Impacts of land use and biophysical properties on soil carbon stocks in southern Yunnan, China

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Summary

For the montane regions of mainland Southeast Asia (Southwest China, Laos, Cambodia, Myanmar, northeast Thailand, and northwest Vietnam) few data is available on soil organic carbon (SOC) stocks in relation to land use, and biophysical properties. For example, despite the regions widespread deforestation for rubber plantations, the impact of this land-use change on SOC stocks is not well understood. Also, it has not yet been studied how terrace construction, a typical phenomenon in rubber plantations in mountainous terrains, affects SOC dynamics. Furthermore, data on the spatial distribution of SOC stocks and the role of potential regulating factors like land use, vegetation, soil texture and topography is limited. This thesis focused on the above listed data gaps and presents three studies conducted in two mountainous landscapes in Xishuangbanna, southern Yunnan, China.

In the first two studies conducted in a tropical landscape, I quantified the changes in SOC stocks (1) due to the conversion from secondary forests to rubber plantations, and (2) caused by terrace construction. In the first study I quantified land-use change effects on SOC stocks using a space-for-time substitution approach. I sampled 11 terraced rubber plantations ranging in age from 5 to 46 years and seven secondary forest plots. The results demonstrated that clearing secondary forests for rubber plantations caused a reduction in SOC stocks by 37.4 Mg C ha^{-1} down to a depth of 1.2 m, which was equal to 19% of the initial SOC stocks in the secondary forests. In the topsoil the changes in SOC stocks followed an exponential decay function; the decline in SOC stocks was strongest in the first 5 years following plantation establishment and a steady state was reached after approximately 20 years. The mean loss in total SOC stocks of 37.4 Mg C ha^{-1} was much larger than the literature-based estimates of changes in above-ground carbon stocks which ranged from a loss of 18 Mg C ha^{-1} to an increase of 8 Mg C ha^{-1}. In contrast to the IPCC tier 1 method, which assumes that SOC changes caused by forest-to-rubber plantation conversions are zero, these findings illustrate that SOC stock losses should be included to avoid potential large errors in estimates of the total ecosystem carbon fluxes.

Terraced rubber plantations typically consist of narrow terraces, with a single row of trees, alternated by original sloping areas. Manual terrace construction, involves cutting the soil from the upper slope to create the terrace's inner edge (cut section), and piling up the removed soil at the down slope position which forms the outer edge of the terrace (fill section). The second study focused on the impacts of terrace construction on SOC stocks in three rubber plantations aged 5, 29, and 44 years old. In each plantation I systematically sampled the terraces according to soil redistribution zones, using the original sloping areas between the terraces as reference. The results

showed that terracing did not affect SOC stocks in the 5-year old plantation. However, in the 29-year and 44-year old plantations the terraces had higher SOC stocks down to 1.2 m compared to the non-terraced reference positions. The positive effect of terracing on SOC stocks in the two oldest plantations was attributed to the observed recovery of SOC stocks in the exposed subsurface soils at the terrace's cut section, and the observed partial preservation of SOC in the buried soil at the terrace's fill section. The recovery of SOC stocks at the cut section in the two oldest plantations was explained by the capacity of the exposed subsurface soil to store new SOC inputs from roots and litter, and by the sedimentation of eroded topsoil material from the upper slopes. Overall, the results of this case study indicate that terracing may reduce the SOC stock losses; without the terraces the SOC stock losses caused by forest conversion to terraced rubber plantations could have been higher.

In the third study, conducted in a subtropical landscape, I quantified the present SOC stocks of the dominant land-use types, and determined the relationships of SOC with land-use types, vegetation, soil texture, and topography. In an area of 10,000 hectares, I selected 28 one-hectare plots, including plots in closed canopy forests, open canopy forests, tea plantations, and shrub lands. The SOC stocks to a depth of 0.9 m were among the highest in the region: 228.6 ± 19.7 (SE) Mg C ha^{-1} in closed canopy forests, 200.4 ± 15.5 Mg C ha^{-1} in open canopy forests, 197.5 ± 25.9 Mg C ha^{-1} in tea plantations, and 236.2 ± 13.7 Mg C ha^{-1} in shrub lands. SOC concentrations and stocks did not differ significantly between land-use types. More than 50% of the overall variance in SOC occurred within the one-hectare sampling plots, and was related to the variation in tree basal area, litter layer carbon stock, and slope gradient. These findings illustrate the importance of local processes on the overall variability of SOC in a mountainous landscape. Overall, the results from these three studies contribute to an improved knowledge on SOC stocks and dynamics in a rapidly changing region, and may serve as a basis for studies on the changes in ecosystem services in montane mainland Southeast Asia.

Zusammenfassung

Für die montanen Regionen kontinental Südostasiens (Südwest China, Laos, Kambodscha, Myanmar, Nordost Thailand, Nordwest Vietnam) gibt es nur wenig Informationen über die organische Bodensubstanz (OBS) und ihre Beeinflussung durch Landnutzung, Bewirtschaftung und biophysikalische Eigenschaften. Zum Beispiel ist trotz großflächiger Entwaldung zu Gunsten von Kautschukplantagen der Einfluss dieser Landnutzungsänderung auf OBS Vorräte kaum bekannt. Auch wurde der Einfluss der Terrassierung, wie sie für den Kautschukanbau in montanen Regionen üblich ist, auf die Dynamik der OBS bislang nicht untersucht. Des Weiteren liegen nur begrenzt Informationen über die räumliche Verteilung von OBS Vorräten und die Rolle potentieller Regulationsfaktoren wie Landnutzung, Vegetation, Bodentextur und Topographie vor. Die vorliegende Arbeit zielte auf die genannten Wissenslücken und präsentiert in diesem Kontext drei Studien aus der montanen Region Xishuangannas, Süd Yunnan, China.

In den ersten beiden, in einer tropischen Landschaft durchgeführten Studien, habe ich die Änderung des OBS Vorrats durch 1) die Umwandlung von Sekundärwald in Kautschukplantagen und 2) durch den Bau von Terrassen, quantifiziert. Um in der ersten Studie Landnutzungseffekte auf die OBS-Vorräte zu quantifizieren, habe ich den Ansatz der unechten Zeitreihe (space-for-time substitution) genutzt. Ich habe 11 terrassierte Kautschukplantagen im Alter von 5 bis 46 Jahren sowie sieben Sekundärwaldparzellen untersucht. Die Ergebnisse zeigten, dass die Umwandlung von Sekundärwald in Kautschukplantagen eine Abnahme der OBS Vorräte von 37.4 Mg C ha^{-1} im Bereich bis zu einer Tiefe von 1.2 m hervorrief; diese Abnahme entsprach 19% des ursprünglichen OBS Vorrats im Sekundärwald. Im Oberboden nahm der OBS Vorrat exponentiell ab; in den ersten 5 Jahren nach der Landnutzungsänderung war die Abnahme am stärksten, nach ca. 20 Jahren hat sich ein Gleichgewicht eingestellt. Der mittlere OBS-Verlust von 37.4 Mg C ha^{-1} war viel höher als literaturbasierte Schätzwerte für Änderungen der oberirdischen Kohlenstoffvorräte, welche zwischen einem Verlust von 18 Mg C ha^{-1} und einer Steigerung von 8 Mg C ha^{-1} liegen. Im Gegensatz zur IPCC tier 1-Methode, die davon ausgeht, dass OBS Vorratsänderungen bei einer Umwandlung von Wald zu Kautschuk gleich 0 sind, zeigen meine Ergebnisse, dass OBS-Verluste in Betracht gezogen werden müssen, um potentiell große Fehler bei der Schätzung von Kohlenstoffflüssen von Ökosystemen zu vermeiden.

Terrassierte Kautschukplantage bestehen aus schmalen Terrassen mit einer Baumreihe, die sich mit ursprünglichen geneigten Flächen abwechseln. Bei der Konstruktion der Terrassen wird Boden vom Hang abgetragen, und so eine innere Kante der Terrasse entsteht (Entnahmebereich); der entfernte Boden wird dann auf den Hang unterhalb der Grabungsfläche aufgehäuft und bildet die äußere Kante der Terrasse (Ablagebereich). Die zweite Studie untersucht den Einfluss der Terrassierung auf OBS Vorräte in 5, 29 und 44 Jahre alten Plantagen. In jeder Plantage habe ich die Terrassen systematisch in den verschiedenen Bodenverteilungszonen beprobt, die ursprünglichen Hangflächen zwischen den Terrassen diente als Referenz. Die Ergebnisse dieser Studie zeigten, dass die Terrassierung die OBS Vorräte der 5 Jahre alten Plantage nicht beeinflusst hat. In den 29 und 44 Jahre alten Plantagen wurden jedoch in 0-1.2 m Tiefe höhere OBS Vorräte auf den Terrassen als auf den Referenzflächen beobachtet. Der positive Effekt der Terrassierung auf die OBS Vorräte in den beiden älteren Plantagen wurde auf die Erholung des OBS Vorrats im freiliegenden Oberboden des Entnahmebereichs, und die teilweise Erhaltung von OBS im begrabenen Boden des Ablagebereichs erklärt. Die Erholung der OBS Vorräte im Entnahmebereichen konnte durch die Aufnahme neuer OBS des freiliegenden Unterbodens in Form von Wurzeln und Laubfall sowie durch die Sedimentation von erodiertem Oberbodenmaterial des Oberhangs erklärt werden. Zusammenfassend zeigen die Ergebnisse, dass Terrassierung die Verluste von OBS verringern kann; ohne die Anlage von Terrassen könnte der Verlust von OBS durch die Umwandlung von Wald zu Kautschukplantagen größer sein.

In der dritten Studie, durchgeführt in einer subtropischen Landschaft, habe ich die aktuellen OBS Vorräte pro dominanter Landnutzung quantifiziert und die Beziehungen zwischen OBS und Landnutzung, sowie Vegetation, Bodentextur und Topographie untersucht. In einem 10.000 Hektar großen Gebiet habe ich 28 ein Hektar große Probeflächen in Wäldern mit geschlossenem und offenem Kronendach, Teeplantagen und Buschland ausgewählt. Die OBS-Vorräte in einer Tiefe von 0-0.9 m waren unter den höchsten der Region: 228.6 ± 19.7 (SE) Mg C ha^{-1} in Wäldern mit geschlossenem Kronendach, 200.4 ± 15.5 Mg C ha^{-1} in Wäldern mit offenem Kronendach, 197.5 ± 25.9 Mg C ha^{-1} in Teeplantagen und 236.2 ± 13.7 Mg C ha^{-1} im Buschland. OBS Konzentrationen und Vorräte unterschieden sich nicht signifikant zwischen den Landnutzungstypen. Mehr als 50% der gesamten Varianz der OBS wurde innerhalb der ein Hektar großen Flächen beobachtet und war abhängig von der Variabilität der Grundfläche der Bäume, Kohlenstoffvorrat der Streuauflage und der Geländeneigung. Diese Ergebnisse illustrieren die Bedeutung lokaler Prozesse auf die Variabilität von OBS Vorräten in einer montanen Landschaft. Die Ergebnisse aller drei hier vorgestellten Studien tragen zu einem besseren Verständnis von OBS Vorräten und deren Dynamik in einer schnellen Änderungsprozessen ausgesetzten Region bei. Darüber hinaus bilden sie eine potentielle Grundlage für weitere Studien über Änderungen von Ökosystemdienstleistungen in montanen Regionen des kontinentalen Südostasiens.

CHAPTER 1

General introduction

1.1. Soil organic carbon and the global carbon cycle

The soil organic carbon (SOC) pool is important for several reasons. SOC is the primary component of soil organic matter, which in turn is a main source of soil nutrients and improves soil structure and soil water holding capacity. Furthermore, SOC plays a key role in the global carbon cycle with ~1500 Pg C to a depth of 1 m (Batjes, 1996; Jobbagy and Jackson, 2000). The SOC pool is larger than the combined total of carbon stored in vegetation (~560 Pg C) and the atmosphere (~597 Pg C) (Denman et al., 2007; Schlesinger, 1997). Together with the carbon stored in oceans (~37,100 Pg C) and fossil fuels (~3700 Pg C) these pools form the global carbon cycle (Denman et al., 2007). The global carbon cycle is characterized by a dynamic exchange of carbon between the atmosphere, oceans and the terrestrial biosphere (soils and vegetation). Prior to human activity, the carbon fluxes among these pools used to be closely balanced (Schlesinger and Andrews, 2000). However, human activities like fossil fuel burning, cement production and land-use changes steadily increased atmospheric CO_2 concentrations (Denman et al., 2007).

Anthropogenic alterations of atmospheric CO_2 levels are a major concern in relation to the observed global warming since the start of the industrial revolution (circa 1750). Global warming has been attributed to the steady rise in atmospheric CO_2 concentrations and other greenhouse gases (methane and nitrous oxide) which disturbed the balance of incoming solar radiation and outgoing infrared radiation (Forster et al., 2007). Since the start of the industrial revolution atmospheric CO_2 concentrations have increased by 35% from 280 ppm to 379 ppm in 2005 (IPCC, 2007).

Recent estimates showed that land-use changes are responsible for ~15% of the current global anthropogenic CO_2 emission with an annual emission rate of ~1.5 Pg C yr^{-1} (including emissions from peat degradation) (van der Werf et al., 2009). CO_2 emissions from land-use changes are primarily caused by deforestation for cropland and pastures in the (sub) tropics (Houghton, 2010; van der Werf et al., 2009). Most of the CO_2 emissions from deforestation are caused by biomass loss and a smaller amount comes from decomposition of SOC (Detwiler, 1986). The estimated contribution of land-use changes to anthropogenic CO_2 emissions of 15% has an uncertainty range from 8-20%. This wide range is attributed to uncertainties in (1) deforestation rates, (2) initial ecosystem carbon stocks before land-use change, and (3) land-use change effects on ecosystem carbon stocks (Houghton, 2010). SOC losses following forest conversion are often due to changes in (1) the quality and quantity of SOC input, (2) SOC decomposition rates driven by microclimatic changes or the breakdown of soil aggregates, and (3) soil erodibility. The majority of the studies on the impacts of land-use changes on SOC, focused on conversions from forest to pasture, pasture to secondary forest, and forests to cropland. However, limited data is available on current important land-use changes (Powers et al., 2011); one example is the conversion of forests to rubber plantations (Ziegler et al., 2009).

1.2. The expansion or rubber plantations in montane mainland Southeast Asia

Land-use cover in montane mainland Southeast Asia, comprising southwest China, Laos, Cambodia, Myanmar, northeast Thailand, and northwest Vietnam, is characterized by a rapid expansion of monoculture rubber plantations (Picture 1) (Li and Fox, 2012). Rubber plantations are mainly replacing secondary forests, and shrub lands historically used for swidden agriculture (also called shifting cultivation or slash and burn) (Fox et al., 2012; Li and Fox, 2012; Ziegler et al., 2009). The recent expansion of rubber plantations started in the late 1950s when the first rubber plantations were successfully established by the Chinese government in the Xishuangbanna prefecture in the southern Yunnan province of China (Figure 1). Rubber trees were traditionally not grown in this region, since the climatic conditions (low temperatures in winter and a distinctive dry season) were considered marginal for rubber trees (Li and Fox, 2012). The establishment of rubber plantations in Xishuangbanna was promoted initially to meet the rubber demand for the Chinese national defence during an international embargo (1950s-1980s), and later on, as an alternative to the practice of swidden agriculture (Xu, 2006). On the one hand the expansion of rubber plantations in the area resulted in a strong economic development (Xu et al., 2005), but on the other hand rubber plantation expansion had dramatic consequences on Xishuangbanna's forest cover. Between 1976 and 2003 the area with rubber plantations, in Xishuangbanna, increased from less than 1% to 11% (Li et al., 2007) followed by a further expansion to more than 22% in 2010 (Xu et al., 2012). Simultaneously the forest cover decreased from 69% in 1976 to less than 50% in 2003 (Li et al., 2007). At present rubber plantations are expanding in montane mainland Southeast Asia, covering an area of more than 1.5 million hectares (Li and Fox, 2012) of which 424,000 hectares is in Xishuangbanna (Xu et al., 2012). The area of rubber plantations is predicted to expand in the coming decades, which may lead to a fourfold increase of the rubber area by 2050 (Fox et al., 2012).

Due to the region's mountainous topography, terrace construction is a typical phenomenon in the rubber plantations (Picture 1). The narrow terraces are built parallel to contours, contain a single row of rubber trees and are alternated by the original sloping areas. Terraces are a soil conservation measure, support water infiltration, and are used as footpaths for plantation workers during rubber tapping. Terraces are often constructed manually with a hoe, by cutting soil from the upslope position and piling up the excavated soil at the downslope position which forms the outer edge of the terrace.

Picture 1. A terraced rubber plantation (left), harvested latex (middle, picture by V.M. Hänsel), and a typical rubber landscape in Xishuangbanna, all hillslopes are covered with terraced rubber plantations (right).

1.3. Research needs

Despite of the geographical importance in montane mainland Southeast Asia of the secondary forest-to-rubber plantation conversion, the impacts of this land-use change on SOC have hardly been studied. Meta-studies of current data reported that forest-to-tree plantation conversions do not affect SOC stocks (Marín-Spiotta and Sharma, 2013; Powers et al., 2011). Conversely, the only three tropical studies on the conversion of forests to rubber plantations reported declines in SOC stocks ranging between 16% and 48% (Araujo et al., 2004; Salimon et al., 2009; Yang et al., 2004). These studies were conducted in rubber plantations aged 3 to 22 years. However, only one out of the three studies focused on the conversion from secondary forest to rubber plantation and that study was conducted in Xishuangbanna (Yang et al., 2004). The other two studies were on primary forest-to-rubber plantation conversion in Brazil (Araujo et al., 2004; Salimon et al., 2009). A main limitation of these three studies was that they had either no (Araujo et al., 2004; Salimon et al., 2009) or only two replicate plots per land-use type (Yang et al., 2004). Hence, more data is needed on this land-use change to improve estimates on the impacts on SOC stocks. Moreover, to our knowledge the impact of terracing on SOC stocks has not yet been studied in detail. However, erosion studies have shown that erosion-induced-soil redistribution not only redistributes SOC within the landscape but also affects SOC dynamics (Gregorich et al., 1998; van Oost et al., 2007). SOC dynamics differ between the eroding sites and depositional sites. At the eroding sites the SOC lost by soil erosion might be rapidly replaced due the storage of newly added organic matter from litter and roots (Harden et al., 1999; Stallard, 1998). The exposed subsurface soil might have a relatively high capacity to store SOC inputs, due to the initially low SOC concentration and the large number of binding sites for carbon from clay particles and Al and Fe hydroxides/oxides. At the depositional site, burial of the former surface soil might inhibit SOC decomposition rates (Doetterl et al., 2012; Stallard, 1998; VandenBygaart et al., 2012). Furthermore, soil erosion might cause the breakdown of soil aggregates, which could enhance SOC decomposition rates (Elliott, 1986). Similar to soil erosion, terrace construction also results in positions where soil is removed (cut section) and positions where soil is deposited (fill section). The impacts of terrace construction on SOC dynamics may thus, to a certain extent, be comparable to mechanisms proposed for erosional impacts.

To improve the model-based estimates of effects of land-use changes on SOC stocks and CO_2 emissions from soils at a regional scale, it is essential to have knowledge on the current spatial distribution of SOC in the landscape (Houghton 2003). Spatial distribution of SOC is related to climate, topography, soil parent material, biota, time and human activity (Jenny, 1941). The importance of each controlling factor for SOC differs with spatial extent and environmental setting. Studies on the spatial distribution of SOC in montane mainland Southeast Asia are limited to northern Thailand (Aumtong et al., 2009; Pibumrung et al., 2008), and Laos (Chaplot et al., 2010, 2009; Phachomphon et al., 2010; Rumpel et al., 2008, 2006). Given the region's mountainous and complex terrain, which may result in large spatial variation in SOC stocks (Chaplot et al., 2009), and the region's dynamic land-use changes, there is a need for more local studies on the spatial distribution of SOC stocks.

1.4. Objectives

This work focused on SOC stocks in two mountainous landscapes in Xishuangbanna, southern Yunnan province of China (21˚31'17.03"N, 100˚37'12.13"E and 21˚29'25.62"N, 100˚30'19.85"E). **In the first study (Chapter 2), conducted in a tropical landscape, I quantified the changes in SOC stocks caused by the conversion of secondary forests to rubber plantations.** I sampled 11 rubber plantations ranging in age from 5 to 46 years and seven secondary forest plots using a space-for-time substitution approach. The objectives were: (i) to quantify changes in SOC stocks following conversion of secondary forests to rubber plantations over a 46 years' time period, and (ii) to determine the biophysical factors which control SOC concentrations, and SOC changes.

The second study (Chapter 3) aimed at quantifying SOC stock changes caused by terrace construction. In three rubber plantations of increasing age, I sampled the terraces according to soil redistribution zones, using the original sloping areas between the terraces as reference. SOC stocks of the terrace were compared with stocks of the original slopes for which I assumed that the SOC stocks were not affected by terrace construction. The following hypotheses were tested: (i) SOC stocks in the cut sections of the terraces are lower than at the reference positions but this difference diminishes with increasing plantation age, and (ii) SOC stocks in the fill sections of the terraces are higher than on the reference positions.

In the third study (Chapter 4), conducted in a subtropical landscape, I quantified SOC concentrations and stocks of the most dominant land-use types and examined the effects of land use, litter layer, vegetation, soil texture and topography on SOC. Here, I applied a probability sampling technique (double sampling for stratification) with a spatially nested structure which allowed us to partition the overall variance in SOC, soil texture, vegetation and topographical factors that could be attributed to variation among land-use types, among sampling plots and within sampling plots. The objectives were (i) to quantify the present SOC stocks of the dominant land-use types, and (ii) to define the relationships of SOC concentrations and stocks with land-use type, vegetation, soil texture and topography.

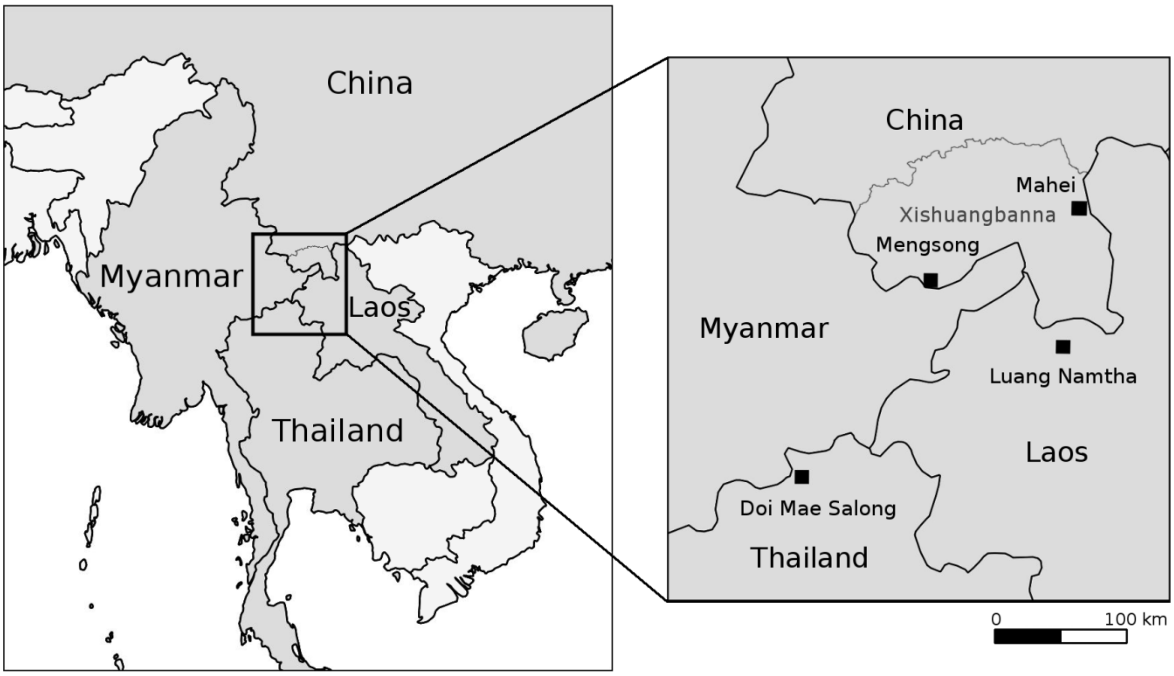

Figure 1. Locations of the four project sites (right) in the Upper Mekong region (left), map from Beckschäfer (2013).

