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Land-use change affects phosphorus fractions in highly weathered tropical soils

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ABSTRACT

Deforestation and land-use change in tropics have increased over the past decades, driven by the demand for agricultural products. Although phosphorus (P) is one of the main limiting nutrients for agricultural productivity in the tropics, the effect of land-use change on P availability remains unclear. The objective was to assess the impacts of land-use change on soil inorganic and organic P fractions of different availability (Hedley sequential fractionation) and on P stocks in highly weathered tropical soils. We compared the P availability under extensive land-use (rubber agroforest) and intensive land-use with moderate fertilization (rubber monoculture plantations) or high fertilization (oil palm monoculture plantations) in Indonesia. The P stock was dominated by inorganic forms (60 to 85%) in all land-use types. Fertilizer application increased easily-available inorganic P (i.e., H_2O-Pi, $NaHCO_3$-Pi) in intensive rubber and oil palm plantations compared to rubber agroforest. However, the easily-available organic P ($NaHCO_3$-extractable Po) was reduced by half under oil palm and rubber. The decrease of moderately available and non-available P in monoculture plantation means that fertilization maintains only the short-term soil fertility that is not sustainable in the long run due to the depletion of P reserves. The mechanisms of this P reserve depletion are: 1) soil erosion (here assessed by C/P ratio), 2) mineralization of soil organic matter (SOM) and 3) P export with yield products. Easily-available P fractions (i.e., H_2O-Pi, $NaHCO_3$-Pi and Po) and total organic P were strongly positively correlated with carbon content, suggesting that SOM plays a key role in maintaining P availability. Ecologically based management is therefore necessary to mitigate SOM losses and thus increase the sustainability of agricultural production in P-limited, highly weathered tropical soils.

1. Introduction

Land-use change and intensification of cultivation are the predominant global changes of this century. This is mainly because of the global socio-economic demand for food, feed, fiber and biofuel driven by population growth (Geissen et al., 2009; Guillaume et al., 2015). Intensification of agriculture involving high-yielding crop varieties, fertilization, irrigation, and pesticides causes soil degradation. As agriculture land becomes degraded, more forests are cut and converted for the needed agricultural production. This has led to a strong decrease of tropical rainforest area worldwide, especially in Southeast Asian countries (Gatto et al., 2015; Tarigan et al., 2015).

Indonesia is one of the tropical countries with highest deforestation rates, surpassing the rate in Brazil in 2012 (Hansen et al., 2009; Margono et al., 2014). Sumatra (Indonesia) lost more than half of its remaining natural rainforest between 1985 and 2007 due to deforestation and land-use intensification (Laumonier et al., 2010; Wilcove and Koh, 2010). Deforestation and agricultural intensification on the island is ongoing. Natural rainforests are converted to extensively managed agroforest (jungle rubber), then to intensively managed monoculture plantations (i.e., oil palm, rubber). These conversions are among the main drivers of deforestation aside from mining, timber and pulp industries (Guillaume et al., 2015; Villamor et al., 2014; Violita et al., 2015). However, extensive transformation of natural ecosystems to plantation leads to the decreased of soil fertility indicators and to subsequent soil degradation in Sumatra (Guillaume et al., 2016a, 2016b).

Land-use change significantly modifies the physical, chemical and biological soil properties, affects soil fertility, and increases erosion and compaction (Geissen et al., 2009; Matson et al., 1997; Moges et al., 2013). Phosphorus (P) is a key nutrient requiring attention in response to land-use change (Garcia-Montiel et al., 2000): it is the most limiting nutrient for plant productivity, especially in tropical regions (Dieter et al., 2010; Holford, 1997; Sanchez, 1976; Spohn et al., 2013;

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Vitousek, 1984). The highly weathered acidic soils and large quantities of sesquioxides adsorb and chemically fix P, leading to P limitations in tropical ecosystems (Bucher et al., 2001; Holford, 1997). Soil available P is mainly supplied by parent material, is recycled by decomposition of organic matter, and added by fertilizer inputs that enrich different P forms (i.e., available, moderately available, non-available inorganic and organic). When available P is depleted, replenishment from other P forms becomes important (Henriquez, 2002).

Land-use changes affect P availability for plant uptake either by increasing P losses or by transforming it to more recalcitrant pools. This leads to potentially significant effects on the distribution of P within chemically-defined pools, in turn determining availability and stability (Wright, 2009). Some studies on the partitioning of total soil P revealed effects of land-use change (Cassanova et al., 2002; Solomon et al., 2002). The fires – forest burning during plantation establishment – also impact soil P. They release P into the available pool, where it can be taken up by microorganisms, sorbed on the mineral matrix, leached or removed by runoff (Sanchez, 1976). At medium to high fire intensities (>300 °C), however, P mobilization is restricted and fixation increases. This is due to a heat-induced increase in mineral surface area, the production of Fe oxides free of organic matter and high affinity for P sorption (Ketterings et al., 2002). Short-term P fertilization is enhanced by ash of forest fires coupled with root decomposition of the original vegetation (Groppo et al., 2015). Nonetheless, fertility is not sustainable. Nutrient depletion occurs as plantations age (Numata et al., 2007; Townsend et al., 2002), reflecting nutrient removal with yield products. Furthermore, tremendous changes in plant biomass production and nutrient cycling due to vegetation conversions have a great negative or positive influence on soil properties and nutrient availability (Chen et al., 2003). The conversion of P from available to non-available (e.g., Al-P, Fe-P) and organic forms occurs in <50 years after land-use change, much faster than the thousands of years required under natural conditions (Garcia-Montiel et al., 2000). The conversion of forest to cropland decreased the P amount and increased the proportion of non-available P forms (Chacon and Dezzeo, 2004). Accelerated soil erosion due to land-use change reduced organic matter by half or more (Zheng et al., 2005), which is a source of organic substrate for nutrient release such as available P (Groppo et al., 2015; Pimentel et al., 1995).

Various approaches have been developed to study the forms, amount and dynamics of P cycling (Bowman and Cole, 1978; Chang and Jackson, 1957; Hedley et al., 1982; Tiessen and Moir, 1993). The sequential chemical fractionation developed by Hedley et al. (1982) has been widely used in recent decades to study soil P fractions and thus soil P dynamics (Chimdi et al., 2014). The chemical fractionation method evaluates the location and bonding type of P within the soil matrix (Guo et al., 2000; Yang and Post, 2011), and investigates the effects of land-use change on the distribution of P fractions. Hedley fractionation assumes that extractants of varying strength estimate inorganic phosphorus (Pi) and organic phosphorus (Po) fractions of different availability and chemical bindings (Guo et al., 2000; Hedley et al., 1982). The following fractions respond to extractants and are available: (i) H_2O-Pi and $NaHCO_3$-Pi, which are considered the most biologically and readily available Pi form. (ii) $NaHCO_3$-Po, which is easily mineralizable and may contribute to the plant-available Pi. (iii) NaOH-P, which is associated with P and is strongly adsorbed via a covalent bond between phosphate oxygen and the aluminum (Al) and iron (Fe) in clays, which are involved in long-term P transformations. (iv) HCl-Pi, which is relatively insoluble P, often associated with Ca-P. HCl-Po has not been measured in most sequential P fractionation studies. It is reported that this fraction is Ca-bound hydrolysable Po.

