Journal of Environmental Management 281 (2021) 111959

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: http://www.elsevier.com/locate/jenvman

Research article

Jungle rubber facilitates the restoration of degraded soil of an existing rubber plantation

Huanhuan Zeng [a,b,c], Junen Wu [a,b,**], Xiai Zhu [a,b], Ashutosh Kumar Singh [a,b], Chunfeng Chen [a,b], Wenjie Liu [a,b,*]

[a] CAS Key Laboratory of Tropical Forest Ecology, Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences, Menglun, Yunnan, 666303, China
[b] Center of Plant Ecology, Core Botanical Gardens, Chinese Academy of Sciences, Menglun, Yunnan, 666303, China
[c] University of Chinese Academy of Sciences, Beijing, 100049, China

ARTICLE INFO

Keywords:
Land-use change
Rubber plantation
Soil physical properties
Soil nutrients

ABSTRACT

Conversion of forest to rubber plantation is one of the most common land-use change in the humid tropical region. It is one of the fastest expanding farms that lead to various socioenvironmental issues. We investigated the effect of this land-use change on soil physico-chemical properties by surveying different succession stage rubber plantations, including monoculture and a mixture derived by mixing jungle rubber and a reference tropical rainforest. We also assessed the impact on stoichiometric ratios and allocation relationships of soil carbon (C), nitrogen (N), and phosphorus (P). Our results demonstrated that conversion of tropical rainforest to rubber monoculture resulted in serious soil degradation, with a lower level of water content, water holding capacities, total porosity, pH, and soil nutrients, and a higher level of soil bulk density. However, after transforming a rubber monoculture into a jungle rubber, the concentrations of soil total C, N, P, Ca, and Mg significantly increased, by 28%, 24%, 23%, 17%, and 39%, respectively. Meanwhile, soil salinity declined by 15%. Jungle rubber also exerted some desirable effects on soil physical properties, such as decreased soil bulk density, increased field capacity and non-porosity by 6%, 2%, and 33%, respectively. Like other tropical regions, soils in the present study areas are mainly under P limitation, but jungle rubber increased soil P turnover and thereby increases P availability. In conclusion, jungle rubber correcting the soil degradation resulted from rubber plantation on tropical forest soil. Given the improvements in soil quality, constructing multiple-strata and multi-species rubber agroforestry (e.g., jungle rubber) can be a promising approach to facilitate the restoration of the existing monoculture rubber plantations.

1. Introduction

Land-use change and associated anthropogenic management directly affect ecosystem services, biological diversity, nutrient cycles, and biomass production (Vitousek et al., 1997b; Lu et al., 2019). In the past century, natural forest cover has decreased dramatically in southwestern China. Simultaneously, a large part of natural forest was transformed into agricultural cropland (e.g., plantations of pitaya, banana, pineapple, rubber) (Cao et al., 2017). In particular, the cultivation areas of rubber monoculture increased by nearly sixfold from 1987 to 2018 (Fox et al., 2014; Xiao et al., 2019). These large-scale land-use transformations have caused many eco-environmental problems, such as biodiversity and biomass loss (Yi et al., 2014), excessive soil erosion and acidification (Liu et al., 2015; Zhu et al., 2018), and decline in soil fertility and carbon sequestration (Liu et al., 2019), which have attracted lots of attention from all over the world.

As an important component of the terrestrial ecosystem, the soil is the primary determinant of functional recovery and ecosystem maintenance which closely associated with sustainable develop of regional eco-environment (Bronick and Lal, 2005). Plants take up water and nutrients from soil via roots and return carbon and various nutrients to the soil through their residues, such as deciduous leaf, shoot, root, and root

* Corresponding author. Key Laboratory of Tropical Forest Ecology, Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences, Menglun, Mengla, Yunnan, 666303, China. Tel.: 86-691-8715070
** Corresponding author. CAS Key Laboratory of Tropical Forest Ecology, Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences, Menglun, Yunnan, 666303, China.
E-mail addresses: wujunen@xtbg.ac.cn (J. Wu), lwj@xtbg.org.cn (W. Liu).

https://doi.org/10.1016/j.jenvman.2021.111959
Received 21 September 2020; Received in revised form 25 December 2020; Accepted 3 January 2021
Available online 9 January 2021

exudates (Marschner, 2011). The quality of soil was determined by both their physical and chemical properties. Among soil chemical properties, soil nutrients are the main factors affect biomass allocation for both aboveground and belowground (John et al., 2007). The concentration of essential nutrient elements in the soil is a useful tool to predict the status of agroforestry production efficiency (Li et al., 2019; Schilling et al., 2009). For example, nitrogen (N) and phosphorus (P) are the building block of the primary productivity of an ecosystem (Elser et al., 2007), potassium (K) regulate osmosis in plant cells, calcium (Ca) is the primary structural material of cell walls, and magnesium (Mg) participates in the binding of the chlorophyll molecule (Marschner, 2011). Besides, soil physical properties regulate air content, the movement of water and dissolved chemicals within the soil. Healthy soil is characterized by lower bulk density and high soil pore spaces or porosity. The soil pore size and abundance determine the soil's ability to store root zone water and air necessary for plant growth; the higher the soil porosity, the better air and water transfer in soil (Katsvairo et al., 2007). The soil water is an important factor in plant growth; the higher the water-related parameters, the less water limitation for plant growth (Chen, 2005; Melesse et al., 2019). The degradation of soil quality in Xishuangbanna could restrict vegetation growth and threaten the security of natural ecosystem and environment. Thereby, soil quality assessment has been a hot topic of research in the last few decades. For instance, Chen et al. (2019) demonstrated that the conversion of natural forest to rubber monoculture resulted in decline in soil physical properties (e.g., soil structure, bulk density, porosity, and water holding capacity), and soil chemical properties (e.g., soil acidity, nutrients, and salt content).

It's worth to mention that most rubber plantations in southwest China were established in 1990s, they could not produce natural latex with high quantity and quality in recent years as a predictable consequence after 30 years tapping (Priyadarshan, 2011). However, little work has so far focused on the existing aged rubber trees management, although aged rubber trees also exhibited benefits to environment, such as increasing soil microbial and macrofaunal biomass (Peerawat et al., 2018), providing shelter and understory plants for animals, and improving the production of rubber seeds for oil industry (Zhu et al., 2014). Moreover, with the price drop of natural latex in the global market, current rubber plantation has been replaced partly by the other cash crops (e.g., pineapple, banana, grapefruit) (Langenberger et al., 2017; Zhang et al., 2019). Planting these crops generally involves cutting down the existing rubber trees and applying a large amount of fertilizer, resulting in even more severe eco-environmental damages (Langenberger et al., 2017). In this scenario, protecting these aged rubber trees and mixing other perennial plants (i.e., agroforestry systems) could be a potential way to meet environmental and economic demands. Regarding this, jungle rubber could be a wise option owing to their proposed compatibility with the commercial rubber trees (Warren-Thomas et al., 2020). However, only few studies concerning jungle rubber are available, especially regarding its long-term ecological benefits.

To explore the influence of jungle rubber on existing soils of rubber plantation, we conducted a soil survey in a rubber monoculture (RM) and a jungle rubber (JR). We were also included a tropical rainforest (TRF) site in the survey considering as a primary reference site. To assess the impact on soil physico-chemical properties, we measured soil physical properties (e.g., water content, bulk density, water holding capacities, three phase ratio and soil porosity) and nutrient concentrations (including total C, N, P, K, Ca, and Mg), pH, and electrical conductivity of soil samples collected from different depths (0–5, 5–15, 15–30, 30–50, 50–75, 75–105 cm) from these sites. To further understand the relationships between the soil nutrients and related nutrient limitation (mainly for C, N and P), the stoichiometric ratios (e.g., C/N, N/P, and C/P) were calculated in this study (Cleveland and Liptzin, 2007). And allometric scaling of soil C, N and P were examined using standardized major axis (SMA) regression, it's widely applied for predicting one variable from the other and fits a slope and an intercept for the model (Warton and Weber, 2002).

We hypothesize that the conversion of tropical rainforest to rubber plantation would alter soil physico-chemical properties in a way that leads to soil degradation, and mixing of jungle rubber with existing rubber would improve soil physico-chemical properties. Therefore, our objective is to assess the impact of rubber monoculture, rubber mixture (jungle rubber), and tropical rainforest effect on (1) soil physical properties, (2) soil chemical properties, and (3) stoichiometric ratios and allometric relationships of soil C, N, P. We hope this information would provide applied suggestions to establish sustainable managements of rubber plantation in Xishuangbanna.

