and extractable P as compared to the frond-stacked and inter-row areas (all $p \leq 0.05$; Table 3.1). Also in this landscape we did not detect any difference between soil nutrient content of the frond-stacked and inter-row areas, and thus we will only discuss differences in soil parameters between fertilized and frond-stacked areas in the rest of this manuscript. Nutrient concentrations (i.e. Na and K) in the soil solution were also higher (all $p \leq 0.06$) in the fertilized than frond-stack areas (Table 3.2). This also resulted in an increased K leaching flux in the fertilized area compared to the frond-stacked area ($p = 0.06$) that only received pruned fronds (Table 3.3). We observed a strong correlation ($p \leq 0.01$) among base cations (K, Na, Ca, and Mg) in the soil solution (Table S3.1) within the fertilized and frond-stacked areas in the clay Acrisol, whilst the annual leaching fluxes of K were marginally correlated with base saturation across sampling areas in the clay Acrisol ($p = 0.06$, r = 0.83).

Table 3.1. Soil characteristics at the top 0.1 m depth within the fertilized, near the frond-stacked, and inter-row areas in smallholder oil palm plantations in the loam and clay Acrisol soils, Jambi, Sumatra, Indonesia.

Characteristics	fertilized area	frond-stacked area	inter-row area
	loam Acrisol soil landscape		
pH (1:4 H_2O)	[a]4.91 (0.2)$_{a\ A}$	4.43 (0.05)$_b$	4.49 (0.07)$_b$
Soil organic C (kg C m^{-2})	1.87 (0.09)$_B$	1.86 (0.33)$_B$	1.80 (0.18)$_B$
Total N (g N m^{-2})	159.60 (15.44)$_B$	148.70 (23.38)$_B$	141.90 (12.21)$_B$
C:N ratio	11.79 (0.54)$_B$	12.38 (0.31)	12.58 (0.62)
Effective cation exchange capacity ($mmol_{charge}$ kg^{-1} soil)	51.43 (8.56)$_B$	37.91 (8.02)$_B$	39.04 (8.01)$_B$
Base saturation (%)	66.06 (17.76)$_a$	23.28 (5.39)$_b$	25.63 (4.91)$_b$
Potassium (g K m^{-2})	3.09 (0.81)	2.12 (0.75)$_B$	1.98 (0.77)
Sodium (g Na m^{-2})	5.06 (0.14)$_A$	4.05 (1.71)	3.61 (1.13)
Calcium (g Ca m^{-2})	63.29 (23.92)$_{a†}$	12.89 (6.32)$_{b†}$	15.75 (6.77)$_{b†}$
Magnesium (g Mg m^{-2})	3.68 (1.92)	1.32 (0.54)	1.68 (0.89)
Aluminum (g Al m^{-2})	12.60 (4.77)	24.62 (4.38)$_B$	24.40 (3.16)$_B$
Iron (g Fe m^{-2})	0.37 (0.17)	0.58 (0.11)	0.54 (0.04)
Manganese (g Mn m^{-2})	0.68 (0.41)	0.56 (0.31)$_B$	0.41 (0.18)$_B$
Bray-extractable phosphorus (g P m^{-2})	2.30 (0.62)$_a$	0.65 (0.14)$_{b\ B†}$	0.71 (0.14)$_{b\ B†}$
	clay Acrisol soil landscape		
pH (1:4 H_2O)	4.28 (0.09)$_B$	4.43 (0.05)	4.44 (0.05)
Soil organic C (kg C m^{-2})	3.84 (0.34)$_A$	3.68 (0.10)$_A$	3.45 (0.37)$_A$
Total N (g N m^{-2})	262.69 (23.60)$_A$	277.02 (12.88)$_A$	260.10 (28.84)$_A$
C:N ratio	14.62 (0.38)$_{a\ A}$	13.33 (0.42)$_b$	13.28 (0.19)$_b$
Effective cation exchange capacity ($mmol_{charge}$ kg^{-1} soil)	110.16 (11.18)$_{a\ A}$	72.05 (6.17)$_{b\ A}$	69.48 (8.55)$_{b\ A}$
Base saturation (%)	71.17 (13.53)$_a$	26.35 (2.85)$_b$	28.63 (7.93)$_b$
Potassium (g K m^{-2})	9.61 (4.94)	4.39 (0.52)$_A$	3.93 (0.71)
Sodium (g Na m^{-2})	1.45 (0.94)$_B$	1.91 (1.80)	1.48 (1.21)
Calcium (g Ca m^{-2})	152.10 (46.20)$_a$	28.03 (6.57)$_b$	35.00 (15.44)$_b$
Magnesium (g Mg m^{-2})	4.54 (0.57)	3.36 (1.04)	3.15 (1.23)
Aluminum (g Al m^{-2})	19.63 (9.39)$_b$	42.29 (1.76)$_{a\ A}$	38.12 (0.79)$_{a\ A}$
Iron (g Fe m^{-2})	0.75 (0.31)	0.81 (0.31)	0.67 (0.27)
Manganese (g Mn m^{-2})	1.95 (1.13)	4.42 (1.69)$_A$	3.70 (1.50)$_A$
Bray-extractable phosphorus (g P m^{-2})	21.18 (12.18)$_a$	1.33 (0.29)$_{b\ A†}$	1.47 (0.35)$_{b\ A†}$

[a]Means (SE, n = 4) followed by different lower case letters indicate significant differences between sampling areas (fertilized, frond-stacked, and inter-row areas) within each landscape and different upper case letters indicate significant differences between landscapes for each sampling areas (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$)

Table 3.2. Nutrient concentrations in soil solution from a depth of 1.5 m within the fertilized area and under the frond-stacked area in smallholder oil palm plantations in the loam and clay Acrisol soils, Jambi, Sumatra, Indonesia