1.5. The MMC project – Making the Mekong Connected

This work was part of the project "Making the Mekong Connected" (MMC) funded by the German Agency for International Cooperation (GIZ) and the German Ministry for Economic Cooperation (BMZ). A main aim of the project was to study the impacts of forest conversions to cash crops (e.g. rubber or tea plantation) on ecosystem carbon stocks, biodiversity, and other ecosystem services in the Upper Mekong Region (Figure 1). Another important objective was to estimate the existing values of the dominant land-use types in four project sites in terms of biodiversity and ecosystem carbon stocks. The four project sites were located in Mengsong and Mahei (both in Xishuangbanna, China), Luang Namtha (Laos), and Doi Mae Salong (Thailand) (Figure 1). The studies in this thesis are based on data collected in Mengsong (Chapter 4) and in a study area situated at a 12-km distance from Mengsong (Chapters 2 and 3). The project was a collaboration between the World Agroforestry Centre (ICRAF, China), Xishuangbanna Tropical Botanical Garden (XTBG, China), Chiang Mai University (CMU, Thailand), Asian Institute of Technology (AIT, Thailand), National Agriculture and Forestry Research Institute in Laos (NAFRI) and the Faculty of Forest Sciences at the Georg-August-Universität Göttingen (Germany).

CHAPTER 2

Soil carbon stocks decrease following conversion of secondary forests to rubber (Hevea brasiliensis) plantations

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Abstract

Forest-to-rubber plantation conversion is an important land-use change in the tropical region, for which the impacts on soil organic carbon (SOC) stocks have hardly been studied. In montane mainland Southeast Asia, monoculture rubber plantations cover 1.5 million hectares and the conversion from secondary forests to rubber plantations is predicted to cause a fourfold expansion by 2050. Our study, conducted in southern Yunnan province, China, aimed to quantify the changes in SOC stocks following the conversion from secondary forests to rubber plantations. We sampled 11 rubber plantations ranging in age from 5 to 46 years and seven secondary forest plots using a space-for-time substitution approach. We found that forest-to-rubber plantation conversion resulted in losses of SOC stocks by an average of 37.4 ± 4.7 (SE) Mg C ha^{-1} in the entire 1.2-m depth over a time period of 46 years, which was equal to 19.3 ± 2.7% of the initial SOC stocks in the secondary forests. This decline in SOC stocks was much larger than differences between published above-ground carbon stocks of rubber plantations and secondary forests, which range from a loss of 18 Mg C ha^{-1} to an increase of 8 Mg C ha^{-1}. In the topsoil, carbon stocks declined exponentially with years since deforestation and reached a steady state at around 20 years. Although the IPCC tier 1 method assumes that SOC stock changes from forest-to-rubber plantation conversions are zero, our findings show that they need to be included to avoid errors in estimating overall ecosystem carbon fluxes.

2.1 Introduction

Deforestation and forest degradation in the tropics have been estimated to contribute 12-15% of the global anthropogenic CO_2 emissions (van der Werf et al., 2009). The majority of land-use induced CO_2 emissions arise from the loss of above-ground biomass and to a lesser extent from decomposition of soil organic carbon (SOC) (Detwiler, 1986; Houghton and Hackler, 1999). Currently, the estimations of land-use change effects on above-ground carbon stocks are improving due to remote sensing techniques. However, estimates of land-use change effects on SOC stocks remain inconclusive (Aalde et al., 2006; Powers et al., 2011).

In tropical regions, the magnitude and direction of land-use induced changes in SOC stocks are largely determined by mean annual rainfall and clay mineralogy (de Koning et al., 2003; Powers et al., 2011). A large number of studies exist on the impact of tropical land-use changes on SOC stocks, especially on the conversion from forest to pasture, pasture to secondary forest, and forest to cropland (Powers et al., 2011). However, the published field observations are unequally distributed over the tropics with regards to an area-weighted distribution of the above-mentioned biophysical conditions. In addition, little research has been done on currently important land-use changes, one of which is forest-to-rubber (*Hevea brasiliensis*) plantation conversion. These limitations in available field observations hamper the estimates of land-use change effects on SOC stocks in the tropics (Powers et al., 2011).

Our present study focuses on the land-use change from secondary forests to rubber plantations in Xishuangbanna, the southernmost prefecture of Yunnan Province in the southwest of China. The area of monoculture rubber plantations is rapidly expanding in montane mainland Southeast Asia, spanning southwest China, Laos, Cambodia, Myanmar, northeast Thailand, and northwest Vietnam, causing a large decrease in the region's forest cover (Ziegler et al., 2009). Rubber trees were traditionally not grown in this region, since environmental conditions (low temperatures in winter and a distinctive dry season) were considered marginal for rubber trees (Li and Fox, 2012). The first rubber plantations were successfully established in Xishuangbanna by the Chinese government in the late 1950s, and the subsequent expansion of rubber plantations resulted in a strong economic development (Xu et al., 2005). At present, rubber plantations in montane mainland Southeast Asia cover an area of more than 1.5 million hectares (Li and Fox, 2012) of which 424,000 hectares are in Xishuangbanna (Xu et al., 2012). By 2050 the area of rubber plantations is predicted to increase fourfold, mainly by replacing secondary forests, and swidden-related bushes and shrublands (Fox et al., 2012).

Forest-to-rubber plantation conversion is an important recent land-use change for which environmental impacts have hardly been studied. Meta-analyses of current data have shown that changes of SOC stocks after conversion of forests to tree plantations are variable: no effects were reported for tropical tree plantations (Marín-Spiotta and Sharma, 2013; Powers et al., 2011) whereas in another review 0-30% decrease in SOC stocks were reported for intensified rubber plantations compared to swidden fields in Southeast Asia (Bruun et al., 2009). However, this review was based on studies which did not have a clear reference land-use type for the rubber plantations but merely compared land-use types and therefore any detected difference cannot directly be attributed to changes in land use. To our knowledge there are only three published tropical studies that investigated the effects of forest-to-rubber plantation conversions on SOC stocks. Two out of the three studies focused on the conversion from primary forests to rubber plantations in Brazil (Araujo et al., 2004; Salimon et al., 2009) and both reported declines in the SOC stocks in 17- and 22-year-old plantations. The other study focused on the conversion from secondary forests to rubber plantations and happened to be conducted in Xishuangbanna (Yang et al., 2004). Yang et al. (2004) reported a 20% decline in the SOC stock in the top 0.6-m depth in a 3-year-old plantation and a 16% decline in a 7-year-old plantation but these estimates were not corrected for changes in soil bulk density with land-use change. It is also important that studies include older rubber plantations as SOC stocks may reach equilibrium after several decades, and older plantations will allow us to quantify the long-term effects of this land-use change. SOC losses after deforestation are often related to (i) changes in the quality and quantity of SOC input, (ii) accelerated SOC decomposition rates driven by changes in microclimatic conditions or the breakdown of soil aggregates, and (iii) enhanced soil surface erosion. The magnitude of SOC losses depends furthermore on site-specific conditions such as soil texture, soil mineralogy, topography and climate.

Improved estimates of the effects of this important land-use change on SOC stocks are essential to the national greenhouse gas inventories from the Conference of Parties of the United Nations Framework of Climate Change. Although the Intergovernmental Panel of Climate Change (IPCC) provides guidelines for the estimations of the ecosystem carbon fluxes arising from land-use changes, the IPCC tier 1 method assumes SOC stock changes to be zero for the conversion from forests to rubber plantations because of limited scientific knowledge (Aalde et al., 2006; Lasco et al., 2006).

We conducted the present study in Xishuangbanna, Yunnan Province, China using a space-for-time substitution approach. Our objectives were: (i) to quantify changes in SOC stocks following conversion of secondary forests to rubber plantations over a 46 years' time period, and (ii) to determine the biophysical factors which control SOC concentrations, and SOC stock changes. We hypothesized that conversion from secondary forests to rubber plantations would result in a decrease of SOC stocks. This decrease is expected to be related to management practices commonly employed in rubber plantations such as terrace construction and removal of the vegetation understory.

2.2 Material and Methods

Study area and site characteristics

The study area of 4500 hectares was located in Menglong township, Jinghong county of Xishuangbanna prefecture in Yunnan province, China (21˚31’17.03”N, 100˚37’12.13”E) (Figure 1). The climate is tropical monsoon and is characterized by a dry season from November to April and a wet season from May to October. The mean annual rainfall is 1377 mm and the mean annual temperature is approximately 22.7°C (Xu et al., 2005). The topography is hilly to mountainous, with an elevation that varies between 650 and 1450 m above sea level (Xu et al., 2005). The study plots were located between 700 and 830 m above sea level. The soils at the plots are dominated with low activity clays and were classified as Ferralsols having an effective cation exchange capacity (ECEC) of less than 12 $cmol_c$ kg^{-1} clay and as (hyper) ferralic Cambisols with an ECEC of less than 24 $cmol_c$ kg^{-1} clay (IUSS Working Group WRB., 2006) (Table 1, Table S2.1.A, Table S2.1.B).

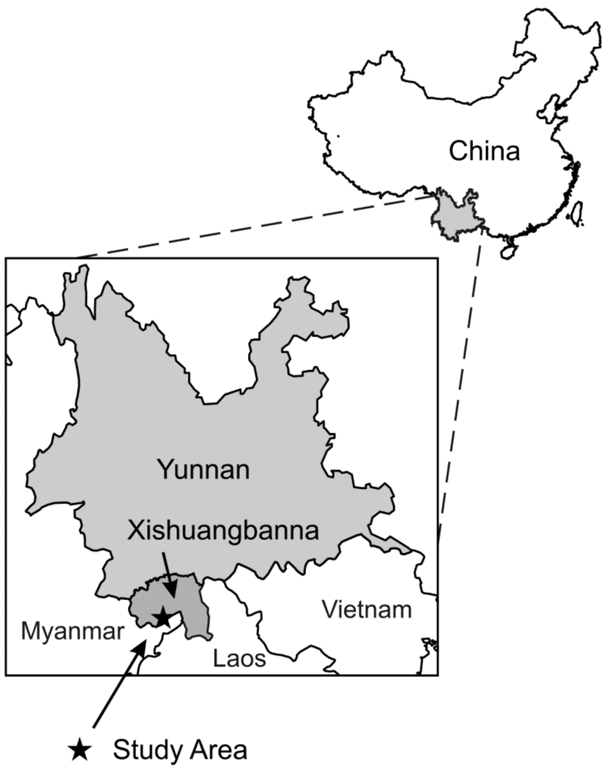

Figure 1. Location of the study area in Xishuangbanna prefecture, Yunnan province, China.

Table 1. Means[1](± SE) of soil characteristics of land-use types.

Characteristic	Depth (m)	Rubber plantation (n = 11)	Secondary forest (n = 7)
Sand (%)	0-0.15	32.5 ± 3.8	34.7 ± 4.6
	0.15-0.3	30.5 ± 3.9	34.4 ± 5.0
	0.3-1.2	27.8 ± 3.4	30.2 ± 4.4
Silt and clay (%)	0-0.15	67.5 ± 3.8	65.3 ± 4.6
	0.15-0.3	69.5 ± 3.9	65.6 ± 5.0
	0.3-1.2	72.2 ± 3.4	69.8 ± 4.4
Bulk density (g cm^{-3})	0-0.15	1.1 ± 0.1	1.0 ± 0.1
	0.15-0.3	1.1 ± 0.0	1.2 ± 0.0
	0.3-1.2	1.3 ± 0.0	1.3 ± 0.0
C:N ratio	0-0.15	12.2 ± 0.3	12.8 ± 0.5
	0.15-0.3	12.1 ± 0.4	12.5 ± 0.4
	0.3-1.2	9.5 ± 0.4	9.7 ± 0.5
pH (H_2O)[2]	0-0.15	4.8 ± 0.1 a	4.7 ± 0.1 b
	0.15-0.3	4.8 ± 0.1	4.9 ± 0.1
	0.3-1.2	5.0 ± 0.1	4.9 ± 0.1
pH (KCl)	0-0.15	3.9 ± 0.0	3.9 ± 0.0
	0.15-0.3	3.8 ± 0.1	3.9 ± 0.0
	0.3-1.2	4.0 ± 0.0	4.0 ± 0.0
ECEC[2] ($mmol_c$ kg^{-1} soil)	0-0.15	46.4 ± 1.8 a	55.5 ± 2.4 b
	0.6-0.9	28.8 ± 1.2 a	36.9 ± 3.9 b
Base saturation (%)	0-0.15	24.4 ± 4.2	24.1 ± 6.0
	0.6-0.9	10.9 ± 1.6	11.3 ± 1.5

[1]Means of the 0.3-1.2 m depth interval are means of the 0.3-0.6-m, 0.6-0.9-m and 0.9-1.2-m depth intervals.
[2]Effective cation exchange capacity, within a row means followed by different letters differ significantly between rubber plantation and secondary forest (linear mixed effects model at P ≤ 0.05).

Current and past land use

The major land-use types in Menglong township include rubber plantations, secondary forests, and farmland. According to local plantation owners the dominant land-use change trajectories in Menglong township were: (i) Primary forest - swidden agriculture - secondary forest - rubber plantation, and (ii) primary forest - swidden agriculture - rubber plantation. Swidden agriculture was the dominant land-use type in the region for centuries (Xu, 2006); this involved cutting and burning of vegetation patches in the forest, thereby creating fields for use as rotation of cropping phases (1-3 years) and fallow periods (5-20 years) during which secondary vegetation regrows (Fox et al., 2012; Mertz, 2009). The widespread practice of swidden cultivation in the past resulted in loss and degradation of primary forests (Fox and Vogler, 2005). Nowadays, almost all swidden fields have been replaced by monoculture rubber plantations. Since primary forest and swidden agriculture are not present anymore, we focused on the more recent land-use change from secondary forest towards rubber plantations. Based on information from local plantation owners, we selected rubber plantations that all went through this land-use change trajectory. Forest clearing was done by hand and no heavy machinery was used. After forest clearing the sites were usually burnt. During the first four years after planting, rubber trees may have been intercropped with maize, upland rice, peanuts and beans. In our study area, four forest remnants remain, including three collective forests and one "watershed protection" forest, which we used as our reference. These broadleaf forests are highly degraded due to the collection of firewood at present and timber extraction in the past. The forests have been cleared from primary forest, after which they were used for swidden agriculture and finally secondary forests were established (information from local farmers). The age of each forest remnant was estimated between 40-60 years. The size of the forest patches ranges from 20-60 hectares.

Management practices in rubber plantations

Management practices commonly applied in rubber plantations are terrace establishment and maintenance, fertilizer application, pest control, management of the vegetation understory and rubber tapping. The terrace benches are constructed by hand using a hoe, cutting soil from upper parts of the slope and moving it to lower parts. During terrace maintenance, weeds are removed from the terrace steps by scraping off the soil, which is subsequently evenly distributed over the entire terrace bench. This process is repeated once or twice per year, resulting in older plantations having wider and deeper incised terraces. Rubber trees are planted in a row on the terrace benches and have a tree spacing ranging from 2-3 m. The step height of each individual terrace ranges from 0.3-0.8 m and bench width is typically ~2.5 m. The horizontal distance between two adjacent terrace benches is 5-7 m, depending on the slope of the hill. Between some of the rubber trees pits are dug which have various uses: (i) as a measure to reduce runoff and retain soil moisture, and (ii) to apply fertilizer and collect leaf litter and cut herbs. The dimensions of these pits vary per plantation and range from: 0.4-1.3-m length x 0.2-0.5-m width x 0.2-0.35-m depth. Chemical fertilizers (NPK-compounds) are applied one to two times per year. The management of vegetation understory varies per rubber plantation; some plantation owners use herbicides to control the vegetation understory while others cut the vegetation understory back. Rubber tapping is usually done between April and October and latex collection is done every second day.

Sampling design

We used the space-for-time substitution approach to quantify changes in SOC stocks following conversion from secondary forests to rubber plantations. SOC stocks were measured in clusters consisting of one reference secondary forest plot and one to three plots in rubber plantations. Clusters were established around randomly selected secondary forest plots. To avoid edge effects, the forest plots were selected at least 20 m from the forest edge. Within each cluster, the rubber plantations were chosen based on biophysical conditions, land-use history and distance to the reference plot. We only selected rubber plantations that were established immediately after forest clearing. To keep biophysical conditions within a cluster as similar as possible, we selected rubber plantations with similar altitude, slope, aspect, soil colour and soil texture as the reference plot. The maximum distance between plots within a cluster was 3 km. In total we selected seven clusters, with a total of 11 rubber plantations and seven reference forests. The rubber plantations within each cluster differed in age ranging from 5 to 46 years. Selected rubber plantations were both state-owned rubber plantations and plantations belonging to smallholder farmers.

A critical assumption of the space-for-time substitution approach is that plots within a cluster were initially similar with regard to soil characteristics, SOC stocks, and land-use history such that measured differences in SOC stocks between the reference land-use type and the converted land-use type can be attributed to recent land-use change (Veldkamp, 1994). To test this assumption we compared land-use independent soil characteristics (i.e. soil texture) of plots within a cluster. Since we did not detect significant differences in soil texture between the secondary forest and rubber plantations within a cluster (Table 1), we assumed that the soils were originally similar and that observed SOC changes can be attributed to changes in land use.

Fieldwork permission

Our research was part of the project “Making the Mekong Connected (MMC)”. This project had been officially registered at the Kunming Institute of Botany, Chinese Academy of Sciences, which provided rights for access to field research in China. We received permission from the owners and managers of the rubber plantations to conduct the described fieldwork in their plantations. The secondary forests are part of local collective forests belonging to the villages. The local forestry station of Menglong Township, Jinghong County, has been informed a priori about our fieldwork in the secondary forests. No official permits were required for the described fieldwork since the secondary forests are not part of a national nature reserve. The fieldwork did not involve endangered or protected species.

Plot layout and soil and litter sampling

In each land-use type, we established a plot with a size of 20 × 20 m, corrected for slope. Within each plot we established five parallel transects with 5-m spacing in between. Transects had a fixed north-south orientation. We randomly positioned four sampling points along each transect, resulting in 20 sampling points per plot.

Soil samples were collected down to 1.2-m depth from five depth intervals: 0-0.15 m, 0.15-0.3 m, 0.3-0.6 m, 0.6-0.9 m, and 0.9-1.2 m. The upper three depth intervals were sampled using a Dutch auger at the 20 sampling points. The 20 soil samples were pooled in the field to form one composite sample for each depth interval. Soil samples for the 0.6-0.9-m and 0.9-1.2-m depth were sampled in a soil pit which was positioned on the slope between two adjacent terraces in the middle of each plot. The soil samples were air dried for five days and sieved through a 2-mm sieve prior to laboratory analyses. Bulk density samples were taken in the soil pit for each of the five depth intervals, using the core method (Blake and Hartge, 1986). Very few of the bulk density samples contained stones or rock fragments and thus we did not correct for the gravel content.

Litter layer samples were taken from every second sampling point, resulting in 10 litter samples per plot. Leaf litter and organic material (including seeds and twigs) were collected from a 0.04-m^2 quadrant sample frame. The collected material was oven dried at 60 ˚C for 48 hours and weighed. Subsamples of each sample were pooled by plot and analysed for total carbon and nitrogen concentration. The carbon stock of the litter layer was calculated with the carbon concentration (%), the mass of the litter layer, and the sample-frame area.

Tree inventory, topographical and land use data

In the rubber-plantation plots, we measured for all the trees the diameter at breast height (DBH) at 1.3 m above the soil surface. In the forest plots, we measured the DBH for trees with a DBH > 4 cm, and the DBH of bamboos. For bamboos, we measured one stem DBH per clump and we recorded the number of stems per clump. Here we report both the tree basal area and total basal area, which is the sum of the basal area of trees and bamboos. Other site characteristics that were collected of each plot included: slope, aspect, altitude, and GPS coordinates. Information on current and past land use and their management was collected through interviews with land owners and elders in the villages.

Laboratory analyses and soil organic carbon stock calculations

Total carbon and nitrogen concentrations were measured from ground soil and litter samples by dry combustion using CNS Elemental analyzer (Elementar Vario EL, Hanau, Germany). As soil pH was below 5.5, carbonates were not expected in these soils and we made no attempt to remove them. Soil pH (H_2O) and pH (KCl) were measured on air dried soil for all individual soil samples in a 1:2.5 soil-to-solution ratio. ECEC was measured on soil samples of the 0-0.15-m and 0.6-0.9-m depth. The soil samples were percolated with unbuffered 1 M NH_4Cl and the percolates were analysed for exchangeable cations using ICP-AES (Spectroflame, Spectro Analytical Instruments, Kleve, Germany) (König and Fortmann, 1996). Soil texture analyses were determined for all depth intervals with the pipette method, distinguishing the fractions: clay (< 0.002 mm), silt (0.002-0.063 mm), and sand (0.063-2 mm). SOC stocks (Mg C ha^{-1}) in each depth interval were calculated using the following equation:

$$SOC\ stocks(Mg\ C\ ha^{-1}) = \frac{\%C}{100} \times BD\ (Mg\ m^{-3}) \times \Delta\ D\ (m) \times 10{,}000\ m^2 ha^{-1}, \qquad (1)$$

where, BD is the bulk density and ΔD refers to the thickness of the depth interval. Total SOC stocks down to 1.2-m depth were calculated as the sum over all depth intervals. Land-use changes often coincide with changes in bulk density due to management practices which may compact or loosen the soil. In order to be able to compare the same soil mass and to avoid the interference of bulk density changes with SOC stocks changes, we used the bulk density data of the reference plots to calculate the SOC stock of the rubber plantation plots (de Koning et al., 2003).

Statistical analyses

All statistical analyses were done using the open source statistical software R version 2.15.0 (R Development Core Team, 2012). To make statistical inferences on the differences in SOC stocks and soil characteristics between rubber plantations and secondary forest, we applied linear mixed effects models (LME) using the nlme package (Pinheiro et al., 2012). Response variables were the SOC stocks and soil characteristics and we included land-use type, depth interval, and the interaction between land-use type and depth interval as fixed effects. Cluster was included as a random factor. Comparisons of SOC stocks and soil characteristics between land-use types at each depth interval were obtained by defining and testing contrasts with the generalized linear hypothesis test in the multcomp package (Hothorn et al., 2008). For the multiple comparisons of SOC stock changes between depth intervals, the P values were adjusted using Holm's correction. For each LME, assumptions on normality and homogeneity of variance were checked by visual inspection of plots of residuals against fitted values. In cases of unequal variances of residuals, we included a variance function that allows for unequal variances (Zuur et al., 2009).

To examine monotonic trends of SOC concentrations and relative SOC stock differences with potential explanatory variables, we did spearman rank correlation tests. Relative SOC stock differences were calculated as carbon stock in rubber plantation minus carbon stock of the reference secondary forest divided by carbon stock of the reference secondary forest multiplied by 100. Relative SOC stock differences were correlated with explanatory variables of the rubber plantations. As potential explanatory variables we included litter carbon stock, litter C:N ratio, total basal area, sum of silt and clay content, slope, and altitude. Correlation tests were done for each depth interval.

The trend between SOC and rubber plantation age was examined using non-linear regression. We tested the fit of both a mono-exponential model and a bi-exponential model according to Lobe et al. (Lobe et al., 2001). The mono-exponential model assumes a single SOC pool which following land-use change tends towards a new equilibrium:

$$X_t = X_e + (X_0 - X_e) \times exp\,(-k \times t), \quad (2)$$

where, X_0 is the initial SOC stock of the secondary forest plots (t = 0), X_t is the SOC stock in the rubber plantation plots at age t, X_e is the SOC stock at steady state, k is the decay rate per year, and t is year since land-use change. The age t at which steady state was reached was calculated as the point where the proportion of carbon remaining in the soil (X_t) did not differ more than 1% of the calculated steady state value X_e (Lobe et al., 2001). The bi-exponential model considers both labile and stable SOC pools:

$$X_t = X_1 \times exp(-k_1 \times t) + X_2 \times exp(-k_2 \times t), \quad (3)$$

where, X_1 is the proportion of carbon in the labile pool, and X_2 is the proportion of carbon in stable pool ($X_2 = 100 - X_1$), k_1 is the decay rate per year of the labile pool, k_2 is the decay rate per year of the stable pool. We expressed SOC as the proportion of the SOC stock in the rubber plantation to the initial amount in the reference secondary forest. The exponential models were fitted to the data using nonlinear least-squares estimations. The goodness of the fit was assessed by Pearson's correlation coefficient (r) showing the relationship between the observed and fitted values.

2.3 Results

Soil characteristics, litter layer, and tree basal area in rubber plantations and secondary forests

Soil texture, bulk density, soil C:N ratio, pH (KCl), and base saturation did not differ between rubber plantations and secondary forests (Table 1). The pH (H_2O) in the top 0.15-m depth was higher in rubber plantations than in secondary forests. The ECEC in all depth intervals was lower in rubber plantations than in secondary forest. Litter carbon concentration, litter C:N ratio, and litter carbon stock did not differ between rubber plantations and secondary forests (Table 2). The tree basal area in rubber plantations ranged from 3.2 to 42.4 and was positively correlated with plantation age (spearman's rho = 0.93, p ≤ 0.001); the mean tree basal area was twice that of the secondary forests (Table 2). However, the total basal area (sum of trees and bamboos) did not differ between rubber plantations and secondary forests.