Most studies in Sumatra (Indonesia) on the effects of land-use change and deforestation deal with soil carbon contents and stocks. This reflects the importance of low-carbon agriculture, climate change and general soil fertility issues. Nonetheless, only few studies focus on the effect of land-use change on soil P availability; no studies are available on P fractionation of various forms of inorganic and organic P. Our study is designed to assess the effects of land-use change on inorganic and organic P forms of different availability and on the P stocks in highly weathered tropical soils. We hypothesized that inorganic and organic P fractions of different availability will strongly decrease after land-use change. Likewise, P stocks – the total of all P fractions – will also decrease.

2. Materials and methods

2.1. Study area and soil sampling

The study was carried out in Jambi Province in Sumatra, Indonesia. The climate is tropical humid with an average temperature of 27 °C and an average precipitation of 2200 mm $year^{-1}$ and 112–259 mm $month^{-1}$ (Guillaume et al., 2015). Aside from tropical rainforest, the area had three dominating land-use types: (1) jungle rubber, (2) rubber plantation, and (3) oil palm plantation. Jungle rubber is an extensively-managed agroforest (minimum age of 16 years) in which rubber trees are planted in a partially logged forest. Tree species namely: *Alstonia* spp., *Artocarpus* spp., *Fabaceae* sp., *Macaranga* spp., *Porterandia* sp., and *Hevea* sp. are the most common tree species in the agroforest system. On the other hand, rubber (*Hevea brasiliensis*) and oil palm (*Elaeis guineensis*) plantations were intensively managed monocultures of similar average age (14 yrs), ranging from 12 to 17 years (Guillaume et al., 2016a). Rubber and oil palm plantation received high NPK fertilization at a rate of 100–300 kg ha^{-1} $year^{-1}$ and 300–600 kg ha^{-1} $year^{-1}$, respectively. Fertilization happens twice a year once in the rainy season (October to March) and once in dry season (April to September). Herbicides were also applied in both plantations every 6 months (Kotowska et al., 2015).

To assess the effects of land-use change on P fractions, three replicate sites for each land-use type were selected within a distance of 16 km with an elevation varied between 50 and 100 m a.s.l. The soil was Acrisols with sandy loam texture. It is a highly weathered soil with strongly acidic soil pH ranged between 3.9 and 5.1. The base saturation ranged between 16 and 28% and effective CEC ranged between 40 and 46 $mmol_c$ kg^{-1} (Allen et al., 2015). At each site, samples were collected in one pit by horizons down a maximum depth of 1 m. Soils were air-dried and sieved at 2 mm. Plant debris and stones were removed. Soils were brought to the laboratory of the Department of Soil Science in Temperate Ecosystem in Göttingen University, Germany, for further analysis. A detailed description of the study area and soil sampling are available in Guillaume et al. (2015). Further information on land-use history, management and soil characteristics can be found in Allen et al. (2015) and Kotowska et al. (2015).

2.2. Soil incubation and preparation

Five grams of air-dried soil was placed in a glass bottle and incubated at field capacity at 24 ± 2 °C for 14 days prior to the sequential extraction in order to reach equilibrium after sampling, drying and sieving disturbances (Hedley et al., 1982). After the incubation, soils were stored at 4 °C and equilibrated at room temperature overnight prior to P sequential fractionation analysis.

2.3. Phosphorus sequential fractionation

The Hedley et al. (1982) sequential fractionation method as modified by Tiessen and Moir (1993) was used to fractionate soil P. This method uses a sequence of increasingly strong extractants that removed labile inorganic phosphorus (Pi) and organic phosphorus (Po) forms first, then stable P forms (Fig. A.1).

One gram of soil was placed into a 50 ml screw cap centrifuge plastic tube and sequentially extracted with the following extractants in

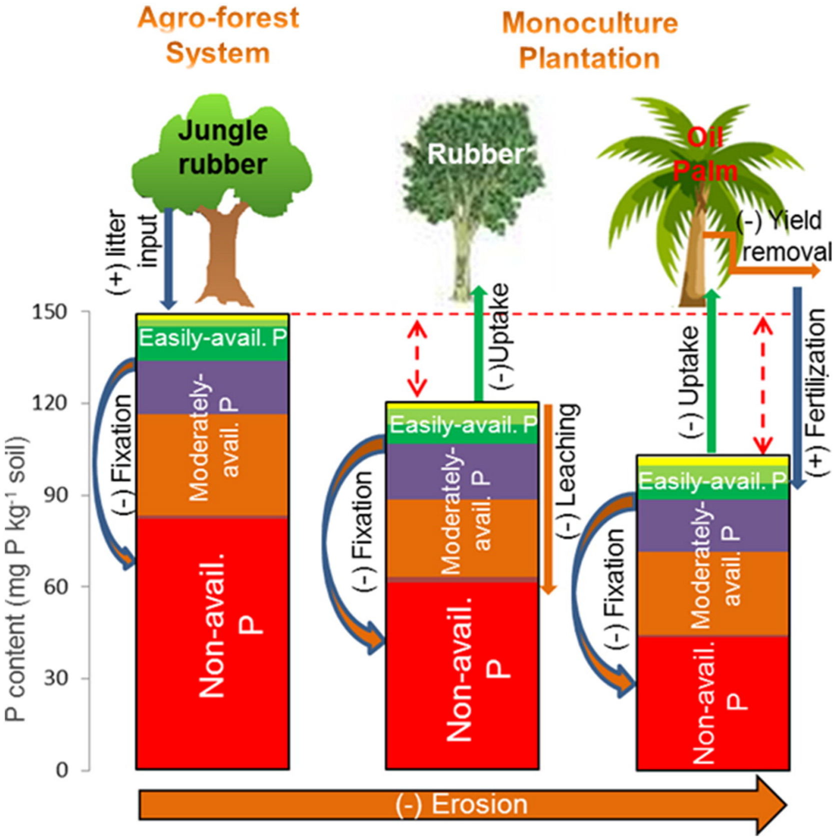

Fig. 1. Effects of land-use change on soil P. (+) Increase P availability; (−) decrease P content and availability for plant uptake. Colors: yellow = H_2O-Pi; yellow-green = $NaHCO_3$-Pi; green = $NaHCO_3$-Po; violet = NaOH-Pi; light brown = NaOH-Po; dark brown = HCl-Pi; red = residual P. Red dashed line (——) shows P losses after land-use change. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

sequential order: (i) 30 ml deionized water, which extracts fairly labile Pi (mobile P) that is directly exchangeable with the soil solution, (ii) 30 ml 0.5 M $NaHCO_3$ at pH 8.5, which extracts relatively labile Pi and Po sorbed onto soil surfaces, plus a small amount of microbial P, (iii) 30 ml 0.1 M NaOH, which extracts amorphous and some crystalline Fe and Al phosphates, as well as P strongly bound by chemisorption to Fe and Al compounds; ultrasonification of soil for 2 min at 75 W in 0.1 M NaOH enabled extraction of P held at the internal surfaces of soil aggregates, (iv) 30 ml 1 M HCl, which extracts relatively insoluble Ca-P minerals including apatite, Al-P and Fe-P in more weathered soils. After every addition of extractant, samples were shaken continuously for 16 h using an end-to-end shaker, and the soil suspension was centrifuged at 3500 rpm for 15 min. Supernatant was filtered using Whatman no. 42 filter paper into small vials and stored at 4 °C for phosphate determination. Finally, soil residues were digested using concentrated H_2SO_4 and 30% H_2O_2 to extract more chemically stable Po forms and relatively insoluble Pi forms (Residual-P). We omitted the addition of an anion exchange membrane and chloroform from the original methodology.