2. Materials and methods

2.1. Study sites

Xishuangbanna prefecture is one of the largest rubber plantation area in China. Our study was carried out in the Xishuangbanna Tropical Botanical Garden (XTBG, 21°55′39″N, 101°15′55″E) in the southwest of Yunnan Province, China (Fig. 1). The present region is located in East Asian monsoon region, which receives tropical southern monsoons from the Indian Ocean (May to October) and continental dry air masses of subtropical origin for the remaining period (November to April). Hence, this area experiences a hot and humid rainy season from May to October and a cool and foggy dry season from November to April (Zhang, 1988). Climate data recorded by Xishuangbanna Station for Tropical Rainforest Ecosystem Studies showed that the average annual air temperature was 21.6 °C, the mean relative humidity was 85% and the mean annual precipitation was 1487 mm from 1978 to 2018. Approximately 80–90% of the precipitation befallen in the rainy season, and only 10–20% of the precipitation befallen during the dry season. During the sampling period (2017–2018), the monthly mean air temperature was 22.6 °C, the relative humidity was 86%, and the total precipitation was 1626 mm. The monthly maximum and minimum air temperature mean values were 33.6 °C (May 2017) and 11.2 °C (February 2018), respectively. The study area is located at the junction of the Indian and Burmese plates of Gondwana and the Eurasian plate of Laurasia and experienced the rising/descending of the Himalayas, with the topography of the area comprising hills, slopes, and valleys (Zhu et al., 2008). The soil is developed from alluvial deposits derived from sandstone and features an ochric A horizon and a cambic B horizon with ferralic properties (Vogel et al., 1995; Liu et al., 2015).

These experimental sites belong to a long-term observation station established for ecological research by Xishuangbanna Station for Tropical Rainforest Ecosystem Studies. The study sites map and their community illustration is given in Fig. 1. The topographical characteristics, such as elevation and slope of these sites, are comparable. Rubber trees (clone PB86) at RM site were planted in 1990. Farmers feed N–P–K synthetic fertilizers in middle October at a rate of ~200 kg ha^{-1} per year. Rubber tapping activities are carried out at one-day intervals, from the last week of April to mid-November. The JR community was established in an aged rubber plantation (planted in 1962) and experienced natural vegetation succession for ten years. TRF is natural vegetation in Xishuangbanna commonly occurs in the humid valleys, low-altitude lands and low hills. The community structure of TRF is more complex compared to RM and JR. The information about plant species composition of all study sites is shown in Supplementary Table A1 and A2. There was no fertilizers application in JR and TRF. We conducted soil sampling for nutrient measurements in November 2017 (early dry season, fruit ripening of rubber trees), January (middle of the dry season, dormant stage of rubber trees), February (middle of the dry season, rubber trees with no leaves), and March 2018 (late dry season, leaf expansion of rubber trees) from three study sites.

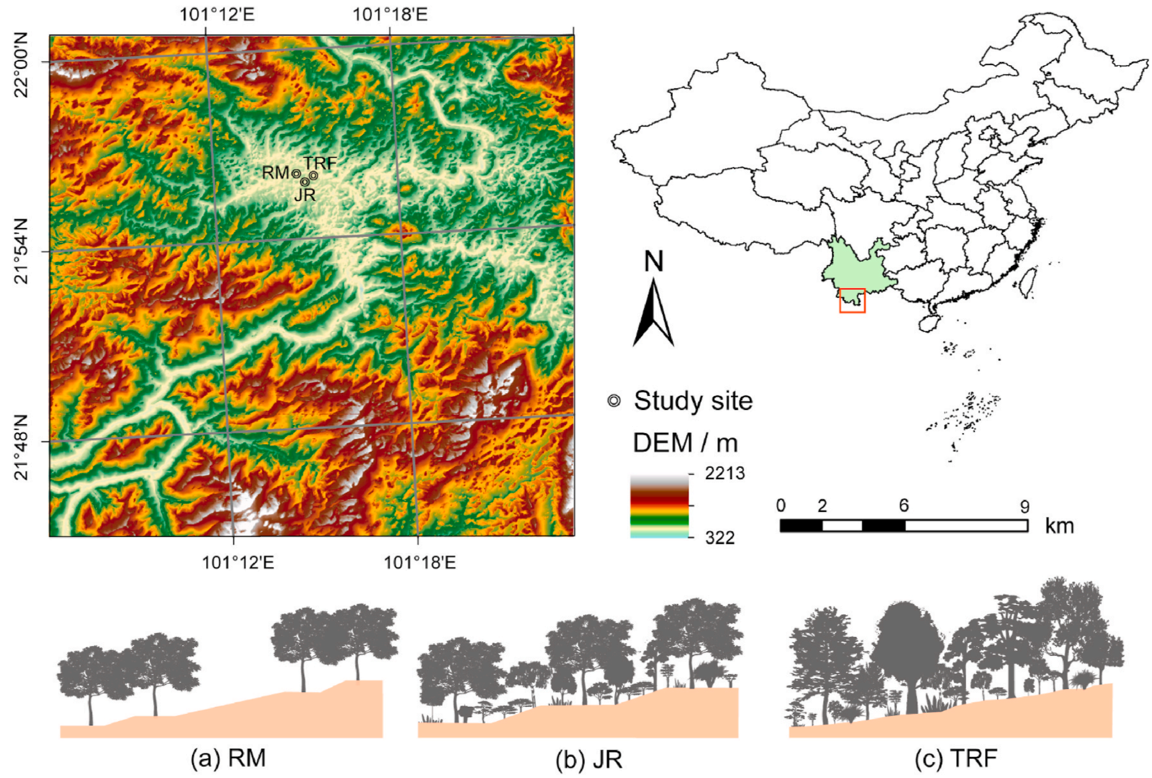

Fig. 1. Location, land uses and land cover map of different study sites. Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest.

2.2. Soil physico-chemical properties measurement

We carried out the measurement of soil physical properties for one single time because they would generally remain stable for a short period (Rezaei and Gilkes, 2005). The bulk soil was collected via pre-weighed (W_{ecc}) metal cutting cylinders (volume: 200 cm^3) from each study site. Soil sampling was carried out during a non-rainy period for at least one week. The soil was sampled at six unequal soil profiles (i.e., 0–5, 5–15, 15–30, 30–50, 50–75, 75–105 cm) in four randomly chosen plots. In the laboratory, the cutting cylinders containing bulk soil were weighed W_{wscc} (g). The samples were then placed in a flat bowl filled with water to the top border of the cutting cylinders. After immersion for 24 h, the cutting cylinders were removed from the water for immediate weighed (W_{swc}, g). Then, the cutting cylinders containing saturated soil were put on a dry sand layer, and the cutting cylinders were weighed after 2 h and 5 d, respectively (W_{2hcc}, W_{5dcc}, g). Finally, the cutting cylinders were weighed after drying in an oven for at least for 48 h at 105 °C to a constant weight (W_{dscc}, g). The soil water content (%), bulk density (g cm^{-3}), three-phase ratio, soil water holding capacities (%) (e.g., saturated water capacity, capillary water capacity, and field capacity), and soil porosity (%) (e.g., non-capillary porosity, capillary capacity and total porosity) were determined by the following formulas (Chen, 2005):

$$Soil\ \ Water\ \ Content(\%) = \frac{W_{wscc}(g) - W_{dscc}(g)}{W_{dscc}(g) - W_{ecc}(g)} \times 100\% \tag{1}$$

$$Bulk\ \ Density\ \ (g\ \ cm^{-3}) = \frac{W_{wscc}(g) - W_{dscc}(g)}{200(cm^3)} \tag{2}$$

$$Saturated\ \ water\ \ capacity(\%) = \frac{W_{swc}(g) - W_{dscc}(g)}{W_{dscc}(g) - W_{ecc}(g)} \times 100\% \tag{3}$$

$$Capillary\ \ water\ \ capacity(\%) = \frac{W_{2hcc}(g) - W_{dscc}(g)}{W_{dscc}(g) - W_{ecc}(g)} \times 100\% \tag{4}$$

$$Field\ \ Capacity(\%) = \frac{W_{5dcc}(g) - W_{dscc}(g)}{W_{dscc}(g) - W_{ecc}(g)} \times 100\% \tag{5}$$

$$Capillary\ \ Porosity(\%) = Capillary\ \ Water\ \ Capasity(\%) \times Bulk\ \ Density(g\ \ cm^{-3}) \times 100\% \tag{6}$$

$$Non-capillary\ \ Porosity(\%) = \frac{Saturated\ \ Water\ \ Capacity(\%) - Capillary\ \ Water\ \ Capacity(\%)}{Bulk\ \ Density(g\ \ cm^{-3})} \times 100\% \tag{7}$$

$$Total\ \ Porosity(\%) = Capillary\ \ Porosity(\%) + Non-Capillary\ \ Porosity(\%) \tag{8}$$

$$Solid\ \ Phase(\%) = 1 - Total\ \ Porosity(\%) \tag{9}$$

$$Gas\ \ Phase(\%) = Total\ \ Porosity(\%) - Soil\ \ Water\ \ Content(\%) \tag{10}$$

$$Three-Phase\ \ Ratio = Solid\ \ Phase : Soil\ \ Water\ \ Content : Gas\ \ Phase \tag{11}$$

For the soil chemical properties measurement, we collected soil profile samples from four random plots via a manual soil auger (inner diameter: 4 cm). Further, we homogenize each soil sample by manually removing roots, earthworms, detritus, and large plant residues. Soil samples were air dried, grind to pass through a 0.6 mm mesh sieve, and stored in air-tight zip-loc bags for further analysis. Soil pH was determined from a 1:5 mixture of soil and deionized water using a digital pH meter (FE28, Mettler Toledo; Switzerland). A part of the 1:5 soil-water mixture was filtered to determine the soil EC (μs cm^{-1}) from a clear solution using a conductivity meter (DDS–307 A, INESA; China). For soil nutrients measurement, soil samples were oven-dried at 65 °C until constant weight, pass through a 0.15 mm mesh sieve, and stored separately in zip-lock bags for further measurement.

