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Riparian wetland properties counter the effect of land-use change on soil carbon stocks after rainforest conversion to plantations

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ABSTRACT

Progressive conversion of tropical rainforests to agricultural monocultures in South East Asia increasingly affects landscape types such as riparian areas. The impacts of conversions on soil organic matter (SOM) vary with changing landforms. However, this was often not accounted for in previous studies where SOM in soils in riparian areas was combined with SOM from well-drained adjacent slopes. Because riparian areas have a high carbon (C) storage potential, our objectives were i) to assess their C stocks after conversion to rubber and oil palm plantations in Sumatra (Indonesia) and ii) to compare the impacts of land use conversion on C stocks between riparian and well-drained areas. Average soil C stock losses from the top 30 cm were about 14% and 4% following conversion of riparian forest to rubber and oil palm plantations, respectively, indicating a high resistance of C to mineralization. C losses from well-drained areas were twice as high as from riparian areas after the respective conversion. $\delta^{13}C$ values from riparian areas showed clear heterogeneity down soil profiles that was explained i) by alternating oxic and anoxic conditions, leading to reduced SOM and litter decomposition in riparian areas and ii) by mineral sediments and organic matter deposition and accumulation by erosion from adjacent slopes covered by plantations. We conclude that riparian areas are more resilient in terms of soil C storage towards land-use change than well-drained areas because of sediment deposition and reduced oxygen availability. On this basis, we developed a conceptual model of the effects of land-use change and various ecotone characteristics on SOM mineralization in the top- and subsoil of riparian areas.

1. Introduction

The growing demand for cash crop products, such as palm oil and rubber, causes the conversion of tropical rainforests into agricultural land. This land-use change towards intensively managed plantations has major impacts on the global carbon (C) cycle, since it reduces soil organic carbon (SOC) stocks and increases carbon dioxide emissions (Don et al., 2011; Harris et al., 2012). The tropics play a crucial role in global C reservoirs, as they store one third of the global soil C (Scharlemann et al., 2014). Due to the described land-use change the total anthropogenic carbon dioxide emissions increased by 48% (Ciais et al., 2013).

In 2012, Indonesia had the highest deforestation rate in the world (0.84 Mha) (Margono et al., 2014). Deforestation is particularly severe on the island of Sumatra, where vast expanses of lowland rainforest have been converted into oil palm and rubber plantations (Austin et al., 2019; Margono et al., 2014). Various studies showed that the forest conversion into agricultural land is accompanied with C losses from mineral soil in this region (Borchard et al., 2019; de Blécourt et al., 2013; Guillaume et al., 2015). In mineral soils in well-drained areas, land-use changes induce SOC losses mainly by decreased C input, accelerated decomposition, or increased erosion (Guillaume et al., 2015). The conversion into plantations reduces above- and belowground biomass up to five fold (Guillaume et al., 2018; Kotowska et al., 2015)

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leading to a reduced C input (Powers, 2004). In combination with mineral fertilization, which reduces the nutrient limitation of soil microbial communities, this accelerates biogeochemical cycles and triggers the decomposition of soil organic matter (SOM) (Becker et al., 2015; Becker and Kuzyakov, 2018). Additionally, reduced vegetation cover in plantations strongly increases soil erosion, which in turn leads to a loss of the C-rich topsoil (Don et al., 2011; Guillaume et al., 2015; Labrière et al., 2015). Forest soils in tropical wetlands have a large C-stock potential (Wantzen et al. 2012) and particularly large C losses are expected from conversion into agricultural land. However, most research with specific focus on riparian areas has been conducted in temperate regions (Guyette et al., 2002; Zehetner et al., 2009). The impact of land-use change in non-peat riparian areas on C losses has hardly been investigated; yet in the tropics (Wantzen et al., 2012). The distinction between the terms 'riparian area' and 'wetland' mainly depends on three main differences: first, riparian areas are commonly transitional zones between terrestrial ecosystems and water bodies. These areas are temporarily inundated or water-logged (Wantzen et al., 2008) with a high water table (McCormick, 1978) and can experience seasonal wet and dry cycles, i.e. temporal changes between oxic and anoxic conditions (Décamps et al., 2004). The term 'wetland', however, describes soils that are often water-saturated but do not need to be adjacent to a river. Second, related to the divergent locations, the terms describe different energy and temporal dynamics of the affecting water body. Riparian areas often reflect higher water flow dynamics where sediments are transported and material accumulation and erosion occur. On the contrary, wetlands are associated with a water table that is close to or above the surface and are characterized by only a scarce flow of water (Brown et al., 1978). Third, as mineral and organic soils both occur within wetlands, the level of organic content is not the only indicator to identify riparian areas as a certain type of wetland. The riparian SOM balance is further influenced by two specific factors: the redistribution of fluvial sediment (Rieger et al., 2014) and oxygen deficiency in water-saturated soil. The effects of these processes on riparian C balances and stabilization can offset erosion and SOM mineralization. Deposits that enter (and partly leave) the ecotone contain high amounts of allochthonous organic material from terrestrial sources and dissolved organic matter from aquatic sources (Moore et al., 2013). Flooding duration and frequency also play crucial roles for long term C accumulation (Bendix and Hupp, 2000; Graf-Rosenfellner et al., 2016).

Depth profiles of natural ^{13}C abundance ($\delta^{13}C$) serve as helpful tools to evaluate the decomposition state of SOM (Alewell et al., 2011; Guillaume et al., 2015; Krüger et al., 2014). The $\delta^{13}C$ value describes the ratio of ^{13}C:^{12}C abundance relatively to the international PDB limestone standard (Craig, 1953). SOM and litter decomposition lead to an ^{13}C enrichment of SOM and, thus, they induce a shift to less negative $\delta^{13}C$ values. In well-drained mineral soils, an increase of $\delta^{13}C$ values was observed with soil depth and soil age (Andreeva et al., 2013; de; Zang et al., 2018). Apparently here, aerobic decomposition processes dominate and lead to ^{13}C enrichment. In contrast, soils in riparian areas experience wet and-dry cycles, and thus alternating oxic and anoxic conditions, which change the ^{13}C fractionation with depth. The delayed decomposition under anoxic conditions, might slow down the ^{13}C fractionation. Here, $\delta^{13}C$ values show a more stable and uniform pattern. If shifts to lighter $\delta^{13}C$ values are detected, this will hint to an enrichment of recalcitrant organic substances during anaerobic decomposition which are depleted in ^{13}C (Alewell et al., 2011; Drollinger et al., 2019).

For the assessment of erosion, it is required that the $\delta^{13}C$ values in the plantation subsoil are similar to those in the forest subsoil prior conversion. Assuming that subsoil $\delta^{13}C$ values are unaffected by land-use change or surface matter inputs, a shift of topsoil $\delta^{13}C$ towards subsoil $\delta^{13}C$ values may be interpreted as erosion dynamics. The respective layer experienced a vertical shift towards the soil surface after an erosional loss of the upper layer (Guillaume et al., 2015). In addition, physical mixing and deposition of fresh material with differing isotopic composition, can influence the profileś ^{13}C signature, particularly in dynamic landscapes like riparian areas (de Junet et al., 2005; Kelleway et al., 2017). Therefore, riparian properties may lead to unique C storage mechanisms, which are expressed as specific dynamics in the $\delta^{13}C$ value. It is essential to understand the combined effects of land-use change and riparian water dynamics on soil C balance to predict probable climate change responses and the mitigation potential of riparian areas. This is of special importance because riparian areas have a high C storage potential (Hazlett et al., 2005) which makes them a valuable asset for regional and global carbon management (Rieger et al., 2014).