Table 2. Means (± SE) of litter and tree characteristics of land-use types.

Characteristic	**Rubber plantation (n = 11)**	**Secondary forest (n = 7)**
Litter carbon concentration (%)	41.1 ± 0.7	40.0 ± 0.7
Litter C : N ratio	46.1 ± 3.8	44.9 ± 3.6
Litter carbon stock (Mg C ha^{-1})	2.1 ± 0.2	2.7 ± 0.4
Tree basal area[1] (m^2 ha^{-1})	18.6 ± 3.8 a	9.7 ± 2.4 b
Total basal area[2] (m^2 ha^{-1})	18.6 ± 3.8	15.3 ± 1.7

[1]Within a row, means followed by different letters differ significantly between rubber plantation and secondary forest (linear mixed effects model at P ≤ 0.05).
[2]Total basal area is calculated as the sum of the basal area of trees and bamboos.

Soil organic carbon concentrations and stocks in rubber plantations and secondary forests

All rubber plantations had a lower SOC stock in the total soil profile (0-1.2-m depth) than secondary forests (P ≤ 0.01) (Table 3). The differences in SOC stocks between rubber plantations and secondary forests ranged from -15.4 to -59.4 Mg C ha^{-1} with a mean of -37.4 ± 4.7 Mg C ha^{-1}, equivalent to a 19.3 ± 2.7% loss of the initial SOC stock. The SOC losses were depth dependent as was shown by a significant interaction between land-use type and soil depth (P ≤ 0.001). The decrease in SOC concentrations and SOC stocks was significant for the three depth intervals in the top 0.6-m depth (Table 3). The largest decrease was found in the top 0.15-m of the soil ($P \leq 0.01$) accounting for 32% of SOC losses.

For the top 0.15-m depth, the proportion of carbon remaining in the soil exponentially decreased with the years since land-use change, as described by the mono-exponential model (Equation 2) (Figure 2a). The largest decrease could be observed in the first 5 years following land-use change, when the SOC stocks had declined to approximately 80% of the original amount. A steady state was reached after approximately 20 years, when SOC stocks had declined to 68% of the original amount. At 0.15-0.3-m depth, SOC had the tendency to exponentially decrease with time but the estimated decay rate of the mono-exponential model was not significant; a steady state after approximately 20 years showed a SOC stock decline of 25% of the original amount (Figure 2b). At 0.3-0.6-m depth (Figure 2c), a mono-exponential trend was not detectable. Bi-exponential model (Equation 3) fitting resulted in insignificant decay rates for both the labile and stable SOC pool for all soil depths (data not shown). Furthermore, the fitted curves of the bi-exponential model and mono-exponential model were identical. Together these results indicate that the mono-exponential model was most suitable to describe the observed SOC changes in relation to years since land-use change.

Correlations of soil organic carbon concentrations and soil organic carbon stock changes with environmental factors

In rubber plantations, SOC concentrations in the top 0.6 m of the soil showed positive correlations with altitude and with the sum of clay and silt content (and a negative correlation with sand content) (Table 4). However, at 0.15-0.3-m depth the correlation with the sum of silt and clay content was only marginally significant ($P = 0.1$). Rubber plantation age was not correlated to SOC concentrations in the top 0.6-m depth. However, for 0.9-1.2-m depth a positive correlation was observed between SOC concentrations and plantation age (spearman's rho = 0.66, $P \leq 0.05$). In secondary forests, SOC concentrations at 0.15-0.3-m depth were positively correlated with the sum of clay and silt content and at 0.3-0.6-m depth with the total basal area of the forest (Table 4). The trends with soil texture and total basal area were also apparent at 0.6-0.9-m depth (data not shown). Relative differences in SOC stocks between rubber plantations and secondary forests in the top 0.15 m of the soil were negatively correlated with total basal area and rubber plantation age. In the top 0.6 m of the soil, relative differences in SOC stocks were positively correlated with altitude, but for the top 0.15 m of the soil this correlation was marginally significant ($P=0.06$).

Table 3. Means (± SE) of soil organic carbon concentrations and stocks and absolute[1] and relative[2] differences between land-use types.

	Rubber plantation (n = 11)		Secondary forest (n = 7)		Difference (n = 7)	
Depth (m)	**C (%)**	**C (Mg ha^{-1})**	**C (%)**	**C (Mg ha^{-1})**	**Absolute (Mg C ha^{-1})**	**Relative (C %)**
0-0.15	2.1 ± 0.1	30.3 ± 1.9	2.9 ± 0.1	43.9 ± 2.6	-11.8 ± 1.1***	-26.9 ± 2.8***
0.15-0.3	1.7 ± 0.1	29.8 ± 1.6	2.2 ± 0.1	38.9 ± 1.5	-8.2 ± 1.4 ***	-21.4 ± 3.2***
0.3-0.6	1.2 ± 0.1	43.6 ± 2.6	1.4 ± 0.1	52 ± 1.6	-8.0 ± 3.0*	-15.4 ± 5.6*
0.6-0.9	0.7 ± 0.1	28.0 ± 1.9	0.9 ± 0.1	35.2 ±3.7	-6.5 ± 3.6	-16.0 ± 8.0
0.9-1.2	0.6 ± 0.0	23.2 ± 1.3	0.7 ± 0.0	26.0 ± 1.0	-2.9 ± 1.8	-11.2 ± 7.0
Total	-	154.9 ± 6.2	-	196.0 ± 3.5	-37.4 ± 4.7**	-19.3 ± 2.7**

Significant at *P≤0.05, ** P ≤0.01, and *** P ≤ 0.001, (linear mixed effects model).
[1]Absolute differences in stocks were calculated as means of rubber plantations within a cluster minus reference forest.
[2]Relative differences in stocks were calculated as means of rubber plantations within a cluster minus reference forest divided by reference forest multiplied by 100.

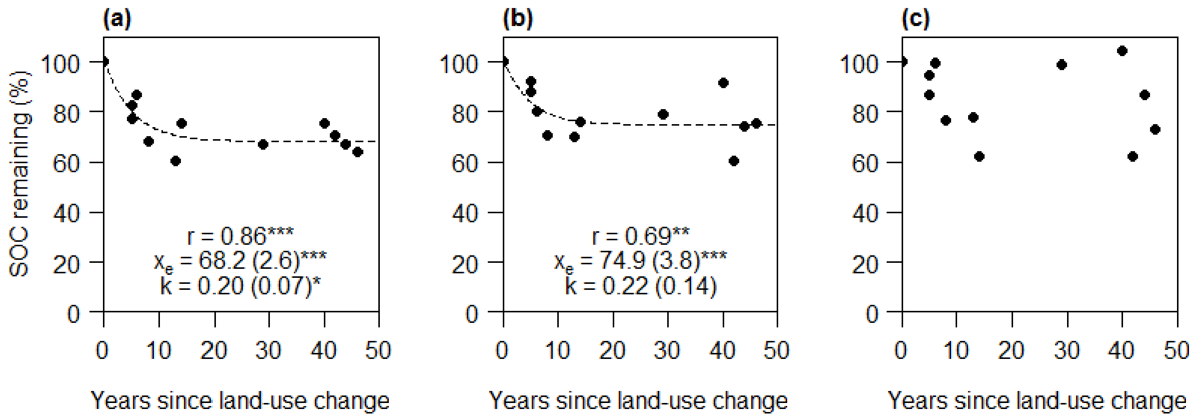

Figure 2. Soil organic carbon (SOC) remaining after land-use change at (a) 0-0.15-m, (b) 0.15-0.3-m, and (c) 0.3-0.6-m depth. SOC remaining is expressed as the proportion of SOC in rubber plantations relative to the secondary forest. The dashed lines represent fitted mono-exponential models (see Equation 2). r = Pearson's correlation coefficient between observed and fitted values; k = decay rate (year^{-1}) and X_e= equilibrium ratio (%), and values in brackets are SE. Pearson's r and parameter estimates are significant at *P ≤ 0.05, **P ≤ 0.01, and ***P ≤ 0.001.

Table 4. Correlation coefficients[1] of soil organic carbon (SOC) concentrations and relative SOC stock differences[2] with explanatory variables at three depths.

Explanatory variable	Rubber plantation C (%) (n = 11)			Secondary forest C (%) (n=7)			Relative soil C differences (%) (n=7)		
	0-0.15m	**0.15-0.3m**	**0.3-0.6m**	**0-0.15 m**	**0.15-0.3m**	**0.3-0.6m**	**0-0.15m**	**0.15-0.3m**	**0.3-0.6m**
Litter C stock (Mg ha^{-1})	0.21	0.39	0.40	-0.07	-0.18	0.36	-0.10	0.52	0.49
Litter C : N ratio	-0.40	-0.23	0.15	-0.43	-0.57	-0.11	-0.35	-0.25	0.04
Total basal area (m^2 ha^{-1})	-0.39	-0.47	-0.24	0.18	0.57	0.89**	-0.64*	-0.55†	-0.48
Silt and clay (%)	0.72*	0.53†	0.66*	0.36	0.93**	0.61	0.13	0.20	0.49
Rubber plantation age (year)	-0.26	-0.34	-0.03	-	-	-	-0.65*	-0.46	-0.24
Slope (%)	-0.08	0.07	0.14	-0.46	-0.04	0.11	0.15	0.35	0.08
Altitude (m)	0.75**	0.76**	0.71*	-0.50	0.00	0.21	0.59†	0.70*	0.70*

[1]Spearman rank correlation test, marginally significant at $^{\dagger}P \leq 0.1$, and significant at $*P \leq 0.05$, and $**P \leq 0.01$.

[2]Relative SOC stock differences were calculated as SOC stock in rubber plantation minus SOC stock of the reference secondary forest divided by SOC stock of the reference secondary forest multiplied by 100. Relative SOC stock differences were correlated with explanatory variables of the rubber plantations.

2.4 Discussion

Impact of land-use changes on soil organic carbon stocks

Our findings of decreased SOC stocks under rubber plantations, not only in the top 0.6-m depth but also when considering the whole 1.2-m depth, are consistent with published studies of paired comparisons and chronosequences that all reported SOC losses following the conversion from primary or secondary forests to rubber plantations (Araujo et al., 2004; Salimon et al., 2009; Yang et al., 2004).

To estimate the effects of land-use changes on SOC stocks it is crucial to account for changes in soil bulk density. The importance of correction for bulk density changes has been emphasized by many authors (Davidson and Ackerman, 1993; de Koning et al., 2003; Detwiler, 1986; Veldkamp, 1994), but it was not applied in the published studies on forest-to-rubber plantation conversions (Araujo et al., 2004; Salimon et al., 2009; Yang et al., 2004). We examined the effects of bulk density changes on the estimated land-use change effects for our own data and for the cited studies that reported data on depth, bulk density, and SOC concentrations (Salimon et al., 2009; Yang et al., 2004). Comparison of corrected and uncorrected estimates showed that in these studies, not accounting for bulk density changes resulted in overestimations up to 5% and underestimations as high as 18% of the relative SOC stock difference (Table S2.2). Errors were greatest for the top 0.3-m depth. This comparison illustrates again that ignoring bulk density changes potentially causes large errors in land-use induced SOC stocks changes. For the following discussion of forest-to-rubber plantation conversions, we used the corrected values of relative SOC stock changes.

The only published study on secondary forest-to-rubber plantation conversion was conducted in Xishuangbanna on an Udic Ferrisol (Chinese classification system) (Yang et al., 2004). Yang et al. (2004) observed SOC losses (corrected for bulk density changes) of 24% in a 3-year-old rubber plantation and 21% in a 7-year-old rubber plantation in the top 0.6-m depth. The SOC stocks in rubber plantations and secondary forests reported by Yang et al. (2004) were comparable to our estimated SOC stocks (Table 3). In Brazil, the conversion from primary forests to rubber plantations resulted in SOC losses of 21% down to 0.5-m depth in a 22-year-old rubber plantation on an Ultisol (Araujo et al., 2004), and of 48% down to 1.0-m depth in a 17-year-old rubber plantation on an Oxisol (Salimon et al., 2009). The SOC losses observed by Araujo et al. (2004) and Yang et al. (2004) correspond well with our observed declines of 24% at 0-0.3-m depth and 21% at 0-0.6-m depth (Table 3). However, the 48% decline down to 1-m depth reported by Salimon et al. (2009) is much larger than the 19% we observed down to 1.2-m depth.

Important methodological differences exist between our study and the cited studies. The large decrease in SOC stocks reported by Salimon et al. (2009) should be interpreted with care, since in their study soil samples below 0.10-m depth were taken from the middle of each depth interval instead of sampling the entire depth interval, which can lead to inaccurate estimations of the SOC stocks. Furthermore, the studies from Araujo et al. (2004) and Salimon et al. (2009) consisted of only one forest and one rubber plantation and the study from Yang et al. (2004) consisted of two rubber plantations and two forests. Results from such case studies with no or insufficient replications should not be extrapolated to a large scale. Finally, Yang et al. (2004) and Salimon et al. (2009) established in each land-use type one plot where they took no more than three replicate soil samples per depth interval.

At first sight, our observed decline in SOC stocks of 24 ± 2.5% in the top 0.3-m depth seems to be much larger than the insignificant effects reported for the compiled studies of the whole tropics for forest-to-tropical tree plantation conversions (Powers et al., 2011). However, analysis of the supplementary dataset from Powers et al. (2011) revealed that 12 out of the 15 tree plantation types included in their meta-analysis showed SOC losses of 19 ± 4.4% down to 0.3-m depth. The overall insignificant effect reported in their study resulted from the large number of observations from the other three plantation types (comprising 36% of all observations) that showed SOC accumulation, thereby offsetting the SOC losses. Our observed decline was thus within the same magnitude as carbon losses reported for many other tropical tree plantations. Taken as a whole, tropical tree plantations established after forest conversion appear to be more prone to SOC losses than previously reported.

The space-for-time substitution approach used in our study has as advantage that long-term SOC stock dynamics can be studied within a relatively short time period. However, this approach has also disadvantages: (i) It requires the untestable assumption that the land-use changes were random in the landscape regarding forest SOC stocks, and (ii) the spatial variation in other biophysical conditions of plots within a cluster (i.e. each cluster included a reference secondary forest and one to three rubber plantations) may interfere with land-use change effects and time trends. To deal with these limitations we carefully selected plots for comparison within a cluster and the clusters were replicated spatially. We included seven replicated clusters, which reduces the chance that SOC stock differences between rubber plantations and secondary forests are not due to land use.

Soil organic carbon losses related to years since land-use change

The observed exponential decrease in SOC stocks in the top 0.15-m depth in rubber plantations with years since land-use change is similar to trends often reported for forest-to-agriculture conversions (Dalal and Mayer, 1986; Jenny, 1941; Zingore et al., 2005). For our study, the potential drivers of the rapid SOC losses during the first five years following land-use change are: (i) soil disturbances during site preparation and terrace construction, which may accelerate soil surface erosion and increase SOC decomposition, and (ii) the sparse vegetation cover in young plantations, which reduces the SOC input and may change the microclimatic conditions, the latter could in turn result in enhanced SOC decomposition rates. The reduced SOC losses in older plantations may be explained by the denser vegetation cover, thereby increasing SOC input and soil stability. Management practices that are likely to affect the SOC balance during the entire rotation period of the rubber plantation are terrace maintenance, rubber tapping, fertilization and the removal of the vegetation understory.

Although a SOC steady state was reached approximately 20 years after conversion, we expect the land-use change induced SOC losses in this region to continue for a longer period of time. This is because the lifespan of rubber plantations in this region ranges between 30-50 years, and it is a common practice to establish new rubber plantations after felling the previous plantation. Site preparation for the new rubber trees involves the establishment of new terraces, which would be accompanied with a further decline in the SOC stock.

Environmental controls on soil organic carbon concentrations and soil organic carbon losses

In rubber plantations and secondary forests, the positive correlation of SOC concentrations with silt and clay content (Table 4) is consistent with literature and can be explained by the chemical and physical stabilization mechanisms of clay and silt particles (Schimel et al., 1994). The correlation between altitude and SOC concentrations, as we observed for rubber plantations, is often related to temperature effects. However, a temperature gradient is probably not the cause of the observed correlation, considering the relatively small altitude range of the sampled rubber plantations (700-830 m). Most likely the observed correlation reflects the marginally significant relationship between altitude and the sum of silt and clay content in the rubber plantations (data not shown). In secondary forests, the positive correlation of SOC concentrations with total basal area (which reflects forest productivity) suggests that increases of above-ground biomass could increase SOC input through increased input of leaf litter and root residues. This implies that in this region, restoration of the above-ground biomass in degraded secondary forests may increase SOC concentrations. The negative correlation between SOC stock differences and plantation age in 0-0.15-m depth attests that SOC stocks in rubber plantations progressively declined with increasing plantation age. This trend was also reflected in the negative correlation between SOC stock differences and total basal area due to the collinearity between plantation age and total basal area. The positive correlation between SOC stock differences and altitude may reflect the previously described soil texture gradient in rubber plantations with altitude.

Changes in above-ground carbon stocks versus soil organic carbon losses

It is often assumed that SOC emissions arising from deforestation and forest degradation in the tropics are relatively small compared to above-ground carbon losses (Detwiler, 1986; Fearnside and Imbrozio Barbosa, 1998; Houghton and Hackler, 1999). In our case, comparison of the observed SOC losses with the estimates of above-ground carbon changes based on regional data (Li et al., 2008) reveals that forest conversion to rubber plantations had a much larger effect on SOC stocks than on above-ground carbon stocks. Above-ground carbon stock estimates for forests outside nature reserves range from 32.2-71.0 Mg C ha^{-1} with a mean of 53.2 ± 2.1 Mg C ha^{-1} (Li et al., 2008). We assumed that these forests may reflect the conditions of our sampled forests, which were also situated outside nature reserves. For rubber plantations, means of above-ground carbon stock estimates range from 61.4 Mg C ha^{-1} for plantations < 800-m altitude to 35.1 Mg C ha^{-1} for plantations between 800 -1000-m altitude (Li et al., 2008). We used these estimates as our sampled plantations were situated between 700 and 830-m altitude. Together, this indicates that land-use change may results in above-ground carbon stock changes ranging from a loss of 18 Mg C ha^{-1} to an increase of 8 Mg C ha^{-1}. These estimates are in agreement with our data on total basal area (Table 2), for which we did not detect differences between the total basal area of forests and rubber plantations. Such changes in above-ground carbon stocks were much lower than the SOC loss of 37.4 ± 4.7 Mg C ha^{-1} for the entire 1.2-m depth.

Consequences of observed soil organic carbon losses

We showed that the conversion from secondary forests to rubber plantations leads to diminishing SOC stocks and that this decline is much larger than the changes in above-ground carbon stocks. Given the clear pattern of our locally collected data, it is likely that in montane mainland Southeast Asia on comparable soils, this land-use change may cause declines in the SOC stock, the magnitude of which will depend on site-specific biophysical conditions and management practices. The size of the observed losses has implications for the estimates of ecosystem carbon fluxes arising from land-use changes in the national inventories based on the IPCC guidelines. The IPCC tier 1 method does not include SOC fluxes for the forest-to-rubber plantation conversions (Aalde et al., 2006; Lasco et al., 2006). Our findings show that SOC changes need to be included to avoid possibly large errors in the estimates of the overall ecosystem carbon fluxes.

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CHAPTER 3

Soil redistribution by terracing alleviates soil organic carbon losses caused by forest conversion to rubber plantation

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Abstract

Secondary forest-to-rubber (*Hevea brasiliensis*) plantation conversion is an important recent land-use change in montane mainland Southeast Asia. This land-use conversion caused a reduction of soil organic carbon (SOC) stocks by on average 19% down to 1.2 m over 46 years. Due to the mountainous topography of the region, most rubber plantations include narrow terraces parallel to contours. Manual terrace construction involves cutting of the soil from the upper slope and piling up the removed soil on the soil surface downslope. Soil redistribution by terrace construction may affect SOC dynamics through exposure of the subsurface soil at the terrace inner sides (cut section) and soil burial at the terrace outer edges (fill section).

Our study, conducted in southern Yunnan province of China, aimed to quantify SOC stock changes induced by terrace construction. In three rubber plantations aged 5, 29 and 44 years, we systematically sampled the terraces according to soil redistribution zones, and the original sloping areas in between the terraces were used as reference.

At the cut section of the terrace, topsoil removal caused a depletion of SOC stocks in the youngest plantation followed by SOC stock recovery in the two oldest plantations. The recovery of SOC stocks at the cut section in the two oldest plantations was attributed to the capacity of the exposed subsurface soil to store new organic carbon inputs from roots and litter, and to sedimentation of eroded topsoil materials from the upper slope. At the fill section of the terrace, soil deposition resulted in higher total SOC stocks compared to the reference position in all plantations. This was due to the deposition of redistributed soil material on top of the original soil surface combined with the partial preservation of carbon in the buried soil. Overall, the increase of SOC in the exposed subsurface soil at the cut sections, and the partial preservation of SOC in the buried soil at the fill sections resulted in higher SOC stocks down to 1.2 m at the terraces compared to the reference positions in the two oldest plantations. Our results imply that terracing may alleviate SOC losses caused by the conversion of secondary forest to terraced rubber plantation.

3.1. Introduction

Land-use changes from natural to managed ecosystems have been estimated to contribute 12-15% to global anthropogenic CO_2 emissions through biomass removal and decomposition of soil organic carbon (SOC) (van der Werf et al., 2009). Soil disturbance with land-use changes is a major factor influencing the carbon flux between soil and the atmosphere (Amundson, 2001). Although terraces are regarded as conservation measures to retain soil and water, terrace construction and maintenance are typically accompanied by severe soil disturbances (Sidle et al., 2006) and thus have the potential to modify SOC dynamics.

Terracing is a widely distributed practice in rubber (*Hevea brasiliensis*) plantations in montane mainland Southeast Asia, spanning southwest China, Laos, Myanmar, northeast Thailand and northwest Vietnam. Originally, rubber trees were not grown in this region but rubber plantations have rapidly expanded in the last decades. At present, rubber plantations cover approximately 1.5 million hectares (Li and Fox, 2012), and this area is predicted to quadruple by 2050 (Fox et al., 2012). Existing rubber plantations were mainly established on areas which used to be forests or swidden land (Li and Fox, 2012; Xu, 2006). In a previous study (de Blécourt et al., 2013), we showed that the conversion from secondary forests to terraced rubber plantations led to SOC losses of, on average, 37.4 ± 4.7 (SE) Mg C ha^{-1} down to 1.2 m over 46 years, which was equal to 19.3 ± 2.7 (SE)% of the initial SOC stocks in the secondary forests. However, the mechanisms responsible for the observed losses and the role of terracing on SOC dynamics were unclear.

Terraced rubber plantations typically consist of narrow terraces parallel to contours, which contain a single row of rubber trees, alternated by the original sloping areas. Terrace establishment involves cutting of the soil from the upper slope and piling the removed soil on the surface of the down slope position. This results in the exposure of the subsurface soil at the terrace's inner edge (cut section) and burial of the original topsoil at the terrace's outer edge (fill section, Figure 1a).

To our knowledge, the impact of terracing on SOC stocks has not yet been studied in detail. However, studies on the effects of soil erosion on SOC stocks have shown that soil redistribution not only redistributes SOC within the landscape but also affects SOC dynamics (Gregorich et al., 1998; van Oost et al., 2007) - SOC dynamics differ between the eroding sites and depositional sites. Similar to soil erosion, terrace construction also results in positions where soil is removed (cut section) and positions where soil is deposited (fill section). The impacts of terrace construction on SOC dynamics may thus, to a certain extent, be comparable to mechanisms proposed for erosional impacts. First, topsoil removal at the cut section will cause an initial depletion of SOC. However, the SOC in the exposed subsurface soil may rapidly increase due to a relatively large capacity of the exposed soil to store newly added organic matter (Harden et al., 1999; Stallard, 1998). The relatively large storage capacity for carbon of the exposed subsurface soil is related to the initially low SOC concentration of this exposed soil and large number of binding sites for carbon from clay surfaces, and Al and Fe hydroxides/oxides. Second, since decomposition rates typically decrease with depth (Van Dam et al., 1997), burial of the original topsoil at the fill section may lead to a reduction in SOC decomposition and to the stabilization of SOC (Doetterl et al., 2012; Stallard, 1998; VandenBygaart et al., 2012). Mechanisms responsible for the reduced SOC decomposition in subsurface soils are still an issue of debate; hypothesized mechanisms are, among others, the occlusion of organic matter in soil aggregates (Moni et al., 2010) and the limited supply of fresh organic matter (plant litter and root exudates) which serves as an energy source to microbes (Fontaine et al., 2007). Third, the breakdown of soil aggregates by soil excavation and redistribution may accelerate SOC decomposition due to the exposure of previously protected SOC (Elliott, 1986). Fourth, terraces may trap eroded soil material from upper slopes, resulting in SOC input. Whether terracing leads to a net SOC loss or gain depends on the magnitude by which these four potential mechanisms alter SOC dynamics.

