Geoderma 375 (2020) 114484

Contents lists available at ScienceDirect

Geoderma

journal homepage: www.elsevier.com/locate/geoderma

Radiocarbon evidence of the impact of forest-to-plantation conversion on soil organic carbon dynamics on a tropical island

Yishan Jiang[a,f], Chunling Luo[a,b,*], Dayi Zhang[c], Nicholas J. Ostle[d], Zhineng Cheng[a], Ping Ding[e], Chengde Shen[e], Gan Zhang[a,*]

[a] State Key Laboratory of Geochemistry, Guangzhou Institute of Geochemistry, Chinese Academy of Sciences, Guangzhou 510640, China
[b] Joint Institute for Environmental Research and Education, South China Agricultural University, Guangzhou 510642, China
[c] School of Environment, Tsinghua University, Beijing 100084, China
[d] Lancaster Environment Centre, Lancaster University, Lancashire LA14YQ, UK
[e] State Key Laboratory of Isotope Geochemistry, Guangzhou Institute of Geochemistry, Chinese Academy of Sciences, Guangzhou 510640, China
[f] University of Chinese Academy of Sciences, Beijing 100049, China

ARTICLE INFO

Handling Editor: Yvan Capowiez

Keywords:
Radiocarbon analysis
Soil organic carbon dynamics
Microbial communities
Forest-to-plantation conversion

ABSTRACT

Tropical soils are critical terrestrial carbon reservoirs with abundant biodiversity that respond rapidly to environmental change. Globally, the expanding conversion of natural tropical forest to plantations in recent decades, to meet economic demands, has markedly influenced the cycling of soil organic carbon (SOC) pools; however, the mechanisms underlying the changes in SOC dynamics are poorly understood. In this study, we examined the SOC dynamics and soil microbial communities at five adjacent tropical forest sites characterized by different logging and plantation practices over the past few decades on Hainan Island, China, by applying natural abundance radiocarbon (^{14}C) and phospholipid fatty acid (PLFA) analysis. At a > 35-year rubber plantation site and a > 50-year eucalyptus plantation site, an abnormal up-profile decrease in radiocarbon abundance was observed in the upper 30 cm soil layer. This could be indicative of continued soil organic matter decomposition long after forest–plantation conversion, and was consistent with the SOC inventories in the upper 30 cm soil layer at the two sites being significantly lower than those of NF. Both the SOC apparent radiocarbon ages and SOC inventory at a eucalyptus plantation site in which tillage was stopped 20 years ago were similar to those of NF, indicating that a recovery process had occurred. The soil microbial biomass was generally lower at the plantation sites than at the NF site. Both the radiocarbon abundance and SOC inventories in the upper 30 cm soil layer showed positive correlations with the soil microbial biomass, suggesting that microbes may have played a key role in the fate of SOC. This study provided evidence that forest–plantation conversion may facilitate the dissimilation of SOC, and also demonstrated that radiocarbon can serve as a powerful tool for assessing the potential changes of soil carbon dynamics resulting from forest-to-plantation conversion.

1. Introduction

Forest conversion from natural forest (NF) to plantation is increasing globally. The proportion of planted forests increased from 4% to 7% of all forests worldwide, from 1990 to 2015 (FAO, 2018). Forest conversion significantly influences soil properties and biochemical processes (Ross, 1993; McGrath et al., 2001; Pabst et al., 2013), potentially leading to decreased fertility, soil erosion, reduced biodiversity, and frequent occurrences of pests and diseases (Liu et al., 2017).

One emerging concern related to forest-to-plantation conversion is the influence of the anthropogenic activity on the fate of soil organic carbon (SOC). Changes in forest cover, which affect the carbon inventory, may increase the exportation of carbon from soils to the atmosphere and hydrosphere (Foley et al., 2005), in turn influencing the SOC sequestration and thus the likelihood of mitigating global climate change and food security (Lal, 2004).

The fate of SOC influenced by the anthropogenic activity is particularly important in tropical regions, wherein millions of hectares of forest are cleared annually for agriculture, pasture, cultivation, and timber (Abood et al., 2015; Guillaume et al., 2015; Laumonier et al., 2010; Margono et al., 2014; Wilcove and Lian, 2010). Additional

* Corresponding authors at: State Key Laboratory of Geochemistry, Guangzhou Institute of Geochemistry, Chinese Academy of Sciences, Guangzhou 510640, China (C. Luo).
E-mail addresses: clluo@gig.ac.cn (C. Luo), zhanggan@gig.ac.cn (G. Zhang).

https://doi.org/10.1016/j.geoderma.2020.114484
Received 27 July 2019; Received in revised form 21 May 2020; Accepted 27 May 2020
Available online 09 June 2020
0016-7061/

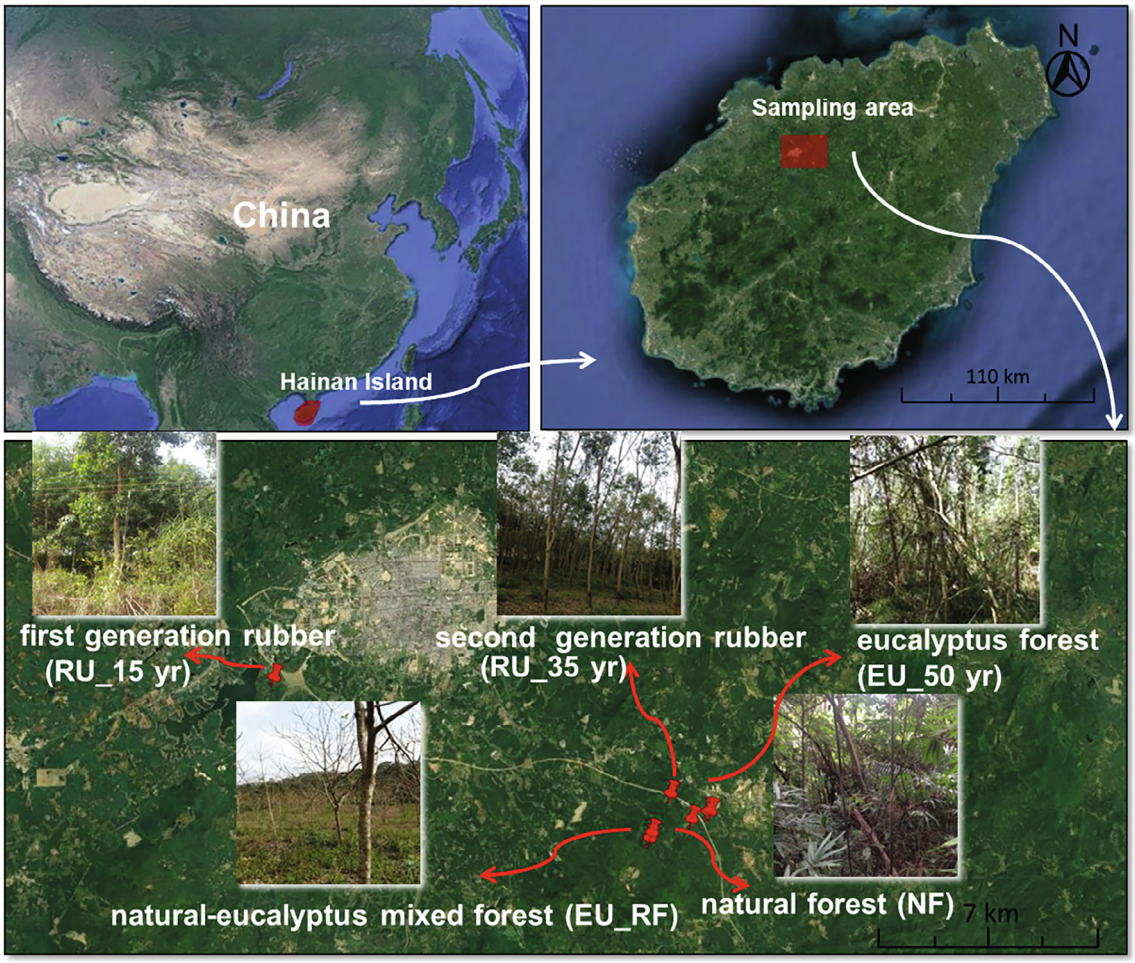

Fig. 1. Location and forest type of sampling forest sites in Hainan Island (The map is based on Google Earth).