Soil total C and N (g kg^{-1}) were measured using an elemental analyzer (vario MAX CN; Elementar Analysen systeme GmbH; Hanau, Germany). Soil total P, K, Ca, and Mg concentrations (g kg^{-1}) were measured by inductively coupled plasma-atomic emission spectrometry (iCAP6300, Thermo Fisher Scientific; Waltham, MA, USA) after the samples were pre-digested with HNO_3–HF–$HClO_4$ in the graphite furnace (DEENA II, Thomas Cain; USA). The digested solution was transferred into a 50 ml volumetric flask, brought to volume by ultrapure water, and stored in a 4 °C refrigerator for at least 12 h. All elemental analyses were conducted in the central laboratory, XTBG.

2.3. Statistical analyses

All the data are expressed with mean value ± standard deviation. The data were tested for normal distribution before analysis. Where needed, the data were log-transformation to meet normal distribution. One-way analysis of variance (ANOVA, $p < 0.05$) coupled with the least significant difference (LSD) test was carried out to identify the significant differences in the soil physico-chemical properties among study sites at each sampling depth. The effects of time, depth, and site on soil physico-chemical properties at each study site were analyzed using general linear models (GLMs), and significant effects were tested at $p < 0.05$, $p < 0.01$, and $p < 0.001$, respectively. All the above analyses were performed in SPSS 25 (IBM Inc. Chicago, USA).

The stoichiometric ratios of chemical elements (mainly for C, N and P) were calculated to further understand the relationships between the soil nutrients and related nutrient limitation. In addition, the standardized major axis regression (SMA) was used for fitting bivariate lines to examine the covariation in soil C, N and P. For allometric relationships, the observed variables (y, x) are assumed related to the equation: $Y = \alpha X^b$. This equation becomes linear after log-transformation:

$$\log Y = \log \alpha + b \log X \tag{12}$$

where coefficient log α is the intercept and exponent *b* is the regression slope of the line, the X and Y are the different soil nutrients (e.g., soil total C and N). When the slope *b* did not significantly differ from 1, the relationship of X to Y was described as isometric; otherwise, regarded as allometric. When $b > 1$, it is assumed that the linear variation in Y is faster than X. Whereas $b < 1$ indicates the linear variation in X is faster than Y. Allometric relationships among soil total C, N, and P concentrations were calculated using the package 'smatr' in R 4.0 (Warton and Weber, 2002).

3. Results

3.1. Changes of soil physical properties

Soil physical properties varied significantly among different study site in all soil layers ($p < 0.05$; Table A1, Table 1). The mean soil water content was the highest in TRF, followed by RM and JR. This difference was most obvious for 0–5 cm soil layer, which increased by 60% and 30% for TRF and JR, respectively (Fig. 2a). In the 0–5 cm soil layer, the decreasing order of soil BD was JR > RM > TRF; while for the deeper soil layers, the decreasing order was RM > TRF > JR. The mean soil bulk density was lowest in the JR, followed by TRF, and highest in the RM ($p < 0.05$; Table A1, Fig. 2b). In the case of soil porosity, JR increased soil non-capillary porosity compared to RM by 33%, and TRF has trebled soil non-capillary porosity compared to RM ($p < 0.05$). The minimum total soil porosity was observed in JR, followed by RM and TRF ($p < 0.05$). For profile variations, we found soil total porosity for the 0–30 cm was lower in JR, but higher for the deeper soil layers, compared to those observed in RM (Fig. 2d–f). JR also improved the soil water capacities (e.g., saturated water capacity, field capacity) compared to RM ($p < 0.05$; Table A1), especially for soil layers deeper than 5 cm. However, the highest values of soil water capacities, especially for top soil layer (0–5 cm), were found in TRF compared to RM and JR ($p < 0.05$; Table A1).

Table 1
Results of a general linear model testing the effects of study sites (*d.f.* = 2) and soil depths (*d.f.* = 5) on soil physical properties.

Tested effects	Sites		Depth		Sites * Depth	
	F values	*p* values	*F* values	*p* values	*F* values	*p* values
Soil water content (%)	11.32	<0.001	5.42	<0.001	2.51	0.01
Three-phase ratio	8.89	<0.001	11.16	<0.001	3.88	<0.001
Bulk density (g cm^{-1})	12.09	<0.001	30.47	<0.001	7.20	<0.001
Saturated water capacity (%)	0.43	0.65	36.81	<0.001	9.63	<0.001
Capillary water capacity (%)	2.07	0.14	18.69	<0.001	7.76	<0.001
Field capacity (%)	46.12	<0.001	0.64	0.67	4.50	<0.001
Non-capillary porosity (%)	7.82	<0.001	19.94	<0.001	4.90	<0.001
Capillary porosity (%)	26.08	<0.001	30.03	<0.001	4.13	<0.001
Total porosity (%)	16.42	<0.001	20.92	<0.001	1.71	0.10

3.2. Soil pH and EC

Soil pH showed significant differences with land-use change, and increased in the order of JR < RM < TRF ($p < 0.05$; Table 2, Fig. 3a). Soil pH was also influenced by soil depth ($p < 0.05$; Table 2), and it precipitously decreased with increasing the soil depth. The soil EC also decreased with increasing soil depth like pH, irrespective of treatments (Table A3). However, the EC values among the treatments were highest in RM followed by JR and TRF. The EC values did not differ between JR and TRF ($p > 0.05$; Fig. 3b). Both the soil pH and EC decreased with increasing soil depth. The pH and EC values remained stable with the sampling time, except in RM where EC values differ with sampling time.

3.3. Variations in soil nutrient concentrations

The concentration of soil nutrients varied significantly with land-use change ($p < 0.05$; Table 2, Fig. 4). The concentrations of soil total C in JR and TRF significantly higher by 39% and 38%, respectively, relative to those in RM ($p < 0.05$; Fig. 4a and b). The JR and TRF also exhibited higher total N values by 32% and 38%, respectively, compared to RM ($p < 0.05$). The soil total C and N had the same variation trend with sampling time. Both the parameters did not differ with sampling time for any treatments except JR where soil total C and N values were higher in February than the other three months (Fig. 4 a). The order of variation of soil total C and N with sampling time was the same for JR and RM;