Here we present a case study in Xishuangbanna prefecture in southern Yunnan province of China where we quantified the SOC stock changes in rubber plantations induced by terrace construction. In three rubber plantations of increasing age, we sampled the terraces according to soil redistribution zones, using the original sloping areas between the terraces as reference. SOC stocks of the terrace were compared with stocks of the original slopes for which we assumed that the SOC stocks were not affected by terrace construction. We tested the following hypotheses: (1) SOC stocks in the cut sections of the terraces are lower than at the reference positions but this difference diminishes with increasing plantation age, and (2) SOC stocks in the fill sections of the terraces are higher than on the reference positions.

3.2. Materials and Methods

Site description and rubber plantation management

The study was conducted in Menglong township, Jinghong county of Xishuangbanna prefecture in Yunnan province of China (21˚31’17.03”N, 100˚37’12.13”E). The region has a mountainous topography with elevations between 650 m and 1450 m above sea level. The climate is tropical monsoon; the dry season usually occurs from November to April and the wet season from May to October. Mean annual temperature is approximately 22.7 ˚C and mean annual precipitation is 1370 mm (Xu et al., 2005). The sampling plots had elevations between 751-779 m above sea level and slopes of 38-44%. The deeply weathered soils at the sampling plots were dominated by low activity clays and were classified as Ferralsols (IUSS Working Group WRB., 2006) with thin A horizons, an effective cation exchange capacity in the subsurface soil of less than 12 $cmol_c\ kg^{-1}$ clay, soil texture ranging from clay loam to clay and soil pH (H_2O) between 4.6 and 5.6 (Table 1).

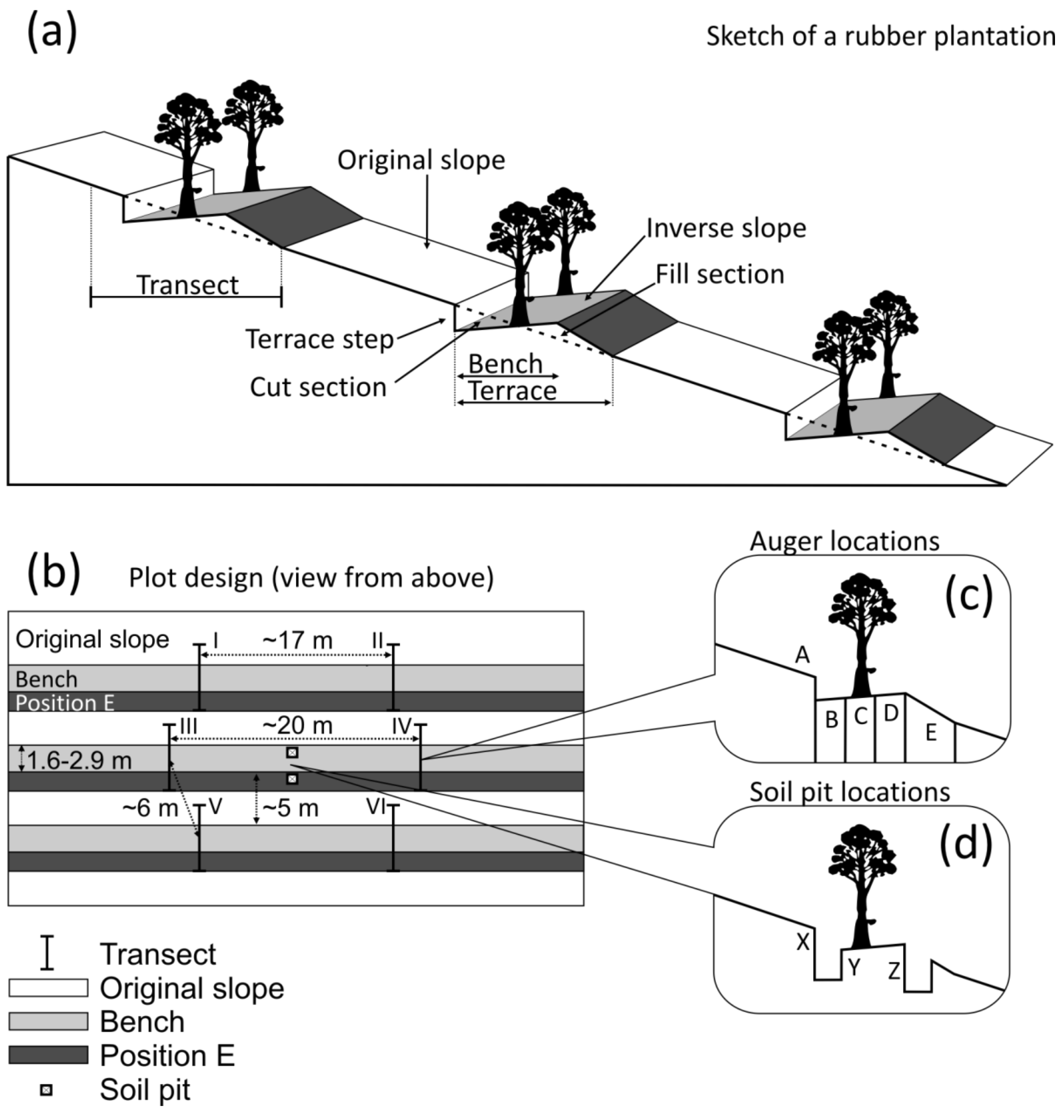

Figure 1. Experimental design. (a) Sketch of a rubber plantation and terminology used in the text. (b) Plot design: Each plot consists of six transects (I-II-III-IV-V-VI) on three terraces with a distance ranging between 6-20 m. (c) Auger locations: Each transect was stratified into five terrace positions (A-B-C-D-E). (d) Soil pit locations: Two soil pits were dug in the center of the six transects from which three profile walls (X-Y-Z) were used for soil and bulk density sampling.

Table 1. Mean (SE) soil characteristics at the original slope as terrace position A[1] in three rubber plantations.

Characteristics	Depth (m)	5-year old plantation	29-year old plantation	44-year old plantation
Sand (%)[2]	0-0.10	42.2 (0.4)	41.7 (1.0)	25.6 (0.6)
	0.45-0.60	37.0 (1.0)	38.2 (0.2)	21.6 (2.6)
	0.90-1.20	37.4	34.2	12.6
Silt and clay (%)[2]	0-0.10	57.8 (0.4)	58.3 (1.0)	74.4 (0.6)
	0.45-0.60	63.0 (1.0)	61.8 (0.2)	78.4 (2.6)
	0.90-1.20	62.6	65.8	87.4
Bulk density ($g\ cm^{-3}$)[3]	0-0.10	0.89	1.09	1.16
	0.45-0.60	1.32	1.35	1.25
	0.90-1.20	1.41	1.29	1.19
pH (H_2O)[2]	0-0.10	4.6 (0.1)	4.6 (0.0)	4.7 (0.1)
	0.45-0.60	4.7 (0.1)	4.9 (0.1)	5.0 (0.0)
	0.90-1.20	5.1	5.2	5.6
ECEC ($mmol_c\ kg^{-1}$ soil)[4]	0-0.30	52.6	68.7	45.5
	0.90-1.20	30.7	40.8	26.9

[1]Terrace positions are described in Figure 1c.
[2] n = 3 for 0-0.10 m and 0.45-0.60 m, and n = 1 for 0.90-1.2 m.
[3] n = 1 for all depths.
[4] n = 1 for ECEC, Effective Cation Exchange Capacity, measured at an adjacent plot in the same plantation (de Blécourt et al., 2013).

According to local plantation workers, management practices typically applied in sloped rubber plantations include: (1) terrace construction and maintenance, (2) fertilizer application, (3) pest control, (4) removal of the vegetation understory and (5) rubber tapping. Rubber tapping is done in plantations with an age of more than 7 years and is usually done every second day from April to October. Terraces are built by hand with a hoe as a soil conservation measure, to support water infiltration, and as footpaths for plantation workers to facilitate rubber tapping. In the first decades, terrace maintenance involves cleaning of the terrace step (Figure 1a) by scraping of several centimeters of the soil (~0.03 m $year^{-1}$) and redistributing the soil material over the terrace bench. Since this process is repeated once or twice a year, older plantations have wider terrace benches and deeper terrace steps. Only in the oldest plantations is terrace maintenance no longer applied. In some plantations, soil pits are dug between rubber trees as a measure to reduce runoff and retain soil moisture. These pits are also used by farmers as a place to apply fertilizer and collect cut herbs, and the pits function as a trap for leaf litter. The size of these pits is within these ranges: 0.4-1.3-m length x 0.2-0.5-m width x 0.2-0.4-m depth. Terrace bench widths range from 1.6-2.9 m, and terrace step heights range from 0.3-0.8 m. In some plantations, terraces are slightly inverse sloping with gradients between 0-11% (Figure 1a). The horizontal distance between two adjacent terraces is ~5 m, depending on the slope. Rubber trees were planted in a single row on the terraces and tree spacing ranged from 2-3 m.

Sampling design

We selected three rubber plantations of 5, 29 and 44 years old. All selected plantations were established immediately after clearing of secondary forests. We selected one plot in each rubber plantation, and all plots of the three rubber plantations had similar slope, aspect, and soil type. Under the assumption that these biophysical conditions of the plantations did not substantially differ, we attributed observed differences in terracing effects on SOC across plantations to plantation age. Since the ages of the rubber plantations were not replicated the observed trends cannot be generalized to a larger area.

Within each plot we established six transects which covered three sequential terraces, the distances between the terraces ranged from 6 to 20 m (Figure 1b). The six transects in each rubber plantation were used as replicates. Transects were oriented perpendicularly to the terraces. We stratified each transect into five terrace positions (A to E, Figure 1c) according to soil redistribution zones, covering positions on the original sloping area between two adjacent terraces (position A), and on the cut (positions B and C) and fill sections (positions D and E) of the terrace. We defined a terrace as the inverse-sloping terrace bench and the outer side of the fill section situated downslope of the bench (Figure 1a). Terrace position A was our reference and was situated on the original slope, 1 m from the terrace step; this position showed no evidence of terrace construction and maintenance. We choose our reference position within the same plantation since plantations without terraces do not occur in the study area and this also reduced problems with inherent spatial variability, which is high in this mountainous area. Position B was situated at the terrace bench 0.2 m from the terrace step. Position C was situated in the middle of the terrace bench. Position D was located at the terrace bench 0.2 m from the outer edge. Position E was situated down slope of the terrace bench, and the distance of position E to the bench was a third of the bench width.

We dug two soil pits in the center of the six transects (Figure 1b, Figure 1d). Soil pit 1 was located on the terrace bench immediately next to the terrace step; the side of the soil pit facing upslope (soil profile X) represented terrace position A, and the side of the soil pit facing downslope (soil profile Y) represented terrace positions B and C. Soil pit 2 was located at the fill section situated downslope of the terrace bench; the upslope-facing side (soil profile Z) represented terrace positions D and E. We were not able to dig a separate soil pit for each terrace position, as this would have caused major disruptions of the terraces in the rubber plantations.

Soil and litter sampling and auxiliary data

Soil samples were taken over a depth of 1.2 m at seven sampling depths: 0-0.1 m, 0.1-0.2 m, 0.2-0.3 m, 0.3-0.45 m and 0.45-0.6 m, 0.6-0.9 m, and 0.9-1.2 m. The top five depths were sampled at each terrace position in each of the six transects per rubber plantation (n = 6, for each depth per terrace position and plantation age), using a Dutch auger. The soil samples of 0.6-0.9-m and 0.9-1.2-m depth were taken at three terrace positions (A, combined B and C, and combined D and E) once in each plantation (n = 1, for each depth per terrace position and plantation age). The samples were taken from soil profiles X, Y, and Z corresponding to positions A, combined B and C, and combined D and E, respectively (Figure 1d). By combining terrace positions, we assumed that the variation in soil characteristics within the combined terrace positions was minimal at depths > 0.6 m. Since it was not possible for the soil profile Y to be dug deeper than 0.9 m, we sampled the 0.9-1.2-m depth with a soil auger in the center of the soil pit. Independent of the sampling depths, we also sampled the buried Ah horizons which we observed in soil profile Z. Soil bulk density was measured for all sampling depths from the soil profiles X, Y and Z using the core method (Blake and Hartge, 1986). Bulk density was not measured for 0.9-1.2-m depth in soil profile Y. The bulk density samples did not contain stones > 2 mm.

Samples from the litter layer (including leaves, seeds and twigs) were collected from each terrace position (n = 6, for each terrace position per plantation age) from a 0.04-m^2 quadrant sample frame. The collected material was oven dried at 60 ˚C for 48 h and weighed.

Finally, we measured the following terrace characteristics at each transect in each plantation: terrace width, distance between the terraces, inclination of terraces, and step height. All measures of distance were corrected for slope. The topographical data collected included: slope, aspect, altitude, and Global Positioning System coordinates.

Laboratory analyses

We analyzed the soil samples for soil texture, pH (H_2O), and total carbon and nitrogen concentration. Samples were air dried (five days) and sieved (mesh size: 2 mm) prior to analyses. Carbon and nitrogen were measured on each sample. Samples were ground and analyzed for carbon and nitrogen by dry combustion using a CNS Elemental analyzer (Elementar Vario EL, Hanau, Germany). As soil pH (H_2O) was below 5.6, carbonates were not expected in the samples and carbonate removal was not necessary. For soil texture and pH, samples from the two transects per terrace were pooled for each terrace position and sampling depth, resulting in three replicates (n =

3) for each depth per terrace position and plantation age. Soil pH (H_2O) was analyzed in a 1:2.5 soil-to-water ratio. Soil texture was analyzed using the pipette method distinguishing the fractions clay (< 0.002 mm), silt (0.002-0.063 mm), and sand (0.063-2 mm).

Calculations

SOC stocks in each sampling depth were calculated using the bulk density (BD) data and thickness of the sampled depth intervals (Δ D):

$$SOC\ stocks(kg\ C\ m^{-2}) = \frac{\%C}{100} \times BD\ (kg\ m^{-3}) \times \Delta\ D\ (m),$$

We calculated total SOC stocks over a depth of 0-1.2 m as the cumulative SOC stocks of all sampling depths. Terrace construction is expected to cause differences in soil bulk density across terrace positions. To compare SOC stocks between terrace positions based on equal soil mass, and to avoid the interference of bulk density changes with SOC stock changes, we used the average bulk density for each depth from the three soil profiles per rubber plantation to calculate the SOC stock.

SOC stocks down to 1.2 m at the terrace were calculated as area-weighted average of positions B-C-D-E for each transect. Positions B-C-D were assumed to have equal area over the terrace bench. Their weights were defined accordingly as the ratio of a third of the bench width to the total terrace width, in which terrace width was defined as the sum of the bench width (positions B-C-D) and the width of the fill section downslope of the bench (position E) (Figure 1a). The weight of position E was defined as the ratio of the width of position E to the total terrace width.

Litter layer carbon stocks were calculated using the litter carbon concentration (%), the mass of the litter layer, and the sample frame surface. We used a litter carbon concentration of 41%, which is the mean litter carbon concentration in rubber plantations in the same study area (de Blécourt et al., 2013).

Statistical analyses

Prior to statistical analyses, we tested all data for normality (Shapiro-Wilk test) and equality of variances (Levene's test). If these assumptions were not met, data were log-transformed and the analysis was repeated. If transformation did not approximate normality and equality of variance, we used non-parametric tests. We tested the differences between terrace positions (A to E) in litter carbon stocks, in total SOC stocks down to 1.2 m, and in SOC concentrations for each sampling depth using either one-way analysis of variance (ANOVA) followed by a Tukey-HSD test or Kruskal-Wallis ANOVA followed by a pairwise Wilcoxon test with Holm's correction for multiple comparisons.

Differences in total SOC stocks down to 1.2 m between the terrace (weighted values from positions B to E) and reference position A were tested using Independent t tests.

To make inferences on the changes in SOC concentrations in the exposed subsurface soil at the cut section (position B) relative to the original SOC concentrations prior to terracing (reference position A), we compared the exposed subsurface soil at the cut section with the equivalent subsurface soil at reference position A, using either an Independent t test (for the top 0.6 m), a One-sample t test (for the depths below 0.6 m) or a Mann-Whitney Wilcoxon test. For this, we estimated the original soil depth of the exposed subsurface soil at the cut section by the sampling depth plus the height of the terrace step, assuming that the step height equaled to the depth of soil excavation. Inferences on SOC changes in the buried soil at the fill section (position E) relative to its original SOC concentrations (reference position A) were based on the comparison of the SOC concentrations of the buried Ah horizon in soil profile Z (positions D and E) with the SOC concentrations in the top 0.10-m depth at reference position A, using a One-sample t test.

To test the impact of plantation age on the differences in total SOC stocks between the terrace and reference position A, we calculated the ratios of total SOC stocks at the terrace to the total SOC stocks at the reference position for each of the three rubber plantations. The differences in ratios between the plantation ages were tested using a one-way ANOVA followed by a Tukey-HSD test. To test whether the terrace-to-reference position A ratio of SOC stocks was different from 1, we used One-sample t tests.

In all tests, significant differences were accepted at $P \leq 0.05$. We considered differences with P-values from 0.06-0.08 as marginally significant. All statistical analyses were carried out using the open source software R version 2.15.0 (R Development Core Team, 2012).

3.3. Results

Soil redistribution patterns, soil characteristics and litter layer

The magnitude of soil redistribution progressed with plantation age as shown by the height of the terrace step at the cut section (position B), terrace width (positions B to E) and the depth of the buried Ah horizon at the fill section (positions D and E). The terrace step height, which we assumed to be equal to the depth of soil excavation, was 0.33 ± 0.02 (SE) m in the 5-year old, 0.79 ± 0.08 (SE) m in the 29-year old, and 0.71 ± 0.06 (SE) m in the 44-year old rubber plantations. Terrace width increased from 2.61 m, 3.63 m, to 4.65 m in the 5-year old, 29-year old and 44-year old plantations. At the fill section, the buried Ah horizon started at a depth of 0.1 m in the 5-year old, 0.45 m in the 29-year old, and 0.8 m in the 44-year old plantations. Soil texture differed between terrace positions in the 5-year old and 44-year old plantations, but the trends of differences were not consistent across the two plantations (Table S3.1). In the 29-year old plantation, soil texture was the same across terrace positions. Soil pH did not differ between terrace positions in each plantation (Table S3.1). The litter layer carbon stocks at the cut section (position B) tended to be the highest in each plantation (Figure 2). However, these differences were only statistically significant for comparisons between the cut section (position B) and the fill section (positions D and E) in both the 5-year old and 29-year old plantations ($P < 0.01$-0.03). No differences between terrace positions were detected in the 44-year old plantation.

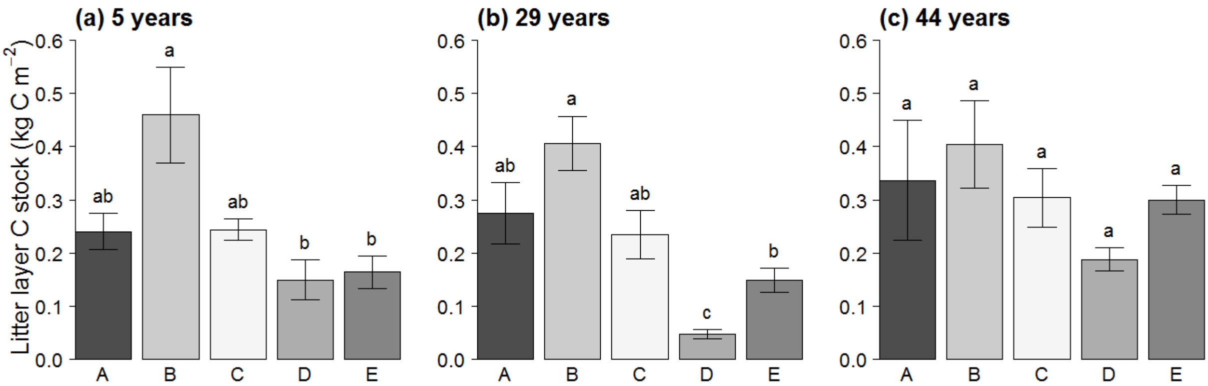

Figure 2. Litter layer carbon stocks at terrace positions A-B-C-D-E in a (a) 5-year old rubber plantation, (b) 29-year old rubber plantation, and (c) 44-year old rubber plantation. Terrace positions are described in Figure 1c. Means (SE bars, n = 6) having different letters are significantly different between terrace positions (one-way ANOVA with Tukey HSD, or Kruskal-Wallis ANOVA with pairwise Wilcoxon test at $P \leq 0.05$).

Soil organic carbon concentrations and stocks at each terrace position

For the entire depth of 1.2 m, SOC stocks at the cut section (position B) were lower than at reference position A in the 5-year old plantation ($P < 0.01$) but no differences between these terrace positions were detected for the two oldest plantations (Figure 3). At the terrace center (position C) in the two youngest plantations, SOC stocks did not differ from reference position A, while in the 44-year old plantation SOC stocks at the terrace center were lower than at reference position A ($P = 0.02$). In all three plantations, SOC stocks were highest at the fill section, represented by positions D and E ($P < 0.01$-0.02).

The cut section (position B) in the youngest plantation had lower SOC concentrations down to 0.3 m than reference position A ($P < 0.01$-0.05) (Figure 4a). However, in the two oldest plantations, the depth of 0-0.1 m at the cut section tended to have higher SOC concentrations compared to reference position A ($P = 0.07$ for the 29-year old and $P = 0.01$ for the 44-year old plantation) (Figure 4b and 4c). Comparisons of SOC concentrations based on the original soil depth (i.e., the soil depth prior to terracing) showed that in the 5-year old plantation, SOC concentrations in the exposed subsurface soil at the cut section did not change relative to the subsurface soil at reference position A (Figure 5a). In contrast, the 29-year old and 44-year old plantations showed SOC concentrations at the cut section that were higher than the equivalent depths of subsurface soil at reference position A ($P < 0.01$-0.04) (Figure 5b and 5c).

At the fill section, a subsurface peak in SOC concentrations was observed at depths of 0.1-0.2 m in the 5-year old plantation (Figure 4a, position E), 0.3-0.45 m in the 29-year old plantation (Figure 4b, position E) and 0.6-0.9 m in the 44-year old plantation (Figure 4c, position E). These peaks were also observed at position D in each plantation (data not shown). The depths of these peaks corresponded to the starting depths of the buried Ah horizons at the fill sections (observed from soil profile Z). In the two youngest plantations, the upper 0.1 m of deposited soil, situated above the buried Ah horizon, had lower SOC concentrations compared to the top 0.1 m of reference position A ($P < 0.01$-0.05) (Figure 4a and Figure 4b). In the 44-year old plantation, the deposited soil (0-0.6 m) had similar SOC concentration as reference position A at the same depths (Figure 4c).

The buried Ah horizon at the fill section (soil profile Z) had SOC concentrations of 2.63% in the 5-year old, 2.08% in the 29-year old, and 1.56% in the 44-year old plantation (not shown in Figure 4). In the 5-year old plantation, SOC concentration in the buried Ah horizon did not differ from the top 0.10 m at reference position A. Conversely, in the 29-year old and 44-year old plantations the SOC concentrations in the buried Ah horizons tended to be lower than the top 0.10 m at reference position A ($P = 0.01$-0.06). There was a trend that the soil below the buried Ah horizon (subsurface peak in Figure 4) had higher SOC concentrations compared to reference position A at the same depths; however, this pattern was only statistically significant for the 29-year old plantation ($P < 0.01$).

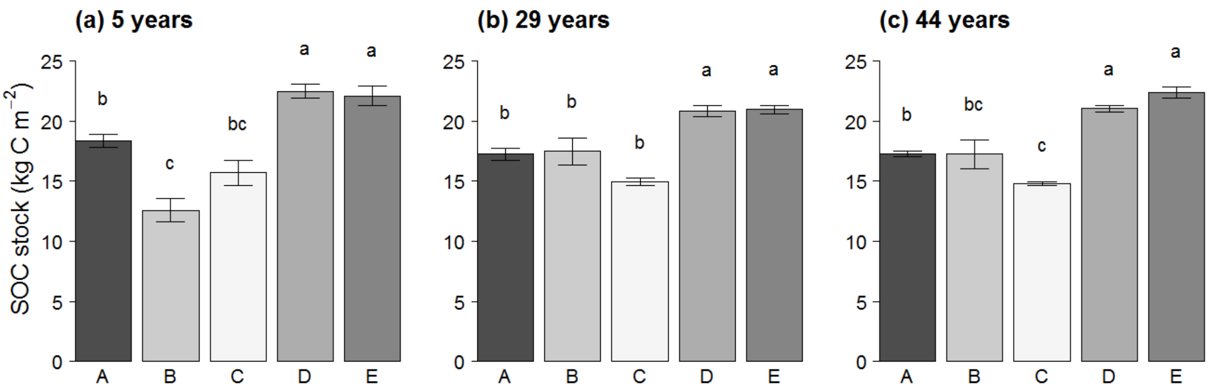

Figure 3. Soil organic carbon stocks over a depth of 0-1.2 m at terrace positions A-B-C-D-E in a (a) 5-year old rubber plantation, (b) 29-year old rubber plantation, and (c) 44-year old rubber plantation. Terrace positions are described in Figure 1c. Means (SE bars, n = 6) having different letters are significantly different between terrace positions (one-way ANOVA with Tukey HSD, or Kruskal-Wallis ANOVA with pairwise Wilcoxon test at $P \leq 0.05$).