2.4. Phosphate determination

Total P (TP) and Pi were measured directly from the extracts, while Po was calculated as the difference between the TP and Pi. TP was determined in 5 ml aliquots of each extract after ammonium-persulfate and H_2SO_4 digestion to oxidize dissolved Po to Pi forms, and TP was measured as soluble reactive P (Environmental Protection Agency, 1997). Another 5 ml aliquot of each extract was acidified using 0.9 M H_2SO_4 to precipitate organic matter and then used to measure Pi. The pH of final extracts for both TP and Pi was adjusted to pH 3 using dinitrophenol (2,4-DNP) as indicator. If yellow coloration formed after 2,4-DNP addition, diluted HCl was added drop by drop until the extracts turned colorless. Otherwise, if the extract remained colorless after 2,4-DNP addition, NaOH was added first until the colors changed to yellow and then dilute HCl until it became colorless again. Standards with increasing P concentrations were prepared. Phosphate concentrations for both standards and soil extracts were determined by molybdate colorimetry (Murphy and Riley, 1962) at 880 nm using a calibrated spectrophotometer (Specord 40). Po in the deionized water fractions and in the HCl-extractable fraction was not determined because preliminary investigations had shown values below the detection limit.

2.5. Bulk density and carbon content determination

Bulk density has been measured by another research group. It was done by core method using cylinders measuring 250 cm^3. Cylinders were inserted horizontally at 5, 20, 40 and 75 cm depth from the side of the pit. Samples were air-dried and weighed. Carbon content was measured using an Elemental Analyser (Eurovector) after weighing 5 to 40 mg of grinded soil in tin capsules.

2.6. Data analysis

P fractions are expressed as the mean of the three field replicates and are presented in mg P kg^{-1} of the fine earth (<2 mm) fraction. All results are expressed based on oven-dried soil (105 °C, 24 h). In order to facilitate interpretations, P fractions were classified into three main groups (Tiessen et al., 1984): easily-available P, moderately-available P and non-available P. Easily-available Pi is the sum of H_2O-extractable and $NaHCO_3$-extractable phosphate. Moderately-available P is the sum of P extracted from 0.5 M NaOH before and after sonification. Likewise, non-available P is the sum of P extracted by 1 M HCl and concentrated H_2SO_4 and 30% H_2O_2 during soil digestion. P stocks expressed in kg P ha^{-1} were calculated for the soil depth intervals 0–20 and 0–60 cm using the following equation:

$$S = x.\rho.z \tag{1}$$

where S is the P stock for fixed depths and (x) is the soil P content at the designated depth (z), and ρ is the soil bulk density.

Differences in the P fractions, total P, P stocks and C/P ratio between agroforest and monoculture plantations were tested by one-way analysis of variance (ANOVA). Correlations between soil carbon content and P fractions were assessed by calculating linear regression. All analysis was performed using Statistix v8.1 statistical software.

3. Results

3.1. Concentrations of sequentially extracted phosphorus fractions

We recorded clear effects of land-use change of jungle rubber to oil palm and rubber plantations on inorganic and organic P fractions (Fig. 1). The content of easily-available, moderately-available and non-available Pi and Po varied markedly among land-use types (Fig. 2). The change of land-use strongly decreased the content of P fractions in the topsoil (Ah horizon; 0–10 cm).

Easily-available Pi, the sum of H_2O-extractable and $NaHCO_3$-Pi, was the only P fraction that increased after land-use change ($P < 0.05$), indicating the effect of NPK fertilization applied twice a year. The oil palm plantation, which received 2–3 times more NPK fertilizer than rubber, had more easily-available Pi (8.5 ± 1.2 mg P kg^{-1}) in the top horizon than the rubber plantation (7.1 ± 0.7 mg P kg^{-1}). The P increase due to fertilization was more notable in available Pi (labile-Pi) than in other P fractions. In contrast, jungle rubber, which did not receive any fertilization, had only 5.0 ± 0.9 mg P kg^{-1}. However, easily-available Po ($NaHCO_3$-extractable Po), which is the most important P reserve,

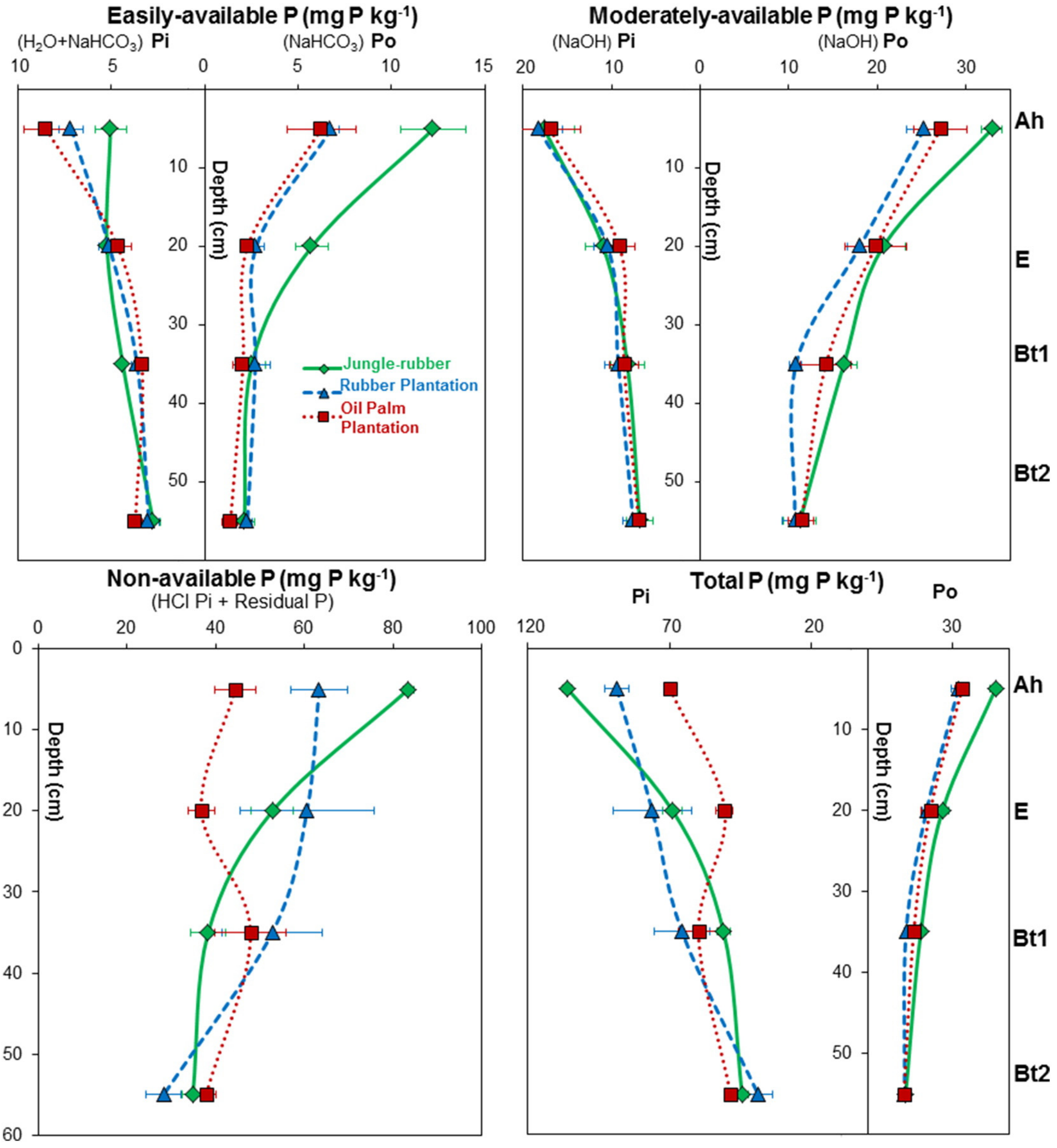

Fig. 2. Inorganic and organic soil phosphorus fractions (mg P kg^{-1}) depending on land use. Values represent means ± SE (n = 3).