Element	loam Acrisol landscape		clay Acrisol landscape	
	fertilized area	frond-stacked area	fertilized area	frond-stacked area
pH	[a]4.11 (0.11)$_{b\dagger\ B\dagger}$	4.32 (0.03)$_{a\dagger\ B}$	4.56 (0.14)$_{A\dagger}$	4.64 (0.06)$_{A}$
Ammonium (mg N l^{-1})	0.17 (0.01)	0.16 (0.01)	0.15 (0.01)	0.14 (0.00)
Nitrate (mg N l^{-1})	0.32 (0.15)	0.08 (0.04)	0.90 (0.88)	0.02 (0.00)
Dissolved organic nitrogen (mg N l^{-1})	0.11 (0.03)$_{A}$	0.07 (0.01)$_{A\dagger}$	0.04 (0.01)$_{B}$	0.04 (0.01)$_{B\dagger}$
Total dissolved N (mg N l^{-1})	0.60 (0.18)	0.31 (0.04)$_{A}$	1.09 (0.89)	0.20 (0.01)$_{B}$
Dissolved organic carbon (mg C l^{-1})	4.19 (0.10)$_{a}$	3.58 (0.07)$_{b}$	4.79 (0.88)	4.40 (1.14)
Sodium (mg Na l^{-1})	7.20 (3.88)$_{a\dagger}$	2.32 (0.27)$_{b\dagger}$	4.63 (1.20)$_{a\dagger}$	2.50 (0.54)$_{b\dagger}$
Potassium (mg K l^{-1})	0.39 (0.14)	0.36 (0.14)	0.38 (0.05)$_{a}$	0.18 (0.06)$_{b}$
Calcium (mg Ca l^{-1})	2.74 (0.91)$_{a}$	0.72 (0.07)$_{b\ A\dagger}$	0.77 (0.17)	0.51 (0.07)$_{B\dagger}$
Magnesium (mg Mg l^{-1})	0.49 (0.11)$_{a\ A\dagger}$	0.24 (0.03)$_{b}$	0.43 (0.10)$_{B\dagger}$	0.21 (0.07)
Total aluminum (mg Al l^{-1})	1.24 (0.71)$_{a\ A\dagger}$	0.14 (0.01)$_{b}$	0.21 (0.11)$_{B\dagger}$	0.08 (0.03)
Total iron (mg Fe l^{-1})	0.02 (0.00)$_{A}$	0.08 (0.06)	0.01 (0.00)$_{B}$	0.05 (0.04)
Total manganese (mg Mn l^{-1})	0.013 (0.005)$_{a}$	0.006 (0.001)$_{b\ B}$	0.08 (0.06)	0.02 (0.01)$_{A}$
Total phosphorus (mg P l^{-1})	0.005 (0.001)	0.005 (0.001)	0.004 (0.000)	0.01 (0.006)
Total sulfur (mg S l^{-1})	0.14 (0.01)	0.12 (0.00)	0.13 (0.01)	0.12 (0.01)
Total silica (mg Si l^{-1})	0.31 (0.13)	0.17 (0.02)$_{B}$	1.03 (0.39)	0.68 (0.17)$_{A}$
Chloride (mg Cl l^{-1})	20.99 (2.72)$_{a\ A}$	6.20 (0.79)$_{b}$	7.19 (2.10)$_{B}$	4.63 (0.77)

[a]Means (SE, n = 4, except for the clay Acrisol soil and the fertilized area in the loam Acrisol n = 3; see section 3.2.2) followed by different lower case letters indicate significant differences between sampling areas (fertilized vs frond-stacked areas) within each landscape and different upper case letters indicate significant differences between landscapes for each sampling areas (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

Table 3.3. Annual leaching fluxes from a depth of 1.5 m within the fertilized area and under the frond-stacked area in smallholder oil palm plantations in loam and clay Acrisol soils, Jambi, Sumatra, Indonesia.

Element	loam Acrisol landscape		clay Acrisol landscape	
	fertilized area	frond-stacked area	fertilized area	frond-stacked area
Ammonium (g N m^{-2} $year^{-1}$)	[a]0.32 (0.01)$_{a\ A}$	0.20 (0.02)$_{b}$	0.20 (0.02)$_{B}$	0.18 (0.00)
Nitrate (g N m^{-2} $year^{-1}$)	0.59 (0.29)	0.11 (0.05)	1.14 (1.12)	0.03 (0.01)
Dissolved organic nitrogen (g N m^{-2} $year^{-1}$)	0.22 (0.06)$_{A}$	0.11 (0.02)$_{A†}$	0.05 (0.00)$_{B}$	0.05 (0.01)$_{B†}$
Total dissolved N (g N m^{-2} $year^{-1}$)	1.13 (0.31)$_{a}$	0.42 (0.08)$_{b}$	1.39 (1.12)	0.27 (0.02)
Dissolved organic carbon (g C m^{-2} $year^{-1}$)	7.29 (0.25)$_{a}$	4.18 (0.35)$_{b}$	6.25 (1.38)	5.61 (0.99)
Sodium (g Na m^{-2} $year^{-1}$)	13.06 (7.58)$_{a†}$	3.07 (0.54)$_{b†}$	6.26 (1.77)	3.29 (0.60)
Potassium (g K m^{-2} $year^{-1}$)	0.70 (0.22)	0.42 (0.13)	0.49 (0.06)$_{a†}$	0.24 (0.07)$_{b†}$
Calcium (g Ca m^{-2} $year^{-1}$)	4.63 (1.34)$_{a\ A†}$	0.99 (0.20)$_{b}$	1.02 (0.23)$_{B†}$	0.66 (0.07)
Magnesium (g Mg m^{-2} $year^{-1}$)	0.88 (0.21)$_{a}$	0.33 (0.05)$_{b}$	0.57 (0.14)	0.25 (0.06)
Total aluminum (g Al m^{-2} $year^{-1}$)	2.32 (1.34)$_{a\ A†}$	0.23 (0.02)$_{b\ A}$	0.27 (0.12)$_{B†}$	0.10 (0.03)$_{B}$
Total iron (g Fe m^{-2} $year^{-1}$)	0.04 (0.00)$_{A}$	0.13 (0.11)	0.01 (0.00)$_{B}$	0.06 (0.05)
Total manganese (g Mn m^{-2} $year^{-1}$)	0.03 (0.01)$_{a†}$	0.01 (0.00)$_{b†\ B†}$	0.09 (0.07)	0.02 (0.01)$_{A†}$
Total phosphorus (g P m^{-2} $year^{-1}$)	0.01 (0.00)	0.005 (0.001)	0.01 (0.00)	0.02 (0.01)
Total sulfur (g S m^{-2} $year^{-1}$)	0.24 (0.02)$_{a†\ A}$	0.15 (0.03)$_{b†}$	0.17 (0.02)$_{B}$	0.17 (0.00)
Total silica (g Si m^{-2} $year^{-1}$)	0.43 (0.11)	0.29 (0.06)$_{B†}$	1.29 (0.58)	0.80 (0.31)$_{A†}$
Chloride (g Cl m^{-2} $year^{-1}$)	38.01 (6.73)$_{a\ A}$	7.83 (1.21)$_{b}$	9.77 (3.00)$_{B}$	5.62 (0.62)

[a]Means (SE, n = 4, except for the clay Acrisol soil and the fertilized area in the loam Acrisol n = 3; see section 3.2.2) followed by different lowercase letters indicate significant differences between sampling areas (fertilized vs frond-stacked) within each landscape and different uppercase letters indicate significant differences between landscapes for each sampling area (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