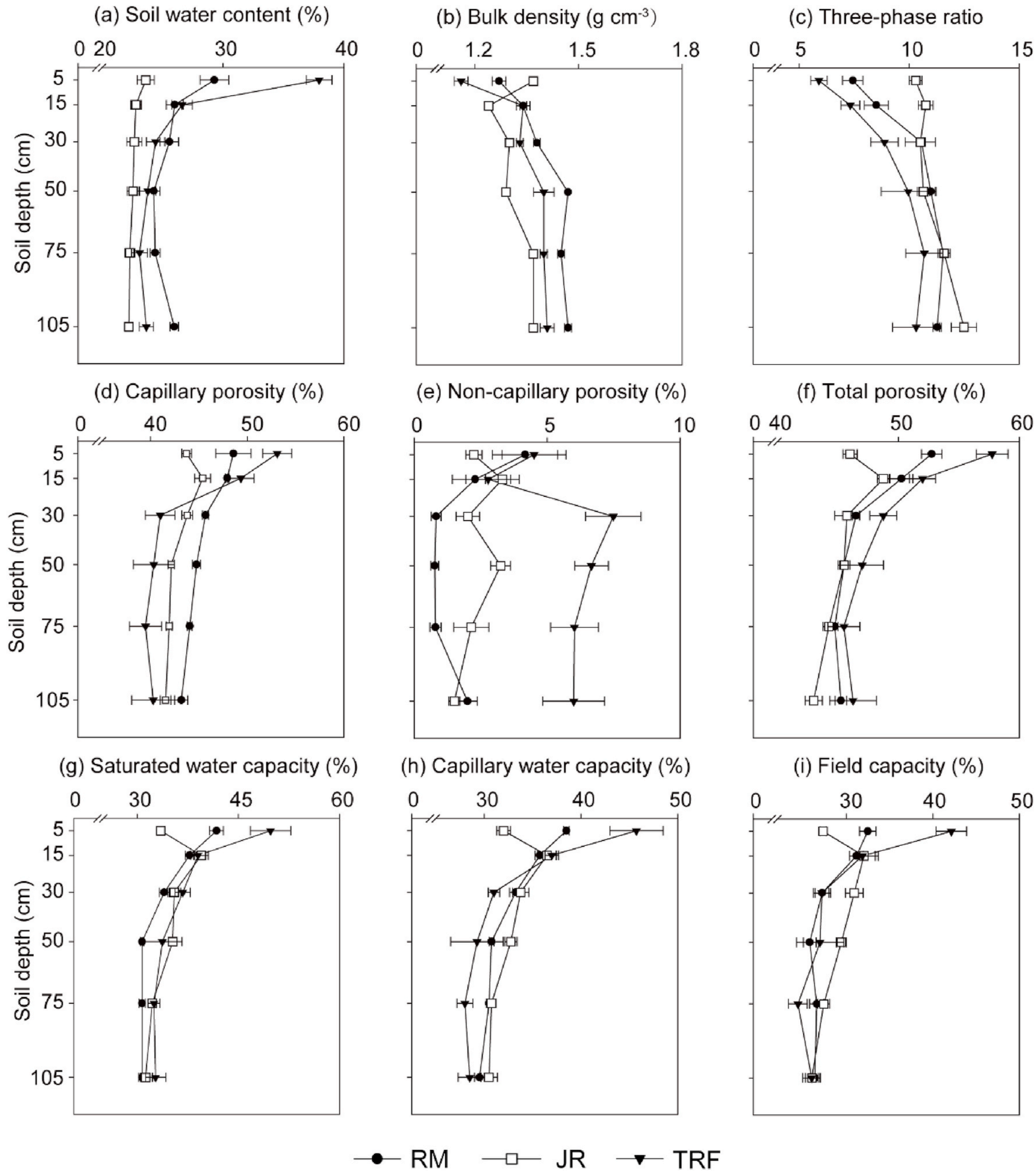

Fig. 2. Profile variations in soil physical properties of different study sites. (a) Soil water content; (b) soil bulk density, (c) three phase ratio, (d) capillary porosity, (e) non-capillary porosity, (f) total porosity; (g) saturated water capacity, (h) capillary water capacity, (i) field capacity. Data are expressed as the mean ± SD (n = 4). Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest.

February > January > November > March. However, TRF had a different decreasing trend than rubber treatments (JR and RM), with the lowest values appeared in February (Fig. 4 a, b). The highest total P concentrations appeared in February for JR and TRF, while the highest value in RM was in March. Except for these peak months, the remaining months of sampling showed no differences for P in all treatments. The soil total Ca showed the same temporal high peaks among the treatments observed for total P (February for JR and TRF, and March for RM), with a more evident variation among other sampling months ($p < 0.05$; Fig. 4c, e).

The soil total Mg concentration varied with the land-use change, with the highest values observed in TRF, followed by JR and RM ($p < 0.001$; Fig. 4f). The soil total Mg concentration was highest in November between sampling months in all land uses, and the lowest was during February in JR and TRF and during March in RM. The rest of the period of sampling was mainly showing similar Mg values in all land uses. The soil total K concentration also varied with land-use change, with the higher values were found in RM followed by TRF and JR ($p < 0.001$; Fig. 4d). Among the sampling months, the total K values were highest during November in RM and TRF, and during February in JR. However, the lowest total K was found during February in RM and TRF, and during January in JR. The rest of the periods of sampling were showing mainly a similar range in all land uses.

The soil nutrient concentrations were also significantly affected by soil depth, except for total Ca concentration ($p < 0.05$; Table 2). The total C, N, and P concentration decreased with increasing soil depth ($p < 0.05$; Table A4). The greatest extent of variation appeared in 0–5 and 5–15 cm soil layers, with the total C, N, P, concentrations decreased by

Table 2
Results of a general linear model testing the effects of soil depth and sampling time on soil chemical properties and its stoichiometric ratios (C/N, N/P, C/P) of each study site.

Tested effects	d. f.	pH	EC (μS cm^{-1})	Total C (g kg^{-1})	Total N (g kg^{-1})	Total P (g kg^{-1})	Total K (g kg^{-1})	Total Ca (g kg^{-1})	Total Mg (g kg^{-1})	C/N	N/P	C/P
RM												
Depth	5	28.06***	2.19	161.05***	109.17***	32.70***	23.11***	2.34	5.34***	76.44***	2.88*	36.62***
Time	3	19.19***	47.98***	5.57**	4.67**	66.54***	222.93***	23.82***	516.14***	2.04	114.92***	116.65***
Depth * Time	15	0.67	0.80	2.32**	1.56	2.41**	2.08*	0.96	3.71***	1.54	2.54**	3.42***
JR												
Depth	5	4.48**	39.40***	119.99***	98.59***	19.41***	10.75***	0.27	2.30	23.74***	63.86***	106.65***
Time	3	482.81***	5.16*	34.03***	35.02***	135.16***	257.01***	126.10**	440.13***	2.81	168.45***	156.47***
Depth * Time	15	3.69***	5.20***	21.41***	13.40***	16.71***	1.19	0.97	4.94***	2.26*	7.43***	6.99***
TRF												
Depth	5	18.42***	79.37***	196.50***	168.15***	2.19	12.07***	22.27***	2.26	22.00***	64.21***	93.39***
Time	3	217.76***	2.49	9.68***	11.51***	3.29*	404.05***	58.71**	41.43***	4.36**	47.95***	33.61***
Depth * Time	15	3.35***	4.79***	8.55***	6.51***	2.72**	0.99	5.14***	0.56	1.32	7.01***	5.98***

Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest. EC: electrical conductivity.
*, **, and *** represent significant difference among the tested effects at $p < 0.05$, 0.01 and 0.001 level, respectively.

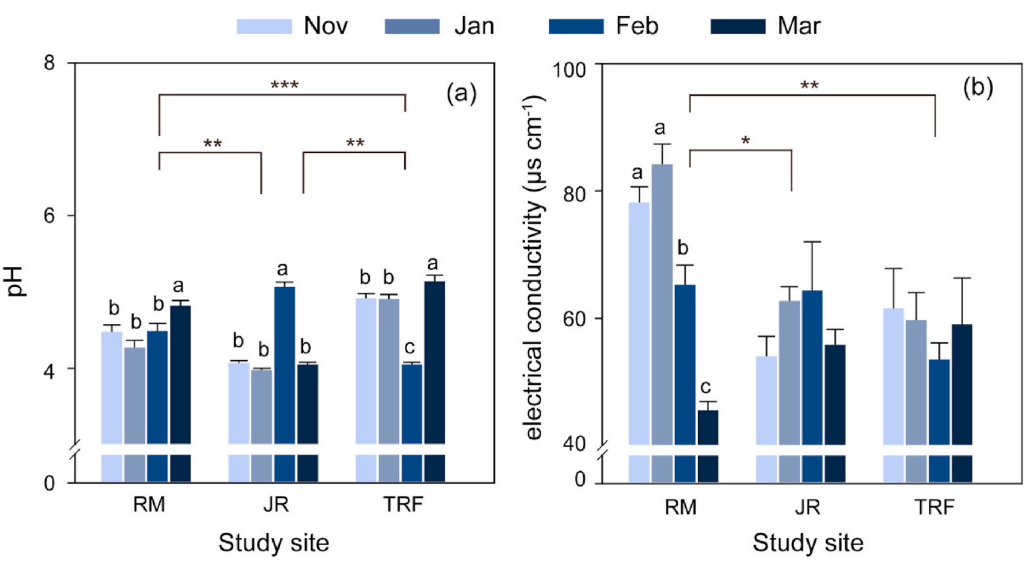

Fig. 3. Seasonal variations in soil pH and electrical conductivity of different study sites. Data are expressed as the mean ± SD (n = 4). Bars with different lowercase letters indicate significant differences among sampling time ($p < 0.05$); if there were significant differences among study sites, the asterisk are marked in the bars (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$), the tested method was one-way analysis of variance (ANOVA) followed by the least significant difference (LSD) test. Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest.

31–57%, 23–47%, and 17–47%, respectively. On the other hand, the values of total K and Mg increased with increasing soil depth (Table A4).

3.4. Relationships among soil C–N–P in different land uses

Land-use change also greatly influenced soil C–N–P stoichiometric ratios, on average, C/N and C/P were highest in JR, and N/P were highest in TRF. Soil N/P and C/P ratios in RM were always significantly lower than those in JR and TRF, and the soil C/N ratio decreased in the order of JR > RM > TRF ($p < 0.05$; Fig. 5). The decreasing order of soil N/P and C/P ratios among the sampling months in RM was February > November > January > March, and in JR and TRF was March > January > November > February (Fig. 5b and c). The C/N ratio in TRF showed an increasing trend over sampling time, while there was no significant difference in RM and JR among different sampling times ($p > 0.05$; Table 2). The C/N ratio in TRF significantly varied with sampling time, with a decreasing order was March ≥ February > January > November. The C/N ratio did not differ between sampling time in the case of RM and JR ($p > 0.05$; Fig. 5a). The soil C–N–P stoichiometric ratios also varied with soil depth in all land uses. The soil C/N, N/P and C/P ratios gradually decreased from the surface to the deeper soil layer in all land uses ($p < 0.05$; Table A3, Table 2).

In all land uses, the slopes of SMA regression for soil total C versus N were significantly lower than 1 ($p < 0.001$; Table 3), while for soil total N versus P, the slopes were larger than 1. For soil total C versus P, the slopes were lower than 1 in both RM and TRF, but higher than 1 in JR. All slopes of SMA regression for JR was higher compared to RM and TRF ($p < 0.05$; Table 3).