knowledge of the mechanisms and magnitude of the response of the SOC pool in tropical soils to forest–plantation conversion is urgently needed (Van der Kamp et al., 2009). Although some studies have indicated that loss of surface and subsurface SOC has occurred during NF conversion to agriculture in humid tropical regions, such as Indonesia (Noordwijk et al., 1997; Don et al., 2011), there have been few studies on the influence of forest–plantation conversion on the fate of SOC in other tropical regions.

The fate of forest SOC is often difficult to predict due to the effects of various factors driving SOC mineralization, such as forest type, climate, and soil properties (IPCC, 2007). It is important to understand and quantify the effects of these factors on carbon flux between the soil and atmosphere, and on feedbacks within the global carbon cycle (Mahecha and Schmidtlein, 2010). Earth system models (ESMs) have been used to ascertain the role of tropical soils in the carbon balance following forest–plantation conversion, but simulated carbon dynamics have not yet been systematically tested against observations (Trumbore et al., 1995; Schmidt et al., 2011). A reduction of the SOC inventory has frequently been observed in agriculturally modified soils, attributed mostly to reduced annual input of plant residues, increased decomposition, and accelerated soil respiration. Increases in soil temperature, aeration, moisture, and mechanical stirring have been suggested as the factors driving reduction of the SOC inventory (Harrison et al., 1993). Soil organic matter (SOM) persists not only because of its intrinsic properties, but also due to physicochemical and biological influences from the surrounding environment that modify the probability and rate of decomposition (Schmidt et al., 2011). This highlights the key role of soil microbes in the cycling of SOC (Figuerola et al., 2015).

Natural abundance radiocarbon (^{14}C) is useful for tracking the fate of SOC, but data on soil radiocarbon are still limited. Radiocarbon is a cosmogenic radionuclide with a half-life of 5,730 years, which makes the $^{14}C/^{12}C$ ratio a useful parameter reflecting the rates of SOM decomposition and radioactive decay (Trumbore, 2000). The thermonuclear weapons testing conducted in the early 1960s nearly doubled the $^{14}C/^{12}C$ ratio in the atmosphere, producing a “bomb peak” that has since served as a global isotopic tracer for carbon exchange among different carbon reservoirs (Levin and Hesshaimer, 2000). Radiocarbon serves as a “clock” indicating the amount of time that carbon has spent in the ecosystem, and can be helpful for quantifying carbon residence time in undisturbed systems. Therefore, by reference to natural abundance radiocarbon, the source and turnover rate of SOC can be calculated, enabling evaluation of the forest–plantation conversion occurring at the decadal scale on SOC dynamics. Previous studies on soil radiocarbon have provided evidence that SOC sequestration may be substantially overestimated by many ESMs (He et al., 2016), and that fast cycling SOC pools would be more sensitive to environmental changes than slow cycling SOC pools (Harrison et al., 1993; Trumbore, 1993; Trumbore et al., 1999; Mathieu et al., 2015). In natural soil horizons, radiocarbon analysis has indicated decreasing radiocarbon abundance from the surface to the deep soil, which may be attributable to different biochemical processes through which soil microbe by-products are produced, and to the stabilization of SOM (Trumbore, 2009).

Through radiocarbon analysis, this study examined the potential influence of forest–plantation conversion on the fate of SOC in a tropical region. Hainan Island was the field study location. With an area of 3.22 million hectares, Hainan Island is the largest tropical island in

China. Widespread forest conversion, from natural tropical forests to rubber tree and eucalyptus plantations, has taken place since the 1960s, and there is a good record of the logging and plantation activities conducted over the decades, which is ideal for the purposes of this study.

2. Materials and methods

2.1. Study area

Hainan Island is located in a typical tropical monsoon climate region. It is the only major island in the Indo-Burma biodiversity hotspot (Zhai et al., 2012), showing very high levels of overall diversity and floral endemism. After World War II, the island became the biggest rubber production area in China due to its geographical advantages (Hainan Provincial Bureau of Statistics, 2018). Rubber tree and eucalyptus are planted on a large scale. With a 30-year life cycle, most first-generation rubber trees were generally replaced by second-generation trees, which will be further replaced by the third generation (Fig. 1). To harvest rubber latex, local people make annual cuts on rubber tree trunks at 30° from the horizontal (rubber tapping) to create a latex vessel. The tapping usually starts from 6 years after plantation and lasts for 30 years until felling. Therefore, the tap marks left on the trunk can help confirming the age of rubber trees.

Our study area is one of the earliest rubber plantation areas in the northwest of Hainan Island (19°26′–19°27′ N, 109°38′–109°39′ E), c.a. 10 km from the centre of Danzhou City (Fig. 1). The annual average temperature, precipitation, and relative humidity are 23.5 °C, 1705 mm, and 83%, respectively (Donghai et al., 2006). The soil in the studied region is classified as Ultisol by the US Soil Taxonomy system and Latosol by Chinese geographic classification system, i.e. a mineral soil including argillic horizon, showing reddish dark brown on the surface, presenting strong acidic. The annual average soil temperature was over 8 °C (Matson et al., 1999; Uexküll and Mutert, 1995).

2.2. Experimental design

According to the logging and plantation history, five forest sites were selected for measuring natural abundance radiocarbon, SOC inventory and soil microbial communities. Among the five sites (Fig. 1),

- NF is at a naturally formed secondary forest growing where a primitive forest had been logged in the 1950 s, dominate by *Psychotria* L;
- RU_15 yr is at a 'newly' planted first-generation rubber tree forest for 15 years, in place of a logged natural secondary forest, and is referred here as a > 15-year rubber plantation site.;
- RU_35 yr is at a second-generation rubber tree forest, in place of a logged first generation rubber plantation, and is referred as a > 35-year rubber plantation site;
- EU_50 yr is at a forest with sustaining eucalyptus forest for 50 years, on the basis of a logged natural secondary forest, and is referred as > 50-year eucalyptus plantation site;
- EU_RF is at an eucalyptus forest abandoned 20 years ago, featuring an unmanaged eucalyptus mixed with natural invading native broadleaf species (*Psychotria* L. et al.), and is referred as as a recovering eucalyptus plantation site.