3.3.2 Differences between loam and clay Acrisol soils in each sampling areas

In the fertilized area, we observed lower soil pH and exchangeable Na, and higher soil organic C, total N, C:N ratio, and ECEC (all $p \leq 0.05$) in the clay Acrisol soil as compared to the loam Acrisol soil (Table 3.1). The clay soil also had higher soil solution pH and lower concentration of DON, Mg, total Al, total Fe and Cl (all $p \leq 0.05$, except $p \leq 0.09$ for pH, Mg and total Al) in the soil solution than the loam soil in the fertilized area (Table 3.2). Also annual leaching fluxes of NH_4^+-N, DON, Ca, total Al, total Fe, total S and Cl were lower (all $p \leq 0.05$, except $p \leq 0.09$ for Ca and total Al) in the clay soil compared to the loam soil (Table 3.3). The ionic strengths of the soil solution were lower ($p = 0.00$) in the clay Acrisol (317 ± 83 μmol_{charge} l^{-1}) than the loam Acrisol (648 ± 306 μmol_{charge} l^{-1}) within the fertilized area, while the ionic strengths of the frond-stack area were comparable ($p = 0.46$) between the clay Acrisol (173 ± 37 μmol_{charge} l^{-1}) and the loam Acrisol (190 ± 23 μmol_{charge} l^{-1}) (Fig. 3.1). We observed positive correlations of base cations (Na^+, K^+, Ca^{2+}, Mg^{2+}), total Al, and total Fe with Cl under the fertilized area in the clay Acrisol (Table S3.1). Annual leaching fluxes of Ca, total Al and DON were negatively correlated with ECEC (all $p = 0.08$, r = -0.90) whilst NH_4^+-N leaching flux was negatively correlated with C:N ratio ($p = 0.08$, r = -0.90). Soil organic C was negatively correlated with total Fe leaching flux ($p = 0.08$, r = -0.90) across soil landscapes, while soil pH was positively correlated with total S leaching flux ($p \leq 0.01$, r = 0.99).

In the clay Acrisol soil, soil nutrient stocks (i.e. organic C, total N, K, Al, Mn and extractable P) and ECEC under frond stacks were higher than in the loam Acrisol soil ($p \leq 0.05$, except $p \leq 0.09$ for extractable P) (Table 3.1). The soil solution from the frond-stacked area in the clay Acrisol soil also showed higher pH and concentrations of total Mn and total Si in soil solution (all $p \leq 0.05$; Table 3.2) as well as higher annual leaching fluxes of total Mn and total Si compared to the loam Acrisol soil (all $p \leq 0.09$; Table 3.3). In contrast, concentrations of DON, TDN and Ca in soil solution (all $p \leq 0.09$, except $p \leq 0.05$ for TDN; Table 3.2) and annual leaching fluxes of DON and total Al ($p = 0.09$ for DON and $p = 0.02$ for total Al; Table 3.3) were lower in the clay Acrisol than loam Acrisol soils. This was supported by strong correlations ($p \leq 0.01$) of base cations (K^+, Na^+, Ca^{2+}, and Mg^{2+}), total Al and total Mn with Cl^- in the soil solution under the frond-stacked area within the clay Acrisol soil (Table S3.1). Annual leaching fluxes of total Mn were positively correlated with soil exchangeable Mn ($p = 0.01$, r = 0.86).

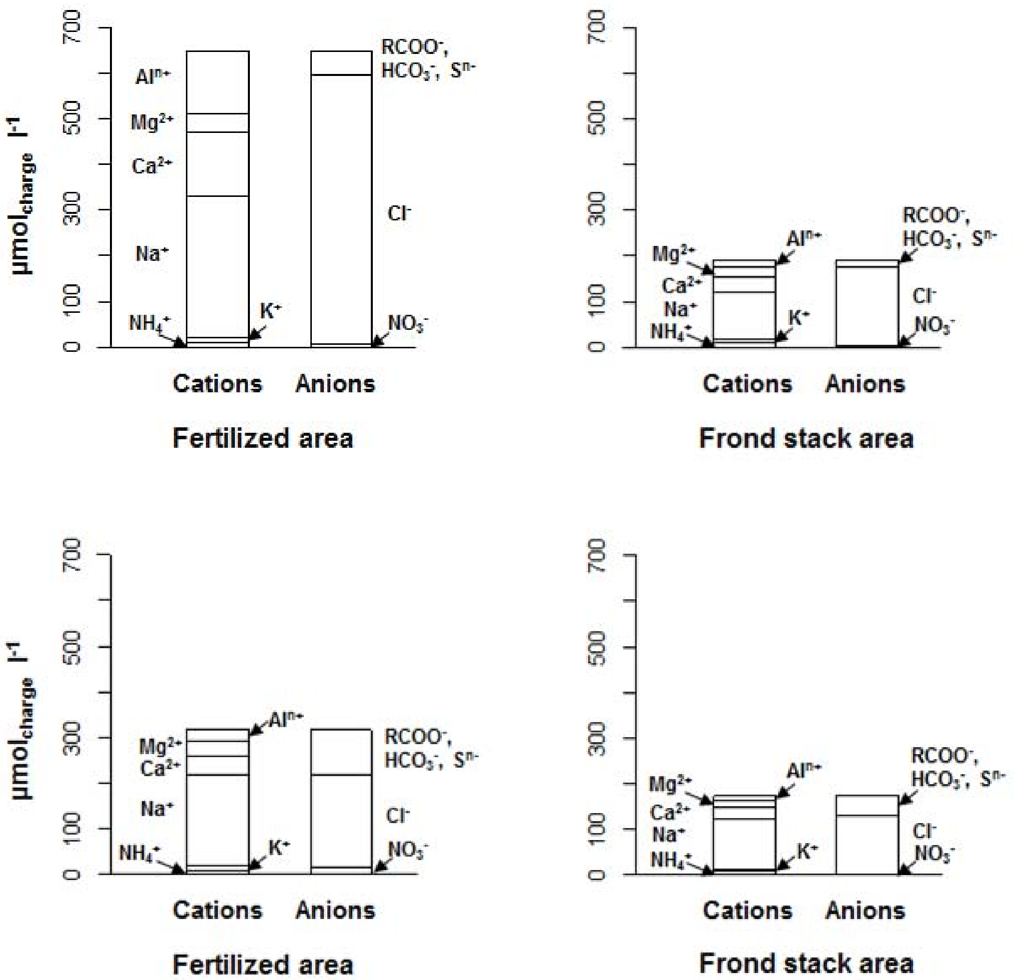

Fig. 3.1. Partial cations-anions charge balance of the major solutes (solutes with concentrations > 0.03 mg l^{-1}) in soil water at a depth of 1.5 m within the fertilized area and under the frond-stacked area in smallholder oil palm plantations in the loam (top) and clay (bottom) Acrisol soils, Jambi, Sumatra, Indonesia.