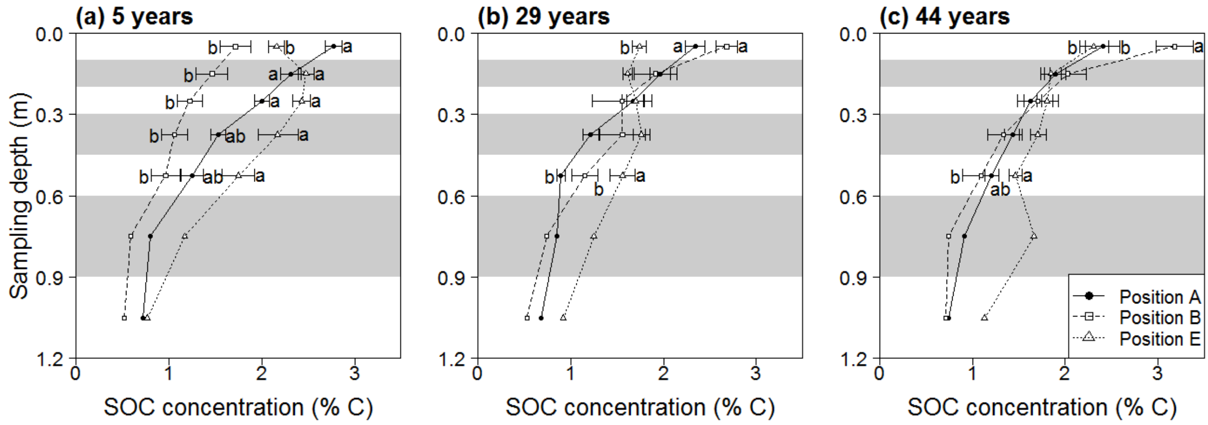

Figure 4. Soil organic carbon concentrations in relation to sampling depth. Results are shown for the terrace positions A (black line), B (coarse dashed line) and E (fine dashed line) in a (a) 5-year old rubber plantation, (b) 29-year old rubber plantation, and (c) 44-year old rubber plantation. Terrace positions are described in Figure 1c. Alternating white and grey bands show the sampling depths. For each depth, means (SE bars, n = 6) having different letters are significantly different between terrace positions; means without letters are not significantly different (one-way ANOVA with Tukey HSD, or Kruskal-Wallis ANOVA with pairwise Wilcoxon test at P ≤ 0.05). For the depths of 0.6-0.9 m and 0.9-1.2 m, soil samples were taken from one soil profile (n = 1) per terrace position.

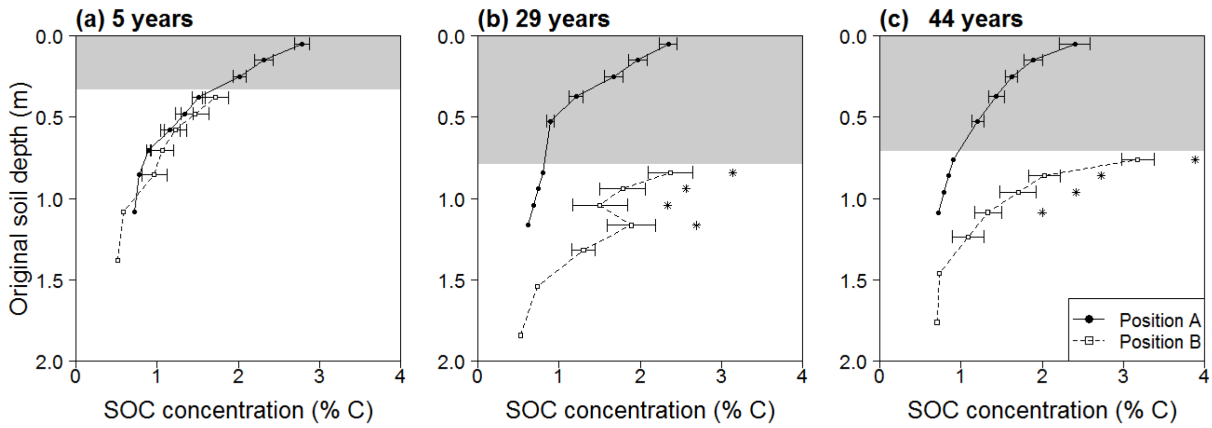

Figure 5. Soil organic carbon concentrations in relation to the original soil depths (prior to terracing) for positions A (black line) and B (coarse dashed line) in a (a) 5-year old rubber plantation, (b) 29-year-old rubber plantation, and (c) 44-year old rubber plantation. Terrace positions are described in Figure 1c. At position A, the original soil depth was the same as the sampling depth. At position B, the original soil depth was estimated as the sampling depth plus the depth of soil excavation. The depth of soil excavation is indicated by the grey shades. * Indicates significant differences between the means (SE bars, n = 6) of positions A and B (either One-sample t test, Independent t test, or Mann-Whitney Wilcoxon test at P-value ≤ 0.05). Means without SE bars correspond to the sampling depths of 0.6-0.9 m and 0.9-1.2 m, where soil samples were taken from only one soil profile (n = 1) per terrace position.

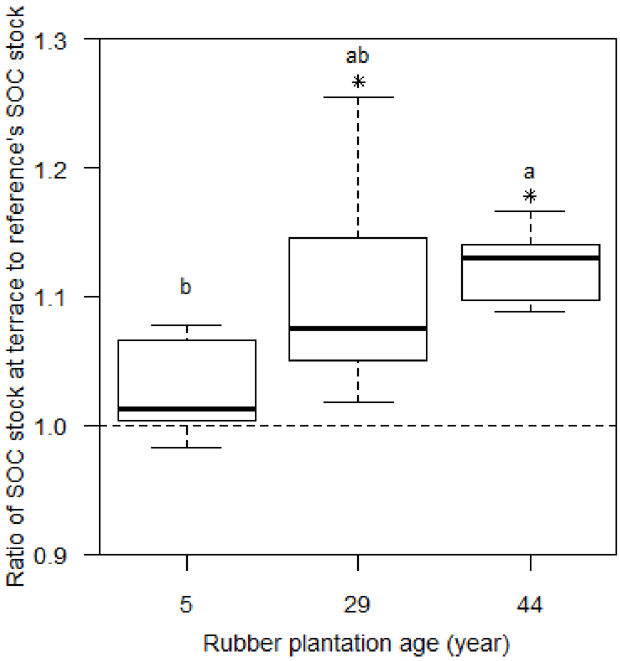

Figure 6. Ratio of soil organic carbon stocks over a depth of 0-1.2 m at the terrace (positions B-E) to carbon stocks at the original slope (position A) as reference position. Terrace positions are described in Figure 1c. In the box and whisker plot, the box shows the 25th and 75th percentiles and the median, and the whiskers extend to 1.5 times the interquartile range above or below the box. The horizontal dashed line marks the ratio at which soil organic carbon stocks at the terrace and at the reference position are equal. Box and whisker plots (n = 6) having different letters are significantly different between rubber plantations (one-way ANOVA with Tukey HSD test at $P \leq 0.05$). * Indicates that the mean ratio (n = 6) of a rubber plantation is higher than 1 (One-sample t test at $P = 0.06$ for 29-year old plantation and at $P < 0.01$ for the 44-year old plantation).

Table 2. Mean (SE, n = 6) soil organic carbon stock (kg C m^{-2}) in 0-1.2-m depth at the original slope as reference position (position A) and at the terrace (positions B-E)[1] in three rubber plantations.

Terrace position	5-year old plantation	29-year old plantation	44-year old plantation
Reference position A	18.34 (0.54) a	17.22 (0.50) b	17.28 (0.25) b
Terrace[1]	18.85 (0.82) a	19.13 (0.19) a	19.44 (0.39) a

Within a column, means having different letters are significantly different between reference position A and terrace (independent t test at $P \leq 0.05$).
[1]Area-weighted average of positions B-C-D-E. Terrace positions are described is Figure 1c.

Soil organic carbon stocks at the terrace

In the 5-year old plantation, SOC stocks down to 1.2 m were comparable between reference position A and the terrace (area-weighted average of positions B-C-D-E) (Table 2). In the 29-year old and 44-year old plantations, SOC stocks at the terrace were higher compared to reference position A ($P < 0.01$). The increased SOC stocks at the terrace were also indicated by the ratios of SOC stock at the terrace to SOC stock at reference position A, which were higher than 1 for the 44-year old plantation ($P < 0.01$), and marginally higher than 1 for the 29-year old plantation ($P = 0.06$) (Figure

6). Comparison of these ratios between plantations showed that the positive effect of terracing on SOC stocks was distinguishable only between the 5-year old and 44-year old plantations (P = 0.02).

3.4. Discussion

Soil organic carbon recovery in the exposed subsurface soil

Our hypothesis that SOC stocks in the exposed subsurface soil at the cut section of the terrace (position B) would be lower as compared to reference position A was only true for the 5-year old plantation (Figure 3). The total recovery of SOC stocks at the cut section that we detected in the two oldest plantations (Figure 3b and Figure 3c) was probably caused by continuous SOC sequestration in the exposed subsurface soil. The latter was shown by the increase in SOC concentrations at the cut section compared to the original depths at reference position A (Figure 5b and Figure 5c). Similarly, results from an erosion study on cropland in Belgium showed total recovery of SOC stocks at eroded slopes, which had been under agricultural use for centuries (Doetterl et al., 2012). However, much lower percentages of SOC stock recovery were estimated for eroding slopes in 10 agricultural watersheds in Europe and USA, ranging between 11-55% (van Oost et al., 2007). SOC stock recovery at the cut section in the two oldest plantations was in line with the concept of dynamic replacement of carbon in eroding landscape positions (Harden et al., 1999; Stallard, 1998). The dynamic replacement of carbon is often explained by the relatively large capacity of the exposed subsurface soil to store new organic carbon inputs from roots and litter (Harden et al., 1999; van Oost et al., 2007; VandenBygaart et al., 2012). Additionally, SOC input at the cut section might be larger compared to other terrace positions because of the relatively large amounts of accumulated leaf litter (Figure 2) and possibly also because of trapping eroded topsoil materials from the upper slope.

Impacts of soil redistribution and burial on soil organic carbon concentrations and stocks

The lower SOC concentrations in the deposited soil on top of the buried Ah horizon at the fill section (position E) in the two youngest plantations compared to reference position A (Figure 4a and Figure 4b) has two possible explanations: (1) the mixing of the topsoil with the underlying soil (which usually has lower SOC concentration) during soil excavation, and (2) soil excavation and redistribution may cause aggregate disruption and thereby accelerated SOC decomposition due to enhanced access of microbes to SOC (Elliott, 1986). In the oldest plantation, similar SOC concentrations in the deposited soil and the topsoil of reference position A (Figure 4c) indicated that the SOC in deposited soil had recovered to the original levels. This also coincided with the cessation of terrace maintenance in the oldest plantation, i.e. no new soil redistribution took place in the 44-year old plantation. The lower SOC concentrations in the buried Ah horizon (fill section, soil profile Z) compared to the topsoil of reference position A in the two oldest plantations, indicate that part of the carbon in the buried soil has been decomposed. The deeper burial of the Ah horizons in older plantations (Figure 4) was due to terrace maintenance, which caused an enhanced accumulation of redistributed soil at the fill sections with increasing plantation age. The trend towards higher SOC concentrations in the soil below the buried Ah horizon compared to reference position A at the same depths (Figure 4) suggests that carbon in the buried soils has been partially preserved. The higher SOC concentrations from 0.9-1.2 m (Figure 4b and Figure 4c) may also indicate preservation of SOC below a depth of 0.9 m. Partial preservation of the carbon in buried soils was also observed at depositional sites in eroding landscapes (Doetterl et al., 2012; VandenBygaart et al., 2012). Overall, these findings indicate that the higher total SOC stocks at the fill section (positions D and E) compared to reference position A (Figure 3), which is consistent with our second hypothesis, were due to the deposition of redistributed soil material on top of the original soil surface, combined with partial preservation of carbon in the buried soil.

Impacts of terracing on soil organic carbon stocks

The higher SOC stocks down to 1.2 m at the terraces in the two oldest plantations compared to the reference positions (Table 2 and Figure 6) can be attributed to the recovery of SOC at the cut sections and the partial preservation of SOC in the buried soil at the fill sections. The progressive increase of SOC stocks at the terrace, relative to the reference position, with increasing plantation age was in line with the recovery of SOC at the cut section, which was observed only in the 29-year old and 44-year old plantations (Figs. 3 and 5). It is important to place our results in the context that with conversion of secondary forest to terraced rubber plantation, SOC stocks down to 1.2 m decrease by an average of 19% across rubber plantations with ages between 5-46 years (de Blécourt et al., 2013). Our present study indicates that terracing may alleviate SOC stock losses; without the terraces the SOC stock losses caused by the conversion of secondary forest to rubber plantation could have been higher.

Acknowledgements

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CHAPTER 4

The importance of local processes on soil carbon stocks in a mountainous landscape in southern Yunnan, China

Marleen de Blécourt, Marife D. Corre, Rainer Brumme, Ekananda Paudel, Rhett D. Harrison, Edzo Veldkamp

Abstract

Information on soil organic carbon (SOC) distribution in relation to land use and biophysical properties is limited in montane mainland of Southeast Asia. Our study area is in the southern Yunnan province of China at the border with Myanmar. In an area of 10,000 hectares, we sampled 28 one-hectare plots from which 10 plots were closed canopy forests, 11 plots were open canopy forests, and seven plots were open lands utilized as tea plantations and shrub lands. We used a sampling design with a spatially nested structure that allows partitioning of the overall variance in SOC, vegetation, soil and topography to be attributed to the variation among land-use types, among sampling plots, and within sampling plots.

The SOC stocks to a depth of 0.9 m were 228.6 ± 19.7 (SE) Mg C ha^{-1} in closed canopy forests, 200.4 ± 15.5 Mg C ha^{-1} in open canopy forests, 197.5 ± 25.9 Mg C ha^{-1} in tea plantations, and 236.2 ± 13.7 Mg C ha^{-1} in shrub lands and were among the highest in the region. SOC concentrations and stocks did not differ across land-use types. In this mountainous terrain, more than 50% of the overall variance in SOC occurred within the one-hectare sampling plots. This shows that local processes are important for the overall variability of SOC in the landscape. Determinants for the short distance variability in SOC were tree basal area, litter layer carbon stocks, and slope gradient. In forests, we found higher SOC stocks at sampling plots located at higher elevation. In open lands, the variation in SOC concentrations among sampling plots was affected by land-use type; SOC concentrations in shrub lands were higher compared to tea plantations when controlling for the effects of litter layer carbon stocks and slope gradient. Our results on SOC concentrations and stocks in a mountainous landscape show the importance of local processes on the overall variance in SOC, and provide valuable baseline information for estimations of SOC distributions at a landscape scale.

4.1. Introduction

Worldwide, soils are the largest pool of terrestrial organic carbon, storing more carbon than the combined total of carbon in the atmosphere and vegetation (Schlesinger, 1997). The carbon pools in soil and atmosphere are tightly linked through photosynthesis and the decomposition of soil organic matter. The feedbacks between the soil organic carbon (SOC) pool and atmospheric CO_2 are sensitive to changes in land use and climate. Besides its role in the global carbon cycle, SOC is a key control of soil fertility, soil structure, and soil water holding capacity. SOC distribution is controlled by climate, topography, soil parent material, biota, time and human activity (Jenny, 1941). The importance of each controlling factor for SOC differs with spatial extent and environmental setting. Information on spatial distribution of SOC stocks is essential for investigations on how the SOC pool responds to global change and for the development of strategies that aim at enhancing SOC stocks.

In our present study, we quantified SOC concentrations and stocks and examined the effects of land-use type, vegetation, soil texture and topography on SOC, in a subtropical mountainous landscape in southern Yunnan province of China at the border with Myanmar. The area's land-use has been characterized by a long history of swidden agriculture (also called shifting cultivation or slash and burn) (Xu, 2006), which resulted in a mosaic of secondary forests, agricultural fields, paddy rice, tea plantations, and shrub lands. These mountainous landscapes extend throughout mainland Southeast Asia comprising southwest China and the northern areas of Laos, Myanmar, Thailand and Vietnam, covering approximately 180 million hectares (Garrity, 1993). Despite the region's large spatial extent, knowledge on SOC stocks and controlling factors are limited to northern Thailand (Aumtong et al., 2009; Pibumrung et al., 2008), and Laos (Chaplot et al., 2010, 2009; Phachomphon et al., 2010; Rumpel et al., 2008, 2006). We sampled SOC stocks, vegetation and soil, and recorded topographical parameters in an area of 10,000 hectares along a disturbance gradient from intensified tea plantation and shrub lands to open canopy forest and closed canopy forests. Our sampling design had a spatially nested structure which allowed us to partition the variation in SOC, soil, vegetation and topographical parameters that could be attributed to variation among land-use types, among sampling plots and within sampling plots. Our objectives were (1) to quantify the present SOC stocks of the dominant land-use types, and (2) to define the relationships of SOC concentrations and stocks with land-use type, vegetation, soil texture, and topography.

4.2. Material and Methods

Study area

The study area (10,000 hectares) was located in Mengsong township, Xishuangbanna prefecture in the southern Yunnan province of China at the border with Myanmar (21˚29'25.62"N, 100˚30'19.85"E) (Figure 1). The topography is mountainous with elevations of 800-2000 m above sea level. The climate is subtropical monsoon and has a mean annual temperature (MAT) of 18 °C. Mean annual precipitation (MAP) ranges from 1600-1800 mm, from which 80% falls in the wet season lasting from May to October (Xu et al., 2009). The sampling plots have elevations between 1147-1867 m above sea level, with slope gradients up to 49% (Table 1). The soils at the sampling plots vary from haplic and ferralic Cambisols in narrow valleys, to cambic and ferralic Umbrisols, umbric and haplic Acrisols and Ferralsols at midslope and upslope positions (IUSS Working Group WRB., 2006). Soil texture ranges from sandy clay loam to clay, soil pH (H_2O) from 3.2-6.2, and the effective cation exchange capacity (ECEC) in the subsurface soil from 4.8-45.8 $cmol_c$ kg^{-1} clay (Table 2).

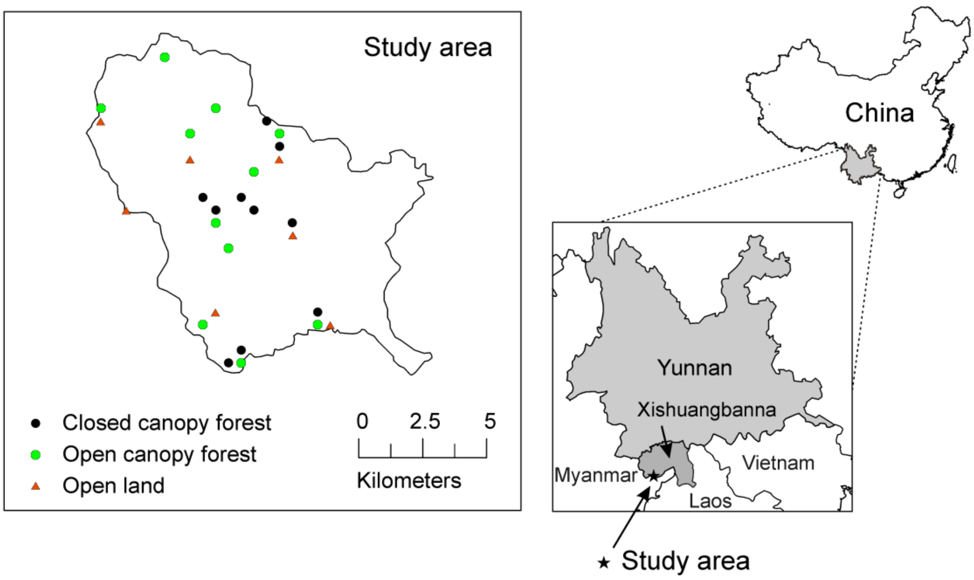

Figure 1. Location of the study area (right), and the sampling plots locations (left) in Xishuangbanna prefecture, Yunnan province, China.

The sampling plots included closed canopy forests, open canopy forests, tea plantations, and shrub lands. The forests are classified as seasonal tropical montane rainforest in valleys and seasonal evergreen broadleaf forest on hill slopes and ridges (Zhu et al., 2005). The most dominant tree families are Lauraceae, Fagaceae, Pentaphylaceae, Euphorbiaceae, and Rubiacea (unpublished data). Closed canopy forest plots mainly consisted of old growth forests, and open canopy forest plots consisted of secondary regrowth. Both closed canopy forests and open canopy forests experienced some disturbances due to timber extraction. Sampling plots in tea plantations included tea shrub plantations as well as tea tree plantations. Shrub lands consist of small shrubs, grasses and some trees, and resulted from frequent fire occurrences. We observed that some of our shrub land plots burnt at least two times between 2010 and 2013. Evidence of fire events in the past was also observed for sampling plots in the other land-use types by pieces of charcoal in the collected soil samples.

Sampling design

We selected 28 one-hectare sampling plots from which 10 plots in closed canopy forests, 11 plots in open canopy forests, and seven plots in open land which included tea plantations and shrub lands. In each sampling plot nine circular subplots with a 10-m radius were established on a square grid with 50-m spacing. Plots were selected using double sampling for stratification, also known as two-phase sampling. In phase 1 we classified the land-use type of a point grid (400 points with 500-m spacing) placed on a 2010 RapidEye satellite image of the study site in either, closed canopy forest, open canopy forest, open land, or other. As open land we considered grassland, tea plantations, orchards, and agricultural fields. In phase 2 we selected our plots from the classified points by stratified random sampling. Thereafter the selected open land plots were classified in four plots in tea-plantations and three plots in shrub land. Of the selected plots, three plots included a maximum of four out of the nine subplots which did not belong to the original land-use classifications. To reduce noise in the data we removed these subplots from the dataset.

Soil and litter sampling

Soil samples were taken down to 1.2 m at five sampling depths: 0-0.15 m, 0.15-0.3 m, 0.3-0.6 m, 0.6-0.9 m and 0.9-1.2 m. At each subplot, we collected samples for the top three depths from four systematically (2 m east, 2 m north, 2 m west and 2 m south of the subplot center) positioned points using a Dutch auger. The samples collected from each subplot were mixed thoroughly in the field to form one pooled sample per sampling depth per subplot. The soil samples of 0.6-0.9-m and 0.9-1.2-m depth were taken in soil pits at respectively four subplots and one subplot per sampling plot. These pits were also used to take soil bulk density samples for each sampling depth using the core method (Blake and Hartge, 1986). The bulk density measures were corrected for gravel content (> 2 mm). The leaf litter layer (including leaf, twigs, and seeds) was sampled at each subplot with a 0.04-m^2 quadrant sampling frame. Samples of the leaf litter layer were collected between May-August 2010. This one-time sampling of the leaf litter layer coincided with the start of the rainy season and does not reflect seasonal or annual fluctuations in litter fall.

Table 1. Vegetation, litter layer and topographic characteristics[1] (means ± SE) of land-use types.

Characteristics	Closed canopy forest	Open canopy forest	Shrub land	Tea plantation	P value
Litter layer C concentration (%)	40.0 (1.1)	40.1 (1.1)	42.8 (0.2)	40.3 (1.7)	0.6
Litter layer C:N ratio	29.7 (1.5) b	36.4 (2.1) a	43.2 (6) a	33.9 (2.7) ab	0.03
Litter layer carbon stock (Mg C ha^{-1})	5.6 (0.6) a	4.2 (0.5) a	1.7 (0.2) b	2.3 (0.9) b	<0.01
Tree basal area (m^2 ha^{-1})	29 (2.5) a	18.2 (1.9) b	3 (0.7) c	5.9 (4.4) c	<0.01
Slope gradient (%)	29.7 (1.6)	26.7 (1.1)	31 (3.8)	23.1 (3.3)	0.11
Elevation (m a.s.l)	1664 (66)	1559 (67)	1719 (59)	1606 (91)	0.49
Compound topographic index	9.9 (0.4)	8.9 (0.2)	8.4 (0.2)	9.5 (0.7)	0.32

[1]Within a row, means followed by different letters differ significantly between land-use types, means without letters are not significantly different (linear mixed effects model at P ≤ 0.05).