4. Discussion

4.1. Changes in soil physical properties

Land-use change affects soil physical properties by directly or indirectly changing the biological factors (e.g., species abundance, forest management practices, litterfall inputs) (Jiang et al., 2019; Li et al., 2019). For example, an increase in species abundance would improve litterfall inputs and root distributions, and ultimately contribute to soil structure (Drewry, 2006; Zhu et al., 2019). Our results suggested that after ten years of natural succession, soil physical properties of JR improved. For example, compared to RM, soil bulk density was significantly decreased, and soil non-capillary porosity was significantly increased in JR. These results probably appeared due to the presence of understory plants (i.e., tree saplings and shrubs) that improved plant

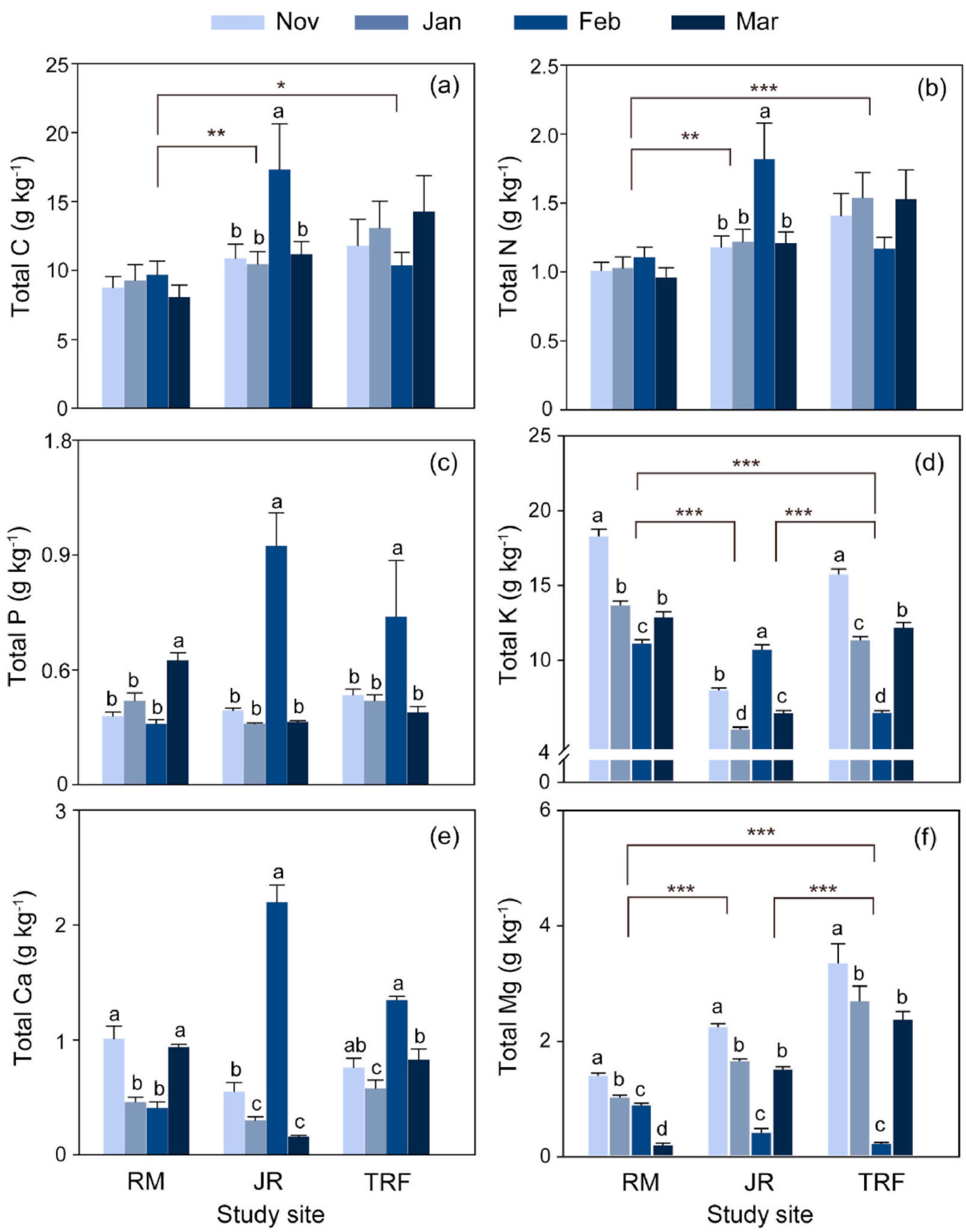

Fig. 4. Seasonal variations in soil nutrient concentrations of different study sites. Data are expressed as the mean ± SD (n = 4). Bars with different lowercase letters indicate significant difference among sampling time ($p < 0.05$); if there were significant differences among study sites, the asterisks are marked in the bars (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$), the tested method was one-way analysis of variance (ANOVA, $p < 0.05$) followed by the least significant difference (LSD) test. Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest.

litterfall inputs and root distribution in JR. Moreover, higher bulk density or soil compaction in RM, particularly in the surface soil layer (0–5 cm), likely resulted from the high-frequency tread by local farmers' tapping and field management activities. (Fig. 2b; Rezaei and Gilkes, 2005; Zaher et al., 2020). Non-capillary porosity is positively related to soil aeration (Gupta and Gupta, 1979), suggesting there are fewer spaces between soil particles in RM that do not favor root development, water retention, and nutrient stability. Compared to RM, the lower bulk density and higher non-capillary porosity in TRF are likely due to similar reasons associated with higher plant density and negligible anthropogenic disturbances. In agreement with these results, Chen et al. (2019) also found that the forest or agroforestry's complex vegetation structure decreases soil bulk density and improves soil porosity and other physical properties.

Soil is an integrated multiphase system that mainly consists of a solid, liquid, and gaseous phases that control the internal movements and storage of heat and flows, which affect plant roots development, edaphic environment and water infiltration (Horn et al., 1994; Wang et al., 2015a, 2015b). The soil three-phase ratio generally varied with the land-use change, with a higher value indicates a better soil structure (Wang et al., 2015a, 2015b). In our study, the transformation of RM to JR improved soil three-phase ratio, indicating soil in JR has a better structure (Fig. 1; Table A1). Land-use change also influence soil hydrological processes (Jiang et al., 2019). In this case, we found JR increased soil water holding capacities (e.g., saturated water capacity, field capacity) compared to RM, meaning that JR improves soil water availability. Besides, rubber trees develop a robust root system in which most feeder roots and taproot occur in the top 30 cm soil layer and grow horizontally. Thereby, rubber trees mainly depend on surface soil water (Priyadarshan, 2011) and preferably use it as found in a stable isotope study (Wu et al., 2017). Latex production by rubber trees leads to a large amount of soil water loss (Priyadarshan, 2011; Tan et al., 2011),

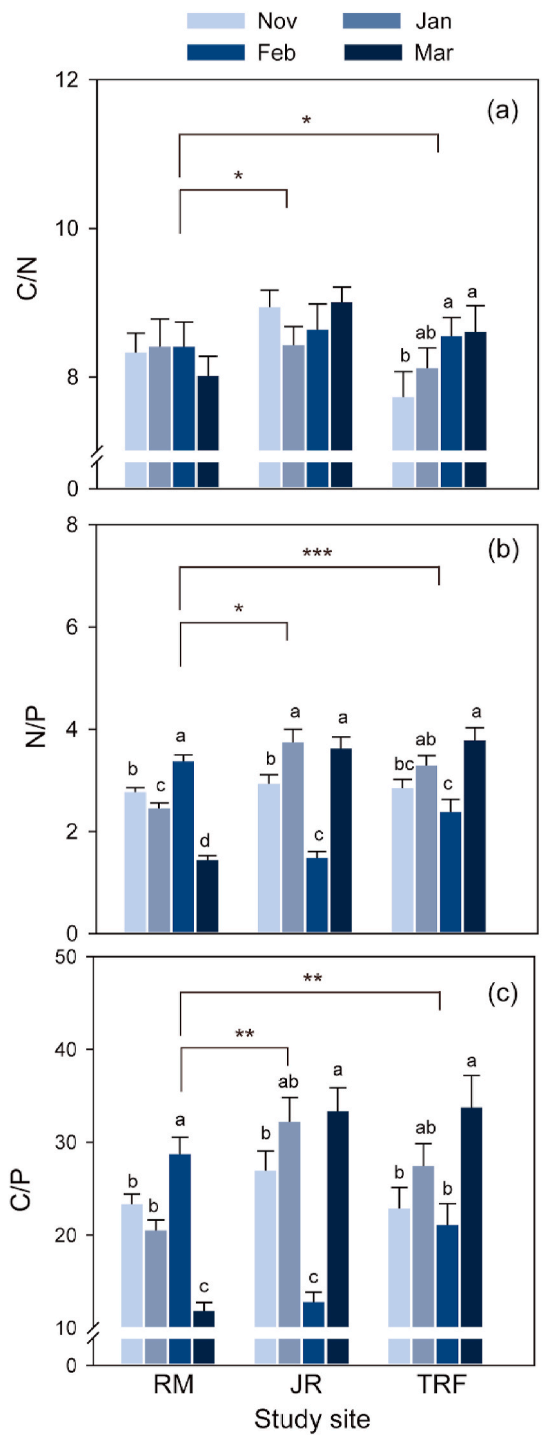

Fig. 5. Seasonal variations in soil C, N, P stoichiometric ratios of different study sites. Data are expressed as the mean ± SD (n = 4). Bars with different lowercase letters indicate significant difference among sampling time ($p < 0.05$); if there were significant differences among study sites, the asterisk are marked in the bars (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$), the tested method was one-way analysis of variance (ANOVA, $p < 0.05$) followed by the least significant difference (LSD) test. Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest.

possibly a reason for the result of lower soil water capacities (e.g., saturated water capacity, capillary water capacity, and field capacity) in JR and RM compared to TRF.