Therefore, the objectives of this study were i) to quantify SOC losses in riparian areas after land-use change from forest to oil palm and rubber plantations, and ii) to determine whether the impacts of this land-use change are comparable for riparian and well-drained areas. We further aimed iii) to develop a conceptional understanding of C sequestration and storage in riparian areas by using the $\delta^{13}C$ values to disentangle SOM decomposition from erosion and deposition. We hypothesize that (1) the conversion of riparian forest to plantations has a more negative impact on soil C stocks than for well-drained mineral soils because anoxic conditions slow down decomposition and thus lead to higher C storage. We hypothesize that (2) not SOC mineralization but SOM transport and deposition are the factors that predominantly influence SOC stocks.

2. Materials and methods

2.1. Study sites

The study was carried out in the Province of Jambi in Sumatra, Indonesia. The region has a tropical humid climate with an average annual temperature of 27.6 °C. Average annual rainfall is 2235 mm. The rainy season has two peak periods (March and December) and an average monthly rainfall of 261 mm, followed by a drier period between April and September with a monthly rainfall of 161 mm (Drescher et al., 2016). The natural vegetation is a mixed dipterocarp lowland rainforest (Laumonier, 1997). Nowadays, however, Jambi Province is a region dominated by large rubber (*Hevea brasiliensis*) and oil palm (*Elaeis guineensis*) monocultures. Plantations managed by smallholder farmers were selected, which varied between 8 and 18 years (rubber) and 10–16 years (oil palm) in stand age (Drescher et al., 2016). Study plots have been established in two landscape types: well-drained areas and riparian areas. In each land-use and landscape type four 50 × 50 m study plots have been established. Additionally, four well-drained sites and four riparian sites from the Harapan Rainforest, an ecosystem restoration area, were used as references, resulting in a total number of 24 plots (Table 1). SOC data from these plots allowed for SOC stock change calculations after land-use change and an additional comparison between landscape types. Well-drained areas and SOC data are described in detail in Guillaume et al. (2015).

2.2. Sampling and analysis

We used an identical sampling design for riparian areas as well as for well-drained areas, sampling from four spatially independent plot replicates. At each replicate plot, a 1 m deep soil profile was established. The soils were classified according to the IUSS Working Group WRB, 2014 by the FAO as Acrisols with a sandy loam texture in the well-drained areas (Guillaume et al., 2015) and Gleysols, Stagnosols and stagnic Acrisols with a loamic and clayic texture in the riparian areas (Table 1). Soil samples in the riparian areas were collected in depth intervals (0–5, 5–10, 10–20, 20–30. 30–50, 50–70, 70–90, 90–100 cm). In case of horizon boundaries within the selected intervals, sampling was adapted to avoid a mixture of horizons within the sample. Sampling was evenly distributed to cover the whole depth interval. Samples for bulk density were collected in five replicates from each

Table 1
Sampling site characteristics.

Landscape type	Land use type	Local name	Geographical coordinates	Soil group	Texture	Stand age	sampling depths (cm)	Sampling year	Field replicates
riparian	Dipterocarp lowland rainforest	Hutan primer	S 02°10′24.4″E 103°21′56.1″	Gleysol	Loamic	Degraded primary (Margono et al., 2014)	0–5; 5–10; 10–20; 20–30; 30–50; 50–70; 70–90; 90–100	2016	3
			S 02°10′51.9″E 103°20′07.8″	Acrisol	Siltic				
			S 02°11′23.9″E 103°20′39.5″	Stagnosol	Clayic				
riparian	Rubber Monoculture	Kebun Karet	S 01°44′18.9″E 103°18′50.0″	Gleysol	Loamic/ Clayic	8–18 years	0–5; 5–10; 10–20; 20–30; 30–50; 50–70; 70–90; 90–100	2016	4
			S 01°53′14.3″E 103°17′29.2″	Acrisol	Clayic				
			S 01°51′42.3″E 103°18′20.4″	Acrisol	Loamic				
			S 01°42′39.6″E 103°17′23.3″	Acrisol	Loamic				
riparian	Oil Palm Monoculture	Kebun Sawit	S 01°54′07.7″E 103°22′53.3″	Stagnosol	Clayic	10–16 years	0–5; 5–10; 10–20; 20–30; 30–50; 50–70; 70–90; 90–100	2016	4
			S 01°52′40.5″E 103°21′23.0″	Stagnosol	Clayic				
			S 01°51′40.2″E 103°18′20.2″	Stagnosol	Loamic				
			S 01°42′39.5″E 103°17′31.1″	Acrisol	Clayic				
well-drained	Dipterocarp lowland rainforest	Hutan primer	S 01°54′35.6″E 103°15′58.3″	Acrisol	Sandy loam	Degraded primary (Margono et al., 2014)	By horizon and recalculated see Guillaume et al. (2015)	2012	4
			S 01°53′00.7″E 103°16′03.6″	Acrisol	Sandy loam				
			S 01°51′28.4″E 103°18′27.4″	Acrisol	Sandy loam				
			S 01°47′12.7″E 103°16′14.0″	Acrisol	Sandy loam				
well-drained	Rubber Monoculture	Kebun Karet	S 01°54′39.5″E 103°16′00.1″	Acrisol	Sandy loam	8–18 years	By horizon and recalculated see Guillaume et al. (2015)	2012	4
			S 01°52′44.5″E 103°16′28.4″	Acrisol	Sandy loam				
			S 01°51′28.4″E 103°18′27.4″	Acrisol	Sandy loam				
			S 01°48′18.2″E 103°15′52.0″	Acrisol	Sandy loam				
well-drained	Oil Palm Monoculture	Kebun Sawit	S 01°54′35.6″E 103°15′58.3″	Acrisol	Sandy loam	10–16 years	By horizon and recalculated see Guillaume et al. (2015)	2012	4
			S 01°53′00.7″E 103°16′03.6″	Acrisol	Sandy loam				
			S 01°51′28.4″E 103°18′27.4″	Acrisol	Sandy loam				
			S 01°47′12.7″E 103°16′14.0″	Acrisol	Sandy loam				

depth interval: 100 cm^3 soil rings were inserted horizontally, weighed and dried at 105 °C until constant weight. Guillaume et al. (2015) sampled well-drained soil on horizon basis. These data were merged with our data set by recalculating horizon data, based on their relative contribution to each depth interval.

Soil samples for organic C and total N as well as $\delta^{13}C$ (‰ VPDB) analyses were dried for two weeks at 40 °C, sieved to 2 mm and ground before analyses. Due to the absence of inorganic C and N, total C and N correspond to organic C and N. Measurements were performed at the Center for Stable Isotope Research and Analysis (KOSI) of the University of Göttingen using an Elemental Analyser (NA1110, CE-Instruments, Rodano, Milano, Italy) coupled via a ConFlow III to an isotope ratio mass spectrometer (Delta Plus, Finnigan MAT, Bremen, Germany). Before soil texture analysis iron oxides were removed by using sodium dithionite.