3.4 Discussion

3.4.1 Nutrient leaching losses between frond stack area and fertilized area

In the loam Acrisol soil landscape, total N in the top 0.1 m soil of our fertilized area (Table 3.1; ~1.55 ± 0.15 g N kg^{-1}) was lower than total N reported for a smallholder oil palm plantation (2.1 g N kg^{-1}; 1-17 years old) in Sarawak, Malaysia on a more fertile Typic Dystrudept soil (USDA classification; Tanaka et al., 2009), but comparable to total N (top 0.15 m) measured in industrial oil palm plantations on comparable soils in the neighbouring province of Riau (2.6 ± 1.5 g N kg^{-1}; Comte et al., 2013). Also exchangeable Ca (~3.07 ± 1.16 $cmol_{charge}$ kg^{-1}) was comparable to the values reported by Tanaka et al. (2009) for a smallholder oil palm plantation (1.07 $cmol_{charge}$ kg^{-1}) and oil palm estate (1.57 $cmol_{charge}$ kg^{-1}) in Sarawak, Malaysia. Compared to leaching fluxes measured at 0.6 m depth in the fertilized area of a 22 year old oil palm plantation in Nigeria (Acrisol soils with sand to sandy clay texture and 1342 mm rainfall), leaching fluxes in the fertilized area were comparable for NH_4^+-N and NO_3^--N (3.6 and 2.5 kg N ha^{-1}), whereas they were higher for Ca and Mg and total S (~46.4 ± 13.4 kg Ca ha^{-1} $year^{-1}$, 8.8 ± 2.1 kg Mg ha^{-1} $year^{-1}$, 2.40 ± 0.20 kg S ha^{-1} $year^{-1}$; Omoti et al. 1983). However, Omoti et al. (1983) measured only for six months during the rainy season and did not use a soil water model to calculate leaching fluxes; instead they assumed that the amount that they sampled with their tension plate lysimeters was representative for the amount leached, which makes comparison of our values with this study problematic. Elevated NH_4^+ leaching losses (measured at 1.2 m depth) were also observed from a 26-year old oil palm plantation on Acrisol soil in Malaysia with NPK fertilizer treatment compared to the control area (Tung et al., 2009). However, this study had application rates that were two to three times higher as compared to the rates applied by the smallholders in our study. Since no previous data have been reported on DOC leaching losses from the subsoil of oil palm plantations, we can only compare our values with measurements from a tropical wet forest in Costa Rica (5.1 ± 1.2 g m^{-2} $year^{-1}$; Schwendenmann and Veldkamp, 2005) which were comparable to our fluxes.

The higher annual leaching fluxes of NH_4^+-N and tendentially of NO_3^--N from the fertilized area (Table 3.3) in the loam Acrisol landscape were probably the result of rapid dissolution of chemical fertilizers, leading to a period with elevated NH_4^+ and NO_3^- concentrations in the soil, during which the availability of mineral N surpassed the N uptake by the oil palm trees. This, together with the precipitation surplus and a decrease in NH_4^+ and NO_3^- -immobilization due to fertilization (Keuter et al., 2013; Allen et al., 2015) probably lead to the elevated NH_4^+ and NO_3^- leaching losses. The generally low ECEC especially in the loam Acrisol soils may have exacerbated this effect for NH_4^+ leaching. In contrast, the NH_4^+ released during mineralization of pruned fronds did not result in elevated concentration peaks of dissolved NH_4^+ probably due to quick uptake by palm roots and consequently NH_4^+ leaching losses were low. The lower soil pH in the frond stack area may also have contributed to slow mineralization of pruned frond resulting in slow release of NH_4^+-N.

The higher soil pH, exchangeable Ca and base saturation (Table 3.1) together with the higher concentrations of Ca, Mg, total Al, total Mn, Na, and Cl in soil solution (Table 3.2), and annual leaching fluxes of Ca, Mg, total Al, total Mn, Na, total S and Cl (Table 3.3) of the fertilized compared to the frond-stacked areas within the loam Acrisol landscape show that management practices resulted in large spatial variability in soil fertility and nutrient leaching. Especially the application of dolomite ($Ca\,Mg\,(CO_3)_2$) in the fertilized area increased Ca^{2+} and Mg^{2+} concentrations in soil water and produced hydroxyl ions, resulting in higher soil pH. Furthermore, the higher Ca^{2+} and Mg^{2+} concentrations probably replaced exchangeable Al, which is high in the loam Acrisol soil (Al saturation between 67-80%; Hassler et al., 2015), and exchangeable Mn causing the elevated annual leaching fluxes of total Al and total Mn

(Table 3.3) in the fertilized area compared to the frond-stacked area. Application of mineral N fertilizer may have exacerbated the Al solubility and lowered the pH in soil solution in the fertilized area within the loam Acrisol (see section 3.2.1; Table S3.1) since N fertilizer can cause soil acidification if NO_3^- is being leaching. Application of other fertilizers such as borate, KCl, and sulfur-containing fertilizer (i.e. kieserite; section 3.2.1) explains the higher Na, total S and Cl leaching losses in the fertilized compared to the frond-stacked areas (Table 3.2 and 3.3).

The higher annual leaching fluxes of DOC that we detected in the fertilized area compared to the frond-stacked area of the loam Acrisol soils may be related to lower DOC retention and/or higher DOC production. Positive correlations of DOC leaching fluxes with soil pH across sampling areas in the loam Acrisol landscape (section 3.3.1) suggests that either an increase in soil pH may have caused higher root activity or higher microbial activity which both can result in higher DOC production, or an increase in pH may have resulted in a shift from a more positive charge in favor of a more negative charge in these soils which are dominated by variable charge. Since in heavily weathered soils negatively charged organic acids are an important part of DOC (Hedin et al., 2003), this would reduce the DOC retention capacity and potentially result in higher annual leaching fluxes of DOC.

For the clay Acrisol landscape the differences in soil characteristics between fertilized and frond-stacked areas were very similar to the effects for exchangeable Ca, extractable P and base saturation that we discussed for the loam Acrisol landscape (Table 3.1). Fertilizer applications probably decrease exchangeable Al and increase ECEC in the fertilized compared to the frond-stacked areas (Table 3.1). However, even though the fertilized area received inorganic fertilizers, we only detected significant differences in annual leaching fluxes of K between the fertilized and frond-stacked areas (Table 3.3). The tendency towards higher soil exchangeable K (Table 3.1), the correlation of annual leaching fluxes of K and base saturation, and the correlation among base cations in soil solution from each sampling areas in the clay Acrisol (Table S3.1) suggest that higher annual leaching fluxes of K in the fertilized areas were most likely the result of dissolved K fertilizer (i.e. NPK, KCl) and the replacement of K on the negative exchange sites with other cations (i.e. Ca^{2+}, Mg^{2+}). The absence of significant differences in annual leaching fluxes of other elements was probably caused by the higher clay content which can retain larger amounts of nutrients against leaching losses (see also below).

3.4.2 Nutrient leaching losses between loam and clay Acrisol soil

We attribute the higher soil nutrient stocks (i.e. soil organic C, total N, K, Na, total Al, extractable P) and ECEC (Table 3.1), and lower DON leaching losses in clay Acrisol soil compared to the loam Acrisol soil (Tables 3.2 and 3.3) to the clay content in these heavily weathered soils which supports our second hypothesis. Higher nutrient stocks are often related to higher clay content, since clay is known increase organic matter contents of soils (van Straaten et al., 2015) while a higher clay content also increases the cation exchange capacity. Furthermore, an ancillary study conducted in the same smallholder oil palm plantations showed that the clay Acrisol had higher rates of NH_4^+ immobilization. The lower annual DON leaching fluxes in the clay Acrisol soil compared to the loam Acrisol soil suggests higher sorption of organic N caused by higher clay content (see section 3.2.1) as well as higher sesquioxide content and exchangeable Al (i.e. frond stack; Table 3.1) in the clay soil compared to the loam Acrisol soil. Since clay and sesquioxide particles combine large surface areas with charge characteristics, a higher clay content can lead to increased sorption of organic components that have a negative charge (i.e. DON) resulting in lower DON concentrations and annual leaching fluxes. Such a mechanism was also reported in a lowland forest in Costa Rica with Ferralsol soil where low DON leaching losses were explained by high sorption capacity (Schwendenmann and Veldkamp, 2005).