2.3. Soil sampling and physicochemical properties

The field sampling was carried out in January 2015. At each forest/plantation site, five replicated soil profiles in a range of 10 × 10 m area were collected. Three soil pits (~1 m depth) were hand dug using a clean stainless steel spade. Soils from the O (0–10 cm), A (10–30 cm), B (30–50 cm), and C (50–100 cm) horizons were distinguished and collected, respectively. At sites where the upper-layer soil (O and A horizons) was thicker than 30 cm along the 1 m excavated profile, only the upper 30 cm soil layer (O and A horizons) were collected; whereas samples from the upper 30 cm layer (O and A horizons) and subsurface soil (B or C horizon) were collected at the rest sites. Upon soil collection, superficial litter, large roots and nonorganic materials if any were hand removed. Soil samples were sealed in plastic bags, ice-cooled in portable cool boxes, and immediately transported to the laboratory.

After passing through a 2 mm sieve, a portion of the upper 30 cm soil layer (O and A horizons), with the most abundant microbes and representing the active SOC pool, were employed for SOC inventory and microbial analysis, respectively. Specifically, about 10 g soil sample from the O and A horizons (0–30 cm) from each soil profile were stored at 4 °C for soil PLFA analysis within 2 days. The remaining soils were dried at 30 °C to a constant weight. A well-mixed ~ 20 g of O and A horizons (0–30 cm) soils was ball-milled to homogenise for chemical analysis. The soil samples were characterised for soil density, moisture, SOC, total nitrogen (TN), total phosphate (TP) and soil pH, respectively, following existing lab protocols. Briefly, soil density was measured using a bulk density ring approach; moisture was by conventional dry-wet weight method; SOC and TN were analysed with a Vario EL III elemental analyser (Elementar, Germany); TP was analysed with a Seal continuous flow auto-analyser (type AA3) with soil being digested in sulphuric acid/hydrogen peroxide reagent; soil pH was measured using a PHSJ-4F pH meter (Rex Electric Chemical, Shanghai, China) with soil being dissolved in KCl solution (1.0 M) at a ratio of 1:2.5 (m/v).

2.4. PLFA analysis

PLFA extraction was performed following the Bligh–Dyer method (Bligh and Dyer, 1959). The retrieved PLFAs were trans-methylated in mild alkaline methanol to yield their fatty acid methyl esters (FAMEs) for gas chromatographic (GC) separation. Briefly, the dried phospholipid fractions were dissolved in 1 mL MeOH/toluene (1:1, v/v), followed by adding 1 mL KOH in methanol (0.2 M, freshly prepared), and incubated in a water bath at 37 °C for 15 min. Then the liquid phase was split by adding 2 mL hexane/chloroform (4:1, v/v) solution, 0.3 mL acetic acid, and 2 mL milli-Q water. The upper organic phase was transferred to a test tube. The bottom phase was washed again with 2 mL hexane/chloroform (4:1, v/v) solution. The supernatant was combined in the same test tube, and then evaporated under nitrogen stream until dry.

Qualitative analyses of FAMEs was mainly performed using the GLC reference standard GRS617 prepared by Brian Nutter (Nu-Chek-Prep, Elysian Township, MN, USA) with a combination of pure fatty acid standards (Supelco 37 Component FAME Mix and 26 component Bacterial Acid Methyl Ester Mix 47080-U; Sigma, Deisenhofen, Germany). In addition, a pure C19:0 FAME purchased from Dr. Ehrenstorfer (Germany) was dissolved in hexane and used as an internal standard for the quantification of FAMEs. Microbial biomass was calculated as the sum of all peaks (μg PLFA/g soil) of PLFAs identified < 20.5 carbon atoms long (Vestal and White, 1989; Zelles, 1999). PLFAs representing broad taxonomic groups such as fungi and Gram-positive, Gram-negative, methanotrophic, and anaerobic bacteria have been used successfully to describe soil microbial community compositions, particularly where the influence of plant compounds and waxes were removed (Frostegård et al., 2011; Ruess and Chamberlain, 2010; Zelles, 1997). Here, we calculated the specific PLFA biomarkers and their relative abundances to represent various bacterial and fungal groups: 15:0 iso (Kaur et al., 2005; Zelles, 1997, 1999) (Gram-positive bacteria), 16:0 10-methyl (Ratledge, 2008) (actinobacteria); 16:1 w7c (Ratledge, 2008; Zelles, 1999) (Gram-negative bacteria), 16:1 w5c (Olsson, 1999; Olsson et al., 1995) (arbuscular mycorrhizal fungi), 18:1 w9c (Bardgett et al., 1996; Frostegård et al., 2011) (saprotrophic fungi), 18:2 w6,9c (Frostegard and Baath, 1996; Joergensen and Wichern, 2008; Kaiser et al., 2010) (saprotrophic fungi) and 19:0 cyclo (Vestal and White, 1989) (anaerobic, gram-negative bacteria).

2.5. SOC inventory estimation

We estimated only the carbon inventories in the upper 30 cm soil layer of all soil profiles. The amounts of SOC stored in soil were calculated by measured SOC content and bulk density (BD) in the soil horizons, as shown in Eq. (1) (Torn et al., 2009).

$$\text{SOC inventory}(\text{g C/cm}^2) = \text{BD}(\text{g soil/cm}^3) \times (\text{g SOC/g soil}) \times \text{depth}(30\,\text{cm}) \quad (1)$$

2.6. Radiocarbon analysis of SOC

To minimize the cost for radiocarbon analysis, we measured only soil-layer composite samples, which were carefully prepared by mixing the same-layer samples from the five replicated soil profiles, on the basis of equal SOC mass contribution at each forest/plantation site. Even this experimental design does not include necessary replicates to make generalisable inference about land-use influence on the SOC dynamic, the radiocarbon analysis of soil-layer composite samples from the five forest sites can provide useful information about the forest-to-plantation conversion influence. The sub-soil mass in composite soil was calculated to meet the detection limit of radiocarbon analysis (~100 μg carbon) and SOC content. Thus, a radiocarbon data would represent an average of the five replicated sub-soils. Combustion and graphitisation of soil samples were performed in Guangzhou Institute of Geochemistry (GIG) using published procedures (Ding et al., 2010). The graphitised samples were measured at the AMS facility in the State Key Laboratory of Nuclear Physics and Nuclear Technology, Peking University, with a precision above 0.5‰.

Radiocarbon abundance is reported as $\Delta^{14}C$ values per mille (‰) relative to the ^{14}C activity of the oxalic acid standard with 1-σ measurement uncertainty (Reimer et al., 2004; Stuiver and Polach, 2006). Negative $\Delta^{14}C$ values corresponded to radiocarbon levels before 1949, and positive $\Delta^{14}C$ values indicated the contribution of radiocarbon derived from atmospheric nuclear weapons testing in the 1950s and 1960s.

2.7. Statistical analysis

Redundancy analysis (RDA) was performed by the "Bioenv" function in the "vegan" package (vegan v2.4–4) of R software (R Development Core Team, Vienna, Austria), using UniFrac weighting and the normalised distance matrix with default parameters. Correlation analysis was conducted using SPSS for Windows (ver. 20.0; SPSS Inc., Chicago, IL, USA). One-way ANOVA and multivariate analysis of variance (MANOVA) were used to examine the significance of differences and variability at the 95% confidence level ($P < 0.05$), respectively.

3. Results and discussion

3.1. Abiotic soil properties

Detailed soil density, moisture, pH, SOC, TN, TP, SOC to TN (C/N) ratio are shown in Figs. 2 and S1. Among the five forest sites, the abiotic parameters of soil moisture (12–18%), pH (3.8–4.5), SOC (0.8–1.5%), and total phosphorus (TP; 0.016–0.060%) showed significant differences (Fig. 2), whereas soil density (1.3–1.6 g/cm^3), TN (0.10–0.12%), and the C/N ratio (8–13) were similar (Fig. S1). There results were in general agreement with those reported in previous studies of tropical forest soils in China, including soils on Hainan Island (Liu et al., 2017; Xiankai et al., 2015).