4.2. Soil chemical properties

Soil chemical properties as indicators of soil quality are sensitive to numerous environmental factors, including plant diversity and agriculture management (Larreguy et al., 2017; Singh et al., 2020). In the present study, rubber cultivation influenced both soil acidity and salinity (Table 2). Soil pH is applied extensively as an index of soil acidification or alkalization (Slessarev et al., 2016). The alkaline soil is a characteristic feature of the arid climate and acidic ones of the humid region (Yang et al., 2015). In humid tropical areas, high mean air temperature and abundant rainfall result in the accumulation of iron-aluminum oxides; additionally, soil weathering and leaching activities are intense, and the biological materials cycling is fast. Thus, the soil is base unsaturated, and the pH value is generally in the acidic range 4.5–6 (Slessarev et al., 2016). Our study sites are located in the southwest tropical region of China, resulted in the pH values across the land use treatments were lower than 6. Moreover, the deposition of nitrogen and sulfur oxides also contribute to soil acidification (Alves et al., 2019). And to increase the productivity and resist powdery mildew of rubber trees, a large amount of compound fertilizer and the sulfur powder was applied in the rubber field (Liu et al., 2019; Zakari et al., 2020). These land management practices, together with the precipitation of nitrogen and sulfur oxides, contribute to soil acidification. (Alves et al., 2019). Therefore, the soil pH values were lower in JR and RM compared to TRF (Fig. 3). These results are consistent with those found by Liu et al. (2019) that the conversion of natural forest to rubber plantation accelerates soil acidification (Liu et al., 2019). In the case of soil salinity, EC is applied extensively as a relatively reliable indicator due to the strong positive correlation between soil EC and total soluble salt concentrations (Rhoades et al., 1989). According to previous researches, the EC of soil saturated paste extract >4000 μS cm^{-1} can be defined as saline-alkali soil (Chhabra, 2004). In our study, there was no need for irrigation activities due to the abundant rainfall. Thus, the soil in all study sites was not threatened by high salinity.

We found significant differences in soil nutrients among three land uses, suggesting the present land-use change significantly influences soil nutrients (Table 2, Fig. 3). The previous researches demonstrated that a mixed agriculture ecosystem would increase concentrations and stocks of soil organic matter, C, and N compared to monoculture (Zhang et al., 2013; Tesfaye et al., 2016), and hence these ecosystems has a positive impact on soil quality compared to monoculture ones. These results are attributed to the increases in plant species abundance, understory vegetation, litter input, root growth, and biomass production in the mixed agriculture system (Xiong et al., 2008). These interpretations are an explanation for our results of improvement in soil total C, and N concentrations following RM converted to JR (a mixture of jungle rubber and commercial rubber) (Fig. 4a and b). However, RM exhibited the lowest soil total C and N concentration across the soil profile among the treatments, indicating RM cropping attributed to C and N loss due to decomposition and leaching. These results also proved that a mixed agriculture ecosystem could improve the soil nutrient supply (Blanco-Moure et al., 2016).

We found JR significantly improved concentration of other nutrients including soil total P, Ca, and Mg compared to those found in RM (Fig. 4 c, e, f). The lower values of these nutrients in RM were likely due to lower plant density, uniform structure (a typical feature of monoculture), and lack of understory vegetation cover (Table A1, Table 2). Other researchers also demonstrated similar results. For instance, Liu et al. (2015) found lower plant density in RM increased rainfall drop kinetic energy and elevated soil erosion, which resulted in nutrient loss and worsened soil properties. Zhu et al. (2018) reported that RM exhibited adverse soil physical properties and accelerates surface runoff and sediment yield resulted in lower soil nutrients.

Although previous studies indicated that soil nutrients concentration can be affected by land-use change, lots of studies mainly concerned the surface soil (Yu et al., 2018), while less attention has focused on the

Table 3
Summary of standardized major axis (SMA) regression analysis of $\log_{10}$-transformed soil total C, N and P concentrations.

Study site	Variable		n	Intercept (95% CI)	Slope (95% CI)	R^2	p values	
	x	y						
RM	T.C	T.N	96	−0.63 (−0.65, −0.61)	0.69 (0.66, 0.71)	0.97	<0.001	***
JR	T.C	T.N	96	−0.73 (−0.77, −0.70)	0.80 (0.77, 0.84)	0.95	<0.001	***
TRF	T.C	T.N	96	−0.69 (−0.74, −0.65)	0.78 (0.75, 0.83)	0.94	<0.001	***
RM	T.C	T.P	96	−1.14 (−1.27, −1.01)	0.83 (0.70, 0.98)	0.32	<0.001	*
JR	T.C	T.P	96	−1.48 (−1.68, −1.28)	1.11 (0.93, 1.31)	0.28	<0.001	NS
TRF	T.C	T.P	96	−1.20 (−1.37, −1.02)	0.84 (0.69, 1.01)	0.09	0.003	NS
RM	T.N	T.P	96	−0.37 (−0.41, −0.34)	1.21 (1.02, 1.43)	0.30	<0.001	*
JR	T.N	T.P	96	−0.46 (−0.41, −0.34)	1.38 (1.17, 1.62)	0.35	<0.001	***
TRF	T.N	T.P	96	−0.46 (−0.51, −0.41)	1.07 (0.88, 1.29)	0.12	0.001	NS

Study sites are RM: rubber monoculture, JR: jungle rubber, TRF: tropical rainforest. N: number of soil samples in each study site; R^2: coefficient of determination; p: significance level.
For all the nutrient relationships, the bivariate relationship was significant ($p < 0.05$). *, **, and *** represent significant difference between the slope and 1 at $p < 0.05$, 0.01 and 0.001 level, respectively. NS represent no significant difference between the slope and 1. All soil nutrients concentration data were transformed into a log-log scale.

variations within soil profiles. Our results showed the concentrations of soil total C, N, and P decreased as soil depth increased (Table A.4). Since surface soil directly receives nutrients from the decomposition of plant and animal residue as well as fertilizer application, the higher proportion of nutrients in topsoil layers compared to the deeper layers is possible. We found the soil total P values were relatively stable across the profile than those found for C and N. This could be either due to higher P leaching through the soil profile or due to its translocation from the P-rich soil to P-poor one by microorganisms (Zhang et al., 2005). The total K and Mg concentrations increased as soil sampling depth increased in all study sites, which may be the consequence of leaching activities due to high temperatures and precipitation in the study area (Laird et al., 2010).

The temporal variation in soil nutrients appeared due to plant phenology, which influences the nutrient cycle by mediating plant nutrient uptake, litterfall, and biodiversity (Nord and Lynch, 2009). Moreover, the seasonal distribution of agronomic activities (e.g., rubber tapping and fertilizer application) also affect soil nutrients' spatiotemporal variability (Dea et al., 2001). We can find such kind of temporal variation in our results, with the soil nutrients (total C, N, P, K, and Ca) in February (rubber trees with no leaf) were higher than other sampling time in JR. However, due to the frequent fertilizer application, there was less difference in soil nutrients between sampling times in RM.

4.3. The relationships among soil C–N–P

Soil C, N, and P are the hot topics for learning about global changes, the relationships and balance among soil C, N, and P were coupling in the ecosystem elemental cycles, so the variations in these soil nutrients relationship would influence the ecosystem cycles strongly (Ågren, 2004; Cleveland and Liptzin, 2007). For instance, soil nutrients stoichiometric ratios can be used to explore both the relative availability of soil nutrients for plants and the nutrient balance in ecological processes (Wang et al., 2020). It is also preferentially used as an indicator for assessing the response of soil nutrients to different land uses (Hobbie, 2015). Land-use change affects soil C, N, and P stoichiometric ratios, and shape the cycling and availability of soil nutrients (Bouwman et al., 2009; Vitousek et al., 1997a). We found RM has lower soil N/P and C/P ratios compared to JR and TRF, indicating N and P limitation in soils of RM (Fig. 5 b, c; Zhang et al., 2013). These results were due to the significantly lower soil total C and N concentrations in RM compared to JR and TRF, while the concentrations of soil total P did not significantly differ among RM, JR, and TRF (Fig. 5). This result also suggesting soils in RM are under organic matter deficit owing to lack of plant density, diversity, and litter availability, because plant diversity regulate nutrient use efficiency and availability (Liu et al., 2017). Moreover, plant diversity and density can also modify nutrient use efficiency and nutrients availability (Wang et al., 2015a, 2015b). However, higher plant density and species richness (including tree, shrub, and herb layers) lead to greater nutrients availability and nutrient balance in JR and TRF compared to RM (Table A1). Also, the insufficient supply of P can affect the N fixation process and lead to higher soil N/P ratios (Elser et al., 2007; Peñuelas et al., 2012). However, during the dry season, a low P loss through leaching results in a higher soil P concentration (Zhang et al., 2005), attributed to the lower soil C/P and N/P ratios in February (Fig. 5). The similar soil C/N ratio between RM and TRF indicating that the accumulation rates of soil C and N were synchronous in RM and TRF. The comparable level of a higher positive correlation between the total C and N in these land use treatments are also corroborating this view. These findings also suggest that P is the primary limiting nutrients in our study sites. Therefore, it is essential to improve the P availability for sustainable functioning and productivity of rubber agroforestry ecosystems (Brady and Weil, 2008; Yang et al., 2012).