2.3. Calculations and statistics

The SOC stocks were calculated by multiplying the C content with the respective bulk density of each depth. Bulk Density data between 50 and 100 cm soil depth were only for riparian areas available to calculate SOC stock. All statistics and graphing were performed in R v3.5.1 (R Core Team, 2018) using base, agricolae (de Mendiburu, 2017), car (Fox and Weisberg, 2011), ggplot2 (Wickham, 2016), tidyverse (Wickham, 2017), ggpubr (Kassambara, 2018) and rstatix (Kassambara, 2020) packages. We excluded one riparian forest profile because of very high topsoil C content (26%) which already hinted at peat formation and, therefore, did not represent the targeted mineral soil conditions. The effects of land-use change on C and N contents, C/N ratio and $\delta^{13}C$ signature within the riparian area were assessed by a Two-Way split-plot ANOVA (p-level < 0.05) with depth as interacting factor within individual plots. *P*-values from multiple comparisons were adjusted by Bonferroni correction. The combined effects of the landscape and land-use type on soil C stocks were compared for three depth ranges (0–10, 0–30 and 0–50 cm), using a Two-Way ANOVA, at p-level < 0.05 with factor interaction. Tukey HSD post-hoc analysis was used for group comparisons. Variance between groups were homogeneous (Levene's test (p-level > 0.05). ANOVA model residuals were normally distributed (Shapiro-Wilk test, $p > 0.05$). Occasional deviations from the normality assumption of groups were small and considered acceptable, given the robustness of homoscedastic ANOVA.

3. Results

3.1. Effect of land use on riparian areas

Carbon content (g kg^{-1}) in the topsoil (0–10 cm) of riparian areas were 34% and 11% higher in the forest than in rubber and oil palm plantations, respectively (Fig. 1). Carbon content declined from topsoil to subsoil across all land-use types ($p < 0.05$). In the forest, the high C content of 28 g kg^{-1} C and 29 g kg^{-1} C at 80 and 95 cm soil depth resulted from a buried C-rich layer at one of the sites (Fig. 1).

In the first 0–30 cm of soils in riparian areas, C stocks decreased by 14% after forest conversion to rubber, and by 4% after conversion to oil palm plantations. However, in 0–10 cm depths under riparian oil palm plantations, C stocks were 20% higher than in riparian forest topsoil. Bulk density in forest topsoil was 1.2 times lower than in plantations, indicating higher soil compaction under monoculture cultivation and the absence of well-structured aggregates (Appendix Table A.1). Thus, while stocks in the three land-use types did not differ significantly from each other, they indicated a tendency towards changing soil conditions after forest conversion. The riparian forest subsoil (30–100 cm) constituted 70% of the total C stock. In oil palm and rubber plantations, subsoil stocks made up only 42% and 44% of the total C stock, respectively.

Differences in Nitrogen (N) content between the land-use types were not significant in any depth. Across all land-use types, N contents decreased with depth (Two-Way split-plot ANOVA, $p < 0.05$). Under forests, N content re-increased within 60 to 100 cm depth, associated with an increase in C content over the same depth interval (Fig. 1), reflecting paleosol formation (Fig. 2, right). In riparian areas, the topsoil C/N ratio was highest in the forests and lowest in rubber plantations, varying between 14.0 (forest) and 11.7 g N kg^{-1} (rubber). In oil palm plantations, C/N ratios slightly decreased with increasing depth (down to a minimum of 7.6). In forest subsoil, an increase up to 22.0 at 80 cm soil depth was found. C/N ratios differed ($p < 0.05$) between all three land-use types.

3.2. Effect of landscape type on soil carbon stocks

The landscape type had a significant effect on topsoil (0–10 cm) C stocks in forests (Fig. 3a). The well-drained forest stored 53% more C in its topsoil than in the riparian forest. However, C stocks in other land-use types were not affected by landscape types. The effect of landscape disappeared when including deeper soil layers (Fig. 3b, c). Including depths up to 30 cm led to a C stock increase in riparian forests of 10 Mg C ha^{-1}, compared to 0–10 cm depth. This leveled out landscape effects at all sites, also when comparing 0–50 cm soil layers.

In 10–30 cm soil depths, the riparian areas showed significantly higher C stocks than well-drained areas. At any other depth interval in the subsoil, neither ecosystem type nor the interaction of ecosystem and land-use type affected C stocks (Table 2). The variance in C stocks of riparian plantations from 0 to 30 cm soil depth were up to 10 times higher than in well-drained plantations and differed significantly from each other (Levenés-test, $p = 0.037$) (Fig. 3).

3.3. Carbon isotopic signature in riparian areas

The lowest $\delta^{13}C$ values were found in the forest topsoil's, varied between −29.3‰ and −28.1‰. Under rubber and oil palm plantations, ^{13}C signatures were on average 2.3‰ and 3.1‰ more enriched relative to the forest and ranged between −27.3‰ and −24.9‰ in rubber, and between −27.36‰ and −24.40‰ in oil palm plantations in riparian areas. The conversion of forest to monoculture tended to affect the ^{13}C signature down to 30 cm depth (Two-Way split-plot ANOVA, $p \leq 0.1$). In forest subsoils (30–100 cm), $\delta^{13}C$ values remained relatively stable with increasing depth, ranging between −30‰ and −28‰. In plantations, a heterogeneous pattern was evident: Replicates differed by up to 3.4‰ in oil palm plantations and $\delta^{13}C$ values were more negative in the subsoil than in the topsoil. Land-use effects were also present in well-drained areas. In contrast to riparian areas, well-drained areas showed a uniform increase of ^{13}C abundance with increasing depth, which is common for undisturbed mineral soils. The $\delta^{13}C$ values were lowest in the forest topsoil (-29.6‰ ± 0.2), while $\delta^{13}C$ in plantation topsoil was higher (-28.0‰ ± 0.06 under oil palm and −27.7‰ ± 0.1 under rubber cultivation). $\delta^{13}C$ in deeper soil layers aligned at −26‰, with a maximum difference of 0.4‰ between forest and rubber plantations at 35 cm soil depth and 0.5‰ between forest and oil palm plantations (Guillaume et al. 2015). The variance of $\delta^{13}C$ in the topsoil of the riparian rubber plantations was 6 times larger than in the well-drained rubber topsoil and up to 220 times larger in deep soil. The variance of $\delta^{13}C$ in soil under oil palm plantations was 77 to 220 times higher in riparian than in well-drained sites.