The lower concentrations of Mg, total Al, and total Fe in soil solution and total ionic strength from the fertilized area in the clay Acrisol soil compared to the loam Acrisol soil were probably related to the low anion (especially Cl^-) concentration in soil solution (Table 3.2; Table S3.1) since positively charged cations can only be leached if they are accompanied by negatively charged anions (Fig. 3.1). We attributed the elevated Cl^- anion concentrations of the fertilized area compared to the frond-stacked area to the type and amount of fertilizer applied, and suggest that this is also the most likely explanation for the higher Cl^- concentrations of the fertilized areas in the loam Acrisol soil compared to the clay Acrisol soil. Cl^- is considered a biologically inert anion (since it is not a nutrient) and has e.g. been used to compare evapotranspiration between sites that have similar Cl^- input (Grimaldi et al., 2009). Since the total ionic strengths of the frond-stacked area (where no fertilizer was applied) in both soil were comparable, the water balance is comparable between the sites and differences in Cl^- input (i.e. fertilizer application) are thus the most likely explanation of the large observed differences.

Finally, the higher total Mn concentrations and annual leaching fluxes (Table 3.2 and 3.3) in the frond-stacked area of the clay soil and the higher exchangeable Mn values of that soil were probably related to occasional water stagnation on the B horizons of the clay Acrisol. We observed stagnic properties in some of the clay Acrisol soils. Mn in reduced form is dissolvable and can thus be adsorbed to the cation exchange complex, but it can also be leached more readily.

3.4.3 Consequences for nutrient management in oil palm plantation

Our present study showed that the higher leaching losses in the fertilized area compared to the frond-stacked area especially in the loam Acrisol landscape were mainly caused by the application of fertilizer and dolomite, which was absent from the frond-stacked area. The current practices of smallholders to apply fertilizer in concentrated form around oil palm trees causes a temporary increase in nutrient concentrations that surpass the nutrient demand of the oil palm trees. As long as these elevated nutrient concentrations occur, this can result in high nutrient leaching which potentially affects ground water quality and reduces the nutrient use efficiency of oil palm plantations. Management practices directed at reducing the period with elevated nutrient concentrations (e.g. more frequent fertilizer applications at lower doses) would probably reduce nutrient losses through leaching. The higher soil nutrient stocks and lower nutrient leaching losses in the clay Acrisol soil compared to the loam Acrisol soil both in the fertilized and frond-stacked areas were caused by the higher nutrient retention as a result of increased clay content. Heavily weathered soil with high clay content are thus less susceptible to nutrient leaching losses than heavily weathered soils with low clay content and may thus be preferable locations for the establishment of oil palm plantations. Earlier work on the same sites illustrated the importance of NH_4^+ and NO_3^- immobilization processes for nitrogen retention in these soils (Allen et al., 2015). If this finding can be extrapolated to other nutrients, this would suggest that stacking pruned fronds (with high C / nutrient ratios) near the fertilized area around the oil palm trees may increase soil microbial biomass and consequently nutrient immobilization, which may help to decrease nutrient leaching losses, especially in the fertilized areas in the loam Acrisol soil landscape. Since fine root biomass decreases with increasing distance from the oil palm tree, recycling pruned fronds near the fertilized area would probably also increase nutrient uptake from mineralization and further reduce leaching losses.

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Supplementary information

Table S3.1. Pearson correlations among element concentrations (mg l^{-1}) in soil solution (1.5-m depth) from smallholder oil palm plantation under fertilized and frond-stacked areas in the loam and the clay Acrisol soil landscape, Jambi, Sumatra, Indonesia. Correlations were carried out using monthly averages of four replicate plots for the loam Acrisol and average of three replicates plots for the clay Acrisol per sampling area (n = 12).

Element	NH_4^+-N	NO_3^--N	DOC	Na	K	Ca	Mg	Total Al	Total Fe	Total Mn	Total P	Total S	Cl	pH
							Fertilized area in the loam Acrisol							
DON	-0.28	0.08	-0.18	-0.38	-0.12	0.16	0.31	0.50	0.11	0.35	-0.27	-0.06	0.08	-0.17
NH_4^+-N		0.54†	-0.12	0.0	0.50	0.15	0.37	0.46	0.11	0.64*	-0.16	0.22	0.46	-0.21
NO_3^--N			-0.12	0.14	-0.02	-0.49	0.00	0.63*	0.53†	0.81	-0.05	-0.38	0.10	-0.57*
DOC				-0.22	0.08	0.02	0.29	-0.17	0.04	-0.19	0.58*	0.40	-0.47	0.26
Na					-0.12	-0.45	-0.45	-0.37	0.04	-0.10	-0.01	-0.38	0.22	0.01
K						0.58*	0.43	-0.17	-0.28	0.14	-0.25	0.58*	0.27	0.36
Ca							0.48	-0.19	-0.29	-0.08	-0.33	0.79**	0.45	0.55*
Mg								0.40	-0.04	0.31	-0.30	0.72**	0.41	-0.06
Total Al									0.46	0.78**	-0.04	-0.16	0.27	-0.76**
Total Fe										0.33	0.05	-0.31	0.16	-0.56*
Total Mn											-0.21	-0.07	0.42	-0.40
Total P												-0.15	-0.55†	-0.12
Total S													0.30	0.39
Cl^-														-0.10

Frond stack area in the loam Acrisol														
DON	-0.38	0.38	0.22	-0.38	0.24	-0.47	-0.16	0.47	-0.02	0.38	0.21	-0.43	0.53*	-0.42
NH_4^+-N		0.07	0.23	0.40	0.25	0.04	0.08	-0.17	-0.17	-0.12	-0.08	0.42	0.06	0.24
NO_3^--N			0.61*	0.12	0.56*	-0.26	-0.21	0.11	-0.08	0.04	0.17	0.20	0.02	0.18
DOC				-0.1	0.57*	-0.38	-0.54†	-0.28	-0.23	-0.31	0.79**	0.22	-0.42	0.29
Na					0.09	0.23	0.22	-0.35	0.16	-0.30	-0.47	0.61*	0.09	0.66*
K						-0.27	-0.21	-0.07	-0.26	-0.12	0.14	0.29	-0.06	0.20
Ca							0.83**	0.30	0.70**	0.44	-0.34	-0.15	0.72**	0.03
Mg								0.63*	0.72**	0.75**	-0.52†	-0.41	0.95**	-0.16
Total Al									0.51†	0.96**	-0.22	-0.81**	0.79**	-0.65*
Total Fe										0.59*	-0.13	-0.46	0.67**	0.04
Total Mn											-0.23	-0.80**	0.87**	-0.59*
Total P												-0.12	-0.49†	0.10
Total S													-0.48	0.66*
Cl														-0.25
Fertilized area in the clay Acrisol														
DON	0.02	-0.09	0.49	0.70*	0.69*	0.67*	0.42	0.45	0.25	0.38	0.57*	0.54†	0.63*	0.06
NH_4^+-N		0.08	0.15	0.39	0.37	0.16	0.06	0.06	0.54†	0.06	-0.07	0.46	-0.01	0.26
NO_3^--N			-0.18	0.03	0.46	0.51†	-0.01	0.19	-0.01	0.25	-0.23	0.33	-0.49	-0.43
DOC				0.52†	0.66*	0.56†	0.5	0.56†	0.49	0.51†	0.06	0.25	0.70*	0.03
Na					0.61*	0.61*	0.29	0.21	0.19	0.15	0.24	0.75**	0.55†	0.12
K						0.85**	0.74**	0.78**	0.54†	0.43	0.16	0.52†	0.59*	-0.31
Ca							0.81**	0.74**	0.33	0.66*	0.01	0.69*	0.64*	-0.25
Mg								0.95**	0.49	0.73**	-0.01	0.26	0.74**	-0.54†
Total Al									0.64*	0.80**	0.03	0.15	0.75**	-0.51†