Table 2. Soil characteristics[1] (means ± SE) of land-use types.

Characteristic	Depth (m)	Closed canopy forest (n=10)	Open canopy forest (n=11)	Shrub land (n=3)	Tea plantation (n=4)	P value
Sand (%)	0-0.15	39.8 (3.9)	36.3 (3.1)	47.4 (3.7)	37.9 (7.5)	0.43
	0.6-0.9	40.9 (4.4)	31.5 (4.3)	47.5 (3.9)	35.2 (6.7)	0.24
Silt plus clay (%)	0-0.15	60.2 (3.9)	63.7 (3.1)	52.6 (3.7)	62.1 (7.5)	0.52
	0.6-0.9	59.1 (4.4)	68.5 (4.3)	52.5 (3.9)	64.8 (6.7)	0.24
Bulk density (g cm^{-3})	0-0.15	0.8 (0.05)	0.8 (0.02)	0.8 (0.03)	0.7 (0.05)	0.53
	0.6-0.9	1.1 (0.05)	1.1 (0.03)	1.0 (0.03)	1.1 (0.03)	0.48
Soil C:N ratio	0-0.15	15.1 (0.6)	14.3 (0.4)	16.3 (1.1)	14.1 (0.6)	0.29
	0.6-0.9	10.7 (0.5)	10.4 (0.3)	12.5 (0.9)	10.5 (0.5)	0.18
pH (H_2O)	0-0.15	4.5 (0.1) b	4.8 (0.1) a	5.0 (0.2) a	4.9 (0.2) ab	0.02
	0.6-0.9	5.0 (0.1)	5.0 (0.1)	5.0 (0.2)	4.9 (0.2)	0.87
pH (KCL)	0-0.15	3.6 (0.1) b	3.8 (0.1) ab	3.9 (0.1) ab	4.0 (0.2) a	0.05
	0.6-0.9	3.8 (0.1)	3.9 (0.1)	3.9 (0.2)	4.0 (0.1)	0.57
ECEC[2] ($cmol_c$ kg^{-1} clay)	0-0.15	47.3 (7.6)	32.5 (3.6)	53.6 (4.4)	28.9 (5.3)	0.08
	0.6-0.9	23.6 (4.3)	16.2 (3.2)	17.5 (1.6)	10.1 (2.6)	0.20
Al saturation (%)	0-0.15	72.4 (3.1)	64.2 (6.1)	60.5 (12.2)	59.0 (13.1)	0.54
	0.6-0.9	86.3 (1.4)	80.5 (6.1)	87.8 (1.4)	67.9 (11.7)	0.21
Base saturation (%)	0-0.15	20.5 (3.1)	29.3 (6.0)	35.6 (11.8)	34.3 (12.1)	0.35
	0.6-0.9	8.5 (1.5)	12.0 (5.3)	7.9 (1.5)	24.5 (10.5)	0.14

[1]Within a row, means followed by different letters differ significantly between land-use types, means without letters are not significantly different (linear mixed effects model, one-way ANOVA or Kruskal-Wallis ANOVA at P ≤0.05).
[2]ECEC, effective cation exchange capacity.

Tree inventory and topographical data

At all nine subplots (10-m radius) per plot we measured the diameter at breast height (DBH) at 1.3 m above the soil surface of all trees with a DBH ≥ 10 cm. Within a 5-m radius of the subplot center we measured the DBH of all trees with a DBH ≥ 2 cm. Tree basal area at each subplot was calculated as the sum of the basal area of all trees. Topographical data obtained for each subplot included slope gradient, elevation, and compound topographic index (CTI). We measured the slope gradient from the center of each subplot to a target point situated 5-m downslope of the subplot center using a clinometer. Elevation was derived from a SRTM digital elevation model with a 90-m resolution resampled to 30-m resolution. The CTI (Gessler et al., 1995; Moore et al., 1993), also known as steady state wetness index, quantifies landscape positions based on slope gradient and upstream contributing area orthogonal to flow direction. High CTI values refer to valleys with large catchments and low values to ridges or steep slopes. We calculated the CTI from the 30-m SRTM digital elevation model using ArcGIS.

Laboratory analyses and calculations

We analyzed the soil samples for total carbon and nitrogen concentrations, soil pH, soil texture and ECEC. Litter layer samples were analyzed for total carbon and nitrogen concentrations. Prior to analyses the soil samples were air dried (5 days) and sieved (< 2 mm). Litter layer samples were oven dried at 60 °C for 48 hours and weighed. Total carbon and nitrogen concentrations were measured on ground subsamples of each soil and litter sample by dry combustion using a CNS Elemental analyzer (Elementar Vario EL, Hanau, Germany). As soil pH (H_2O) was below 6.2, carbonates were not expected in these soils and carbonate removal was not necessary. Soil pH (H_2O), pH (KCl) and soil texture were measured on each sample from the 0-0.15-m, 0.15-0.3-m, and 0.9-1.2-m depth, and on a pooled sample per sampling plot for the depth 0.6-0.9 m. Soil pH (H_2O) and pH (KCl) were measured in a 1:2.5 soil-to-solution ratio. Soil texture was determined using the pipette method distinguishing the fractions clay (<0.002 mm), silt (0.002-0.063 mm), and sand (0.063-0.002 mm). ECEC was measured on soil samples of the depth 0-0.15 m and on a pooled sample from each sampling plot for the depth 0.6-0.9 m. The soil samples were percolated with unbuffered 1 M NH_4Cl and the percolates were analyzed for exchangeable cations using ICP-EAS (Spectroflame, Spectro Analytical Instruments, Kleve, Germany) (König and Fortmann, 1996).

The litter layer carbon stocks were calculated with the carbon concentration, the mass of the litter layer and the sample frame area. The SOC stock for each sampling depth was calculated by:

$$SOC\ \text{stocks}(\text{Mg C ha}^{-1}) = \frac{\%C}{100} \times BD\ (Mg\ m^{-3}) \times \Delta\ \text{D}\ (m) \times 10{,}000\ m^2 ha^{-1},$$

where BD is the bulk density and ΔD is the thickness of the sampling depth. Total SOC stocks down to 0.9 m were calculated as cumulative stocks of the top four sampling depths. SOC stocks of 0.9-1.2 m depth were not included since the soil depth of some sampling plots did not reach down to 1.2 m.

Statistical analyses

Statistical analyses were done with the statistical software R version 2.15.0 (R Development Core Team, 2012). Statistical tests were conducted separately for each sampling depth. Prior to analyses we tested the data for normality (Shapiro-Wilk test) and equality of variances (Levene's test). Significant differences were accepted at $P \leq 0.05$, $P \leq 0.1$ were considered as marginally significant.

Data available at subplot level (SOC concentrations and stocks, soil C:N ratio, other soil characteristics down to 0.3 m, tree basal area, litter layer characteristics, and topographical data) were analyzed with linear mixed effects models (LME) with sampling plot included as random intercept, using the package nlme (Pinheiro et al., 2012). We tested if land-use types (fixed effect term) differed in SOC, tree basal area, soil, litter and topographical characteristics (response variables). For multiple comparisons between land-use types we conducted Tukey's HSD test in the package multcomp (Hothorn et al., 2008). To partition the overall variance in each response variable that could be ascribed to the variation occurring among land-use types, among sampling plots, and within sampling plots, we refitted the LME with sampling plot nested within land-use type as random intercept. Subsequently, we tested if both random factors were required in the LME by leaving out the random effect for land-use type, and comparing the two LME using a likelihood ratio test (Crawley, 2007). In forests (closed canopy forest and open canopy forest combined) and open land (tea plantation and shrub land combined), we tested the relationships between SOC concentrations and stocks (response variable) with the following potential explanatory variables (fixed effect terms): land-use type, silt plus clay concentration, ECEC of the subsurface soil (0.6-0.9-m depth), litter layer carbon stock, litter layer C:N ratio, tree basal area, slope gradient, elevation, and compound topographic index. Exploratory data analyses did not indicate strong correlations among the potential explanatory variables. Prior to analyses the elevation data was recalculated to have a minimum value of zero. Minimum adequate models were selected using an automated stepwise model selection based on the Akaikes Information Criterion with the function stepAIC in the package MASS (Venables and Ripley, 2002). Residuals of the selected models were examined for normality and equality of variances. In cases of unequal variances we included variance functions and if the assumption of normality was violated we used a logarithmic transformation of the response variable. The proportion of the variance explained by the fixed effect terms (R^2) of each LME was calculated according to Nakagawa and Schielzeth (2013).

For data available at plot level (soil characteristics below 0.3-m depth other than SOC, and SOC of the 0.9-1.2 m depth), we tested the effect of land-use type using either one-way analysis of variance (ANOVA) (parametric test) followed by Tukey's HSD test, or Kruskal-Wallis ANOVA (non-parametric test) followed by a pairwise Wilcoxon test with Holm's correction for multiple comparisons.

4.3. Results

Soil, vegetation, and topographic characteristics

Comparison of soil characteristics across land-use types only revealed differences in soil pH (Table 2). The pH (H_2O) to 0.3 m was the lowest in closed canopy forest (data 0.15-0.3 m not shown), and the pH (KCl) to 0.15 m was lower in closed canopy forests compared to tea plantations. The vegetation characteristics tree basal area and litter layer carbon stocks were larger in open and closed canopy forests compared to tea plantations and shrub lands (Table 1). The litter layer mainly consisted of fresh and partly decomposed plant material. Litter C:N ratios were lower in closed canopy forest compared to open canopy forests and shrub land. Comparison of topographical characteristics showed that the land-use types were situated at similar slope gradients, altitudes and topographical positions (reflected by CTI) (Table 1).

Table 3. Soil organic carbon stocks[1] (Mg C ha^{-1}) (means ± SE) of land-use types.

Depth (m)	Closed canopy forest (n = 10)	Open canopy forest (n = 11)	Tea plantation (n = 4)	Shrub land (n = 3)	P value
0-0.15	65.5 (6.8)	58.4 (4)	48.7 (6.9)	66 (2.6)	0.56
0.15-0.3	51.7 (4.5)	48.4 (4.3)	44.9 (5.5)	55.1 (5)	0.59
0.3-0.6	73.4 (8)	58.9 (4.8)	68.6 (10.7)	67.7 (2.6)	0.34
0.6-0.9	38 (3.2)	34.6 (3.5)	35.2 (3.9)	47.4 (4)	0.39
Total (0-0.9)	228.6 (19.7)	200.4 (15.5)	197.5 (25.9)	236.2 (13.7)	0.52

[1]Within a row, means followed by different letters differ significantly between land-use types, means without letters are not significantly different (LME at P ≤ 0.05).

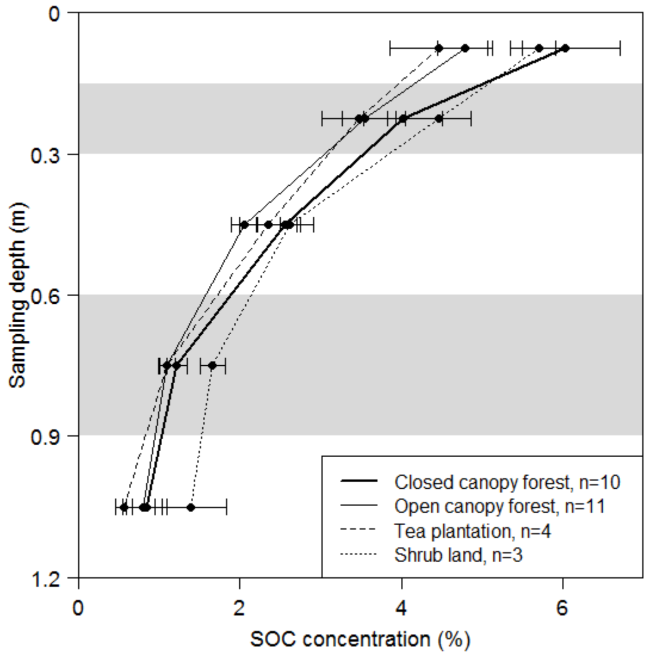

Figure 2*.* Soil organic carbon concentrations in relation to sampling depth for four different land-use types. Alternating white and grey bands show the sampling depths. For each depth, means (SE bars) were the same across land-use types (linear mixed effects model with P = 0.22-0.49 at sampling depths < 0.9 m, and one-way ANOVA with P = 0.37 at 0.9-1.2 m).

Soil organic carbon concentrations and stocks

SOC concentration and stocks were the same across all land-use types for each sampling depths as well as the total SOC stock from 0-0.9 m (Figure 2, Table 3). Additionally we did not detect differences in SOC between forests (open canopy forest and closed canopy forest combined) and open land (tea plantation and shrub land combined) (results not shown). In forests, SOC concentrations and total SOC stocks showed positive trends with litter layer carbon stock, tree basal area, and elevation (R^2 = 0.51 for 0-0.15 m, R^2 = 0.25 for 0.15-0.3 m, R^2 = 0.18 for total SOC stocks) (Table 4). However, the effect of elevation on total SOC stocks was just marginally significant, and at 0.15-0.3-m depth litter layer carbon stock was the only controlling factor of SOC that was statistically significant. In open land, the most important controls of SOC were land-use type, litter layer carbon stock and slope gradient (R^2 = 0.54 for 0-0.15 m, R^2 = 0.67 for 0.15-0.3-m, R^2 = 0.64 for total SOC stock). SOC concentrations and total SOC stocks increased with increasing litter layer carbon stocks and decreased with increasing slope gradient. Furthermore, SOC concentrations in shrub lands were higher compared to tea plantations when controlling for the variation of SOC with litter layer carbon stocks and slope gradient. Tree basal area was also included as explaining factor for SOC concentrations in open land at 0.15-0.3 m and for total SOC stock, but the effect of tree basal on SOC at 0.15-0.3 m was not statistically significant, and the effect of tree basal area on total SOC stocks was marginally significant.

Table 4. Coefficient estimates[1] (± SE) of effects of soil, vegetation and topographic characteristics on SOC concentrations and total SOC stocks in forests (open canopy forest and closed canopy forest combined) and open land (tea plantation and shrub land combined).

Response	Effect	Forest (n = 21)		Open land (n = 7)	
		Estimate	P value	Estimate	P value
SOC concentration (%) at 0-0.15 m	Intercept	2.22 (0.66)	>0.001	6.32 (0.55)	>0.001
	Land-use type[2]		ns	-1.72 (0.37)	<0.01
	Silt plus clay concentration (%)		ns		ns
	ECEC[3] at 0.6-0.9 m (cmol$_c$ kg^{-1} clay)		ns		ns
	Litter layer carbon stock (Mg C ha^{-1})	0.16 (0.04)	>0.001	0.43 (0.07)	<0.01
	Litter layer C:N ratio		ns		ns
	Tree basal area (m^2 ha^{-1})	0.03 (0.01)	>0.001		ns
	Slope gradient (%)		ns	-0.04 (0.01)	<0.01
	Elevation[4] (m a.s.l.)	0.01 (0.001)	>0.01		ns
	Compound topographic Index		ns		ns
SOC concentration (%) at 0.15-0.30 m	Intercept	0.94 (0.86)	0.28	4.99 (0.59)	<0.01
	Land-use type[2]		ns	-1.03 (0.33)	0.05
	Silt plus clay concentration (%)	0.02 (0.01)	0.16		ns
	ECEC[3] at 0.6-0.9 m (cmol$_c$ kg^{-1} clay)		ns		ns
	Litter layer carbon stock (Mg C ha^{-1})	0.17 (0.03)	>0.001	0.33 (0.08)	<0.01
	Litter layer C:N ratio		ns		ns
	Tree basal area (m^2 ha^{-1})	0.01 (0.006)	0.13	0.03 (0.02)	0.13
	Slope gradient (%)		ns	-0.05 (0.02)	<0.01
	Elevation[4] (m a.s.l.)	0.01 (0.001)	0.13		ns
	Compound topographic Index		ns		ns
Total SOC stock (%) at 0-0.9 m	Intercept	109.8 (24.1)	>0.001	257.2 (30.8)	<0.01
	Land-use type[2]		ns	-41.9 (21.9)	0.15
	Silt plus clay concentration (%)		ns		ns
	ECEC[3] at 0.6-0.9 m (cmol$_c$ kg^{-1} clay)		ns		ns
	Litter layer carbon stock (Mg C ha^{-1})	5.3 (1.53)	>0.001	13.2 (3.49)	<0.01
	Litter layer C:N ratio		ns		ns
	Tree basal area (m^2 ha^{-1})	0.89 (0.35)	0.01	1.39 (0.77)	0.09
	Slope gradient (%)		ns	-3.25 (0.79)	<0.01
	Elevation[4] (m a.s.l.)	0.08 (0.05)	0.09		ns
	Compound topographic Index		ns		ns

[1] Linear mixed effects models with sampling plot as random intercept. All effects were included in the full model, model simplification resulted in the minimum adequate model. ns, not significant (i.e, the effects excluded by model simplifications)

[2]The land-use effect in open land is calculated as SOC in tea plantation minus SOC in shrub land.

[3]ECEC, Effective Cation Exchange Capacity.

[4]Prior to analyses elevation was recalculated to have a minimum value of zero.

Variance partitioning of soil, vegetation, litter and topographical characteristics

Variance partitioning showed that in the top 0.3 m of the soil, except for soil pH H_2O (P=0.04), no significant amount of the overall variance in soil characteristics occurred between land-use types (Figure 3a, data 0.15-0.3m not shown). For SOC concentration, total SOC stocks and soil characteristics other than soil texture, both the variation among sampling plots and within sampling plots were the primary components of the overall variance. For soil texture the variation among sampling plots was the most important component of the overall variance. Most of the overall variance in the litter layer carbon stocks and litter layer C:N ratio occurred within sampling plots (Figure 3b). For tree basal area the variation among land-use types was the most important component of the overall variance followed by the variation within sampling plots. The proportion of the overall variance in slope gradient was largest within sampling plots and the overall variance in elevation was almost solely due the variation among sampling plots.

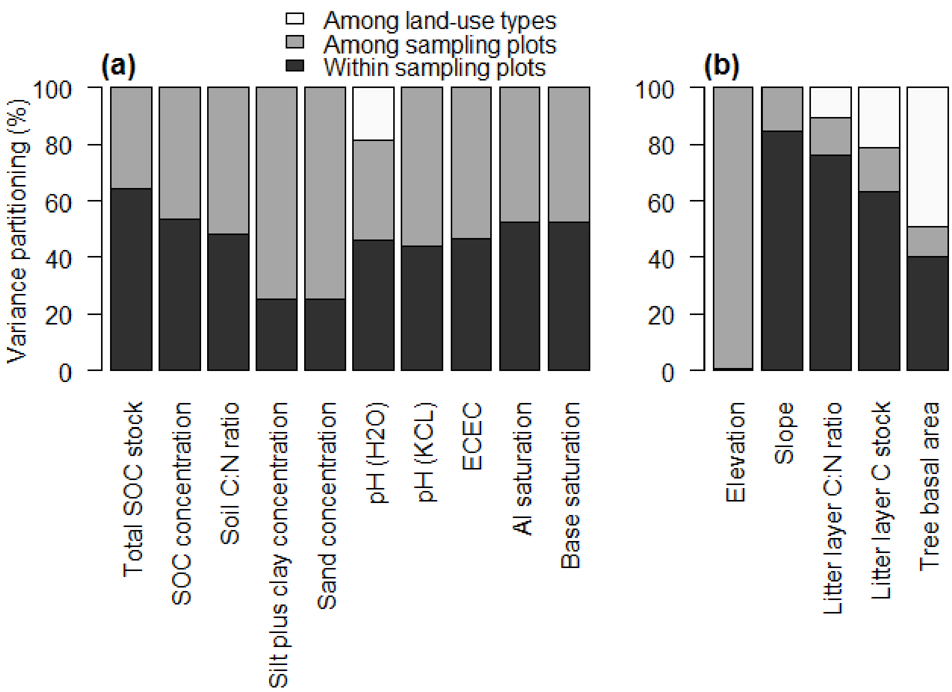

Figure 3. Partitioning of the overall variance in (a) SOC and soil characteristics at 0-0.15 m depth, and (b) topographical and vegetation characteristics, that can be attributed to variation among land-use type, among sampling plots, and within sampling plots (Linear mixed effects model with likelihood ratio test at P ≤ 0.05, the variation in litter C:N ratio among land-use types is marginally significant with P = 0.06)

Table 5. Overview of published soil organic carbon stocks in different land-use types from montane mainland Southeast Asia.

Land use	Country, site	Soil type	Elevation m a.s.l	Climate		Depth (m)	SOC stock (Mg C ha^{-1})	Reference
				MAP (mm)	MAT (˚C)			
Forest	Laos, total country	-	-	-	-	0-0.3	112	Chaplot et al. (2010)
	China, Xishuangbanna	Haplic Acrisol	600	1539	21.7	0-1	84-102	Lü et al. (2010)
	China, Menglong, Xishuangbanna	Ferralsols and(hyper) ferralic Cambisols	700-830	1377	22.7	0-0.9	170	Chapter 2
	Thailand, Nam Hean watershed	Red Yellow Podzolic soils and Reddish Brown Lateritic soils	215-1674	1405	16.9 (DS[1])-32.5 (WS[1])	0-1	196.24	Pibumrung et al. (2008)
	Thailand, Khun Samun Watershed	Hyperalic Alisols (Humic) and Endogleyic Luvisol (Chromic)	300-800	1400	22-29	0-1.2	~170	Aumtong et al. (2009)
	China, Subtropical zone	-	-	-	-	0-1	104.4-111.2	Yu et al. (2011)
Tea	China, Southwest	Haplic Acrisol	-	1000-1700	15-19	0-0.6	132.3- 158.7	Li et al. (2011)
Fallow	Thailand, Khun Samun Watershed	Hyperalic Alisols (Humic) and Endogleyic Luvisol (Chromic)	300-800	1400	22-29	0-1.2	~210	Aumtong et al., (2009)

~ gives an approximate value, read from a graph.

[1]DS, Dry season; WS, wet season.

4.4. Discussion

Effects of land-use type on soil organic carbon concentrations and stocks

Our values of SOC stocks in closed canopy forest, open canopy forests, tea plantations and shrub lands (Table 3) were on the high end of the range of SOC stocks reported for these land-use types in other studies from montane mainland of Southeast Asia (Table 5, comparisons based on equivalent sampling depths). SOC stocks to a depth of 0.3 m in closed canopy forest and open canopy forest were comparable to national estimates of SOC stocks in forests in Laos (Chaplot et al., 2010). However, our SOC stocks to 0.9-m depth were higher than the regional estimates of SOC stocks to 1-m depth in subtropical forests in China (Yu et al., 2011), and the SOC stocks to depths of 0.9 m, 1 m and 1.2 m of tropical forests in Xishuangbanna (de Blécourt et al., 2013; Lü et al., 2010) and northern Thailand (Aumtong et al., 2009; Pibumrung et al., 2008). Data on SOC stocks in tea plantations and shrub lands in the mountainous regions of Southeast Asia is scarce. We found slightly higher SOC stocks to 0.6-m depth in tea plantations compared to regional estimates of SOC stocks in tea plantations in the southwest of China (Li et al., 2011). Our values of SOC stocks to 0.9-m depth in shrub lands were higher than the amounts reported for fallow fields in northern Thailand (Aumtong et al., 2009).

Although land-use type is often considered to be an important controlling factor of SOC, we did not observe differences in SOC concentrations and stocks among land-use types (Table 3, Figure 2). There are several possible explanations for the high SOC levels in shrub lands, which were similar to SOC levels in closed canopy forest. First, due to the abundance of grasses a lot of organic matter is added to the soil as root residues, while in forests that is mainly via the addition of leaf litter from which a major part might be decomposed on the soil surface (Oades, 1988). Second, charcoal input in shrub lands might be relatively high due to the high fire frequencies. However, results from field measurements on impacts of fire and charcoal additions on SOC quantities are controversial ranging from SOC losses (Bird et al., 2000; Fynn et al., 2003) to maintenance or increases in SOC (Eckmeier et al., 2007; Ojima et al., 1994). Our result of similar SOC stocks in forests and shrub lands is in line with a recent meta-study (Don et al., 2011) reporting limited changes in SOC after forest-to-grassland conversions in tropical regions with MAT of 20 °C and MAP of 2000 mm. Conversely, another meta-study of (sub) tropical land-use conversions (Powers et al., 2011) reported an increase in SOC stocks of 26% after forest-to-grassland conversions on low activity clays soils with MAP between 1501-2500 mm. However, comparison of our results with the cited meta-studies is constrained by differences in management regimes; the cited studies synthesized information on managed grasslands in contrast to the unmanaged shrub lands in our study.