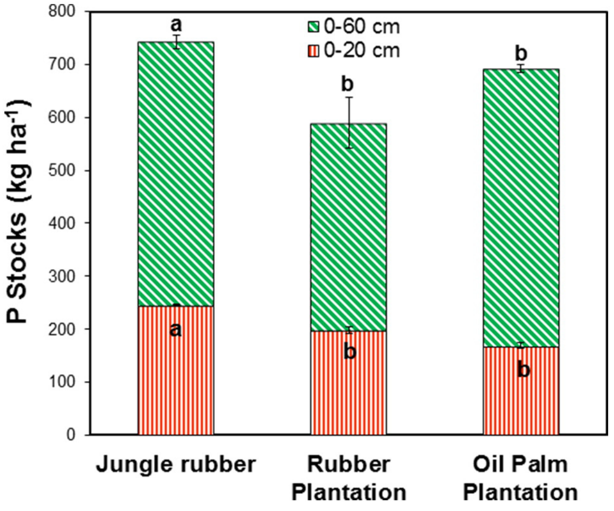

Fig. 3. Soil phosphorus stocks (kg P ha^{-1}) at the 0–20 and 0–60 cm soil depth layer depending on land use. Values represent means ± SE (n = 3). Means followed by different letters within the same depth differ significantly (*t*-test at $P < 0.05$).

decreased ($P < 0.05$) in plantations by almost 50% compared to the jungle rubber.

Moderately-available Pi (NaOH-Pi) had the same content in all soils across all horizons. However, moderately-available Po decreased in the top horizon by 18% under oil palm and by 23% under rubber plantations compared to the jungle rubber. Differences between jungle rubber and the intensive plantations were significant ($P < 0.05$) only in the Ah horizon (0–10 cm) but not at the lower depths. Nonetheless, organic P fractions were almost twice as high as the inorganic P in all soils across depths. Non-available P in the topsoil (extracted by 1 M HCl and Residual-P) also decreased ($P < 0.05$) by 47% under oil palm and by 24% under rubber plantations compared to jungle rubber, but there were no differences below 10 cm. Our results (Figs. 1 & 2) suggest that land-use change caused the redistribution of P forms.

3.2. Total phosphorus content and phosphorus stocks

Most of the total P in soils of each land-use type was inorganic. It ranged between 39 and 106 mg P kg^{-1}, which represented between 68 and 83% of the total P (Fig. 2). Jungle rubber had more total Pi in the Ah horizon than in intensive monoculture plantations (i.e. oil palm and rubber). Total Po was also higher ($P < 0.05$) in the Ah horizon in jungle rubber than in the intensive monoculture plantations.

The soil P stock decreased after conversion of jungle rubber to an intensively managed oil palm plantation at the 0–20 cm depth by about 31%, and slightly increased by 5% at the 0–60 cm depth (Fig. 3). Soil P stocks also gradually decreased in the rubber plantation by 20% from 0 to 60 cm depth. P stocks were lost in both plantation systems ($P < 0.05$) compared to jungle rubber at 0–20 cm depth. Consequently, land-use change strongly decreased P stocks, especially in the topsoil.

3.3. Carbon to phosphorus ratio

The carbon:organic P (C:Po) ratio calculated using total organic P content ranged between 695 ± 342 and 742 ± 88 in surface soils, being greatest in soils with the least total soil Po (Fig. 4). The lowest C:Po ratio (335 ± 181) was measured at 35 cm depth in the rubber plantation. Moreover, the ratio declined from the surface down to 35 cm and stabilized in the subsoil, but not under oil palm. Likewise, the C:TP ratio calculated using the total P (sum of all P fractions) had the same pattern as the C:Po ratio in all land-use types. Both ratios declined from the surface down to 35 cm depth and tended to stabilize in the lower depth. The C:TP ratio ranged between 57 ± 34 and 220 ± 12, being highest in jungle rubber across all soil depths.

3.4. Correlation between phosphorus fractions and soil carbon

Total Po was correlated with soil C ($R^2 = 0.29–0.84$; $P < 0.001$) (Fig. 5). The correlation was strongest in jungle rubber ($R^2 = 0.84$) compared to monoculture plantations. Easily-available P fractions (H_2O, $NaHCO_3$ Pi and Po) were also strongly correlated ($R^2 = 0.79–0.82$; $P < 0.001$) with soil C in all land-use types. Easily-available P was dominated by $NaHCO_3$-extractable Po, and had the strongest correlation to soil C ($R^2 = 0.37–0.68$; $P < 0.001$) compared to H_2O-P and $NaHCO_3$-Pi in all land-use types. Easily-available P correlated well with C in jungle rubber and rubber compared to oil palm plantation. This suggests that SOM contributes to the P fertility in the soil.

4. Discussion

4.1. Effect of land-use change on the forms and distribution of soil P

Inorganic and organic P forms of different availability are influenced by anthropogenic, biotic and abiotic processes that either increase or decrease the soil P content (Fig. 1 & Table 1). The higher concentrations

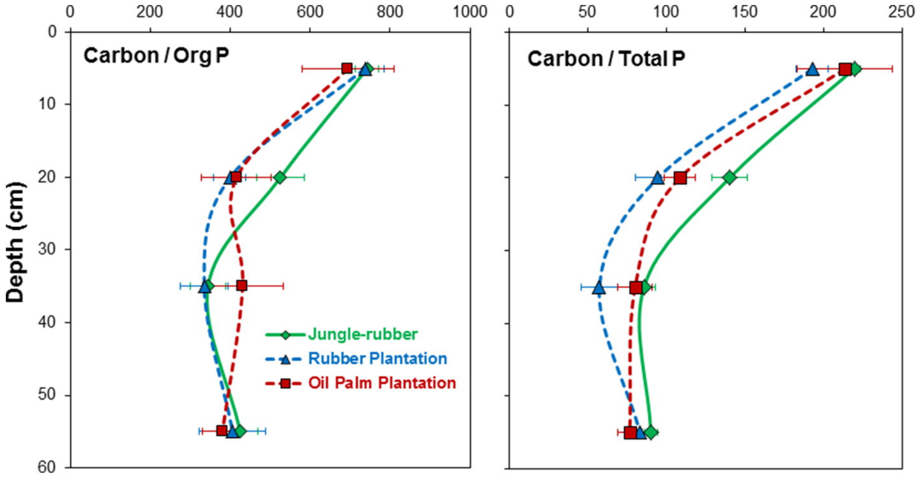

Fig. 4. Ratio of (left) soil carbon to organic phosphorus, and (right) soil carbon to total phosphorus depending on land use. Values represent means ± SE (n = 3).