4. Discussion

4.1. SOC after land-use change in riparian and well-drained areas

The average C content of around 2.3% in the first 20 cm of riparian forest soils was similar to other hydromorphic mineral sites in tropical riverine forests (2.2%) (Scipioni et al., 2019). However, C content was much lower than previously reported for tropical wetland forests (6%

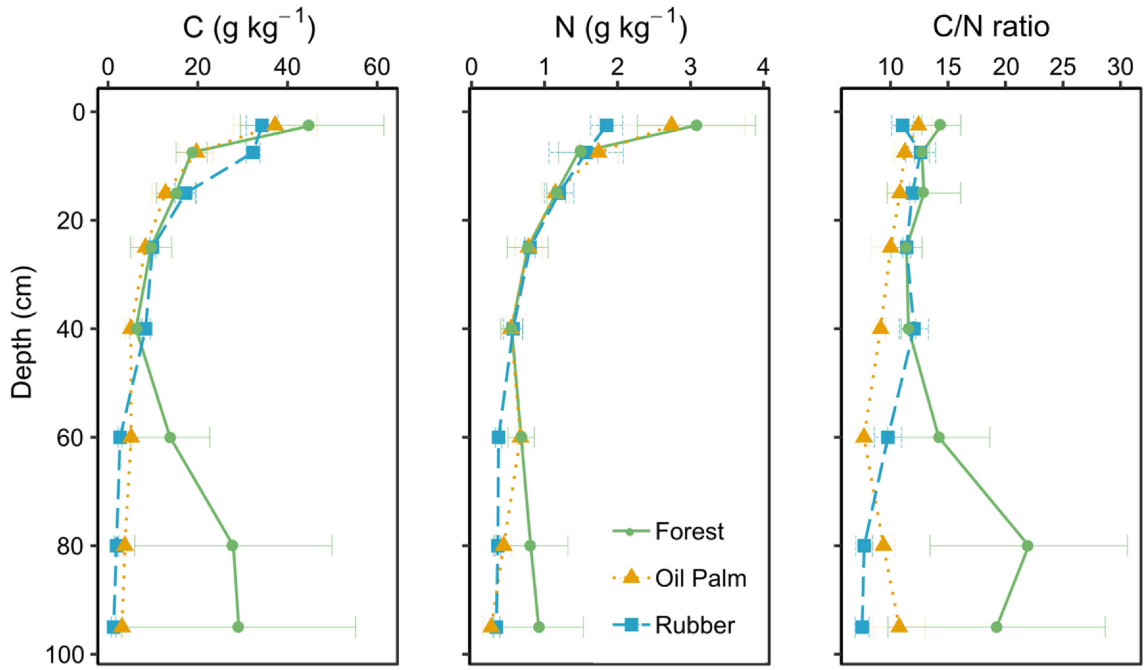

Fig. 1. Depth profiles of C content (g kg^{-1}), N content (g kg^{-1}) and C/N ratio in forest (green), rubber (blue) and oil palm (yellow) plantations. Values represent means (in forest n = 3, in plantations n = 4). Error Bars indicate the Standard Error of Mean (SEM).

Fig. 2. From left to right: Typical well-drained soil profile in the oil palm plantations. Classified as loamic Acrisol (WRB) *(left)*; Typical riparian soil profile in one of the oil palm plantations. Classified as loamic Stagnosol (WRB) *(middle)*; Forest Soil Profile with a buried layer in the riparian forest. This soil profile differs from the usual riparian Stagnosol soil type, due to the paleo soil layer leading to different diagnostic properties *(right)*.

and 35%) (Bernal and Mitsch, 2008). This results from the fact that our riparian areas contain mineral and mucky mineral soils, whereas wetlands also contain organic soils. Consequently, C contents of our riparian areas (Fig. 1) are comparable to C contents in tropical mineral soils that are not associated with wetland conditions (e.g. Chiti et al., 2014; de Blécourt et al., 2013; Guillaume et al., 2015). In temperate riparian forests similar ranges of 1.7 to 3.5% C content were found (Graf-Rosenfellner et al., 2016).

The conversion from riparian forests to riparian plantations led to a small decrease in SOC contents in the upper 30 cm, whereas large reductions of between 35% and 40% under rubber cultivation on upland mineral soils have been reported (Chiti et al., 2014; Guillaume et al., 2015). Forest transformations to oil palm plantations are often accompanied by average SOC losses between 18% and 45% (Chiti et al., 2014; Guillaume et al., 2018; Rahman et al., 2018; van Straaten et al., 2015). All these studies were conducted in well-drained areas; however, our results indicate that riparian C stocks are not that strongly affected by land-use change.

C content and stocks tend to higher values in deeper soil in riparian areas. The higher C contents in the riparian forest subsoil were likely due to buried C-rich layers, which were obvious (Fig. 2). This agrees with findings of Shakhmatova and Korsunov (2008) who have described alluvial subsoil layers with strong C accumulation that exceed the topsoil C content. Further, abrupt changes in soil particle size in some profiles e.g. from 4% sand at 15 cm soil depth to 40% sand at 25 cm (under rubber cultivation), hint at a shift in the organic material source and confirm a spontaneous deposition event.

There is a trend to higher soil C stocks in riparian areas than in well-drained areas. Subsoil stocks (30–100 cm) in the riparian areas show high C stocks and reflect delayed mineralization under periodic anoxic conditions, especially in the forest with the buried layer (Gudasz et al., 2015; Hounkpatin et al., 2018; Sobek et al., 2009). High C accumulation could be further explained by the finer texture of subsoil common in Stagnosols (Zech et al., 2014). Therefore, the increasing subsoil stocks in plantations are more likely a consequence of riparian properties (including e.g. waterlogging and sediment transport), than of land-use change alone. Most of the soils in the investigated riparian areas, especially in plantations, were only periodically water-logged and consequently, in contrast to many wetland soils, anoxic conditions were only temporarily present. We suggest that this leads to lower C contents and stocks than we expected from other wetland ecosystems in these climatic zones (Wantzen et al., 2012).

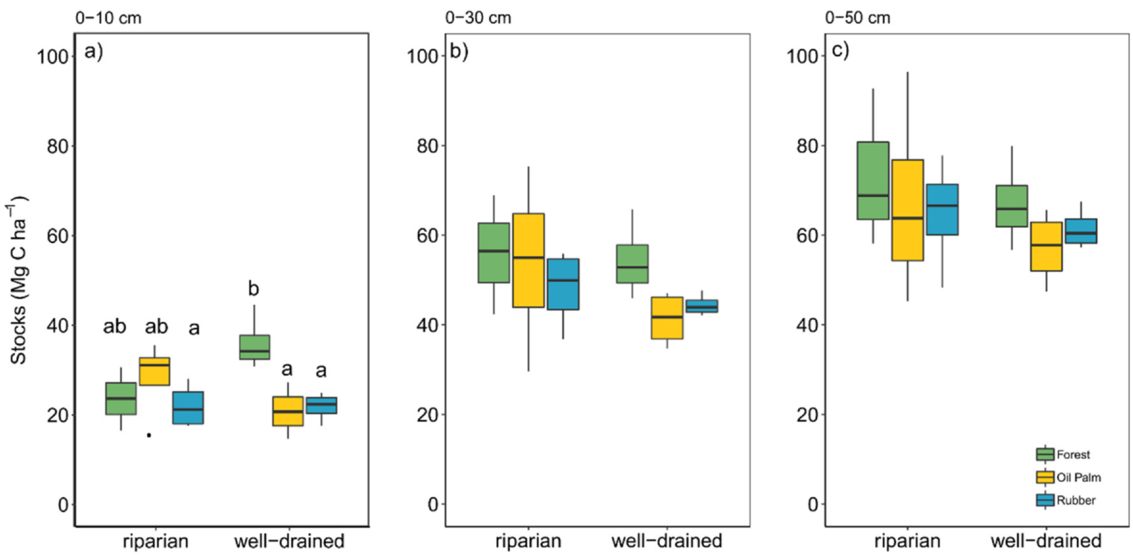

Fig. 3. Carbon stocks of riparian and well-drained areas at 0–10, 0–30 and 0–50 cm soil depth. Different letters between groups indicate significant differences according to Two-Way ANOVA ($p < 0.05$).