The > 15-year rubber plantation site (RU_15 yr) showed higher soil moisture values than the other sites (18% vs. 12–15%) (Fig. 2). This may be attributable to the higher soil density and leaf area index (LAI; Wu et al., 2009) at the RU_15 yr site, as the rubber plantation is currently in its vigorous growth stage. Regarding soil pH, the rubber plantation sites were more acidic (pH 3.7–4.0) than the NF site (pH 4.2) and the eucalyptus plantation sites (pH 4.4–4.5). The RU_35 yr and EU_50 yr sites (0.8%) had significantly lower SOC levels than the NF, RU_15 yr, and recovering eucalyptus plantation (EU_RF) sites (1.1–1.5%), indicating greater soil erosion with longer plantation time. The soil at RU_35 yr (0.06%) had significantly higher TP than the soil at the other sites (0.02–0.03%), which may be related to the large amount of phosphate fertilizer applied in the latter stages of rubber plantation.

3.2. Soil microbial biomass and structures

Microbially regulated SOC pathways are expected to affect the cycling of SOM (Smith et al., 2014; Trumbore, 2009), and the compositions and concentrations of phospholipid fatty acids (PLFAs) have been employed widely as indicators of microbial structure and mass in soil. The PLFA-derived soil microbial community biomass for the different forest sites, including total bacteria, fungi, Gram-positive bacteria, Gram-negative bacteria, and microbes, are compared in Figs. 2 and 3.

The fungi-to-bacteria (F/B) ratio showed significant differences among the five forest sites (Fig. 2), but the Gram positive-to-Gram negative bacteria (GP/GN) ratio did not (see Fig. S1). The F/B and GP/GN ratios are indicative of the microbial community structure in soil. The F/B ratio at RU_35 y (1.0) was significantly higher than the ratios at the other forest sites (0.4–0.6), suggesting that a longer rubber plantation time may result in distinct changes in the microbial community structure.

The total soil microbial biomass, and that of each microbial community, showed similarities among the five forest sites (Fig. 3). The soil microbial biomass at plantation sites EU_50 yr, RU_15 yr and, especially, RU_35 yr was obviously lower than that of the NF. After cessation of tillage for 20 years, the microbial biomass at EU_RF was higher than that at the other plantation sites, and approximately equal to half the biomass of site NF. It can be concluded that, in rubber and eucalyptus plantations in tropical areas, loss of microbial biomass will occur over time. Even after cessation of tillage for 20 years, the microbial biomass in the eucalyptus plantations was not comparable with that in the NF.

A redundancy analysis (RDA) was applied to explore the potential environmental factors driving microbial change. Environmental parameters, i.e., longitude, latitude, mean annual temperature (MAT), and mean annual precipitation (MAP), as well as soil abiotic parameters (density, moisture, pH, SOC, TN, C/N, and TP), were included as variables in the RDA analysis, and PLFA patterns were used to represent the microbial structure. As shown in Fig. 4, soil pH, SOC content and TP displayed significant correlations with PLFA concentrations, suggesting that forest–plantation conversion altered the soil microbial biomass and structure through modification of soil pH, SOC content and TP (Rodrigues et al., 2013; Wang et al., 2017).

3.3. SOC inventories

As shown in Fig. 5, the SOC inventories in the upper 30 cm soil layer of the long-term plantation sites (RU_35 yr and EU_50 yr) were lower than the inventory of the NF. This may be attributable to either enhanced decomposition of surface SOM or to artificial removal of soil protective cover (litter layer and canopy) by the continuous planting practice, which consequently reduced the carbon input to surface soil (Guillaume et al., 2015; Islam and Weil, 2000; Sidle et al., 2006). The SOC inventory at site EU_50 yr was lower than that at RU_35 yr, implying that accelerated SOC consumption due to more amended chemical fertilizer (Batjes, 1996) and longer plantation time of the eucalyptus plantation may result in more intensive carbon loss from the surface soil.

However, the SOC inventory in the upper 30-cm soil of EU_RF was comparable with the inventory in the NF. This indicates that the SOC

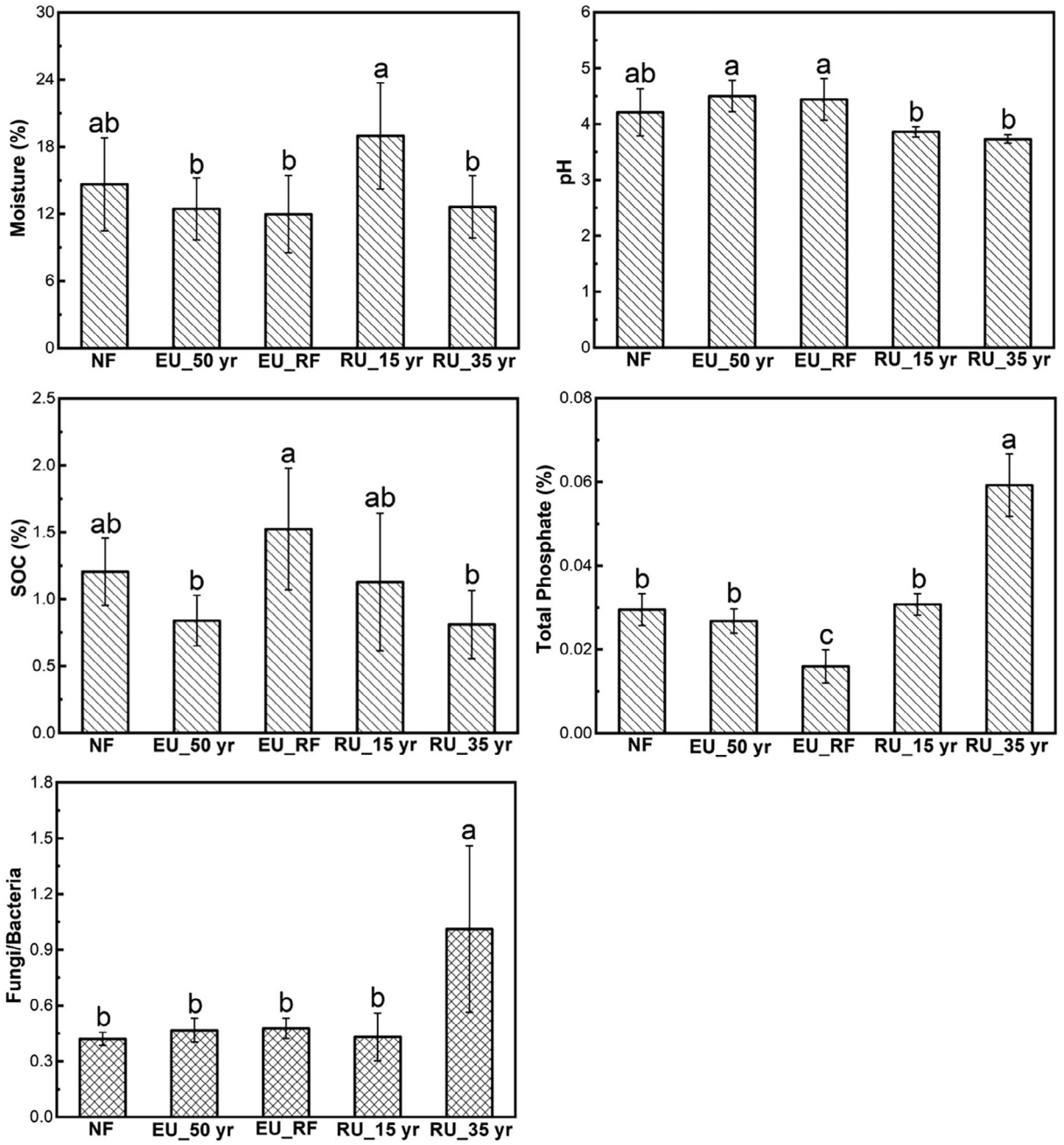

Fig. 2. Moisture, pH, SOC, TP and F/B ratios in the upper 30 cm soil layer at five forest sites.