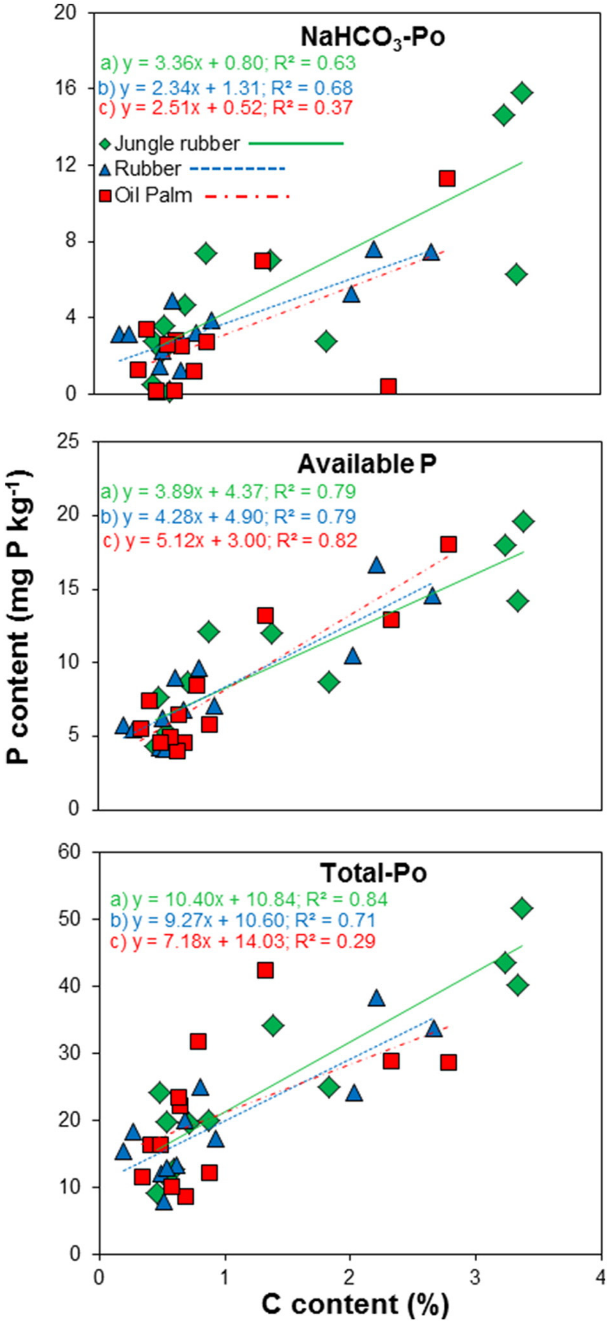

Fig. 5. Relationship between soil carbon content and (top) $NaHCO_3$-Po, (middle) available-P (i.e., H_2O-extractable P, $NaHCO_3$-extractable inorganic and organic P), (bottom) total Po depending on land use. Very close correlation between all land uses for $NaHCO_3$-Po, available-P and total-Po to soil carbon reflects P losses by erosion.

of easily-available Pi (i.e., H_2O–Pi and $NaHCO_3$-Pi) (Fig. 2) that we found in intensive plantations are explained by the adsorption of P from fertilizers in the surface layer (Henriquez, 2002; Neufeldt et al., 2000). The increase in P content due to fertilization is more notable in available Pi (labile-Pi) than in other P fractions (e.g., Neufeldt et al., 2000). At the same time, the depletion of easily-mineralizable Po ($NaHCO_3$-Po) could be attributed to the decrease of litterfall accumulated in the plantation. The aboveground litterfall of rubber (3.84 Mg ha^{-1} $year^{-1}$) and oil palm (6.23 Mg ha^{-1} $year^{-1}$) plantation was lower than jungle rubber litterfall (7.66 Mg ha^{-1} $year^{-1}$) (Kotowska et al., 2016). The high accumulation of litterfall in jungle rubber resulted to high easily-mineralizable Po. Soil without tillage for 20 years builds up a considerable amount of Po from accumulated SOM (Magid, 1993). Furthermore, the vertical distribution of Po is related to the distribution of SOM in the soil profile (Sarapatka, 2003). Vertical trends of easily-available P that declined from the surface to the lower depths is explained by a process termed “nutrient pumping” (Kautz et al., 2013). This means that trees are able to absorb nutrients from lower soil horizons and then redistribute them to the surface soil through litterfall and throughfall (Farley and Kelly, 2004). Easily-available P content was dominated by $NaHCO_3$-Po and strongly correlated with soil C (Fig. 5). This highlighted the potential importance of SOM in maintaining fertility in tropical ecosystems (Tiessen et al., 1994).

P extracted by 0.5 M NaOH before and after ultrasonification is associated with humic compounds and mostly adsorbed to Al and Fe oxides in acidic soils (Hedley et al., 1982; Schoenau et al., 1989). This fraction is considered to be moderately-available P that involves long-term soil P transformation (Hedley et al., 1982; Tiessen et al., 1984) and acts as a buffer for labile Pi in highly weathered soils (Guo et al., 2000). Reduced moderately-available Po (Fig. 2) in plantations is also due to less litter input than in jungle rubber. As litter inputs dropped, the soil capacity to retain P in the form of NaOH-extractable Po diminished (Zamuner et al., 2008). In contrast, P fixation by Al and Fe is repressed, reducing available P in the long term (Dieter et al., 2010; Groppo et al., 2015). Nonetheless, Po fractions were almost twice as more as the Pi fractions in all soils across all depths. This highlights the importance of Po as P reserves that involve long-term soil P transformation and in P cycling when soil Pi reserves are limited (Buehler et al., 2002).

The importance of plant non-available P forms has been frequently reported (Neufeldt et al., 2000; Reddy et al., 1999; Sharpley, 1985; Tiessen et al., 1984; Zheng et al., 2002). Depending on soil type and management, non-available P forms could be mobilized and become available for plant uptake, depending on P form, undergoing desorption, weathering, and mineralization processes. Land-use change decreases non-available P (Fig. 2), which suggests lower P reserves compared with agroforest systems.

4.2. Phosphorus status and long-term sustainability

Soil total P represents the long-term potential of the P supply, whereas easily-available Pi represents the short-term bioavailability. Moderately available and non-available P (Figs. 1 & 2) decreased after land-use change, which contributes to the decrease of total P in intensive monoculture plantations. Land-use change, however, increases only easily-available Pi (H_2O–Pi and NaHCO3-Pi) due to fertilization, which merely maintains short-term soil fertility. Nonetheless, it is not sustainable in the long run due to the depletion of soil P reserve fractions (i.e., $NaHCO_3$-Po, moderately-available P and non-available P). The higher correlation (Fig. 5) between total Po and soil C was mainly attributable to $NaHCO_3$-extractable Po versus NaOH-extractable Po. This further indicates that litter mineralization significantly contributes to the available P content (Fig. 2).

The decreased soil P stocks at 0–20 cm depth (Fig. 3) suggests that the addition of NPK fertilizer did not compensate for the amount of P lost after conversion of extensively managed (jungle rubber) to intensive plantations. The strong soil P loss after intensification is closely linked to soil erosion. This interpretation is supported by Guillaume et al. (2015), who estimated strong soil erosion under rubber and oil palm plantations. Accelerated soil

Table 1
Direct effects of abiotic, biotic and anthropogenic processes on P fractions in soil.