Continuously flooded wetland ecosystems can only be used for rubber or oil palm plantations after drainage, an economically expensive and work-intensive process. In contrast, seasonal drying of riparian sites allows for easier conversion into plantations. Such sites are therefore much more likely to be subjected to large-scale land-use conversions and thus are the warrant closer study. Similar C stocks are reported by Rahman et al. (2018) who found ~50 Mg C ha^{-1} in the upper 30 cm of a forest mineral soil in Borneo/Malaysia. Continuously flooded soils are favorable for peat formation and C stocks can become higher, as we found in one riparian forest plot, which was subsequently excluded from this data set. SOC stocks in the upper 30 cm of soil in riparian forests (Fig. 3) were lower than in published studies from tropical undisturbed wetland sites, e.g. 90.2 Mg C ha^{-1} and 67.0 Mg C ha^{-1} in the Amazon Basin (Wantzen et al., 2012) and 80 Mg C ha^{-1} in the upper 24 cm of a wetland in Costa Rica (Powers, 2004).

4.2. *δ^{13}C elucidation of soil processes*

Using δ^{13}C values Guillaume et al. (2015) estimated 15–20 cm of erosive soil loss from well-drained sites within 17 years after land-use change. They assumed that C content and δ^{13}C values in the plantationś subsoil were similar to the forest subsoil prior to conversion. They deduced that after the erosional loss of the upper layer, subsoil C content and δ^{13}C values were vertically shifted towards the surface (Fig. 5). The δ^{13}C values in the riparian forest topsoiĺs were on average 2.5‰ lower than in the plantations, indicating that more enriched C reaches the soil surface after conversion, induced by soil material transportation. This difference corresponds to Guillaume et al. (2015) who observed a 2‰ increase of δ^{13}C values in the topsoil between forest and plantations. Riparian forests and plantations differ strongly in their isotopic signatures in the top- and the subsoil. Forests show more negative δ^{13}C values with a more uniform depth profile than under plantations. Single replicates in the plantation's sites show a strong erratic pattern with increasing soil depth (Appendix, Fig. A.4), which hints at erosion and deposition bringing organic matter from various sources to the plantations (Davies et al., 2012) but the average values reveal a more homogenous δ^{13}C depth profile.

The topsoiĺs of well-drained plantations (particularly oil palm) were more depleted in ^{13}C isotopes compared to topsoiĺs of riparian plantations (Fig. 4). This contradicts the assumption that material from the elevated well-drained sites is steadily deposited on the lower riparian plantations. Since well-drained sites are located upstream and are heavily eroded (Guillaume et al., 2015), sedimentation on riparian sites in lower reaches would likely align their topsoil δ^{13}C values. Instead, the riparian δ^{13}C depth profiles indicate a more complex source-sink relationship with other processes of C dynamics in the watershed. Therefore, we assume that sedimentation is a secondary pathway for topsoil C inputs in riparian plantations, whose actual contribution and process relationships require further investigation.

We conclude that alternating oxic and anoxic conditions mainly drive the formation of the δ^{13}C depth profiles in riparian plantations. This is supported by the following: first, the riparian forest shows a uniform δ^{13}C pattern (Fig. 4a), which is common under constant water-saturated conditions, where anaerobic decomposition processes are dominant (Alewell et al., 2011; Krüger et al., 2014). This interpretation is supported by studies that reported more heterogeneous patterns, indicating alternating wet and dry cycles (Broder et al., 2012; Loisel et al., 2009). Second, the δ^{13}C depth profile found in rubber plantations hints at aerobic decomposition down to 15 cm, which might be caused by land-use change where the topsoil has been well-drained during

Table 2
Carbon stocks in riparian and well-drained sites (Mg C ha^{-1}) values in brackets represent Standard Error of Means (SEM).

	riparian			well-drained‡		
Land use	0–10 cm ±	10–30 cm ±	30–50	0–10 cm ±	10–30 cm ±	30–50
Forest	23.4 (4.1)^{ab}	32.7 (6.9)	130.1(84.1)	35.8 (3.0)^{b}	18.6 (1.8)	12.7 (0.7)
Rubber	21.8 (2.5)^{a}	26.3 (2.5)	37.5 (2.5)	21.6 (1.6)^{a}	22.8 (2.0)	17.0 (2.0)
Oil Palm	28.1 (4.4)^{ab}	25.6 (6.0)	39.6 (3.9)	20.7 (2.7)^{a}	20.6 (1.8)	15.9 (1.2)

‡ C stocks already published in Guillaume et al. (2015).
± and superscripts represent significant differences according to Two-Way ANOVA with factor interaction ($p < 0.05$) between riparian and well-drained areas at 0–10 and 10–30 cm soil depth.

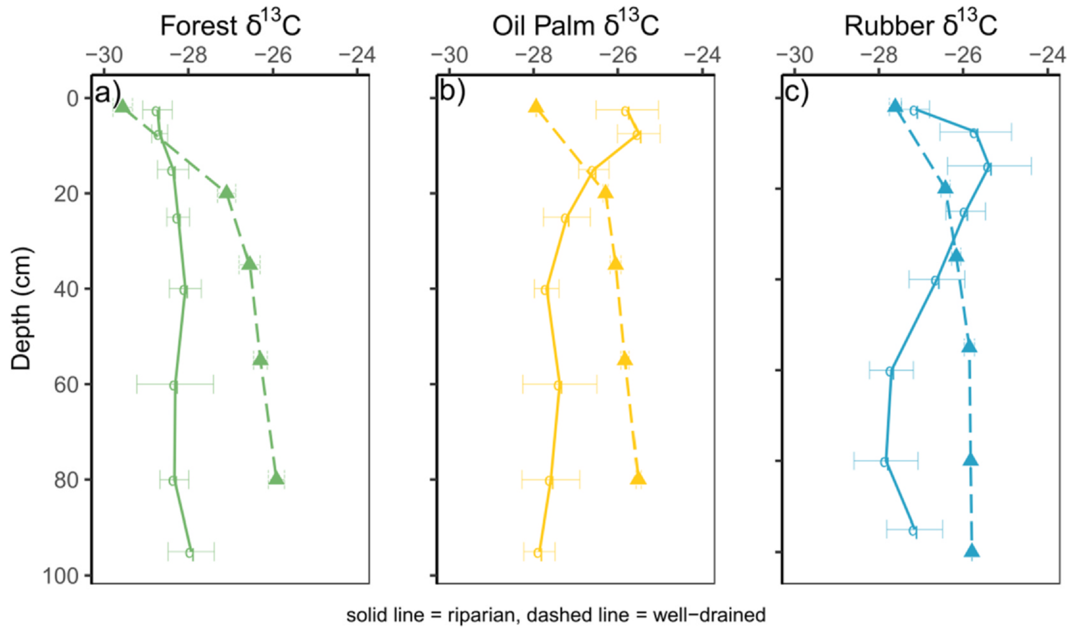

Fig. 4. δ^{13}C depth profiles in riparian (solid lines) and well-drained soils (dashed lines). δ^{13}C values at fixed depth represent means (in forest n = 3, in plantations n = 4). Error bars indicate the Standard Error of Mean (SEM).

conversion. The turning point to lower δ^{13}C at 25 cm values indicates a change to saturated water conditions, and therefore a change to anaerobic decomposition and enrichment of recalcitrant material (Alewell et al., 2011). Similar δ^{13}C depth profile patterns are common in peatlands (Drollinger et al., 2019; Krüger et al., 2015). In agreement with Alewell et al. (2011) and Drollinger et al. (2019), we suggest a preservation of ^{13}C-depleted substances that decompose slowly under anoxic conditions, such as lignin and lipids. Third, the oil palm plantatioń's δ^{13}C depth profile shows a turning point at 7.5 cm depth. Since land-use change primarily affects the upper 30–40 cm, δ^{13}C values in riparian and plantatioń's subsoils should be similar, as it was clearly shown for well-drained areas (Fig. 4).