inventory in the abandoned eucalyptus forest may have recovered to its natural state after cessation of tillage for 20 years. At site RU_15 yr, the SOC inventory in the upper 30 cm soil layer was comparable with that of NF, which may be attributable to application of organic fertilizer during the early stages of rubber plantation.

The SOC inventories in subsurface soils (B and C horizons) did not show significant differences between the plantation sites and NF (Fig. S2). Although forest–plantation conversion could introduce nutrients to the deep soil, via the root system, for example, there was little apparent effect on the soil intrinsic properties or SOC inventories of the subsurface soil in our study.

3.4. SOC dynamics inferred by radiocarbon

The measured SOC radiocarbon abundances of soil profile samples from the five forest sites are depicted in Fig. 6. Each soil sample was generated by mixing five replicate samples obtained from the same forest site. Generally, the SOC $\Delta^{14}C$ values in all samples ranged from –102‰ to 94‰.

In the upper 30 cm soil layer, the radiocarbon abundance was typical of that of modern carbon (6–94‰), consistent with previous reports of surface soil radiocarbon abundance measured elsewhere (Marin-Spiotta et al., 2008; Trumbore, 2000).

The SOC radiocarbon abundance data showed two distinct patterns. As shown in Fig. 6, the SOC $\Delta^{14}C$ values at sites NF and RU_15 yr displayed an up-profile increase in the A–O horizons. However, at sites RU_35 yr, EU_50 yr, and EU_RF, the $\Delta^{14}C$ values decreased toward the top of the profile. In the subsurface soils (30–100 cm), the SOC $\Delta^{14}C$ values at all five sites showed similar up-profile increases in the C–B horizons, as occurs naturally (Trumbore, 2009).

Organic carbon in soil can be divided into fast- and slow-turnover pools. Previous studies indicated that the fast-cycling carbon pool,

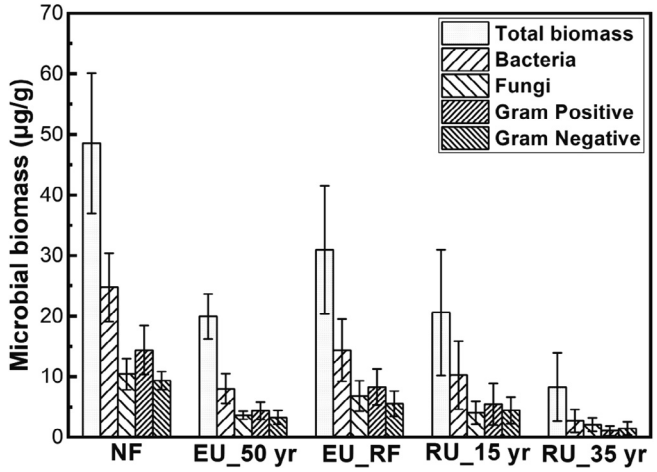

Fig. 3. Comparison of soil microbial biomass at the five forest sites.

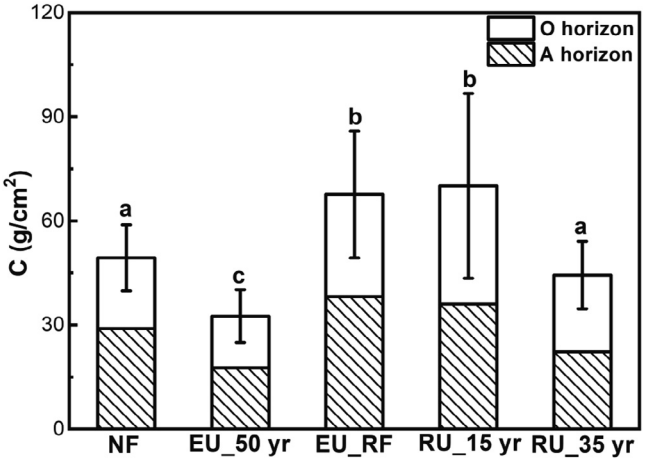

Fig. 5. SOC inventories in the upper 30 cm soils (O and A horizons) at the five forest sites.

characterized by a high $^{14}C/^{12}C$ proportion and arising from modern carbon input, constitutes a larger proportion of surface versus subsurface soil (Harrison et al., 1993; Trumbore, 1993; Trumbore et al., 1999). In our study, this was indicated by the decrease in radiocarbon with soil depth at site NF.

It has been noted that soil carbon disturbed by land-use change falls mainly into the fast-cycling "active" carbon pool in upper-layer soils, due to translocation of fresh and ancient carbon or the mixing of fast and slow carbon pools in the surface soil (Don et al., 2009; Wang et al., 1999). Accordingly, SOC $\Delta^{14}C$ values at plantation sites are expected to be homogenous in upper-layer soils. In our study, the SOC $\Delta^{14}C$ values at the two long-term plantation sites (RU_35 yr, EU_50 yr) were not homogenous as expected. Actually, they showed an opposite trend in the upper 30 cm soil layer, which was distinctly different from those at site NF. This suggests that the decrease of SOC sequestration in the upper 30 cm soil layer at the plantation sites was influenced not only by physical mixing (e.g., tillage) of SOM during long-term plantation, but also by microbial activities accelerating the decomposition of fast-cycling fresh SOM. It should be noted that the SOC $\Delta^{14}C$ value and the SOC radiocarbon age calculated by the $\Delta^{14}C$ value in the O horizon were close to those in the A horizon at site EU_RF. This implies a natural recovery of SOC sequestration in the upper 30 cm soil layer after the 20-year cessation of tillage at the abandoned site. Lastly, at site RU_15 yr, the two samples in the upper 30 cm soils displayed similar SOC radiocarbon ages calculated from the $\Delta^{14}C$ value; this may be due to the effect of physical mixing, considering the earlier forest–plantation conversion.