Processes/mechanism		Easily-available		Moderately-available		Non-available
		Pi	Po	Pi	Po	Pi + Po
Anthropogenic	P fertilization (inorganic & organic)	↑↑↑	↑↑	↑	↑	↑
Biotic	Litter input	↑	↑↑		↑	
	Mineralization	↑↑	↓↓		↓	
	Plant uptake	↓↓↓	↓↓	↓↓	↓	
	Immobilization in microorganism	↓↓↓	↓↓		↑↑	↑↑
Abiotic	Erosion	↓↓↓	↓↓↓	↓↓↓	↓↓↓	↓↓↓
	Leaching	↓↓	↓	↓		
	Fixation/adsorption	↓↓↓	↓↓	↑↑	↑	↑↑
	Desorption	↑↑		↓↓		↓

↑↑↑ strong increase; ↑↑ moderate increase; ↑ slight increase; ↓↓↓ strong decrease; ↓↓ moderate decrease; ↓ slight decrease.

erosion due to land-use change reduced SOM by half or more (Zheng et al., 2005), which is a source of available P when mineralized. About 50% of total P is contained in SOM (Pimentel et al., 1995). The soil removed by erosion is 1.3 to 5 times richer in OM than the remaining soil (Matson et al., 1997; Pimentel et al., 1995). A combination of decreasing litter inputs (when plantation crops are replaced rainforest) and increasing soil erosion leads to a decrease of SOM and P in the long run (Chimdi et al., 2014). Another contribution to the P depletion in plantation soils is nutrient export with the yield (Kotowska et al., 2015). This pertains especially to the reproductive parts of oil palm trees that are removed from the ecosystem and not returned back to the soil (Violita et al., 2015). Indeed, three years of litter removal resulted in a marked reduction of Po in the surface soil (Vincent et al., 2010). At the same time, the slight increase (5%) of P stocks under oil palm at 0–60 cm depth might be attributed to leaching. Some of the fertilizers applied at the soil surface were leached or removed by erosion. Such leached P fertilizers accumulated at lower depths (≥ 35 cm depth) where Al and Fe accumulation were high. This slightly boosted (5%) P stocks of oil palm in the lower depth compared with jungle rubber. The P fixation, however, decreased the availability of P for plant uptake.

4.3. The importance of organic phosphorus for P availability

In tropical ecosystems, the turnover of OM is a very important source of Po that is widely considered to be fundamental in maintaining the P supply (Condron and Tiessen, 2005). Available P in highly weathered soils is generally low and depends on Po mineralization (Tiessen et al., 1984). Po in the available pool ($NaHCO_3$-Po) is very important because it increases the apparent P availability (Johnson et al., 2003). Likewise, Po, which occurred in moderately-available (NaOH-Po), is as important as $NaHCO_3$-Po because it contributes to P reserves. Both Po fractions decrease after land-use change (Figs. 1 & 2), mainly due to soil erosion and yield export.

The C:Po ratios can be used to assess the nutrient status of a site. An adequate supply of available phosphate for plant growth prevents Po mineralization by phosphatases. This result in an accumulation of soil Po and a reduced C:Po ratio (Dieter et al., 2010; Zhao et al., 2008). If available P is insufficient for plant growth, phosphatase synthesis increases (Spohn and Kuzyakov, 2013). This, in turn, enhances Po mineralization compared to C mineralization and increases the C:Po ratio (Dieter et al., 2010; McGill and Cole, 1981). If soil has a high available P content, then the C:Po ratio is <100, whereas the ratio in soils with insufficient available P is >200 (Dieter et al., 2010; Smeck, 1985). Based on the above considerations, the declining C:Po ratio of plantation soils (Fig. 4) indicates the effect of available P from the added NPK fertilizer. Nonetheless, the C:Po ratio in each land-use type (>200) confirms that the soils in the study area are P limited. The low available P in all land-use types is due to the inherently low-P status of the parent material and erosion losses (Henriquez, 2002; Moges et al., 2013).

5. Conclusions

The forms and distribution of P in various land-use systems is determined by anthropogenic, biotic and abiotic processes. The proportion of Pi was higher than Po in soil under each land-use type. Pi was mainly composed of chemically more stable (NaOH-Pi) and relatively insoluble P forms (i.e., HCl-Pi and Residual-P) rather than available-Pi in all land-use types. The effects of the management after land-use change of jungle rubber to oil palm and rubber plantations on Pi and Po fractions were underlined. Short-term high NPK fertilization increased easily-available Pi in intensive monoculture plantations. At the same time, easily-mineralizable Po decreased due to depletion of SOM. Moderately-available P and non-available P also decreased after land-use change of jungle rubber to oil palm and rubber plantations. Land-use change leads to an overall reduction of P stocks due to a strong decrease of SOM caused by erosion and yield export of rubber or of oil palm seeds. Fertilization did not compensate for the P losses. Fertilization increases solely the available Pi in the topsoil. This maintains or increases fertility only over the short term, but decreases it over the long term by depleting P reserves (e.g., moderate or non-available P). The positive strong correlation between soil C and Po and the C:Po ratio indicates that mineralization from SOM strongly contributes to the available P content. This highlights the potential of SOM in maintaining P reserves. This calls for proper plantation management practices to stop SOM losses.

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Appendix A

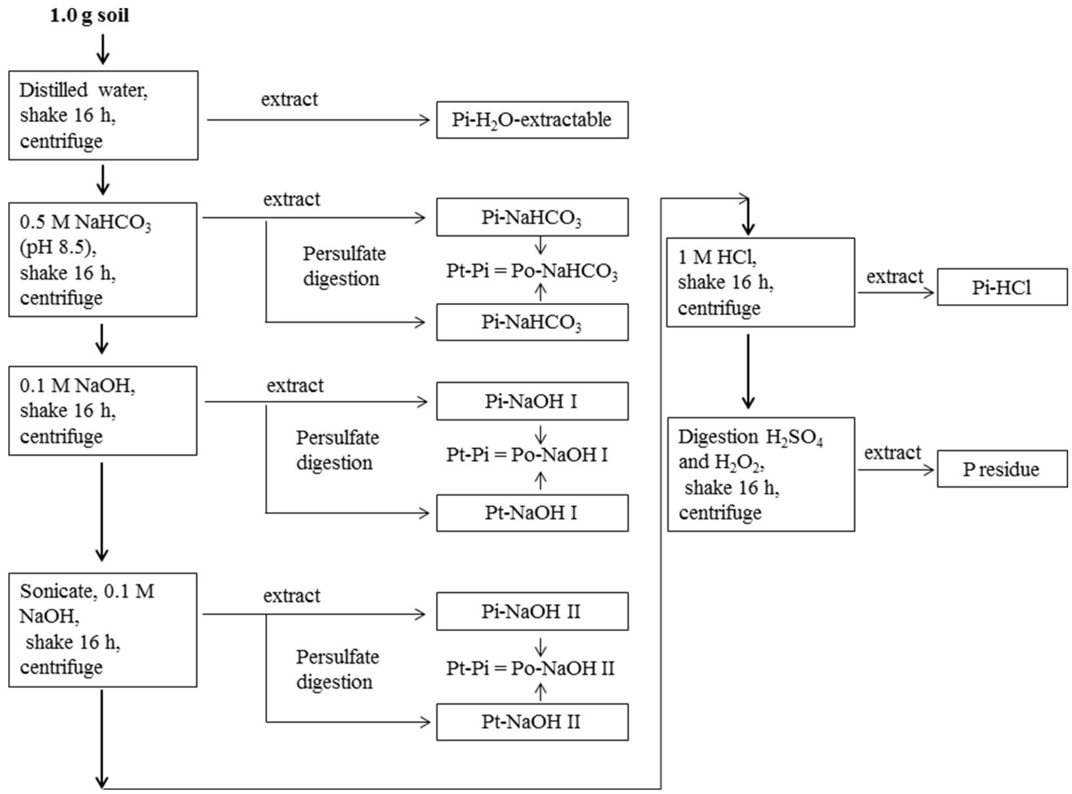

Fig. A.1. Diagram of Hedley P sequential fractionation scheme as modified by Tiessen and Moir (1993).