The allometric relationship of soil nutrients among different land uses can also reveal the relationships among soil nutrients (Legendre and Legendre, 2012). The results of SMA showed there are close relationships among soil C, N, and P concentrations ($p < 0.05$ for all models; Table 3). The relationships between soil total C and N are allometric in all study sites that resulted in the significantly different slope of the models from 1 (Table 3). In RM and TRF treatments, soil total C varied fastest, followed by soil total P and N, while in JR, soil total P varied fastest, followed by soil total C and N. It indicates soil total N is the most stable nutrients in these study sites, and soil total P is the most volatile one. These results also indicate that soil in JR adjusted the P cycling process to acquire the P for plant growth and development, and it reflects a higher turnover efficiency of P in JR (Johnson et al., 2003).

5. Conclusion

This study found that soil physical properties, nutrient concentrations and nutrients relationships could be influenced significantly by land-use change. The conversion of TRF to RM resulted in severe soil degradation, with decreased soil water capacities and soil porosities, and increases in soil bulk density, and soil acidification. The soil nutrients, including total C, N, P, Ca, and Mg concentrations, were also reduced in RM compared to TRF across the vertical soil profile. On the other hand, the conversion of RM to JR resulted in the improvement of soil properties. The concentration of total C, N, P, Ca, and Mg concentrations were significantly improved with JR. Moreover, JR also improved soil P turnover efficiency and hence improved soil nutrient cycle. These land-use change also altered the C–N–P stoichiometric ratio, with P was the primary limiting nutrient. It suggests that more attention should be paid to improve P availability in the present region.

Thereby, constructing JR would be a practical way to restore soil degradation caused by monoculture rubber plantations effectively. This approach will also prove useful in expanding our understanding of the sustainable management of existing aged monoculture rubber plantations economically.

Credit author statement

Huanhuan Zeng: Conceptualization, Methodology, Investigation, Formal analysis, Writing – original draft, Writing – review & editing. Junen Wu: Methodology, Investigation, Data curation, Writing – original draft, Resources, Funding acquisition. Xiai Zhu: Formal analysis, Investigation, Visualization, Resources. Ashutosh Kumar Singh, Investigation, Methodology, Writing – review & editing. Chunfeng Chen: Investigation, Resources, Visualization. Wenjie Liu: Supervision, Project administration, Funding acquisition, Writing – review & editing.

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

We thank Mr. Liu, M.N., Dr. Zakari, S., Xishuangbanna Station for Tropical Rainforest Ecosystem Studies and the Central Laboratory of XTBG for their contribution to this work. This work was supported by the National Natural Science Foundation of China (31800356, 31700560 and 31570622); the National Postdoctoral Program for Innovative Talents (BX201700278); the China Postdoctoral Science Foundation (2018M633437); and the CAS 135 Program (2017XTBG–F01).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2021.111959.

References

Ågren, G.I., 2004. The C: N: P stoichiometry of autotrophs-theory and observations. Ecol. Lett. 7 (3), 185–191. https://doi.org/10.1111/j.1461-0248.2004.00567.x.
Alves, L.A., de Oliveira Denardin, L.G., Martins, A.P., Anghinoni, I., de Faccio Carvalho, P.C., Tiecher, T., 2019. Soil acidification and P, K, Ca and Mg budget as affected by sheep grazing and crop rotation in a long-term integrated crop-livestock system in southern Brazil. Geoderma 351, 197–208. https://doi.org/10.1016/j.geoderma.2019.04.036.
Blanco-Moure, N., Gracia, R., Bielsa, A.C., López, M.V., 2016. Soil organic matter fractions as affected by tillage and soil texture under semiarid Mediterranean conditions. Soil Tillage Res. 155, 381–389. https://doi.org/10.1016/j.still.2015.08.011.
Bouwman, A.F., Beusen, A.H., Billen, G., 2009. Human alteration of the global nitrogen and phosphorus soil balances for the period 1970–2050. Global Biogeochem. Cycles 23, 4. https://doi.org/10.1029/2009GB003576.
Bronick, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124 (1–2). https://doi.org/10.1016/j.geoderma.2004.03.005.
Brady, N.C., Weil, R.R., 2008. Nature and Properties of Soils, fourteenth ed. Prentice Hall, New Jersey.
Cao, H., Liu, J., Fu, C., Zhang, W., Wang, G., Yang, G., Luo, L., 2017. Urban expansion and its impact on the land use pattern in Xishuangbanna since the reform and opening up of China. Rem. Sens. 9 (2), 137. https://doi.org/10.3390/rs9020137.
Chen, C., Liu, W., Wu, J., Jiang, X., Zhu, X., 2019. Can intercropping with the cash crop help improve the soil physico-chemical properties of rubber plantations? Geoderma 335, 149–160. https://doi.org/10.1016/j.geoderma.2018.08.023.
Chen, L., 2005. Soil Test Course. Northeast Forestry University Press, Harbin (in Chinese).
Chhabra, R., 2004. Classification of salt-affected soils. Arid Land Res. Manag. 19 (1), 61–79. https://doi.org/10.1080/15324980590887344.
Cleveland, C.C., Liptzin, D., 2007. C/N/P stoichiometry in soil: is there a "Redfield ratio" for the microbial biomass? Biogeochemistry 85 (3), 235–252. https://doi.org/10.1007/s10533-007-9132-0.
Dea, G.B., Assiri, A.A., Gabla, O.R., Boa, D., 2001. Influence of soil preparation method on root and vegetative growth of rubber tree (*Hevea brasiliensis*) in the southwest Côte d'Ivoire. Soil Tillage Res. 59 (1–2), 3–11. https://doi.org/10.1016/S0167-1987(00)00174-4.
Drewry, J.J., 2006. Natural recovery of soil physical properties from treading damage of pastoral soils in New Zealand and Australia: a review. Agric. Ecosyst. Environ. 114 (2–4), 159–169. https://doi.org/10.1016/j.agee.2005.11.028.
Elser, J.J., Bracken, M.E., Cleland, E.E., Gruner, D.S., Harpole, W.S., Hillebrand, H., Smith, J.E., 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecol. Lett. 10 (12), 1135–1142. https://doi.org/10.1111/j.14610248.2007.01113.x.
Fox, J., Castella, J.C., Ziegler, A.D., Westley, S.B., 2014. Rubber plantations expand in mountainous Southeast Asia: what are the consequences for the environment? Asia Pac. Issues 114, 1–8.
Hobbie, S.E., 2015. Plant species effects on nutrient cycling: revisiting litter feedbacks. Trends Ecol. Evol. 30 (6), 357–363. https://doi.org/10.1016/j.tree.2015.03.015.
Horn, R., Taubner, H., Wuttke, M., Baumgartl, T., 1994. Soil physical properties related to soil structure. Soil Tillage Res. 30 (2–4), 187–216. https://doi.org/10.1016/0167-1987(94)90005-1.
Jiang, X.J., Chen, C., Zhu, X., Zakari, S., Singh, A.K., Zhang, W., Liu, W., 2019. Use of dye infiltration experiments and HYDRUS-3D to interpret preferential flow in soil in a rubber-based agroforestry systems in Xishuangbanna, China. Catena 178, 120–131. https://doi.org/10.1016/j.catena.2019.03.015.
John, R., Dalling, J.W., Harms, K.E., Yavitt, J.B., Stallard, R.F., Mirabello, M., Foster, R. B., 2007. Soil nutrients influence spatial distributions of tropical tree species. Proc. Natl. Acad. Sci. U.S.A. 104 (3), 864–869. https://doi.org/10.1073/pnas.0604666104.
Johnson, A.H., Frizano, J., Vann, D.R., 2003. Biogeochemical implications of labile phosphorus in forest soils determined by the Hedley fractionation procedure. Oecologia 135 (4), 487–499. https://doi.org/10.1007/s00442-002-1164-5.
Katsvairo, T.W., Wright, D.L., Marois, J.J., Hartzog, D.L., Balkcom, K.B., Wiatrak, P.P., Rich, J.R., 2007. Cotton roots, earthworms, and infiltration characteristics in sod–peanut-cotton cropping systems. Agron. J. 99 (2), 390–398. https://doi.org/10.2134/agronj2005.0330.
Langenberger, G., Cadisch, G., Martin, K., Min, S., Waibel, H., 2017. Rubber intercropping: a viable concept for the 21st century? Agrofor. Syst. 91 (3), 577–596. https://doi.org/10.1007/s10457-016-9961-8.
Larreguy, C., Carrera, A.L., Bertiller, M.B., 2017. Reductions of plant cover induced by sheep grazing change the above-belowground partition and chemistry of organic C stocks in arid rangelands of Patagonian Monte, Argentina. J. Environ. Manag. 199, 139–147. https://doi.org/10.1016/j.jenvman.2017.04.086.
Legendre, P., Legendre, L., 2012. Numerical Ecology, third ed. Elsevier, Amsterdam.
Li, J., Wan, X., Liu, X., Chen, Y., Slaughter, L.C., Weindorf, D.C., Dong, Y., 2019. Changes in soil physical and chemical characteristics in intensively cultivated greenhouse vegetable fields in North China. Soil Tillage Res. 195, 104366. https://doi.org/10.1016/j.still.2019.104366.
Liu, C.A., Liang, M.Y., Nie, Y., Tang, J.W., Siddique, K.H., 2019. The conversion of tropical forests to rubber plantations accelerates soil acidification and changes the distribution of soil metal ions in topsoil layers. Sci. Total Environ. 696, 134082. https://doi.org/10.1016/j.scitotenv.2019.134082.
Liu, W., Luo, Q., Li, J., Wang, P., Lu, H., Liu, W., Li, H., 2015. The effects of conversion of tropical rainforest to rubber plantation on splash erosion in Xishuangbanna, SW China. Nord. Hydrol 46 (1), 168–174. https://doi.org/10.2166/nh.2013.109.
Liu, X., Ma, J., Ma, Z.W., Li, L.H., 2017. Soil nutrient contents and stoichiometry as affected by land-use in an agro-pastoral region of northwest China. Catena 150, 146–153. https://doi.org/10.1016/j.catena.2016.11.020.
Lu, M., Ren, Y., Wang, S., Tian, K., Sun, X., Peng, S., 2019. Contribution of soil variables to bacterial community composition following land use change in Napahai plateau wetlands. J. Environ. Manag. 246, 77–84. https://doi.org/10.1016/j.jenvman.2019.05.149.
Marschner, H., 2011. Marschner's Mineral Nutrition of Higher Plants, third ed. Elsevier, London.
Melesse, A.M., Abtew, W., Senay, G., 2019. Extreme Hydrology and Climate Variability: Monitoring, Modelling, Adaptation and Mitigation. Elsevier, London.
Nord, E.A., Lynch, J.P., 2009. Plant phenology: a critical controller of soil resource acquisition. J. Exp. Bot. 60 (7), 1927–1937. https://doi.org/10.1093/jxb/erp018.
Peerawat, M., Blaud, A., Trap, J., Chevallier, T., Alonso, P., Gay, F., Brauman, A., 2018. Rubber plantation ageing controls soil biodiversity after land conversion from cassava. Agric. Ecosyst. Environ. 257, 92–102. https://doi.org/10.1016/j.agee.2018.01.034.
Peñuelas, J., Sardans, J., Rivas-ubach, A., Janssens, I.A., 2012. The human-induced imbalance between C, N and P in Earth's life system. Global Change Biol. 18 (1), 3–6. https://doi.org/10.1111/j.1365-2486.2011.02568.x.
Priyadarshan, P.M., 2011. Biology of *Hevea* Rubber, second ed. CABI, Wallingford.
Rezaei, S.A., Gilkes, R.J., 2005. The effects of landscape attributes and plant community on soil physical properties in rangelands. Geoderma 125 (1–2), 145–154. https://doi.org/10.1016/j.geoderma.2004.07.010.
Rhoades, J.D., Manteghi, N.A., Shouse, P.J., Alves, W.J., 1989. Soil electrical conductivity and soil salinity: new formulations and calibrations. Soil Sci. Soc. Am. J. 53 (2), 433–439. https://doi.org/10.2136/sssaj1989.03615995005300020020x.
Schilling, K.E., Palmer, J.A., Bettis III, E.A., Jacobson, P., Schultz, R.C., Isenhart, T.M., 2009. Vertical distribution of total carbon, nitrogen and phosphorus in riparian soils