Another reason for the higher proportion of enriched ^{13}C in riparian plantations is the limited input of C from aboveground litter providing, less easily available material for decomposers. Microorganisms in the topsoil of riparian plantations have to consume strongly processed C fractions with high ^{13}C enrichment (Fig. 4). In contrast to the erosion of well-drained plantations, soil C losses in riparian plantations are mainly controlled by mineralization processes, with potential additional fluvial erosion–deposition effects. The effect of accelerated mineralization under agricultural use extends over a longer time period and may already be detectable in the very accurate measurable δ^{13}C values but not yet significant changes of the C stock estimates, which suffer from combined variation of bulk density and C content values. Thus, the

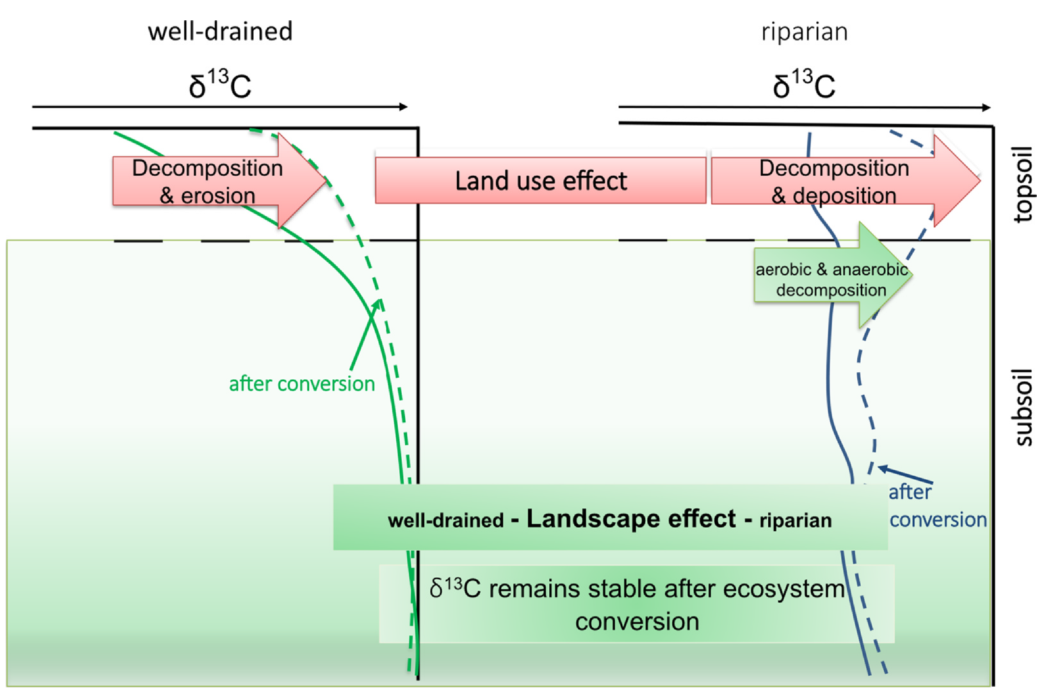

Fig. 5. Impact of riparian forest conversion in combination with specific ecosystem characteristics on δ^{13}C distributions in the soil depth, separated in decomposition and erosion effects in well-drained areas (left) and decomposition and deposition as well as alternating oxic and anoxic conditions in the riparian areas (right).

immediate erosion after converting natural forests to plantations leads to a strong decline of C stocks on well-drained mineral soils, but not in riparian plantations although the riparian plantations have similar age. Hence, C stocks in riparian areas respond more slowly to land-use change due to 1) their overall slower mineralization rates under flooded conditions and 2) less erosion and partial deposition (Fig. 5).

5. Conclusions

Land-use change in tropical regions has severe impacts on soil C stocks, due to erosion and enhanced mineralization. However, the specific geomorphic conditions in riparian areas counter these land-use effects through two dominant processes:

(1) C preservation due to oxygen-limited mineralization under alternating oxic and anoxic conditions and
(2) input and accumulation of allochthonous organic materials with various decomposition degree.

These dynamics of seasonal flooding followed by delayed mineralization are reflected by the heterogeneous $\delta^{13}C$ pattern. Compared to well-drained areas, riparian areas are more resilient to short-term (1–2 decades) soil C loss after land-use change, as topsoil layers show similar C stocks compared to that under natural vegetation. However, accelerated mineralization in the topsoil indicates possible long-term effects on C storage. Riparian areas, especially if drained, have high C loss potential, considering the high amounts of subsoil C in their buried C-rich layers. Our study shows, that further research is required to deepen the understanding of the role of riparian areas in global C storage and the vulnerability of these C stocks under the combined effect of land-use change and riparian properties.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Declaration of Competing Interest

The study has not been published and is not under consideration for publication somewhere else. The manuscript has been approved by all authors and no conflict of interest was stated by any author.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2020.104941.