References

Allen, K., Corre, M.D., Tjoa, A., Veldkamp, E., 2015. Soil nitrogen-cycling responses to conversion of lowland forests to oil palm and rubber plantations in Sumatra, Indonesia. Plos One http://dx.doi.org/10.1371/journal.pone.013332.

Bowman, R.A., Cole, C.V., 1978. An exploratory method for fractionation of organic phosphorus from grassland soils. Soil Sci. 125, 95–101.

Bucher, M., Rausch, C., Daram, P., 2001. Molecular and biochemical mechanisms of phosphorus uptake into plants. J. Plant Nutr. Soil Sci. 164, 209–217.

Buehler, S., Oberson, A., Rao, I.M., Friessen, D.K., Frossard, E., 2002. Sequential phosphorus extraction of a 33P-labeled oxisol under contrasting agricultural systems. Soil Sci. Soc. Am. J. 66, 868–877.

Cassanova, E., Salas, A.M., Toro, M., 2002. Evaluating the effectiveness of phosphate fertilizers in some Venezuela soils. Nutr. Cycl. Agroecosyst. 63, 13–20.

Chacon, N., Dezzeo, N., 2004. Phosphorus fractions and sorption processes in soil samples taken in a forest-savanna sequence of the Gran Sabana in Southern Venezuela. Biol. Fertil. Soils 40, 14–19.

Chang, S.C., Jackson, M.L., 1957. Fractionation of soil phosphorus. Soil Sci. 84, 133–144.

Chen, C.R., Condron, L.M., Davis, M.R., Sherlock, R.R., 2003. Seasonal changes in soil phosphorus and associated microbial properties under adjacent grassland and forest in New Zealand for. Ecol. Manag. 177, 539–557.

Chimdi, A., Esala, M., Ylivainio, K., 2014. Sequential fractionation patterns of soil phosphorus collected from different land use systems of Dire Inchine District, West Shawa Zone, Ethiopia. J. Amer-Eura of Sci. Res. 9 (3), 51–57.

Condron, I.M., Tiessen, H., 2005. Interactions of organic phosphorus in terrestrial ecosystems. In: Turner, B.L., Frossard, E., Baldwin, D.S. (Eds.), Inositol Phosphates: Linking Agriculture and the Environment. CAB Int., Wallingford, UK, pp. 295–308.

Dieter, D., Elsenbeer, H., Turner, B., 2010. Phosphorus fractionation in lowland tropical rainforest soils in Central Panama. Catena 82, 118–125.

Environmental Protection Agency, 1997. Methods of Chemical Analysis of Water and Wastes. Environmental Protection Agency, Cincinnati, Ohio.

Farley, K.A., Kelly, E.F., 2004. Effects of afforestation of a paramo grassland on soil nutrient status. Forest Ecol. Manag. 195, 281–290.

Garcia-Montiel, D.C., Nelly, C.H., Melillo, J., Thomas, S., Steudler, P.A., Cerri, C.C., 2000. Soil phosphorus transformation following forest clearing for pasture in the Brazilian Amazon. Soil Sci. Soc. Am. J. 64, 1792–1804.

Gatto, M., Wollni, M., Qaim, M., 2015. Oil palm boom and land-use dynamics in Indonesia: the role of policies and socioeconomic factors. Land Use Policy 46, 292–303.

Geissen, V., Sanchez-Hernandez, R., Kampichler, C., Ramos-Reyes, R., Sepulveda-Lozada, A., Ochoa-Goana, S., de Jong, B.H.J., Huerta-Lwanga, E., Hernandez-Daumas, S., 2009. Effects of land-use change on some properties of tropical soils – an example from Southeast Mexico. Geoderma 151, 87–97.

Groppo, J.D., Lins, S.R.M., Camargo, P.B., Assad, E.D., Pinto, H.S., Martins, S.C., Salgado, P.R., Evangelista, B., Vasconcellos, E., Sano, E.E., Pavao, E., Luna, R., Martinelli, L.A., 2015. Changes in soil carbon, nitrogen and phosphorus due to land-use changes in Brazil. Biogeosciences 12, 2533–2571.

Guillaume, T., Holtkamp, A.M., Damris, M., Brümmer, B., Kuzyakov, Y., 2016b. Soil degradation in oil palm and rubber plantations under land resource scarcity. Agric. Ecosyst. Environ. 232, 110–118.

Guillaume, T., Maranguit, D., Murtilaksono, K., Kuzyakov, Y., 2016a. Sensitivity and resistance of soil fertility indicators to land-use changes: new concept and examples from conversion of Indonesian rainforest to plantations. Ecol. Indic. 67, 49–57.

Guillaume, T., Damris, M., Kuzyakov, Y., 2015. Losses of soil carbon by converting tropical forest to plantations: erosion and decomposition estimated by δ^{13}C. Glob. Chang. Biol. http://dx.doi.org/10.1111/gcb.12907.

Guo, F., Yost, R.S., Hue, N.V., Evenson, C.I., Silva, J.A., 2000. Changes in phosphorus fractions in soils under intensive plant growth. Soil Sci. Soc. Am. J. 64, 1681–1689.

Hansen, M.C., Stehman, S.V., Potapov, P.V., Arunarwati, B., Stolle, F., Pittman, K., 2009. Quantifying changes in the rates of forest clearing in Indonesia from 1990 to 2005 using remotely sensed data sets. Environ. Res. Lett. 4, 1–12.

Hedley, M.J., Stewart, J.W.B., Chauhan, B.S., 1982. Changes inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations. Soil Sci. Soc. Am. J. 46, 970–976.

Henriquez, C., 2002. Assessing soil phosphorus status under different agronomic land use. Retrospective Theses and Dissertation Paper. 517.

Holford, I.C.R., 1997. Soil phosphorus: its measurement and its uptake by plants. Aust. J. Soil Res. 35, 227–239.

Johnson, A.H., Frizano, J., Vann, D.R., 2003. Biogeochemical implications of labile phosphorus in forest soils determined by the Hedley fractionation procedure. Oecologia 135, 487–499.

Kautz, T., Amelung, W., Ewert, E., Gaiser, T., Horn, R., Jahn, R., Javaux, M., Kuzyakov, Y., Munch, J.C., Pätzold, S., Peth, S., Scherer, H.W., Schloter, M., Schneider, H., Vanderborght, J., Vetterlein, D., Wiesenberg, G., Köpke, U., 2013. Nutrient acquisition from the arable subsoil in temperate climates: a review. Soil Biol. Biochem. 57, 1003–1022.

Ketterings, Q.M., van Noordwijk, M., Bigham, J.M., 2002. Soil phosphorus availability after slash-and-burn fires of different intensities in rubber agroforests in Sumatra, Indonesia. Agric. Ecosyst. Environ. 92, 37–48.

Kotowska, M.M., Leuschner, C., Triadiati, T., Hertel, D., 2016. Conversion of tropical lowland forest reduces nutrient return through litterfall, and alters nutrient use efficiency and seasonality of net primary production. Oecologia 180, 601–618.