Although statistically not significant, SOC concentration and stocks down to 0.6 m in open canopy forests and tea plantations tended to be lower than the SOC levels of closed canopy forests and shrub land (Figure 2, Table 3). Potentially, we failed to detect SOC difference between these land-use types due to the high variability in SOC within land-use types, as reflected by the large standard errors of SOC (Figure 2), and the large proportion of the overall variance in SOC occurring among and within sampling plots (Figure 3a). The large variability in SOC among sampling plots might be a consequence of our probability sampling technique (double sampling for stratification) implemented in a mountainous landscape and a small number of sampling plots.

Effects of biophysical properties on soil organic carbon concentrations and stocks

Our result that more than half of the overall variance in SOC occurred within the one-hectare sampling plots (Figure 3a) indicates that local processes play an important role in the variation of SOC in the landscape. Such a short distance variation in SOC is common in mountainous areas as ours, Chaplot et al. (2009) observed that at a hill slope in Laos 85% of the variation in SOC occurred within a distance of 20 m. In open land, we attribute the variability of SOC within sampling plots to the effects of tree basal area, litter layer carbon stocks and slope gradient on SOC (Table 4), this is because most of the overall variance of these factors occurred within the one-hectare sampling plots (Figure 3b). Variation in SOC among sampling plots in open land was attributed to land-use effects. In forests, tree basal area and litter layer carbon stocks appeared as important controls of the within sampling plot variation of SOC, and elevation was the most important control of the variation in SOC among sampling plots (Table 4, Figure 3b).

The observed increase in SOC in forests and open land with increasing tree basal area and litter layer carbon stocks (Table 4) is in accordance with literature (de Blécourt et al., 2013; Powers and Schlesinger, 2002; Woollen et al., 2012) and is attributed to biomass productivity. Enhanced biomass productivity may increase SOC input through increases in litter fall and root residues. The use of tree basal area and litter layer carbon stock as a proxy for biomass productivity is supported by increases in yearly litter fall with increasing tree basal area, and the positive trend between yearly litter fall and litter layer carbon stocks, which we observed in a subset of our forest plots (Table S4.1.). The observed increase in SOC in forests with increasing elevation is in line with literature (Chaplot et al., 2010; Dieleman et al., 2013; Powers and Schlesinger, 2002). Elevation effects on SOC are often related to changes in precipitation, temperature, soil characteristics, and biomass productivity. However, despite the large elevation gradient of the forest plots (1147-1867 m above sea level) we did not observe elevation effects on silt plus clay concentration, ECEC of the subsurface soil (reflecting clay mineralogy), soil pH H_2O, soil C:N ratio and tree basal area (Table S4.2). Because climate data of our sampling plots is not available, we cannot explain the underlying mechanisms that cause the elevation effect on SOC. The decrease in SOC in open land with increasing slope gradient (Table 4) is most likely due to soil erosion. An erosion study conducted on a steep hill slope covered with swidden fields in northern Laos showed that soil erosion was highest at the hill slope summit and that most of the eroded soil and SOC was deposited within a short distance on the same hill slope (Chaplot et al., 2005). Although soil texture is often regarded as an important control for SOC (Schimel et al., 1994), the silt plus clay concentration only appeared as a controlling factor on SOC in forests, but this trend was not statistically significant (Table 4). Silt plus clay particles may affect SOC stabilization through the adsorption of organic matter on clay surfaces and the physical protection of organic material within soil aggregates against microbial decay (Schimel et al., 1994). The lacking trend between silt plus clay concentration and SOC does not necessarily mean that the chemical and physical stabilization of SOC through silt and clay particles is not important. Potentially the range of the silt-plus-clay content is too small to affect the variation in SOC.

Implications for sampling soil organic carbon stocks

Probability sampling techniques as applied in our study are appropriate for SOC inventories, providing valuable baseline information for estimations of SOC distributions at a landscape scale. However, as illustrated by our results, the establishment of causal relationships between land-use type and SOC based on results from inventory studies might be problematic. In mountainous landscapes, large variability in SOC, soil, vegetation, and topography within and among sampling plots (Figure 3) may confound the land-use effects on SOC. An often used approach that proved to be effective in detecting land-use effects on SOC is space-for-time substitution (e.g., de Koning et al., 2003; de Blécourt et al., 2013). This approach aims to select sampling plots that only differ in land-use type, with soil, vegetation and topographical characteristics being equal. However, in contrast to our probability sampling technique, plot selection using the space-for-time substitution approach is often nonrandom in order to meet the criteria for comparison, and thus SOC stocks measured in those studies cannot easily be extrapolated to a larger scale.

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CHAPTER 5

Synthesis

5.1. Key findings of this thesis

Chapter 2. Clearing secondary forests for rubber plantations caused a loss in SOC stocks of on average 37.4 ± 4.7 Mg C ha^{-1} in the entire 1.2-m depth in plantations aged 5 to 46 years. This loss was equal to 19 ± 2.7% of the initial SOC stocks in the studied secondary forests. In the topsoil the SOC stocks declined exponentially with years since land-use change; the strongest decline was observed in the first 5 years and SOC stocks reached a steady state approximately 20 years after land-use change. The observed losses in total SOC stocks were much higher than the literature-based estimates of the impacts on above-ground carbon stocks, which ranged from a loss of 18 Mg C ha^{-1} to an increase of 8 Mg C ha^{-1}.

Chapter 3. Terracing may reduce the losses in SOC stocks caused by forest clearance for terraced rubber plantations - without the terraces the SOC stock losses could have been higher. Terraces are usually constructed manually, by cutting the soil from the upper slope, this creates the inner edge of the terrace (cut section), and piling up the removed soil down slope which forms the terrace's outer edge (fill section). Although terracing did not affect SOC stocks in the 5-year old plantation, in the 29-year and 44-year old plantations terracing caused a net gain in total SOC stocks (0-1.2 m depth) on the terraces compared to the non-terraced reference positions. The positive effect of terracing on SOC stocks in the two oldest plantations was explained by the observed recovery of SOC stocks at the cut sections, and the observed partial preservation of SOC in the buried soil at the fill sections. The recovery of SOC stocks at the cut section was attributed to the capacity of the exposed subsurface soil to store new SOC inputs from roots and litter, and to the sedimentation of eroded topsoil materials from the upper slope.

Chapter 4. More than 50% of the overall variance in SOC measured across a subtropical landscape occurred within the one-hectare sampling plots, indicating that local processes are important for the variation of SOC in mountainous landscapes. The SOC stocks to a depth of 0.9 m, which did not differ across land-use types, were among the highest in montane mainland of Southeast Asia: 228.6 ± 19.7 (SE) Mg C ha^{-1} in closed canopy forests, 200.4 ± 15.5 Mg C ha^{-1} in open canopy forests, 197.5 ±

25.9 Mg C ha^{-1} in tea plantations and 236.2 ± 13.7 Mg C ha^{-1} in shrub lands. The high variability in SOC within the one-hectare sampling plots was related to the variation in tree basal area, litter layer carbon stocks, and slope gradient. In forests, elevation was the most important driver of the variation in SOC among sampling plots, while in open land (tea plantations and shrub lands combined) variation in SOC among sampling plots was related to land-use type.

5.2. Implications for soil organic carbon assessments

"How do SOC stocks respond to land-use changes?" and "What are the current SOC stocks across a landscape?". Answers on these questions are crucial for reducing the uncertainties in the estimates of CO_2 emissions from land-use change (Houghton, 2010). Moreover, information on the current SOC stocks of land-use types at a landscape scale is of particular interest for initiatives aiming to generate financial compensation for projects that protect ecosystem services and enhance ecosystem carbon stocks. Two relevant examples include the CDM (Clean Development Mechanism according to the Kyoto Protocol) and the REDD+ mechanism (Reducing Emissions from Deforestation and forest Degradation) which is currently being discussed by the United Nations Framework of Climate Change.

Chapters 2 and 4 illustrate two different sampling designs that can be used to address the above-mentioned questions. With the space-for-time substitution approach used in Chapter 2, I was able to answer the question how land-use change affects SOC stocks. However, the selection of sampling plots using the space-for-time substitution approach is often non-random, and therefore measured SOC values and estimated SOC changes do not provide information at a landscape scale. Information on the current SOC stocks of land-use types across a landscape can be obtained through probability sampling techniques like the double-sampling for stratification approach used in Chapter 4. But due to the random selection of sampling plots it cannot be assured that detailed information on land-use history can be obtained with this method. Moreover, probability sampling techniques do not account for land-use types' inherent soil properties. These two drawbacks limit conclusions toward land-use change effects.

The sampling design will affect the variability of the SOC data. For instance, the results from Chapter 4 indicated that the probability sampling technique implemented in a mountainous rugged terrain resulted in high variability in SOC within each land-use type. Potentially due to the high variability in SOC and the small number of sampling plots, I failed to detect differences in SOC between different land-use types (for instance tea plantations and closed canopy forests). In comparison, the space-for-time substitution approach is typically based on the SOC differences between different land-use types within paired or clustered sampling plots. In Chapter 2 clusters consisted of one to three sampling plots in rubber plantations and a corresponding sampling plot in a reference forest. Since the clusters had been selected to have similar soil and topographical characteristics, variation in SOC stocks within each land-use type in a cluster has been minimized. A higher variability in the data requires more sampling plots in order to assure a high probability that a statistical test detects a significant land-use effect on SOC, in case this effect exists. Theoretically then, probability sampling techniques will require more sampling plots to detect a certain land-use effect on SOC, compared to the space-for-time substitution approach.

To demonstrate the impact of sampling design on the minimum sample size required to detect a certain significant difference in SOC stocks between two land-use types I conducted a statistical power analyses (Figure 1). I used the SOC data of the 0-0.15-m depth from Chapter 2 (space-for-time substitution) and Chapter 4 (probability sampling technique). For the probability sampling technique I focused on the comparisons between closed canopy forests and tea plantations, and between closed canopy forests and open canopy forests. Obtained sample sizes refer to the number of paired comparisons in the case of the space-for-time substitution approach, and in the case of the probability sampling technique the sample size is the number of sampling plots in each land-use type. The results (Figure 1) show that in order to detect a certain significant difference in SOC stocks, the space-for-time substitution approach requires a much smaller sample size compared to the probability sampling technique. For instance, to detect a difference in SOC stocks of 11.8 Mg C ha^{-1} (the observed SOC stock difference at 0-0.15-m depth between forest and rubber plantations, Chapter 2), a sample size of n = 4 would be sufficient when using the space-for-time substitution approach, whereas infeasible sample sizes of n = 36 to 47 would be required when using a probability sampling technique. Furthermore, the results illustrate that for instance with a sample size of n = 5, SOC stock differences > 6.5 Mg C ha^{-1} can be detected using the space-for-time substitution approach, while with the probability sampling technique only large differences in SOC stocks can be detected of > 35.6 Mg C ha^{-1} for the comparison between closed- and open canopy forests, and > 40.6 Mg C ha^{-1} for the comparison between closed canopy forest and tea plantations. The results from the statistical power analyses could be used to define the number of sampling plots needed per land-use type in order to detect a certain difference in SOC stocks. However, the results are only valid for studies conducted in areas with similar topography and soil characteristics, and with comparable sized sampling plots (20 x 20 m for the space-for-time substitution approach and 100 x 100 m for the probability sampling technique). Summarizing, these results imply that in a mountainous terrain the land-use impacts on SOC stocks are most effectively assessed by the space-for-time substitution approach; additionally this approach enables conclusions towards land-use change effects.

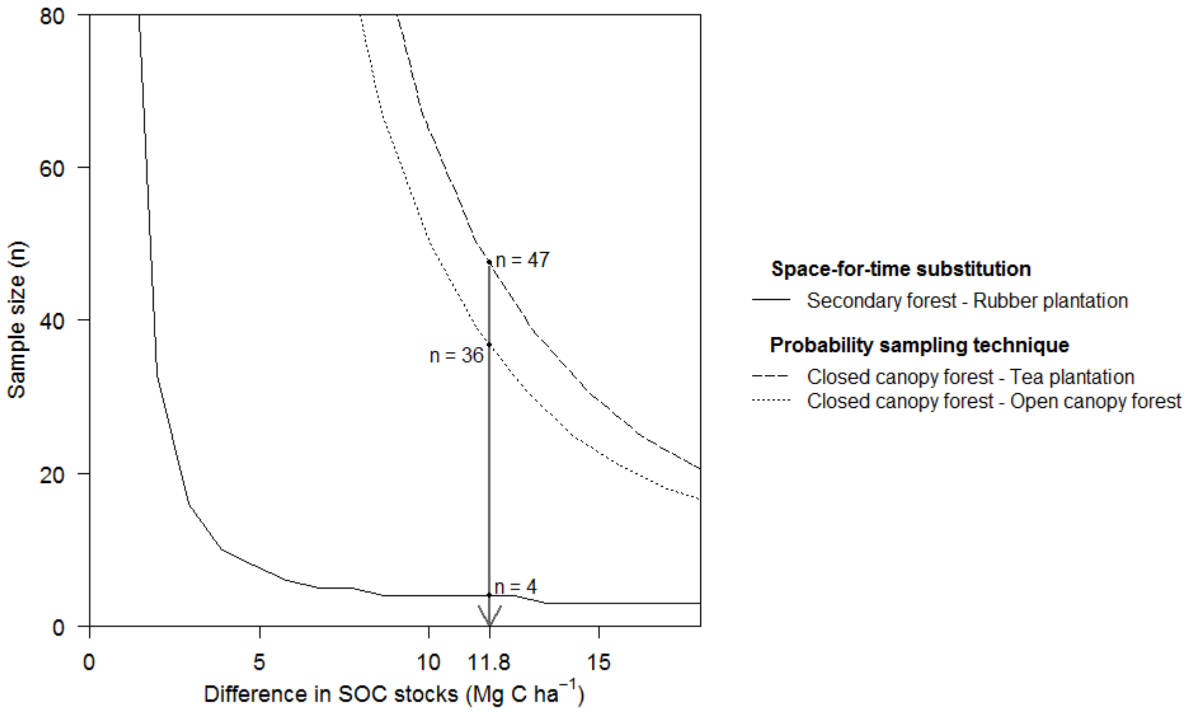

Figure 1. The difference in soil organic carbon (SOC) stocks at 0-0.15 m depth between two land-use types in relation to the sample size needed to indicate effects. Statistical power analyses were done with a power of 0.8 and a significance level of α = 0.05. The statistical power analysis for the space-for-time substitution approach was based on a one-sample t test of SOC stock differences. For the probability sampling technique the statistical power analyses were based on an independent t test of SOC stocks, using the mean SOC stock per sampling plot. Sample size is the number of paired comparisons in case of the space-for-time substitution approach, and in the case of the probability sampling technique the sample size is the number of sampling plots in each of the land-use types. Curves indicate either a space-for-time substitution approach (continuous line), or a probability sampling technique (dashed lines).The grey arrow indicates the observed difference in SOC stocks of 11.8 Mg C ha^{-1} between secondary forests and rubber plantations (Chapter 2).

5.3. Implications of soil organic carbon losses from forest clearance for rubber plantations

The high losses in SOC stocks due to forest clearance for rubber plantations have consequences for estimations of overall ecosystem carbon fluxes from land-use changes. The IPPC guidelines set default factors for the SOC changes in the top 0.3 m of the soil following a change in land use (Aalde et al., 2006; Lasco et al., 2006). For the conversion of forests to rubber plantations the IPCC set a default factor for SOC stock changes of 1 (meaning no change in SOC). In contrast, the results from Chapter 2 indicated that changes in SOC stocks should be included to prevent potential large errors in the overall estimates of ecosystem carbon fluxes from land-use changes.

As a first step towards a revised default value of the SOC stock change factor, I calculated the overall mean of the mean SOC stock changes from Chapter 2, data available from literature, as well as data from an unpublished study (Van Straaten, unpublished data) (Table 1). The in total six studies cover five countries (Brazil, Cameroon, China, Ghana, and Indonesia). In four out of the five countries, all paired comparisons between rubber plantations and forests showed SOC losses with mean reductions in SOC stocks ranging between 24.4% and 43.9%. But in Indonesia SOC stock changes were not significant. The overall mean SOC stock reduction was 28.6%, corresponding with a SOC stock change factor of 0.71. A similar value of 0.72 was obtained when I only included plantations older than 20 years, which is the IPCC's default for the time period needed to reach a new SOC steady state after a change in land use. The estimated stock change factors are based on a limited number of studies. The partly contrasting results from the study in Indonesia (Van Straaten, unpublished data) compared with the other studies (Chapter 2; Chiti et al., 2013, Salimon et al., 2009; Van Straaten, unpublished data; Yang et al., 2004) indicate that more studies are needed across the tropics to improve the SOC stock change factor.

***Table 1**.* Overview of available data across the tropics on relative soil organic carbon (SOC) stock changes due to forest conversion to rubber plantations.

Country	**Soil type**	**MAP (mm)**	**MAT (˚C)**	**Age (years)**	**n[1]**	**Relative SOC stock change (%)[2]**
Brazil (Salimon et al., 2009)	Oxisol	1750-2000[3]	26	19	1	-43.9*
Cameroon (Van Straaten, unpublished data)	Ferralsol	1923	23.7	25-50	6	-34* (-17,5;-67.1)
China (Chapter 2)	Ferralsol, Cambisol	1377	22.7	5-46	7	-24.4* (-15.8;-38.6)
China (Yang et al., 2004)	Udic Ferrisol	1400	22.1	3-7	2	-30.5* (-27.1; -33.8)
Ghana (Chiti et al., 2013)[4]	Oxisol	1700	26	5-49	4	-28.5* (-20.2; -35.8)
Indonesia (Van Straaten, unpublished data)	Agrisol	2646	26.6	10-30	16	-8.1 (-44.3; +24,3)

[1]n is the number of paired comparisons between rubber plantations and forests.
[2]Negative values refer to SOC losses, and positive values to SOC gains. Values between brackets give the range of the SOC stock changes. For the data from Yang et al. (2004) and Salimon et al. (2009) I worked with the SOC changes with bulk density corrections (Chapter 2).
[3]precipitation range instead of the MAP.
[4]recently published study not cited in Chapter 2.
*indicates statistical significant effect of land-use change on SOC stocks according to the cited studies.

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DECLARATION OF ORIGINALITY AND CERTIFICATE OF AUTHORSHIP

I, Marleen de Blécourt, hereby declare that I am the sole author of this dissertation entitled "Impacts of land use and biophysical properties on soil carbon stocks in southern Yunnan, China". All references and data sources that were used in the dissertation have been appropriately acknowledged. I furthermore declare that this work has not been submitted elsewhere in any form as part of another dissertation procedure. I certify that the manuscripts presented in chapters 2, 3 and 4 have been written by me as first author.

Göttingen, November 2013 ______________________________ (Marleen de Blécourt)

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Publications

Roeder, M., McLeish, M., Beckschäfer, P., de Blécourt, M., Paudel, E., Harrison, R. D., Slik, F. 2014. Phylogenetic clustering increases with succession for lianas in a Chinese tropical montane rain forest. Ecography. In Press. DOI: 10.1111/ecog.01051.

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Supporting information

Table S2.1.A. Site and soil characteristics of the secondary forest plots.

Cl.	Depth (m)	Soil type[1]	Alt. (m)	Asp. (˚)	Sl. (%)	Sand (%)	S+C (%)	BD (g cm^{-3})	pH (H_2O)	pH (KCl)	ECEC (mmol$_c$ kg^{-1} soil)	BS (%)	Soil C:N	Soil C (%)	Soil C (Mg ha^{-1})
1	0-0.15	Fl CM (au, hu, hd, cr)	828	187	61	52.1	47.9	1.3	5.3	4.1	61.1	58.4	14.6	2.6	51.2
1	0.15-0.3	Fl CM (au, hu, hd, cr)	828	187	61	52.2	47.8	1.4	5.4	4.0	-	-	11.9	2.0	43.1
1	0.3-0.6	Fl CM (au, hu, hd, cr)	828	187	61	48.6	51.4	1.5	5.4	4.0	-	-	8.7	1.2	55.9
1	0.6-0.9	Fl CM (au, hu, hd, cr)	828	187	61	40.7	59.3	1.5	4.9	3.9	57.9	19.1	6.5	0.7	30.8
1	0.9-1.2	Fl CM (au, hu, hd, cr)	828	187	61	38.8	61.2	1.5	4.9	3.9	-	-	6.0	0.5	22.1
2	0-0.15	ha FR (hu, au, hd, ce)	727	250	23	35.9	64.1	1.2	4.4	3.8	48.5	15.7	12.1	2.6	47.9
2	0.15-0.3	ha FR (hu, au, hd, ce)	727	250	23	46.5	53.5	1.3	4.7	3.9	-	-	11.2	2.0	38.5
2	0.3-0.6	ha FR (hu, au, hd, ce)	727	250	23	42.0	58.0	1.1	4.8	3.9	-	-	10.2	1.3	44.3
2	0.6-0.9	ha FR (hu, au, hd, ce)	727	250	23	41.8	58.2	1.4	4.5	3.9	35.6	12.5	9.0	0.8	33.1
2	0.9-1.2	ha FR (hu, au, hd, ce)	727	250	23	40.6	59.4	1.4	5.2	4.0	-	-	8.8	0.6	26.6
3	0-0.15	ha FR (hu, au, hd, ce)	742	340	30	34.0	66.0	0.9	4.3	3.8	59.3	15.1	11.8	3.0	41.2
3	0.15-0.3	ha FR (hu, au, hd, ce)	742	340	30	33.1	66.9	1.2	4.8	3.8	-	-	12.7	2.2	37.9
3	0.3-0.6	ha FR (hu, au, hd, ce)	742	340	30	33.1	66.9	1.1	4.7	3.9	-	-	11.9	1.5	48.9
3	0.6-0.9	ha FR (hu, au, hd, ce)	742	340	30	31.7	68.3	1.2	4.9	3.9	38.6	7.2	9.9	0.9	31.7
3	0.9-1.2	ha FR (hu, au, hd, ce)	742	340	30	27.6	72.4	1.2	5.0	3.9	-	-	9.6	0.8	28.8
4	0-0.15	vt FR (hu, au, hd, ce)	797	130	40	37.3	62.7	0.9	4.8	3.9	44.4	28.4	11.0	2.4	33.4
4	0.15-0.3	vt FR (hu, au, hd, ce)	797	130	40	38.1	61.9	1.1	5.0	4.0	-	-	11.1	2.0	34.2
4	0.3-0.6	vt FR (hu, au, hd, ce)	797	130	40	35.3	64.7	1.1	4.8	4.0	-	-	11.3	1.6	55.4
4	0.6-0.9	vt FR (hu, au, hd, ce)	797	130	40	35.1	64.9	1.3	5.3	4.0	28.2	12.1	9.6	0.8	29.5
4	0.9-1.2	vt FR (hu, au, hd, ce)	797	130	40	31.9	68.1	1.4	5.2	4.1	-	-	9.7	0.7	29.6

Cl.	Depth (m)	Soil type[1]	Alt. (m)	Asp. (°)	Sl. (%)	Sand (%)	S+C (%)	BD (g cm^{-3})	pH (H_2O)	pH (KCl)	ECEC ($mmol_c$ kg^{-1} soil)	BS (%)	Soil C:N	Soil C (%)	Soil C (Mg ha^{-1})
5	0-0.15	ac FR (hu, au, hd, ce)	750	250	17	45.2	54.8	1.0	4.6	3.8	59.2	15.0	12.8	3.3	49.5
5	0.15-0.3	ac FR (hu, au, hd, ce)	750	250	17	36.1	63.9	1.2	4.8	3.9	-	-	12.7	2.1	36.9
5	0.3-0.6	ac FR (hu, au, hd, ce)	750	250	17	32.8	67.2	1.2	4.5	4.0	-	-	12.1	1.4	53.6
5	0.6-0.9	ac FR (hu, au, hd, ce)	750	250	17	34.3	65.7	1.4	4.6	4.0	37.5	7.2	8.9	0.7	31.2
5	0.9-1.2	ac FR (hu, au, hd, ce)	750	250	17	34.5	65.5	1.4	5.3	3.9	-	-	8.5	0.6	26.0
6	0-0.15	ha FR (hu, au, hd, ce)	790	85	30	19.3	80.7	1.1	4.8	3.9	56.5	15.5	13.5	2.9	47.7
6	0.15-0.3	ha FR (hu, au, hd, ce)	790	85	30	16.6	83.4	1.2	4.6	3.9	-	-	14.5	2.6	45.5
6	0.3-0.6	ha FR (hu, au, hd, ce)	790	85	30	15.3	84.7	1.2	4.9	3.9	-	-	12.1	1.5	52.2
6	0.6-0.9	ha FR (hu, au, hd, ce)	790	85	30	14.8	85.2	1.3	5.2	4.0	35.2	9.8	9.4	0.8	32.6
6	0.9-1.2	ha FR (hu, au, hd, ce)	790	85	30	12.4	87.6	1.2	5.7	4.1	-	-	7.8	0.7	24.6
7	0-0.15	vt FR (hu, au, hd, ce)	768	60	42	18.9	81.1	0.8	4.7	3.9	59.5	20.4	14.0	3.2	36.4
7	0.15-0.3	vt FR (hu, au, hd, ce)	768	60	42	18.3	81.7	1.1	4.6	3.9	-	-	13.1	2.1	36.3
7	0.3-0.6	vt FR (hu, au, hd, ce)	768	60	42	18.0	82.0	1.2	4.7	3.9	-	-	12.6	1.5	54.1
7	0.6-0.9	vt FR (hu, au, hd, ce)	768	60	42	12.4	87.6	1.2	4.7	4.0	25.5	11.4	12.7	1.6	57.1
7	0.9-1.2	vt FR (hu, au, hd, ce)	768	60	42	13.4	86.6	1.2	4.7	4.2	-	-	8.6	0.7	24.4

Cl. = Cluster, Alt. = Altitude, Sl. = Slope, S+C = Silt and Clay concentration, BD = Bulk Density, ECEC = Effective Cation Exchange Capacity, BS = Base Saturation.