Total Fe										0.43	-0.09	0.16	0.59*	0.05
Total Mn											-0.09	0.12	0.42	-0.59*
Total P												-0.11	0.17	0.10
Total S													0.26	0.25
Cl														-0.13
Frond stack area in the loam Acrisol														
DON	0.19	0.34	0.15	0.49†	0.47	0.51†	0.23	0.29	-0.52†	0.48	-0.06	0.28	0.36	-0.12
NH_4^+-N		-0.07	0.27	0.21	0.38	0.11	0.06	0.07	-0.16	0.06	0.02	0.13	0.09	0.12
NO_3^--N			0.34	0.24	0.32	0.13	-0.13	0.09	-0.36	0.03	0.80**	0.56*	-0.05	-0.07
DOC				0.09	0.23	0.25	0.45	0.02	-0.09	0.25	-0.14	-0.46	0.19	0.18
Na					0.91**	0.94**	0.76**	0.91**	0.11	0.85**	-0.04	0.33	0.89**	-0.29
K						0.88**	0.74**	0.80**	0.02	0.81**	0.17	0.21	0.79**	-0.21
Ca							0.90**	0.91**	0.10	0.95**	-0.12	0.10	0.95**	-0.35
Mg								0.81**	0.27	0.04	-0.22	-0.28	0.93**	-0.38
Total Al									0.24	0.91**	-0.23	0.16	0.92**	-0.38
Total Fe										0.04	-0.18	-0.02	0.25	-0.06
Total Mn											-0.27	-0.09	0.94**	-0.44
Total P												0.31	-0.27	0.03
Total S													-0.06	0.18
Cl														-0.47

*, ** - significant at $p \leq 0.05$ and $p \leq 0.01$, respectively, † show marginal significant at $p \leq 0.09$; element that did not show correlation with other elements (total Si) were excluded.

Chapter 4

Synthesis

4.1. Key findings

Chapter 2. In the reference land uses, the higher clay content in the clay Acrisol soil exhibited higher soil nutrient stocks (i.e. SOC, total N, extractable P and exchangeable bases) and retention efficiencies of N and base cations, and lower nutrient leaching losses (e.g. N, Na, Ca, Mg and total Al) than in the loam Acrisol soil. In the converted land uses, management practices (i.e. fertilization, liming) mainly controlled nutrient leaching losses (e.g. N, DOC, base cation) and retention efficiencies of N and base cations.

Chapter 3. The pulse rates of applications of mineral fertilizers and lime around the oil palm tree elevated nutrient concentrations in soil solution, resulting in higher leaching losses (e.g. N, base cations, total Al, total Mn, total S and Cl) compared to the area (inter-rows) stacked with fronds. At the landscape scale, the higher soil nutrient stocks and lower nutrient leaching losses (e.g. N, base cation, total Al and total Fe) in the clay Acrisol landscape compared to the loam Acrisol landscape in both fertilized and frond-stacked areas were caused by the higher nutrient retention as a result of higher clay content.

4.2. Implications

Nutrient leaching is just one process involved in the partial budgets of nutrients in land-use types. Other key processes involved in the partial input-output budgets of nutrients include inputs through deposition from bulk precipitation, fertilization and outputs through harvest export. The magnitudes of these components can provide us with a more holistic view of the major causes of nutrient changes with land-use change. In line with this view, I conducted partial nutrient budgets as synthesis of my studies, incorporating my measured annual leaching fluxes with annual nutrient inputs (i.e. bulk precipitation and fertilization) as well as those results from ancillary studies on nutrient exports through harvest (Kotowska, 2015) and soil nutrient stocks in the top 1-m depth (Allen, 2015). Through these partial nutrient budgets, I am able to quantify the magnitude of changes of nutrients (K, Ca, Mg, and Na) with conversion of forest to rubber and oil palm plantations. This information is important to identify the main pathway of nutrient losses in agricultural systems and the sustainability of converted land uses following years of cultivation.

In the reference land uses, the higher soil nutrient stocks (i.e. extractable P, exchangeable K, Ca and Mg) and base saturation in the top 1-m depth (Table 4.1) of the clay than the loam Acrisol soils were mirrored with the lower nutrient leaching losses (e.g. N, Ca, Mg and Na; Table 2.4; pg. 36). These results suggested a more efficient retention of nutrients (e.g. soil-vegetation cycling) in the reference land uses of the clay than the loam Acrisol soils, which was also supported by higher (all $p \leq 0.05$) annual partial budgets of N, P, and base cation (Table 4.2), higher NPP (Kotowska et al., 2015) and subsequently increased retention efficiency of N and base cations (Chapter 2). Annual leaching fluxes were the main output pathways for N, P and base cations in the reference land uses, except for P in the jungle rubber sites where the major output pathway was the harvest export. Decrease in extractable P stock in the top 1-m depth in jungle rubber compared to forest in the clay Acrisol landscape (Table 4.1) contrasted the positive, albeit low, partial budget of P (Table 4.2). This could be attributed to the fluctuations of harvest export over time, as the present annual partial budget was only based during 2013. Additionally, this result suggests that losses via runoff and/or soil erosion (not measured in this study) could have also contributed to other losses of soil P from this land use.

When forest and jungle rubber are converted to unfertilized rubber plantations, soil nutrient stocks and nutrient leaching losses could decrease over time (Chapter 2). In unfertilized rubber plantations, with continuous harvest export and decrease inputs from litter and root production compared to the reference land uses (Kotowska et al., 2015), there were negative partial budget of nutrients (N, P, K and Mg in either landscapes; Table 4.2). It is noteworthy that the negative partial budget of P in rubber plantations (Table 4.2) was reflected by decrease in P stock in the top 1-m depth (Table 4.1) and lower total P leaching losses in rubber plantations compared to the reference land uses, particularly in the loam Acrisol landscape with older rubber plantations (14-17 years old). Additionally, lower annual NO_3^--N and DOC leaching fluxes in rubber plantations compared to the reference land uses were also detected in the loam Acrisol landscape (Chapter 2), and may be attributed to the low stocks of soil organic C (van Straaten et al., 2015) and soil extractable NO_3^- (Allen et al., 2015). The negative partial budget, low nutrient stocks and leaching losses in unfertilized rubber plantation has implication on the sustainability of yield, as well as on the duration of the existing land use or further conversion to another land use.