Kotowska, M.M., Leuschner, C., Triadiati, T., Meriem, S., Hertel, D., 2015. Quantifying above-and below ground biomass carbon loss with forest conversion in tropical lowlands of Sumatra (Indonesia). Glob. Chang. Biol. http://dx.doi.org/10.1111/gcb.12979.

Laumonier, Y., Uryu, Y., Stüwe, M., Budiman, A., Setiabudi, B., Hadian, O., 2010. Eco-floristic sectors and deforestation threats in Sumatra: identifying new conservation area network priorities for ecosystem-based land use planning. Biodivers. Conserv. 19, 1153–1174.

Magid, J., 1993. Vegetation effects on P fractions in set-aside soils. Plant Soil 149, 111–119.

Margono, B.A., Potapov, P.V., Turubanova, S., Stolle, F., Hansen, M.C., 2014. Primary forest cover loss in Indonesia over 2000–2012. Nat. Clim. Chang. 4, 730–735.

Matson, P.A., Parton, W.J., Power, A.G., Swift, M.J., 1997. Agricultural intensification and ecosystem properties. Sci. http://dx.doi.org/10.1126/277.5325.504.

McGill, W., Cole, C., 1981. Comparative aspects of cycling of organic C, N, S and P through soil organic matter. Geoderma 26, 267–286.

Moges, A., Dagnachew, M., Yimer, F., 2013. Land use effects on soil quality indicators: a case study of Abo-Wonsho Southern Ethiopia. Appl. Environ. Soil Sci. 1–9.

Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27, 31–36.

Neufeldt, H., da Silva, J.E., Ayarza, M.A., Zech, W., 2000. Land-use effects on phosphorus fractions in Cerrado oxisols. Biol. Fertil. Soils 31, 30–37.

Numata, I., Chadwick, O.A., Roberts, D.A., Schimel, J.P., Sampaio, F.F., Leonidas, F.C., Soares, J.V., 2007. Temporal nutrient variation in soil and vegetation of post-forest pastures as a function of soil order, pasture age, and management, Rondonia, Brazil. Agric. Ecosyst. Environ. 118, 159–172.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R., 1995. Environmental and economic costs of soil erosion and conservation. Sci. 267, 1117–1123.

Reddy, D.D., Rao, A.S., Takkar, P.N., 1999. Effects of repeated manure and fertilizer phosphorus additions on soil phosphorus dynamics under a soybean-wheat rotation. Biol. Fertil. Soils 28, 150–155.

Sanchez, P.A., 1976. The Properties and Management of Soils in the Tropics. Wiley, New York.

Sarapatka, B., 2003. Phosphatase Activities (ACP, ALP) in Agroecosystem Soil Doctoral thesis 396. Acta Universitatis Agriculturae Sueciae, Agraria.

Schoenau, J.J., Stewart, J.W.B., Bettany, J.R., 1989. Forms and cycling of phosphorus in prairie and boreal forest soils. Biogeochemistry 8, 223–237.

Sharpley, A.N., 1985. Phosphorus cycling in unfertilized and fertilized agricultural soils. Soil Sci. Soc. Am. J. 49, 905–911.

Smeck, N., 1985. Phosphorus dynamics in soils and landscapes. Geoderma 36, 185–199.

Solomon, D., Lehmann, J., Mamo, T., Fritzsche, F., Zech, W., 2002. Phosphorus forms and dynamics as influenced by land-use changes in the sub-humid Ethiopian highlands. Goederma 105, 21–48.

Spohn, M., Kuzyakov, Y., 2013. Distribution of microbial- and root-derived phosphatase activities in the rhizosphere depending on P availability and C allocation – coupling soil zymography with 14C imaging. Soil Biol. Biochem. 67, 106–113.

Spohn, M., Ermak, A., Kuzyakov, Y., 2013. Microbial gross organic phosphorus mineralization can be stimulated by root exudates – a 33P isotopic dilution study. Soil Biol. Biochem. 65, 254–263.

Tarigan, S.D., Sunarti, Widyaliza, S., 2015. Expansion of oil palm plantations and forest cover changes in Bungo and Merangin Districts, Jambi Province, Indonesia. Procedia Envi. Sci. 24, 199–205.

Tiessen, H., Moir, J.O., 1993. Characterization of available P by sequential extraction. In: Carter, M.R. (Ed.), Soil Sampling and Methods of Analysis. Canadian Soc. Soil Sci., pp. 75–86.

Tiessen, H., Chacon, P., Cuevas, E., 1994. Phosphorus and nitrogen status in soils and vegetation along a toposequence of dystrophic rainforests on the upper Rio Negro. Oecologia 99, 145–150.

Tiessen, H., Stewart, J.W.B., Cole, C.V., 1984. Pathways of phosphorus transformations in soil of differing pedogenesis. Soil Sci. Soc. Am. J. 48, 853–858.

Townsend, A.R., Asner, G.P., Cleveland, C.C., Lefer, M.E., Bustamante, M.C., 2002. Unexpected changes in soil phosphorus dynamics following tropical deforestation to cattle pasture. J. Geophys. Res. http://dx.doi.org/10.1029/2001JD000650.

Villamor, G.B., Pontius Jr., R.G., Van Noordwijk, M., 2014. Agroforest's growing role in reducing carbon losses from Jambi (Sumatra), Indonesia. Reg. Environ. Chang. 14, 825–834.

Vincent, A.G., Turner, B.L., Tanner, E.V.J., 2010. Soil organic phosphorus dynamics following perturbation of litter cycling in a tropical moist forest. Eur. J. Soil Sci. 61, 48–57.

Violita, Kotowska, M.M., Hertel, D., Triadiati, T., Miftahudin, Anas, I., 2015. Transformation of lowland rainforest into oil palm plantations result in changes of leaf litter production and decomposition in Sumatra, Indonesia. J. Biodivers. Environ. Sci. 6, 546–556.

Vitousek, P.M., 1984. Literfall, nutrient cycling and nutrient limitation in tropical forest. Ecology 65, 285–298.

Wilcove, D.S., Koh, L.P., 2010. Addressing the threats to biodiversity from oil-palm agriculture. Biodivers. Conserv. 19, 999–1007.

Wright, A.L., 2009. Soil phosphorus stocks and distribution in chemical fractions for long-term sugarcane, pasture, turfgrass, and forest systems in Florida. Nutr. Cycl. Agroecosyst. 83, 223–231.

Yang, X., Post, W.M., 2011. Phosphorus transformations as a function of pedogenesis: a synthesis of soil phosphorus data using Hedley fractionation method. Biogeosciences 8, 2907–2916.

Zamuner, E.C., Picone, L.I., Echeverria, H.E., 2008. Organic and inorganic phosphorus in Mollisol soil under different tillage practices. Soil Till. Res. 99, 131–138.

Zhao, Q., Zeng, D.H., Fan, Z.P., Lee, D.K., 2008. Effect of land cover change on soil phosphorus fractions in Southeastern Horqin Sandy Land, Northern China. Pedosphere 18 (6), 741–748.

Zheng, F., He, X., Gao, X., Zhang, C., Tang, K., 2005. Effects of erosion patterns on nutrient loss following deforestation on the Loess Plateau of China. Agri. Ecosyst. Environ. 108, 85–97.

Zheng, Z., Simard, R.R., Lafond, J., Parent, L.E., 2002. Pathways of soil phosphorus transformations after 8 years of cultivation under contrasting cropping practices. Soil Sci. Soc. Am. J. 66, 999–1007.