3.5. Role of microbes in SOC dynamics

To explore the potential mechanisms of the SOC dynamics modified

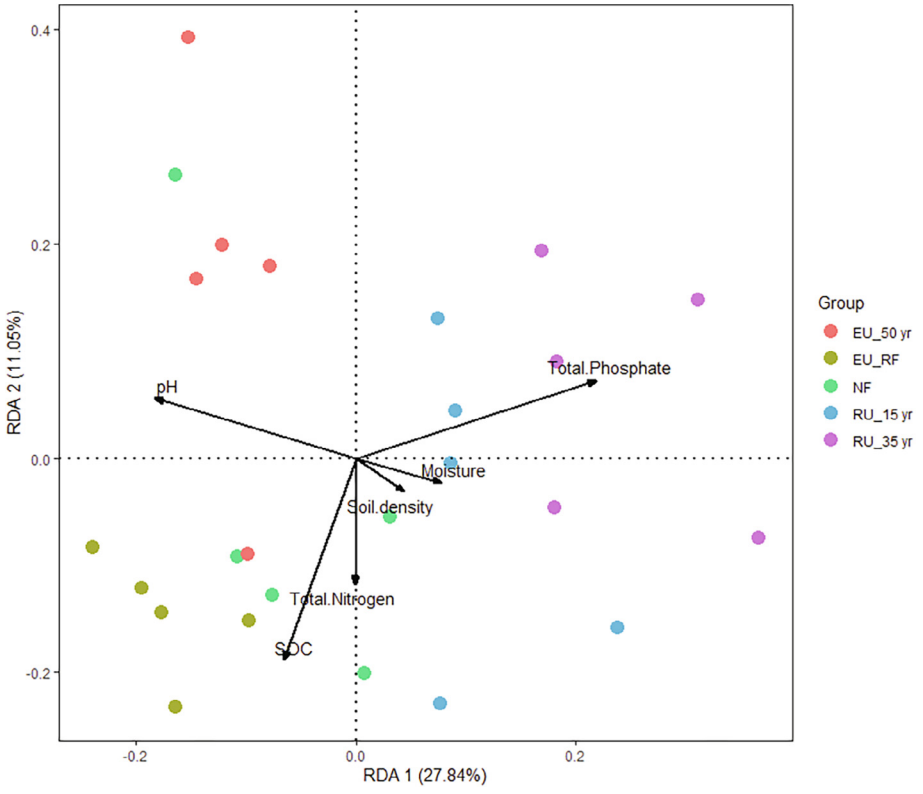

Fig. 4. Bioplot of redundancy analysis, with environmental factors, of the upper 30 cm soil samples based on microbial composition from the five sites. Soil PLFA data were fitted onto the RDAs ordination. Each point represents a profile from the same site. Vectors only indicate environmental variables that were significantly correlated with the ordination ($p < 0.05$).

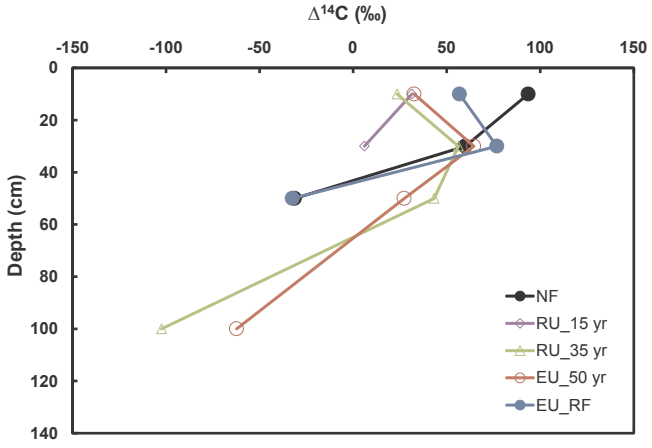

Fig. 6. SOC radiocarbon abundance in soil profiles at the five forest sites.

Table 1
Correlation coefficients between abiotic/biotic parameters (soil properties and microbial composition) and SOC inventory, radiocarbon abundance.

	C inven.-O[1] (N = 35)	C inven.-A[2] (N = 35)	C inven.-O + A[3] (N = 35)	Δ^{14}C-O[4] (N = 5)	Δ^{14}C-A[5] (N = 5)
Soil density	0.197	0.486	0.203	−0.643	0.569
Moisture	0.577	0.182	**0.979****	0.010	0.830
pH	−0.402	0.485	**0.654****	0.386	0.633
SOC	−0.551	−0.266	**0.712****	−0.052	−0.488
TN[a]	0.721	0.395	0.054	−0.362	−0.103
C/N[b]	0.722	0.465	0.233	0.439	−0.305
TP[c]	0.054	−0.633	−0.267	−0.400	−0.087
Bacteria	0.085	0.407	**0.421***	**0.949***	0.170
Fungi	0.041	0.382	**0.374***	**0.981****	0.310
GP[d]	0.068	0.409	**0.366***	**0.956***	0.203
GN[e]	0.129	0.414	**0.506****	**0.929***	0.110
F/B[f]	−0.201	−0.752	**0.399***	−0.478	0.105
GP/GN[g]	−0.061	0.677	**−0.349***	0.743	0.429
Biomass	0.031	0.419	0.003	**0.959***	0.236

*Correlation is significant at the 0.05 level (2-tailed).
**Correlation is significant at the 0.01 level (2-tailed).
[1] C inven.-O: SOC inventory of soil in O horizon; [2]C inven.-A: SOC inventory of soil in A horizon; [3]C inven.-O + A: SOC inventory of soil in O and A horizon; [4]Δ^{14}C-O: SOC Δ^{14}C of soil in O horizon; [5]Δ^{14}C-A: SOC Δ^{14}C of soil in A horizon.
[a] TN.: total nitrogen concentration; [b]C/N: ratio of soil C to nitrogen concentration; [c]TP: total phosphate concentration; [d]GN: Gram-negative bacteria; [e]GP: Gram-positive bacteria; [f]F/B: ratio of fungi to bacteria; [g]GP/GN: ratio of gram-positive to gram-negative bacteria.

by forest-plantation conversions, statistical analysis was carried out using the dataset of soil abiotic/biotic parameters, SOC inventories and radiocarbon abundance as a whole (Table 1, Table S1).

As shown in Table 1 (see also Fig. S3), the SOC inventories in the upper-layer (O and A horizon) soils were significantly positively correlated with soil moisture, pH, and SOC content ($p < 0.01$), suggesting that soil moisture and soil pH are good indicators of SOC inventory (Liu et al., 2017). For subsurface soil, more positive Δ^{14}C indicates younger or more recently produced carbon is being transported to and sequestered into subsoils. A significant positive correlation between pH and Δ^{14}C was observed (see Table S1), which is indicative of the importance of pH in maintaining/nourishing subsurface SOC.

The microbial biomass indices and microbial structure parameters (F/B and GP/GN) were significantly correlated with the SOC inventories in the upper 30 cm soil layer (O and A horizons) ($p < 0.05$). In the topsoil (O horizon), the soil biomass indices showed significant positive correlations with the Δ^{14}C of SOC ($p < 0.01$ for fungi, $p < 0.05$ for other biomasses; Table 1). This suggests that sequestration/capture of bomb radiocarbon (higher Δ^{14}C) in topsoil may occur with the development of soil microbes, especially fungi, which are more sensitive to changes in the ecological environment than other microbial communities (Apostel et al., 2013; Veresoglou et al., 2012). The changes in SOC radiocarbon abundance and inventories in the various soil horizons may have been driven by microbial processing of SOM in environmental carbon pools, with preferential enrichment of microbial products (Rillig et al., 2001; Six et al., 2000).

In summary, strong associations among intrinsic soil properties, microbial parameters, biomasses, and radiocarbon abundances were seen across the five field sites in this study, which are representative of different forest–plantation conversion scenarios on a tropical island. Soil carbon dynamics changed in response to forest cover change, a process in which the soil microbes played an essential role.