The higher soil nutrient stocks (Table 4.1; Allen, 2015) and nutrient leaching in oil palm plantations (Chapter 2) compared to the reference land uses and unfertilized rubber plantations in both soil landscapes confirmed that management practices (i.e. fertilization, liming) controlled soil nutrient levels and leaching losses. An ancillary study conducted in the same sites reported that oil palm plantations had higher nutrient loss through harvest export (Kotowska, 2015) compared to the other land uses. The high leaching losses

(Chapter 2) and harvest export (Kotowska, 2015) resulted in the lowest annual partial budgets (i.e. Ca and Mg) in oil palm plantations than in the other land uses in both landscapes (all $p \leq 0.05$, except $p \leq 0.09$ for Ca in the loam Acrisol) (Table 4.2), as well as the lowest base cation retention efficiency in the loam Acrisol landscape (Table 2.5; pg. 37). The high net annual loss Mg (Table 4.2) contributed to the decrease in Mg stocks in the top 1-m depth in oil palm plantations compared to the other land uses in the loam Acrisol landscape (Table 4.1). Furthermore, the net annual loss of Mg (Table 4.2) contributed to a 73 ± 19% decrease in Mg stocks in the top 1-m depth in oil palm plantations in the loam Acrisol landscape. Thus, addition of Mg through fertilizer (i.e. kieserite) or liming (i.e. dolomite) was indeed needed in this highly weathered soil to slow down degradation of the soil fertility and to maintain the sustainability of yield in smallholder oil palm plantations.

The amount of fertilizer applied also affects leaching losses and partial budget of nutrients in oil palm plantations. For example, the higher N input from fertilizer in oil palm plantations in the loam than clay Acrisol soils (section 2.2.1) resulted in higher leaching losses of N (Table 2.4; pg. 36) and lower N retention efficiency (Table 2.5; pg. 37) than in the other land uses. Additionally, such high N fertilizer application in the loam Acrisol soil also increased acidity of soil solution and concentration of dissolved Al (Table 2.3; pg. 35). The high N fertilization plus the bulk precipitation N input in oil palm plantations in the loam Acrisol landscape were higher than the N output (leaching and harvest export), resulting in a positive partial budget of N (Table 4.2). In contrast, the negative partial budget of N in the oil palm plantations of the clay Acrisol landscape (Table 4.2) was due to lower N fertilization rates, even though N leaching and retention efficiency did not differ among land uses (Table 2.4 and 2.5; pg. 36-37). These findings imply that the sustainability of palm oil yield should take into account the long-term effects of pulse N application on soil acidity, which can result in a dependency on liming input that requires additional capital by smallholders, and its impact on increased nutrient leaching on ground water quality.

Due to the increased nutrient leaching losses with conversion of forest or jungle rubber to fertilized oil palm plantations, it is important to critically observe in more detail the impact of current management practices in smallholder oil palm plantations. Management practices in the smallholder oil palm plantations in our landscapes were characterized by the application of chemical fertilizers around each palm or by stacking pruned fronds on inter-rows of oil palm trees. Pulse rate of fertilization around each oil palm tree increased nutrient concentrations in soil solutions, resulting in higher leaching losses in the fertilized area compared to the area where palm fronds were stacked on inter-rows (Chapter 3). The higher soil nutrient stocks and lower nutrient leaching losses in the clay Acrisol soil compared to the loam Acrisol soil in both fertilized and frond-stacked areas reflected the higher nutrient retention due to the ability of clay to retain more nutrients against the force of gravity. Heavily weathered soil with high clay content are thus less susceptible to nutrient leaching losses than heavily weathered soils with low clay content and may thus be preferable locations for the establishment of oil palm plantations. Also, the amount of fertilizer applied strongly impacted nutrient leaching in the fertilized areas between the loam and clay Acrisol soils. Earlier work on the same sites illustrated the importance of microbial immobilization for N retention in these soils (Allen et al., 2015). If this finding can be extrapolated to other nutrients, this would suggest that stacking pruned fronds (with high C / nutrient ratios) near the fertilized area around the oil palm trees may increase soil microbial biomass and consequently nutrient immobilization, which may help to decrease nutrient leaching losses.

Table 4.1. Soil characteristics[a] in the top 1 m of soil from different land uses in two soil landscapes of Jambi, Sumatra, Indonesia

Characteristics	Forest	Jungle rubber	Rubber plantation	Oil palm plantation
	loam Acrisol soil landscape			
Total nitrogen (kg N ha^{-1})	[b]9642 ± 622	10669 ± $808_{B†}$	11079 ± 1376	7697 ± 586
Extractable phosphorus (kg P ha^{-1})	14 ± 2_{B}	27 ± 12	15 ± 2	21 ± 6
Exchangeable potassium (kg K ha^{-1})	216 ± 3	198 ± $31_{B†}$	226 ± 55	146 ± 36
Exchangeable calcium (kg Ca ha^{-1})	571 ± 165	376 ± $65_{B†}$	634 ± 149	695 ± 246
Exchangeable magnesium (kg Mg ha^{-1})	84 ± 17_{ab}	78 ± $8_{ab\ B†}$	172 ± 69_{a}	42 ± 8 b
Exchangeable sodium (kg Na ha^{-1})	144 ± 69	120 ± 35	216 ± 78	335 ± 67
ECEC ($mmol_c\ kg^{-1}$)	192 ± 20	149 ± 43	209 ± 27	128 ± 36
Base saturation (%, weighted for the top 1 m)	6.4 ± $0.9_{b†\ B}$	8.5 ± $2.0_{ab†}$	9.0 ± $1.5_{ab†}$	13 ± $0.3_{a†}$
	clay Acrisol landscape			
Total nitrogen (kg N ha^{-1})	14018 ± 4578	16940 ± $2682_{A†}$	11658 ± 2944	12889 ± 1564
Extractable phosphorus (kg P ha^{-1})	37 ± $4_{a\ A}$	17 ± 2_{bc}	9 ± 1_{c}	32 ± 12_{ab}
Exchangeable potassium (kg K ha^{-1})	652 ± 412	743 ± $310_{A†}$	201 ± 66	258 ± 64
Exchangeable calcium (kg Ca ha^{-1})	1087 ± 558	1184 ± $344_{A†}$	560 ± 92	1194 ± 337
Exchangeable magnesium (kg Mg ha^{-1})	289 ± 174	569 ± $235_{A†}$	158 ± 51	142 ± 33
Exchangeable sodium (kg Na ha^{-1})	52 ± 7	68 ± 21	40 ± 13	60 ± 8
ECEC ($mmol_c\ kg^{-1}$)	330 ± 181	565 ± 165	279 ± 91	290 ± 57
Base saturation (%, weighted for the top 1 m)	11.1 ± $0.6_{a†\ A}$	8.1 ± $1.7_{ab†}$	7.1 ± $0.6_{b†}$	10.4 ± $1.1_{ab†}$

[a] Allen (2015).