References

Alewell, C., Giesler, R., Klaminder, J., Leifeld, J., Rollog, M., 2011. Stable carbon isotopes as indicators for environmental change in palsa peats. Biogeosciences 8, 1769–1778. https://doi.org/10.5194/bg-8-1769-2011.
Andreeva, D.B., Zech, M., Glaser, B., Erbajeva, M.A., Chimitdorgieva, G.D., Ermakova, O.D., Zech, W., 2013. Stable isotope (δ13C, δ15N, δ18O) record of soils in Buryatia, southern Siberia: Implications for biogeochemical and paleoclimatic interpretations. Quat. Int. 290–291, 82–94. https://doi.org/10.1016/j.quaint.2012.10.054.
Austin, K.G., Schwantes, A., Gu, Y., Kasibhatla, P.S., 2019. What causes deforestation in Indonesia? Environ. Res. Lett. 14, 24007. https://doi.org/10.1088/1748-9326/aaf6db.
Becker, J., Pabst, H., Mnyonga, J., Kuzyakov, Y., 2015. Annual litterfall dynamics and nutrient deposition depending on elevation and land use at Mt. Kilimanjaro. Biogeosciences 12, 5635–5646. https://doi.org/10.5194/bg-12-5635-2015.
Becker, J.N., Kuzyakov, Y., 2018. Teatime on Mount Kilimanjaro: assessing climate and land-use effects on litter decomposition and stabilization using the Tea Bag Index. L. Degrad. Dev. 29, 2321–2329. https://doi.org/10.1002/ldr.2982.
Bendix, J., Hupp, C.R., 2000. Hydrological and geomorphological impacts on riparian plant communities. Hydrol. Process. 14, 2977–2990. https://doi.org/10.1002/1099-1085(200011/12)14:16/17 < 2977::AID-HYP130 > 3.0.CO;2-4.
Bernal, B., Mitsch, W.J., 2008. A comparison of soil carbon pools and profiles in wetlands in Costa Rica and Ohio. Ecol. Eng. 34, 311–323. https://doi.org/10.1016/j.ecoleng.2008.09.005.
Borchard, N., Bulusu, M., Meyer, N., Rodionov, A., Herawati, H., Blagodatsky, S., Cadisch, G., Welp, G., Amelung, W., Martius, C., 2019. Deep soil carbon storage in tree-dominated land use systems in tropical lowlands of Kalimantan. Geoderma 354, 113864. https://doi.org/10.1016/j.geoderma.2019.07.022.
Broder, T., Blodau, C., Biester, H., Knorr, K.H., 2012. Peat decomposition records in three pristine ombrotrophic bogs in southern Patagonia. Biogeosciences 9, 1479–1491. https://doi.org/10.5194/bg-9-1479-2012.
Brown, S., Brinson, M. M., Lugo, A.E., 1978. Structure and Function of Riparian Wetlands, in: Johnson, R. R., McCormick, J.F. (Ed.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. U.S. Department of Agriculture. Forest Service, Washington D.C., pp. 17–31.
Chiti, T., Grieco, E., Perugini, L., Rey, A., Valentini, R., 2014. Effect of the replacement of tropical forests with tree plantations on soil organic carbon levels in the Jomoro district, Ghana. Plant Soil 375, 47–59. https://doi.org/10.1007/s11104-013-1928-1.
Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Quéré, C. Le, Myneni, R.B., Piao, S., Thornton, P., 2013. Carbon and other biogeochemical cycles, Climate Change 2013 the Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. https://doi.org/10.1017/CBO9781107415324.015.
Craig, H., 1953. The geochemistry of the stable carbon isotopes. Geochim. Cosmochim. Acta. https://doi.org/10.1016/0016-7037(53)90001-5.
Davies, S.J., Leng, M.J., MacQuaker, J.H.S., Hawkins, K., 2012. Sedimentary process control on carbon isotope composition of sedimentary organic matter in an ancient shallow-water shelf succession. Geochem. Geophys. Geosystems 13, 1–15. https://doi.org/10.1029/2012GC004218.
de Blécourt, M., Brumme, R., Xu, J., Corre, M.D., Veldkamp, E., 2013. Soil carbon stocks decrease following conversion of secondary forests to rubber (Hevea brasiliensis) plantations. PLoS ONE 8. https://doi.org/10.1371/journal.pone.0069357.
de Junet, A., Abril, G., Guérin, F., Billy, I., de Wit, R., 2005. Sources and transfers of particulate organic matter in a tropical reservoir (Petit Saut, French Guiana): a multi-tracers analysis using delta;^{13}C C/N ratio and pigments. Biogeosciences Discuss. 2, 1159–1196. https://doi.org/10.5194/bgd-2-1159-2005.
de Mendiburu, F., 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-8. https://doi.org/https://cran.r-project.org/package=agricolae.
Décamps, H, Pinay, G, Naiman RH, Petts, GE, McClain, ME, Hillerecht-Ilkowska, A, Hanley, TA, Holmes, RM, Quinn, J, Gibert, J, Planty Tabacchi, A-M, Schiemer, F, Tabacchi, E, Zalewski, M., 2004. Riparian zones: where biogeochemistry meets biodiversity in management practice. Polish J. Ecol.
Don, A., Schumacher, J., Freibauer, A., 2011. Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. Glob. Chang. Biol. https://doi.org/10.1111/j.1365-2486.2010.02336.x.
Drescher, J., Rembold, K., Allen, K., Beckschäfer, P., Buchori, D., Clough, Y., Faust, H., Fauzi, A.M., Gunawan, D., Hertel, D., Irawan, B., Jaya, I.N.S., Klarner, B., Kleinn, C., Knohl, A., Kotowska, M.M., Krashevska, V., Krishna, V., Leuschner, C., Lorenz, W., Meijide, A., Melati, D., Nomura, M., Pérez, K.H., S.S., 2016. Ecological and socio-economic functions across tropical land use systems after rainforest conversion. Philos. Trans. R. Soc. B Biol. Sci. 371, 20150275. https://doi.org/10.1098/rstb.2015.0275.
Drollinger, S., Kuzyakov, Y., Glatzel, S., 2019. Effects of peat decomposition on δ 13 C and δ 15 N depth profiles of Alpine bogs. Catena 178, 1–10. https://doi.org/10.1016/j.catena.2019.02.027.
Fox, J. Weisberg, S., 2011. An {R} Companion to Applied Regression, Third Edition. Thousands Oaks CA:Sage. https://doi.org/https://socialsciences.mcmaster.ca/jfox/Books/Companion/.
Graf-Rosenfellner, M., Cierjacks, A., Kleinschmit, B., Lang, F., 2016. Soil formation and its implications for stabilization of soil organic matter in the riparian zone. Catena 139, 9–18. https://doi.org/10.1016/j.catena.2015.11.010.
Gudasz, C., Sobek, S., Bastviken, D., Koehler, B., Tranvik, L.J., 2015. Temperature sensitivity of organic carbon mineralization in contrasting lake sediments. J. Geophys.