[1]Codes for Reference Soil Group (IUSS Working Group WRB., 2006): CM = Cambisol, FR = Ferralsol, and codes for qualifiers: ac = Acric, au = Alumic, ce = Clayic, cr = Chromic, dy = Dystic, dyo = Orthodystic, ec = Escalic, fl = Ferralic, flh = Hyperferralic, ha = Haplic, hd = Hyperdystic, hu = Humic, vt = Vetic, xa = Xanthic.

Table S2.1.B. Site and soil characteristics of the rubber plantation plots.

Cl.	Age (y)	Depth (m)	Soil type[1]	Alt. (m)	Asp. (°)	Sl. (%)	Sand (%)	S+C (%)	BD (gcm^{-3})	pH (H_2O)	pH (KCl)	ECEC ($mmol_c$ kg^{-1} soil)	BS (%)	Soil C:N	Soil C (%)	Soil C[2] ($Mg\ ha^{-1}$)
1	5	0-0.15	flh CM (au, hd, ec)	797	228	65	47.7	52.3	1.3	5.5	4.1	41.7	46.7	12.0	2.0	39.6
1	5	0.15-0.3	flh CM (au, hd, ec)	797	228	65	48.1	51.9	1.0	5.4	4.0	-	-	12.1	1.9	39.8
1	5	0.3-0.6	flh CM (au, hd, ec)	797	228	65	47.2	52.9	1.4	5.4	4.0	-	-	10.0	1.2	52.8
1	5	0.6-0.9	flh CM (au, hd, ec)	797	228	65	44.3	55.7	1.6	5.3	4.0	31.9	12.7	7.1	0.4	20.6
1	5	0.9-1.2	flh CM (au, hd, ec)	797	228	65	46.6	53.4	1.4	5.3	3.9	-	-	5.6	0.3	13.6
2	8	0-0.15	vt ac FR (au, hd, ce, ec)	711	330	22	43.4	56.6	1.0	4.7	3.7	38.9	12.0	10.6	1.8	32.8
2	8	0.15-0.3	vt ac FR (au, hd, ce, ec)	711	330	22	41.9	58.1	1.0	4.6	3.8	-	-	10.2	1.4	27.3
2	8	0.3-0.6	vt ac FR (au, hd, ce, ec)	711	330	22	38.1	61.9	1.1	4.6	3.9	-	-	10.1	1.0	34.0
2	8	0.6-0.9	vt ac FR (au, hd, ce, ec)	711	330	22	34.9	65.1	1.2	5.3	4.0	26.1	10.7	9.2	0.7	28.8
2	8	0.9-1.2	vt ac FR (au, hd, ce, ec)	711	330	22	36.9	63.1	1.3	5.0	4.0	-	-	8.2	0.5	22.4
3	5	0-0.15	ha FR (hu, au, hd, ce, ec)	767	330	39	39.9	60.1	0.9	4.7	3.8	52.6	13.9	13.3	2.5	34.1
3	5	0.15-0.3	ha FR (hu, au, hd, ce, ec)	767	330	39	38.1	61.9	1.1	4.6	3.8	-	-	13.5	1.9	33.3
3	5	0.3-0.6	ha FR (hu, au, hd, ce, ec)	767	330	39	37.4	62.6	1.2	4.9	3.8	-	-	12.3	1.3	42.4
3	5	0.6-0.9	ha FR (hu, au, hd, ce, ec)	767	330	39	38.3	61.7	1.4	4.7	3.9	30.7	5.9	9.4	0.7	26.1
3	5	0.9-1.2	ha FR (hu, au, hd, ce, ec)	767	330	39	35.9	64.1	1.4	4.8	4.0	-	-	8.8	0.7	24.0
3	44	0-0.15	vt ac FR (au, hd, ce, ec)	730	310	39	26.0	74.0	1.2	4.8	3.9	44.6	29.7	11.1	2.0	27.6
3	44	0.15-0.3	vt ac FR (au, hd, ce, ec)	730	310	39	23.2	76.8	1.2	4.9	3.9	-	-	11.3	1.6	28.1
3	44	0.3-0.6	vt ac FR (au, hd, ce, ec)	730	310	39	20.2	79.8	1.2	4.9	4.0	-	-	10.6	1.3	42.6
3	44	0.6-0.9	vt ac FR (au, hd, ce, ec)	730	310	39	22.7	77.3	1.2	5.0	4.1	26.6	14.0	9.4	0.9	33.2
3	44	0.9-1.2	vt ac FR (au, hd, ce, ec)	730	310	39	20.5	79.5	1.1	5.2	4.3	-	-	8.1	0.7	24.8
3	13	0-0.15	ha FR (au, hd, ce, xa, ec)	697	312	37	44.6	55.4	1.2	4.7	3.8	41.1	16.4	12.3	1.8	25.0

Cl.	Age (y)	Depth (m)	Soil type[1]	Alt. (m)	Asp. (°)	Sl. (%)	Sand (%)	S+C (%)	BD (gcm^{-3})	pH (H_2O)	pH (KCl)	ECEC ($mmol_c$ kg^{-1} soil)	BS (%)	Soil C:N	Soil C (%)	Soil C[2] ($Mg\ ha^{-1}$)
3	13	0.15-0.3	ha FR (au, hd, ce, xa, ec)	697	312	37	41.7	58.3	1.3	4.7	3.3	-	-	12.2	1.5	26.6
3	13	0.3-0.6	ha FR (au, hd, ce, xa, ec)	697	312	37	39.6	60.4	1.3	4.6	3.9	-	-	11.0	1.1	37.9
3	13	0.6-0.9	ha FR (au, hd, ce, xa, ec)	697	312	37	39.3	60.7	1.3	4.9	4.1	28.8	7.5	8.8	0.6	20.8
3	13	0.9-1.2	ha FR (au, hd, ce, xa, ec)	697	312	37	37.2	62.8	1.3	4.9	4.0	-	-	8.6	0.5	18.7
4	42	0-0.15	ac FR (au, dy, ce, ec)	743	123	43	43.1	56.9	1.3	4.9	3.9	40.9	30.4	11.6	1.7	23.6
4	42	0.15-0.3	ac FR (au, dy, ce, ec)	743	123	43	36.1	63.9	1.1	4.8	3.9	-	-	10.9	1.2	20.6
4	42	0.3-0.6	ac FR (au, dy, ce, ec)	743	123	43	28.9	71.1	1.3	5.0	4.0	-	-	10.3	1.0	34.6
4	42	0.6-0.9	ac FR (au, dy, ce, ec)	743	123	43	21.4	78.7	1.2	5.5	4.1	-	-	7.8	0.6	21.0
4	42	0.9-1.2	ac FR (au, dy, ce, ec)	743	123	43	19.1	80.9	1.2	5.4	4.2	-	-	7.7	0.6	24.0
4	14	0-0.15	ha FR (au, hd, ce, ec)	717	104	43	40.2	59.8	1.1	4.8	3.8	43.8	10.7	11.2	1.8	25.3
4	14	0.15-0.3	ha FR (au, hd, ce, ec)	717	104	43	38.2	61.8	1.1	4.9	3.8	-	-	10.2	1.6	26.1
4	14	0.3-0.6	ha FR (au, hd, ce, ec)	717	104	43	37.2	62.8	1.2	4.9	3.8	-	-	9.6	1.0	34.4
4	14	0.6-0.9	ha FR (au, hd, ce, ec)	717	104	43	34.9	65.1	1.4	5.1	3.9	33.5	7.5	8.2	0.6	21.6
4	14	0.9-1.2	ha FR (au, hd, ce, ec)	717	104	43	36.3	63.7	1.4	5.2	3.9	-	-	7.6	0.5	22.1
5	46	0-0.15	vt ac FR (au, hd, ce, ec)	742	253	25	23.2	76.8	1.3	4.7	3.9	45.5	29.1	12.1	2.1	31.8
5	46	0.15-0.3	vt ac FR (au, hd, ce, ec)	742	253	25	26.8	73.2	1.2	4.9	3.9	-	-	11.2	1.6	27.9
5	46	0.3-0.6	vt ac FR (au, hd, ce, ec)	742	253	25	20.5	79.5	1.3	5.1	4.0	-	-	10.2	1.1	39.2
5	46	0.6-0.9	vt ac FR (au, hd, ce, ec)	742	253	25	17.6	82.4	1.3	5.0	4.1	26.9	12.2	9.3	0.9	37.5
5	46	0.9-1.2	vt ac FR (au, hd, ce, ec)	742	253	25	15.4	84.6	1.3	5.2	4.1	-	-	8.2	0.7	30.3
6	6	0-0.15	vt FR (hu,au, hd, xa, ec)	836	80	23	16.0	84.0	0.8	4.7	3.8	53.5	13.5	13.1	2.5	41.5
6	6	0.15-0.3	vt FR (hu,au, hd, xa, ec)	836	80	23	14.1	85.9	1.2	4.7	3.8	-	-	12.9	2.1	36.4
6	6	0.3-0.6	vt FR (hu,au, hd, xa, ec)	836	80	23	12.0	88.0	1.1	4.7	3.8	-	-	12.0	1.5	52.1

Cl.	Age (y)	Depth (m)	Soil type[1]	Alt. (m)	Asp. (°)	Sl. (%)	Sand (%)	S+C (%)	BD (gcm^{-3})	pH (H_2O)	pH (KCl)	ECEC ($mmol_c$ kg^{-1} soil)	BS (%)	Soil C:N	Soil C (%)	Soil C[2] ($Mg\ ha^{-1}$)
6	6	0.6-0.9	vt FR (hu,au, hd, xa, ec)	836	80	23	12.9	87.1	1.2	4.8	3.9	32.0	5.4	8.7	0.8	32.5
6	6	0.9-1.2	vt FR (hu,au, hd, xa, ec)	836	80	23	16.4	83.7	1.3	4.9	4.0	-	-	7.7	0.7	24.7
7	40	0-0.15	vt ac FR (hu, au, dyo, ce, ec)	784	60	40	14.5	85.5	0.9	4.6	3.9	51.4	16.4	12.6	2.4	27.4
7	40	0.15-0.3	vt ac FR (hu, au, dyo, ce, ec)	784	60	40	11.6	88.4	1.1	4.8	3.9	-	-	14.3	2.0	33.4
7	40	0.3-0.6	vt ac FR (hu, au, dyo, ce, ec)	784	60	40	10.9	89.1	1.2	4.9	4.0	-	-	13.6	1.6	56.3
7	40	0.6-0.9	vt ac FR (hu, au, dyo, ce, ec)	784	60	40	10.9	89.2	1.1	5.1	4.2	22.3	22.0	11.5	0.8	29.5
7	40	0.9-1.2	vt ac FR (hu, au, dyo, ce, ec)	784	60	40	42.6	57.5	1.1	5.1	4.2	-	-	9.6	0.7	24.9
7	29	0-0.15	Ac FR (hu, au, dy, ce, ec)	813	70	49	19.3	80.7	1.1	4.9	4.0	56.4	49.3	13.9	2.1	24.4
7	29	0.15-0.3	Ac FR (hu, au, dy, ce, ec)	813	70	49	15.3	84.7	1.1	5.0	3.9	-	-	13.9	1.7	28.8
7	29	0.3-0.6	Ac FR (hu, au, dy, ce, ec)	813	70	49	14.2	85.8	1.2	5.0	3.9	-	-	13.8	1.5	53.6
7	29	0.6-0.9	Ac FR (hu, au, dy, ce, ec)	813	70	49	12.0	88.0	1.2	5.3	4.1	-	-	11.5	1.0	36.6
7	29	0.9-1.2	Ac FR (hu, au, dy, ce, ec)	813	70	49	14.0	86.0	1.3	5.3	4.1	-	-	10.3	0.7	25.1

Cl. = Cluster, Alt. = Altitude, Sl. = Slope, S+C= Silt and Clay concentration, BD = Bulk Density, ECEC = Effective Cation Exchange Capacity, BS = Base Saturation.

[1]Codes for Reference Soil Group (IUSS Working Group WRB., 2006): CM = Cambisol, FR = Ferralsol, and Codes for Qualifiers: ac = Acric, au = Alumic, ce = Clayic, cr = Chromic, dy = Dystic, dyo = Orthodystic, ec = Escalic, fl = Ferralic, flh = Hyperferralic, ha = Haplic, hd = Hyperdystic, hu = Humic, vt = Vetic, xa = Xanthic.

[2]Soil C stocks in rubber plantations were calculated with the bulk density data from the secondary forest (see methods Chapter 2).

Table S2.2. The effect[1] of correction for bulk density changes on estimates of soil organic carbon stock changes. Corrections for bulk density changes were applied for estimates of the soil organic carbon stocks in rubber plantations and for the estimates of absolute[2] and relative[3] differences in soil organic carbon stocks between rubber plantations and secondary forests for our own data[4] (means ± SE), and for the cited studies that reported soil organic carbon concentration and bulk density data.

Country	Age (year)	Depth (m)	Soil C stock (Mg ha^{-1})			Abs. soil C stock difference (Mg ha^{-1})		Rel. soil C stock difference (%)		Effect (%)	Source
			Forest	Rubber Pl. (Without BD correction)	Rubber Pl. (With BD correction)	Without BD correction	With BD correction	Without BD correction	With BD correction		
China	5-46	0-0.15	43.9 ± 2.6	33.7 ± 1.2	30.3 ± 1.9	-10.0 ± 2.7	-11.8 ± 1.4	-21.2 ± 5.5	-26.9 ± 2.8	-6	Chapter 2
China	5-46	0.15-0.3	38.9 ± 1.5	28.5 ± 1.4	29.8 ± 1.6	-10.4 ± 1.5	-8.2 ± 1.1	-26.7 ± 3.9	-21.4 ± 3.2	5	Chapter 2
China	5-46	0.3-0.6	52.0 ± 1.6	45.4 ± 2.1	43.6 ± 2.6	-7.0 ± 2.5	-8.0 ± 3.0	-13.6 ± 4.8	-15.4 ± 5.6	-2	Chapter 2
China	5-46	0.6-0.9	35.2 ± 3.7	27.9 ± 1.8	28.0 ± 1.9	-7.3 ± 3.5	-6.5 ± 3.6	-18.2 ± 7.3	-16.0 ± 8.0	2	Chapter 2
China	5-46	0.9-1.2	26.0 ± 1.0	22.8 ± 1.2	23.1 ± 1.3	-3.6 ± 1.7	-2.9 ± 1.8	-13.7 ± 6.7	-11.2 ± 7.0	2	Chapter 2
China	5-46	0-1.2	196.0 ± 3.5	158.3 ± 5.6	154.9 ± 6.2	-38.3 ± 5.5	-37.4 ± 4.7	-19.7 ± 2.9	-19.3 ± 2.7	0	Chapter 2
China	3	0-0.2	68.7	45.3	41.9	-23.4	-26.8	-34	-39	-5	Yang et al. (2004)
China	3	0.2-0.4	46.3	39.4	37.7	-6.8	-8.6	-15	-19	-4	Yang et al. (2004)
China	3	0.4-0.6	35.9	35.3	34.9	-0.6	-1.0	-2	-3	-1	Yang et al. (2004)
China	3	0-0.6	150.9	120.1	114.6	-30.8	-36.3	-20	-24	-4	Yang et al. (2004)
China	7	0-0.2	68.7	52.5	47.9	-16.2	-20.9	-24	-30	-7	Yang et al. (2004)
China	7	0.2-0.4	46.3	40.3	38.2	-6.0	-8.0	-13	-17	-4	Yang et al. (2004)
China	7	0.4-0.6	35.9	34.2	33.6	-1.7	-2.3	-5	-7	-2	Yang et al. (2004)
China	7	0-0.6	150.9	127.0	119.6	-23.9	-31.2	-16	-21	-5	Yang et al. (2004)
Brazil	17	0-0.05	9.1	6.8	5.1	-2.3	-4.0	-25	-43	-18	Salimon et al. (2009)
Brazil	17	0.05-0.1	7.5	4.7	3.7	-2.8	-3.8	-37	-50	-13	Salimon et al. (2009)
Brazil	17	0.1-0.175	9.3	5.5	4.4	-3.8	-4.9	-41	-52	-12	Salimon et al. (2009)
Brazil	17	0.175-0.275	10.6	7.5	7.2	-3.1	-3.4	-29	-32	-3	Salimon et al. (2009)

Country	Age (year)	Depth (m)	Soil C stock (Mg ha^{-1})			Abs. soil C stock difference (Mg ha^{-1})		Rel. soil C stock difference (%)		Effect (%)	Source
			Forest	Rubber Pl. (Without BD correction)	Rubber Pl. (With BD correction)	Without BD correction	With BD correction	Without BD correction	With BD correction		
Brazil	17	0.275-0.375	9.3	4.9	4.3	-4.4	-5.0	-47	-54	-7	Salimon et al. (2009)
Brazil	17	0.375-0.475	9.8	4.8	4.6	-5.0	-5.2	-51	-53	-2	Salimon et al. (2009)
Brazil	17	0.475-0.625	11.3	7.9	7.3	-3.4	-4.0	-30	-36	-6	Salimon et al. (2009)
Brazil	17	0.625-0.825	14.4	6.8	6.3	-7.6	-8.1	-53	-57	-4	Salimon et al. (2009)
Brazil	17	0.825-1.0	14.5	7.3	6.8	-7.2	-7.7	-50	-53	-3	Salimon et al. (2009)
Brazil	17	0-1.0	95.8	56.2	49.7	-39.6	-46.1	-41	-48	-7	Salimon et al. (2009)

Rubber Pl. = Rubber plantation, Abs. = Absolute, Rel. = Relative, BD = Bulk Density

[1]Effect was calculated as the relative stock differences corrected for BD changes minus the relative stock differences not corrected for BD changes.

[2]Absolute differences in stocks were calculated as rubber plantations minus reference forest.

[3]Relative differences in stocks were calculated as rubber plantations minus reference forest divided by reference forest multiplied by 100.

[4]For our own data, the absolute and relative differences in soil C stocks were based on comparison of the means of rubber plantations with the reference forest.

Table S3.1. Mean[1] (SE) soil characteristics at terrace positions[2] A-B-C-D-E in three rubber plantations.

Charac-teristics	Depth (m)	5-year old plantation					29-year old plantation					44-year old plantation				
		A	B	C	D	E	A	B	C	D	E	A	B	C	D	E
Sand (%)[3]	0-0.1	42.2 (0.4)a	39.1 (1.8)ab	41.8 (0.3)a	40.7 (0.1)ab	39.5 (0.3)b	41.7 (1.0)a	45.9 (1.2)a	45.7 (2.7)a	42.0 (1.8)a	40.7 (1.3)a	25.6 (0.6)ab	23.6 (1.6)abc	21.2 (2.0)bc	20.4 (0.7)c	26.2 (0.2)a
	0.45-0.6	37.0 (1.0)a	39.8 (1.8)a	38.5 (1.5)a	37.8 (0.7)a	38.5 (0.4)a	38.2 (0.2)a	37.1 (1.3)a	37.8 (3.1)a	39.0 (1.7)a	39.3 (0.9)a	21.6 (2.6)a	15.4 (0.4)a	17.4 (1.9)a	21.0 (2.0)a	18.8 (1.0)a
	0.90-1.2	37.4	37.5	37.5	39.1	39.1	34.2	33.9	33.9	35.1	35.1	12.6	14.4	14.4	12.8	12.8
Silt and clay (%)[3]	0-0.1	57.8 (0.4)b	60.9 (1.8) ab	58.2 (0.3)b	59.3 (0.1)ab	60.5 (0.3)a	58.3 (1.0)a	54.1 (1.2) a	54.3 (2.7)a	58.0 (1.8)a	59.3 (1.3)a	74.4 (0.6)bc	76.4 (1.6)abc	78.8 (2.0)ab	79.6 (0.7)a	73.8 (0.2)c
	0.45-0.6	63.0 (1.0)a	60.2 (1.8) a	61.5 (1.5)a	62.2 (0.7)a	61.5 (0.4)a	61.8 (0.2)a	62.9 (1.4) a	62.2 (3.1)a	61.0 (1.7)a	60.7 (0.9)a	78.4 (2.6) a	84.6 (0.4)a	82.6 (1.9)a	79.0 (2.0)a	81.2 (1.0)a
	0.90-1.2	62.6	62.5	62.5	60.9	60.9	65.8	66.1	66.1	65.0	65.0	87.4	85.6	85.6	87.2	87.2
Bulk density (g cm^{-3})[4]	0-0.1	0.89	1.07	1.07	0.96	0.96	1.09	1.24	1.24	1.28	1.28	1.16	1.11	1.11	1.04	1.04
	0.45-0.6	1.32	1.33	1.33	1.06	1.06	1.35	1.33	1.33	1.19	1.19	1.25	1.17	1.17	1.19	1.19
	0.90-1.2	1.41	-	-	1.33	1.33	1.29	-	-	1.30	1.30	1.19	-	-	1.19	1.19
pH H_2O[3]	0-0.1	4.6 (0.1)a	4.9 (0.1) a	4.8 (0.1)a	4.8 (0.0)a	4.8 (0.0) a	4.6 (0.0)a	4.6 (0.0) a	4.6 (0.0)a	4.7 (0.0)a	4.7 (0.0) a	4.7 (0.1) a	4.7 (0.0) a	4.8 (0.1)a	4.9 (0.1)a	4.7 (0.1) a
	0.45-0.6	4.7 (0.1)a	4.7 (0.1)a	4.7 (0.1)a	4.7 (0.0)a	4.8 (0.1)a	4.9 (0.1)a	4.8 (0.1)a	4.8 (0.1)a	4.9 (0.1)a	4.8 (0.1)a	5.0 (0.0)a	4.9 (0.1)a	5.1 (0.1)a	5.0 (0.1)a	4.9 (0.0)a
	0.90-1.2	5.1	4.7	4.7	5.0	5.0	5.2	4.4	4.4	5.1	5.1	5.6	5.3	5.3	4.4	4.4

[1]Within a row and each rubber plantation, means having different letters are significantly different between terrace positions (one-way ANOVA with Tukey HSD, or Kruskal-Wallis ANOVA with pairwise Wilcoxon test at $P \leq 0.05$).

[2]Terrace positions are described in Figure 1c in Chapter 3.

[3]n = 3 for 0-0.10 m and 0.45-0.60 m, and n = 1 for 0.90-1.2 m.

[4] n = 1 for all depths.

Table S4.1. Direction of effects[1] of soil, vegetation, and topographic characteristics on litter layer carbon stock and yearly litter fall in forest.

Response	Effect	Direction of effect	P value
Litter layer carbon stock	Elevation	+	0.05
	Soil pH H2O (0-0.15-m depth)	-	0.07
	Yearly litter fall	+	0.03
Yearly litter fall	Tree basal Area	+	0.002
	ECEC subsoil	+	0.14

[1]Linear mixed effects model with sampling plot as random intercept. Fixed effects included in the full models were elevation, slope gradient, compound topographic index, silt plus clay concentration, soil pH H_2O, tree basal area, litter C:N ratio, yearly litterfall.

Table S4.2. Direction of effects[1] of elevation on silt plus clay concentration, soil C:N ratio, pH H_2O, ECEC[2] of the subsoil, and tree basal area in all land-use types combined.

Response	Depth (m)	P value
Silt plus clay concentration	0-0.15	0.93
	0.15-0.3	0.88
Soil C:N	0-0.15	0.48
	0.15-0.3	0.63
Soil pH H_2O	0-0.15	0.44
	0.15-0.3	0.26
ECEC at 0.6-0.9 m	0.6-0.9	0.94
Tree basal Area	Not applicable	0.61

[1] Linear mixed effects model with sampling plot as random intercept of variation in the response variable with elevation.
[2]ECEC, Effective Cation Exchange Capacity.