[b] Means (± SE, n = 4, except for oil palm n = 3, the same plot with leaching measurement) followed by different lower case letters indicate significant differences among land uses within each landscape and different upper case letters indicate significant differences between landscapes for each reference land use (Linear mixed effects models with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

Table 4.2. Annual (2013) partial nutrient budgets[a] of different land uses (forest, jungle rubber, rubber and oil palm plantations) in two landscapes (loam and clay Acrisol soils) in Jambi, Sumatra, Indonesia

Element	loam Acrisol soil landscape				clay Acrisol soil landscape			
	Forest	Jungle rubber	Rubber plantation	Oil palm plantation	Forest	Jungle rubber	Rubber plantation	Oil palm plantation
Nitrogen (kg N ha^{-1} year^{-1})								
Input	[b]12.90 ± 0.13	12.90 ± 0.13	12.90 ± 0.13	104.90 ± 45.51	16.44 ± 2.56	16.44 ± 2.56	16.44 ± 2.56	32.44 ± 16.00
Output	5.96 ± 0.81	11.59 ± 4.01	12.71 ± 2.47	93.17 ± 4.01	3.44 ± 0.83	4.24 ± 0.62	16.87 ± 2.48	81.28 ± 11.09
Balance	$6.94 \pm 0.81_{B}$	$1.31 \pm 4.01_{B}$	0.19 ± 2.47	11.73 ± 46.37	$13.00 \pm 0.83_{a\ A}$	$12.21 \pm 0.62_{a\ A}$	$- 0.42 \pm 2.48_{b}$	$- 48.84 \pm 15.82_{c}$
Phosphorus (kg P ha^{-1} year^{-1})								
Input	0.42 ± 0.05	0.42 ± 0.05	0.42 ± 0.05	21.42 ± 0.00	0.79 ± 0.01	0.79 ± 0.01	0.79 ± 0.01	7.79 ± 7.00
Output	0.10 ± 0.03	1.37 ± 0.25	3.13 ± 0.84	7.41 ± 0.24	0.05 ± 0.00	0.47 ± 0.19	3.79 ± 0.96	6.09 ± 0.86
Balance	$0.32 \pm 0.03_{b\ B}$	$- 0.95 \pm 0.25_{b\ B}$	$- 2.71 \pm 0.84_{c}$	$14.01 \pm 0.24_{a}$	$0.74 \pm 0.00_{a\ A}$	$0.31 \pm 0.19_{a\ A}$	$- 3.00 \pm 0.96_{b}$	$1.70 \pm 6.14_{ab}$
Potassium (kg K ha^{-1} year^{-1})								
Input	5.49 ± 1.50	5.49 ± 1.50	5.49 ± 1.50	96.09 ± 51.91	9.60 ± 4.87	9.60 ± 4.87	9.60 ± 4.87	75.27 ± 47.10
Output	4.33 ± 0.57	5.98 ± 2.21	8.10 ± 1.57	69.77 ± 4.08	3.07 ± 0.39	3.92 ± 1.00	8.06 ± 1.44	56.51 ± 7.18
Balance	$1.16 \pm 0.57_{B}$	$- 0.50 \pm 2.21_{B}$	- 2.61 ± 1.57	26.32 ± 48.00	$6.53 \pm 0.39_{a\ A}$	$5.68 \pm 1.00_{a\ A}$	$1.54 \pm 1.44_{b}$	$18.76 \pm 48.57_{ab}$
Calcium (kg Ca ha^{-1} year^{-1})								
Input	10.94 ± 0.78	10.94 ± 0.78	10.94 ± 0.78	25.23 ± 14.29	12.41 ± 2.41	12.41 ± 2.41	12.41 ± 2.41	15.97 ± 3.57
Output	10.00 ± 0.70	11.96 ± 3.40	10.14 ± 1.06	84.50 ± 13.70	6.75 ± 0.55	9.12 ± 0.42	9.38 ± 1.15	41.56 ± 5.61
Balance	$0.94 \pm 0.70_{a\dagger\ B}$	$- 1.02 \pm 3.40_{a\dagger}$	$0.80 \pm 1.06_{a\dagger}$	$- 59.27 \pm 10.96_{b\dagger}$	$5.66 \pm 0.55_{a\ A}$	$3.28 \pm 0.42_{a}$	$3.03 \pm 1.15_{a}$	$- 25.59 \pm 2.88_{b}$
Magnesium (kg Mg ha^{-1} year^{-1})								
Input	2.41 ± 0.47	2.41 ± 0.47	2.41 ± 0.47	9.61 ± 7.20	3.01 ± 0.38	3.01 ± 0.38	3.01 ± 0.38	3.91 ± 0.90
Output	4.14 ± 0.26	4.94 ± 1.31	5.75 ± 1.02	39.88 ± 1.12	2.50 ± 0.19	3.51 ± 0.49	4.91 ± 0.52	31.31 ± 4.78
Balance	$- 1.73 \pm 0.26_{a\ B}$	$- 2.52 \pm 1.31_{a}$	$- 3.33 \pm 1.02_{a}$	$- 30.27 \pm 8.12_{b}$	$0.52 \pm 0.19_{a\ A}$	$- 0.50 \pm 0.49_{a}$	$- 1.90 \pm 0.52_{a}$	$- 27.40 \pm 3.92_{b}$
Sodium (kg Na ha^{-1} year^{-1})								
Input	62.99 ± 1.26	62.99 ± 1.26	62.99 ± 1.26	62.99 ± 1.26	66.14 ± 6.33	66.14 ± 6.33	66.14 ± 6.33	66.14 ± 6.33
Output	37.67 ± 3.58	37.00 ± 7.59	30.98 ± 2.65	130.69 ± 75.82	24.89 ± 3.77	32.28 ± 3.19	25.15 ± 0.78	62.73 ± 17.69
Balance	$25.31 \pm 3.58_{B}$	25.99 ± 7.59	32.01 ± 2.65	- 67.70 ± 75.82	$41.25 \pm 3.77_{A}$	33.86 ± 3.19	40.99 ± 0.78	3.41 ± 17.69

[a] Partial nutrient budget = Input (bulk precipitation + fertilizers) – Output (annual leaching fluxes + harvest export)
Element balance that showed: + = net nutrient gain; - = net nutrient loss
[b] Means (SE, n = 4; except oil palm n = 3; see Chapter 2) followed by different lower case letters indicate significant differences among land uses within each landscape and different upper case letters indicate significant differences between landscapes for each reference land use (one-way analysis of variance with Fisher's LSD test at $p \leq 0.05$, except those indicated with † at $p \leq 0.09$).

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DECLARATION OF ORIGINALITY AND CERTIFICATE OF AUTHORSHIP

I, Syahrul Kurniawan, hereby declare that I am the sole author of this dissertation entitled "Conversion of lowland forests to rubber and oil palm plantations changes nutrient leaching and nutrient retention efficiency in highly weathered soils of Sumatra, Indonesia". All references and data sources that were used in the dissertation have been appropriately acknowledged. I furthermore declare that this work has not been submitted elsewhere in any form as part of another dissertation procedure. I certify that the manuscripts presented in chapter 2 and 3 have been written by me as the first author.

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