Res. G Biogeosciences 120, 1215–1225. https://doi.org/10.1002/2015JG002928.
Guillaume, T., Damris, M., Kuzyakov, Y., 2015. Losses of soil carbon by converting tropical forest to plantations: erosion and decomposition estimated by δ13C. Glob. Chang. Biol. 21, 3548–3560. https://doi.org/10.1111/gcb.12907.
Guillaume, T., Kotowska, M.M., Hertel, D., Knohl, A., Krashevska, V., Murtilaksono, K., Scheu, S., Kuzyakov, Y., 2018. Carbon costs and benefits of Indonesian rainforest conversion to plantations. Nat. Commun. 9. https://doi.org/10.1038/s41467-018-04755-y.
Guyette, R.P., Cole, W.G., Dey, D.C., Muzika, R., 2002. Perspectives on the age and distribution of large wood in riparian carbon pools. Can. J. Fish. Aquat. Sci. 59, 578–585. https://doi.org/10.1139/f02-026.
Harris, N.L., Brown, S., Hagen, S.C., Saatchi, S.S., Petrova, S., Salas, W., Hansen, M.C., Potapov, P.V., Lotsch, A., 2012. Baseline map of carbon emissions from deforestation in tropical regions. Science (80-.). 336, 1573–1576. https://doi.org/10.1126/science.1217962.
Hazlett, P.W., Gordon, A.M., Sibley, P.K., Buttle, J.M., 2005. Stand carbon stocks and soil carbon and nitrogen storage for riparian and upland forests of boreal lakes in northeastern Ontario. For. Ecol. Manage. 219, 56–68. https://doi.org/10.1016/j.foreco.2005.08.044.
Hounkpatin, O.K.L., Op de Hipt, F., Bossa, A.Y., Welp, G., Amelung, W., 2018. Soil organic carbon stocks and their determining factors in the Dano catchment (Southwest Burkina Faso). Catena 166, 298–309. https://doi.org/10.1016/j.catena.2018.04.013.
IUSS Working Group WRB, 2014. International soil classification system for naming soils and creating legends for soil maps, World Reference Base for Soil Resources 2014, update 2015. https://doi.org/10.1017/S0014479706394902.
Kassambara, A., 2020. rstatix: Pipe-friendly Framework for Basic Statistical Tests. https://doi.org/https://CRAN.R-project.org/package=rstatix.
Kassambara, A., 2018. ggpubr: 2 "ggplot2" Based Publication Ready Plots. https://doi.org/https://cran.r-project.org/package=ggpubr.
Kelleway, J.J., Saintilan, N., Macreadie, P.I., Baldock, J.A., Ralph, P.J., 2017. Sediment and carbon deposition vary among vegetation assemblages in a coastal salt marsh. Biogeosciences 14, 3763–3779. https://doi.org/10.5194/bg-14-3763-2017.
Kotowska, M.M., H.D., Triadiati T, Selis M, L.C., 2015. Quantifying above- and below-ground biomass carbon loss with forest conversion in tropical lowlands of Sumatra (Indonesia). Glob. Chang. Biol. 21, 3620–3634. https://doi.org/10.1111/gcb.12979.
Krüger, J.P., Leifeld, J., Alewell, C., 2014. Degradation changes stable carbon isotope depth profiles in palsa peatlands. Biogeosciences 11, 3369–3380. https://doi.org/10.5194/bg-11-3369-2014.
Krüger, J.P., Leifeld, J., Glatzel, S., Szidat, S., Alewell, C., 2015. Biogeochemical indicators of peatland degradation – a case study of a temperate bog in northern Germany. Biogeosciences 12, 2861–2871. https://doi.org/10.5194/bg-12-2861-2015.
Labrière, N., Locatelli, B., Laumonier, Y., Freycon, V., Bernoux, M., 2015. Soil erosion in the humid tropics: a systematic quantitative review. Agric. Ecosyst. Environ. https://doi.org/10.1016/j.agee.2015.01.027.
Laumonier, Y., 1997. The Vegetation and Physiography of Sumatra. Springer Netherlands, Dordrecht. https://doi.org/10.1007/978-94-009-0031-8_1.
Loisel, J., Garneau, M., Hélie, J.F., 2009. Modern Sphagnum δ13C signatures follow a surface moisture gradient in two boreal peat bogs, James Bay lowlands. Québec. J. Quat. Sci. 24, 209–214. https://doi.org/10.1002/jqs.1221.
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. https://doi.org/10.1038/nclimate2277.
McCormick, J.F., 1978. A summary ofthe national riparian symposium., in: Johnson, R.R. McCormick, J.F. (Ed.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. Washington D.C., pp. 362–363.
Moore, S., Evans, C.D., Page, S.E., Garnett, M.H., Jones, T.G., Freeman, C., Hooijer, A., Wiltshire, A.J., Limin, S.H., Gauci, V., 2013. Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. Nature 493, 660–663. https://doi.org/10.1038/nature11818.
Powers, J.S., 2004. Changes in soil carbon and nitrogen after contrasting land-use transitions in northeastern Costa Rica. Ecosystems 7, 134–146. https://doi.org/10.1007/s10021-003-0123-2.
R Core Team, 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://doi.org/https://www.R-project.org/.
Rahman, N., De Neergaard, A., Magid, J., Van De Ven, G.W.J., Giller, K.E., Bruun, T.B., 2018. Changes in soil organic carbon stocks after conversion from forest to oil palm plantations in Malaysian Borneo. Environ. Res. Lett. 13. https://doi.org/10.1088/1748-9326/aade0f.
Rieger, I., Lang, F., Kowarik, I., Cierjacks, A., 2014. The interplay of sedimentation and carbon accretion in riparian forests. Geomorphology 214, 157–167. https://doi.org/10.1016/j.geomorph.2014.01.023.
Scharlemann, J.P.W., Tanner, E.V.J., Hiederer, R., Kapos, V., Pw, J., Tanner, E.V.J., Hiederer, R., Kapos, V., 2014. Global soil carbon : understanding and managing the largest terrestrial carbon pool Global soil carbon : understanding and managing the largest terrestrial carbon pool 3004. https://doi.org/10.4155/cmt.13.77.
Scipioni, M.C., de Araújo Pedron, F., Longhi, S.J., Galvão, F., Budke, J.C., Schneider, P.R., 2019. Natural channeling in riverine forests determines variations in their floristic composition, structure, and land use in southern Brazil. Landsc. Ecol. Eng. 15, 347–362. https://doi.org/10.1007/s11355-019-00385-8.
Shakhmatova, E.Y., Korsunov, V.M., 2008. Buried humus horizons of flooded soils of the Selenga river delta. Geogr. Nat. Resour. 29, 338–342. https://doi.org/10.1016/j.gnr.2008.10.012.
Sobek, S., Durisch-Kaiser, E., Zurbrugg, R., Wongfun, N., Wessels, M., Pasche, N., Wehrli, B., 2009. Organic carbon burial efficiency in lake sediments controlled by oxygen exposure time and sediment source. Limnol. Oceanogr. 54, 2243–2254. https://doi.org/10.4319/lo.2009.54.6.2243.
van Straaten, O., Corre, M.D., Wolf, K., Tchienkoua, M., Cuellar, E., Matthews, R.B., Veldkamp, E., 2015. Conversion of lowland tropical forests to tree cash crop plantations loses up to one-half of stored soil organic carbon. Proc. Natl. Acad. Sci. 112, 9956–9960. https://doi.org/10.1073/PNAS.1504628112.
Wantzen, K.M. Yule, Catherine, M., Trockner Klement, J.W.J., 2008. Riparian Wetlands of Tropical Streams. In: Dudgeon, D. (Ed.), Tropical Stream Ecology. Elsevier Inc., Academic Press, pp. 3035–3044. https://doi.org/10.1016/B978-008045405-4.00352-9.
Wantzen, K.M., Couto, E.G., Mund, E.E., Amorim, R.S.S., Siqueira, A., Tielbörger, K., Seifan, M., 2012. Soil carbon stocks in stream-valley-ecosystems in the Brazilian Cerrado agroscape. Agric. Ecosyst. Environ. 151, 70–79. https://doi.org/10.1016/j.agee.2012.01.030.
Wickham, H., 2017. tidyverse: Easily Install and Load the "Tidyverse". https://doi.org/https://CRAN.R-project.org/package=tidyverse.
Wickham, H., 2016. ggplot2: Elegant Graphics for Data Analysis. Springer Verlag, New York.
Zang, H., Blagodatskaya, E., Wen, Y., Xu, X., Dyckmans, J., Kuzyakov, Y., 2018. Carbon sequestration and turnover in soil under the energy crop Miscanthus: repeated 13 C natural abundance approach and literature synthesis. GCB Bioenergy 10, 262–271. https://doi.org/10.1111/gcbb.12485.
Zech, W., Schad, P., Hintermaier-Erhard, G., 2014. Böden der Welt. Ein Bildatlas, Second Edi. ed. Springer Spektrum, Berlin, Heidelberg. https://doi.org/https://doi.org/10.1007.
Zehetner, F., Lair, G.J., Gerzabek, M.H., 2009. Rapid carbon accretion and organic matter pool stabilization in riverine floodplain soils. Global Biogeochem. Cycles 23, 1–7. https://doi.org/10.1029/2009GB003481.