4. Conclusions

To the best of our knowledge, this study is the first to provide vertical profiles of radiocarbon abundance for sites of forest–plantation conversion. We found an abnormal up-profile decrease, compared to NF, in radiocarbon abundance in the upper 30 cm soil layer at sites RU_35 yr and EU_50 yr. We attribute this phenomenon to faster SOM decomposition in the upper 30 cm soil layer long after forest–plantation conversion, which was supported by the lower SOC inventories. Both the SOC apparent radiocarbon age and SOC inventory at a eucalyptus plantation site where tillage had ceased 20 years prior implied potential recovery toward the NF condition. We also found a positive relationship between microbial biomass and both SOC inventory and radiocarbon abundance, which suggests enhanced microbe-mediated decomposition of SOM after forest–plantation conversion. This study demonstrated that radiocarbon is a powerful tool for assessing the changes in soil carbon dynamics caused by forest–plantation conversion.

Declaration of Competing Interest

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

Acknowledgements

Thanks a lot for the help of editors and reviewers in the improvement of the manuscript. Financial support was provided by the Local Innovative and Research Teams Project of Guangdong Pearl River Talents Program (2017BT01Z134) and the Department of Science and Technology of Guangdong Province (2016TQ03Z938).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geoderma.2020.114484.

References

Abood, S.A., Lee, J.S.H., Burivalova, Z., Garcia Ulloa, J., Lian, P.K., 2015. Relative contributions of the logging, fiber, oil palm, and mining industries to forest loss in indonesia. Conserv. Lett. 8, 58–67.
Apostel, C., Dippold, M., Glaser, B., Kuzyakov, Y., 2013. Biochemical pathways of amino acids in soil: assessment by position-specific labeling and 13 C-PLFA analysis. Soil Biol. Biochem. 67, 31–40.
Bardgett, R.D., Hobbs, P.J., Frostegård, Åsa, 1996. Changes in soil fungal:bacterial biomass ratios following reductions in the intensity of management of an upland grassland. Biol. Fertil. Soils 22, 261–264.
Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163.
Bligh, E.G., Dyer, W.J., 1959. A rapid method of total lipid extraction and purification. Can. J. Biochem. Physiol. 37, 911–917.
Ding, P., Shen, C.D., Wang, N., Yi, W.X., Ding, X.F., Fu, D.P., Liu, K.X., Zhou, L.P., 2010. Turnover rate of soil organic matter and origin of soil (CO_2)-C-14 in deep soil from a subtropical forest in Dinghushan biosphere reserve, south China. Radiocarbon 52,

1422–1434.
Don, A., Scholten, T., Schulze, E.D., 2009. Conversion of cropland into grassland: implications for soil organic-carbon stocks in two soils with different texture. J. Plant Nutr. Soil Sci. 172, 53–62.
Don, A., Schumacher, J., Freibauer, A., 2011. Impact of tropical land-use change on soil organic carbon stocks – a meta-analysis. Glob. Change Biol. 17, 1658–1670.
Donghai, L.I., Xiaobo, Y., Yunwu, D., Yuelie, L.I., 2006. Soil physical properties under effects of Eucalyptus understory vegetation and litter. Chin. J. Ecol. 25, 607–611.
FAO, 2018. The State of the World's Forests 2018 - Forest pathways to sustainable development. Rome. Licence: CC BY-NC-SA 3.0 IGO.
Figuerola, E.L.M., Guerrero, L.D., Tuerkowsky, D., Wall, L.G., Erijman, L., 2015. Crop monoculture rather than agriculture reduces the spatial turnover of soil bacterial communities at a regional scale. Environ. Microbiol. 17, 678–688.
Foley, J.A., Ruth, D., Asner, G.P., Carol, B., Gordon, B., Carpenter, S.R., Stuart, F.C., Coe, M.T., Daily, G.C., Gibbs, H.K., 2005. Global consequences of land use. Science 309, 570–574.
Frostegård, Å., Tunlid, A., Bååth, E., 2011. Use and misuse of PLFA measurements in soils. Soil Biol. Biochem. 43, 1621–1625.
Frostegard, A., Baath, E., 1996. The use of phospholipid fatty acid analysis to estimate bacterial and fungal biomass in soil. Biol. Fertil. Soils 22, 59–65.
Guillaume, T., Damris, M., Kuzyakov, Y., 2015. Losses of soil carbon by converting tropical forest to plantations: erosion and decomposition estimated by δ13C. Glob. Change Biol. 21, 3548–3560.
Hainan Provincial Bureau of Statistics, 2018. The 2018 statistical year book. http://stats.hainan.gov.cn/tjj/tjsu/ndsj/.
Harrison, K.G., Broecker, W.S., Bonani, G., 1993. The effect of changing land-use on soil radiocarbon. Science 262, 725–729.
He, Y., Trumbore, S.E., Tore, M.S., Harden, J.W., Vaughn, L.J.S., Allison, S.D., Randerson, J.T., 2016. Science 353, 1419–1424.
IPCC, 2007. Changes in atmospheric constituents and radiative forcing; Observations: Changes in Snow, Ice and Frozen Ground; Observations: Ocean Climate Change and Sea Level. Climate change 2007: the physical science basis, contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change, 129, 337, 385.
Islam, K.R., Weil, R.R., 2000. Land use effects on soil quality in a tropical forest ecosystem of Bangladesh. Agric. Ecosyst. Environ. 79, 9–16.
Joergensen, R.G., Wichern, F., 2008. Quantitative assessment of the fungal contribution to microbial tissue in soil. Soil Biol. Biochem. 40, 2977–2991.
Kaiser, C., Frank, A., Wild, B., Koranda, M., Richter, A., 2010. Negligible contribution from roots to soil-borne phospholipid fatty acid fungal biomarkers 18:2ω6,9 and 18:1ω9. Soil Biol. Biochem. 42, 1650–1652.
Kaur, A., Chaudhary, A., Kaur, A., Choudhary, R., Kaushik, R., 2005. Phospholipid fatty acid – a bioindicator of environment monitoring and assessment in soil ecosystem. Curr. Sci. 89, 1103–1112.
Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.
Laumonier, Y., Uryu, Y., Stüwe, M., Budiman, A., Setiabudi, B., Hadian, O., 2010. Eco-floristic sectors and deforestation threats in Sumatra: identifying new conservation area network priorities for ecosystem-based land use planning. Biodivers. Conserv. 19, 1153–1174.
Levin, I., Hesshaimer, V., 2000. Radiocarbon - a unique tracer of global carbon cycle dynamics. Radiocarbon 42, 69–80.
Liu, X., Wang, S., Jiang, Y., Sun, Y., Li, J., Zhang, G., 2017. Polychlorinated biphenyls and polybrominated diphenylethers in soils from planted forests and adjacent natural forests on a tropical island. Environ. Pollut. 227, 57.
Mahecha, M., Schmidtlein, S., 2010. Revealing biogeographical patterns by nonlinear ordinations and derived anisotropic spatial filters. Glob. Ecol. Biogeogr. 17, 284–296.
Margono, B.A., Potapov, P.V., Turubanova, S., Stolle, F., Hansen, M.C., 2014. Primary forest cover loss in Indonesia over 2000–2012. Nat. Clim. Change 4, 730–735.
Marin-Spiotta, E., Swanston, C.W., Torn, M.S., Silver, W.L., Burton, S.D., 2008. Chemical and mineral control of soil carbon turnover in abandoned tropical pastures. Geoderma 143, 49–62.
Mathieu, J.A., Hatté, C., Balesdent, J., Parent, É., 2015. Deep soil carbon dynamics are driven more by soil type than by climate: a worldwide meta-analysis of radiocarbon profiles. Glob. Change Biol. 21, 4278–4292.
Matson, P.A., Mcdowell, W.H., Townsend, A.R., Vitousek, P.M., 1999. The globalization of N deposition: ecosystem consequences in tropical environments. Biogeochemistry 46, 67–83.
McGrath, D.A., Smith, C.K., Gholz, H.L., Oliveira, F.D.A., 2001. Effects of land-use change on soil nutrient dynamics in Amazônia. Ecosystems 4, 625–645.
Noordwijk, M.V., Carlos, C., Paul, L.W., Kusumo, N., Martial, B., 1997. Soil carbon dynamics in the humid tropical forest zone. Geoderma 79, 187–225.
Olsson, P.A., 1999. Signature fatty acids provide tools for determination of the distribution and interactions of mycorrhizal fungi in soil. FEMS Microbiol. Ecol. 29, 303–310.
Olsson, P.A., Bååth, E., Jakobsen, I., Söderström, B., 1995. The use of phospholipid and neutral lipid fatty acids to estimate biomass of arbuscular mycorrhizal fungi in soil. Mycol. Res. 99, 623–629.
Pabst, H., Kühnel, A., Kuzyakov, Y., 2013. Effect of land-use and elevation on microbial biomass and water extractable carbon in soils of Mt. Kilimanjaro ecosystems. Appl. Soil Ecol. 67, 10–19.
Ratledge, C., 2008. Microbial Lipids.
Reimer, P.J., Brown, T.A., Reimer, R.W., 2004. Discussion: reporting and calibration of post-bomb C-14 data. Radiocarbon 46, 1299–1304.
Rillig, M.C., Wright, S.F., Nichols, K.A., Schmidt, W.F., Torn, M.S., 2001. Large contribution of arbuscular mycorrhizal fungi to soil carbon pools in tropical forest soils. Plant Soil 233, 167–177.
Rodrigues, J.L.M., et al., 2013. Conversion of the Amazon rainforest to agriculture results in biotic homogenization of soil bacterial communities. Proc. Natl. Acad. Sci. U. S. A. 110, 988–993.
Ross, S.M., 1993. Organic matter in tropical soils: current conditions, concerns and prospects for conservation. Prog. Phys. Geogr. 17, 265–305.
Ruess, L., Chamberlain, P.M., 2010. The fat that matters: Soil food web analysis using fatty acids and their carbon stable isotope signature. Soil Biol. Biochem. 42, 1898–1910.
Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kögel-Knabner, I., Lehmann, J., Manning, D.A.C., 2011. Persistence of soil organic matter as an ecosystem property. Nature 478, 49–56.
Sidle, R.C., Ziegler, A.D., Negishi, J.N., Nik, A.R., Siew, R., Turkelboom, F., 2006. Erosion processes in steep terrain—truths, myths, and uncertainties related to forest management in Southeast Asia. For. Ecol. Manage. 224, 199–225.
Six, J., Paustian, K., Elliott, E.T., Combrink, C., 2000. Soil structure and organic matter: I. Distribution of aggregate-size classes and aggregate-associated carbon. Soil Sci. Soc. Am. J. 64, 681–689.
Smith, A.P., Marín-Spiotta, E., Graaff, M.A.D., Balser, T.C., 2014. Microbial community structure varies across soil organic matter aggregate pools during tropical land cover change. Soil Biol. Biochem. 77, 292–303.
Stuiver, M., Polach, H.A., 2006. Discussion; reporting of C-14 data. J. Biol. Chem. 287, 4726–4739.
Torn, M.S., Swanston, W.C., Castanha, C., Trumbore, S., 2009. 2.3 Soil Bulk Density and Carbon Inventory Estimation, Storage and Turnover of Natural Organic Matter in Soil. John Wiley & Sons Inc, pp. 7.
Trumbore, S.E., 2000. Age of soil organic matter and soil respiration: radiocarbon constraints on belowground C dynamics. Ecol. Appl. 10, 399–411.
Trumbore, S.E., 2009. Radiocarbon and soil carbon dynamics. Annu. Rev. Earth Planet. Sci. 37, 47–66.
Trumbore, S.E., 1993. Comparison of carbon dynamics in tropical and temperate soils using radiocarbon measurements. Global Biogeochem. Cycles 7, 275–290.
Trumbore, S.E., Bubier, J.L., Harden, J.W., Crill, P.M., 1999. Carbon cycling in boreal wetlands: a comparison of three approaches. J. Geophys. Res. Atmospheres 104, 27673–27682.
Trumbore, S.E., Davidson, E.A., Nepstad, D.C., Martinelli, L.A., 1995. Belowground cycling of carbon in forests and pastures of Eastern Amazonia. Global Biogeochem. Cycles 9, 515–528.
Uexküll, H.R.V., Mutert, E., 1995. Global extent, development and economic impact of acid soils. Plant Soil 171, 1–15.
Van der Kamp, J., Yassir, I., Buurman, P., 2009. Soil carbon changes upon secondary succession in Imperata grasslands (East Kalimantan, Indonesia). Geoderma 149, 76–83.
Veresoglou, S.D., Barry, T., George, M., Mamolos, A.P., Veresoglou, D.S., 2012. Soil fertilization leads to a decline in between-samples variability of microbial community $\delta^{13}C$ profiles in a grassland fertilization experiment. PLoS One 7, e44203.
Vestal, J.R., White, D.C., 1989. Lipid analysis in microbial ecology—quantitative approaches to the study of microbial communities. Bioscience 39, 535–541.
Wang, J.C., Ren, C.Q., Cheng, H.T., Zou, Y.K., Bughio, M.A., Li, Q.F., 2017. Conversion of rainforest into agroforestry and monoculture plantation in China: consequences for soil phosphorus forms and microbial community. Sci. Total Environ. 595, 769–778.
Wang, Y., Amundson, R., Trumbore, S., 1999. The impact of land use change on C turnover in soils. Global Biogeochem. Cycles 13, 47–57.
Wilcove, D.S., Lian, P.K., 2010. Addressing the threats to biodiversity from oil-palm agriculture. Biodivers. Conserv. 19, 999–1007.
Wu, Z.X., Xie, G.S., Tao, Z.L., Zhou, Z.D., Wang, X., 2009. Characteristics of soil carbon and tatal nitrogen contents of rubber plantations at different age stages in Danzhou, Hainan island. Ecol. Environ. Sci. 18, 1484–1491.
Xiankai, L., Qinggong, M., Jiangming, M., Gilliam, F.S., Guoyi, Z., Yiqi, L., Wei, Z., Juan, H., 2015. Divergent responses of soil buffering capacity to long-term N deposition in three typical tropical forests with different land-use history. Environ. Sci. Technol. 49, 4072–4080.
Zelles, L., 1997. Phospholipid fatty acid profiles in selected members of soil microbial communities. Chemosphere 35, 275–294.
Zelles, L., 1999. Fatty acid patterns of phospholipids and lipopolysaccharides in the characterisation of microbial communities in soil: a review. Biol. Fertil. Soils 29, 111–129.
Zhai, D.L., Cannon, C.H., Slik, J.W.F., Zhang, C.P., Dai, Z.C., 2012. Rubber and pulp plantations represent a double threat to Hainan's natural tropical forests. J. Environ. Manage. 96, 64–73.