of Walnut Creek, southern Iowa. Catena 77 (3), 266–273. https://doi.org/10.1016/j.catena.2009.02.006.

Singh, A.K., Jiang, X.J., Yang, B., Wu, J., Rai, A., Chen, C., Singh, N., 2020. Biological indicators affected by land use change, soil resource availability and seasonality in dry tropics. Ecol. Indicat. 115, 106369. https://doi.org/10.1016/j.ecolind.2020.106369.

Slessarev, E.W., Lin, Y., Bingham, N.L., Johnson, J.E., Dai, Y., Schimel, J.P., Chadwick, O. A., 2016. Water balance creates a threshold in soil pH at the global scale. Nature 540 (7634), 567–569. https://doi.org/10.1038/nature20139.

Tan, Z.H., Zhang, Y.P., Song, Q.H., Liu, W.J., Deng, X.B., Tang, J.W., Liang, N.S., 2011. Rubber plantations act as water pumps in tropical China. Geophys. Res. Lett. 38 (24) https://doi.org/10.1029/2011GL050006.

Tesfaye, M.A., Bravo, F., Ruiz-Peinado, R., Pando, V., Bravo-Oviedo, A., 2016. Impact of changes in land use, species and elevation on soil organic carbon and total nitrogen in Ethiopian Central Highlands. Geoderma 261, 70–79. https://doi.org/10.1016/j.geoderma.2015.06.022.

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Tilman, D.G., 1997a. Human alteration of the global nitrogen cycle: sources and consequences. Ecol. Appl. 7 (3), 737–750. https://doi.org/10.1890/1051-0761(1997)007[0737:HAOTGN]2.0. .CO;2.

Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997b. Human domination of Earth's ecosystems. Science 277 (5325), 494–499. https://doi.org/10.1126/science.277.5325.494.

Vogel, A.W., Wang, M.Z., Huang, X.Q., 1995. People's Republic of China: Reference Soil of Tropical Southern Yunnan Province. Institute of Soil Science-Academica Sinica, Nanjing, and International Soil Reference and Information Centre, Wageningen.

Wang, E., Cruse, R.M., Zhao, Y., Chen, X., 2015a. Quantifying soil physical condition based on soil solid, liquid and gaseous phases. Soil Tillage Res. 146, 4–9. https://doi.org/10.1016/j.still.2014.09.018.

Wang, W., Wang, C., Sardans, J., Min, Q., Zeng, C., Tong, C., Peñuelas, J., 2015b. Agricultural land use decouples soil nutrient cycles in a subtropical riparian wetland in China. Catena 133, 171–178. https://doi.org/10.1016/j.catena.2015.05.003.

Wang, X.G., Lv, X.T., Zhang, H.Y., Dijkstra, F.A., Jiang, Y.G., Wang, X.B., Han, X.G., 2020. Changes in soil C:N:P stoichiometry along an aridity gradient in drylands of northern China. Geoderma 361, 114087. https://doi.org/10.1016/j.geoderma.2019.114087.

Warren-Thomas, E., Nelson, L., Juthong, W., Bumrungsri, S., Brattström, O., Stroesser, L., Dolman, P.M., 2020. Rubber agroforestry in Thailand provides some biodiversity benefits without reducing yields. J. Appl. Ecol. 57 (1), 17–30. https://doi.org/10.1111/1365-2664.13530.

Warton, D.I., Weber, N.C., 2002. Common slope tests for bivariate errors-in-variables models. Biom. J. 44 (2), 161–174. https://doi.org/10.1002/1521-4036. (200203)44:2<161::AID-BIMJ161>3.0.CO;2-N.

Wu, J., Liu, W., Chen, C., 2017. How do plants share water sources in a rubber-tea agroforestry system during the pronounced dry season? Agric. Ecosyst. Environ. 236, 69–77. https://doi.org/10.1016/j.agee.2016.11.017.

Xiao, C., Li, P., Feng, Z., 2019. Monitoring annual dynamics of mature rubber plantations in Xishuangbanna during 1987–2018 using Landsat time series data: a multiple normalization approach. Int. J. Appl. Earth Obs. Geoinf. 77, 30–41. https://doi.org/10.1016/j.jag.2018.12.006.

Xiong, Y., Xia, H., Cai, X.A., Fu, S., 2008. Impacts of litter and understory removal on soil properties in a subtropical *Acacia mangium* plantation in China. Plant Soil 304 (1–2), 179–188. https://doi.org/10.1007/s11104-007-9536-6.

Yang, W., Cheng, H., Hao, F., Ouyang, W., Liu, S., Lin, C., 2012. The influence of land-use change on the forms of phosphorus in soil profiles from the Sanjiang Plain of China. Geoderma 189, 207–214. https://doi.org/10.1016/j.geoderma.2012.06.025.

Yang, Y., Li, P., He, H., Zhao, X., Datta, A., Ma, W., Fang, J., 2015. Long-term changes in soil pH across major forest ecosystems in China. Geophys. Res. Lett. 42 (3), 933–940. https://doi.org/10.1002/2014GL062575.

Yi, Z.F., Cannon, C.H., Chen, J., Ye, C.X., Swetnam, R.D., 2014. Developing indicators of economic value and biodiversity loss for rubber plantations in Xishuangbanna, southwest China: a case study from Menglun township. Ecol. Indicat. 36, 788–797. https://doi.org/10.1016/j.ecolind.2013.03.016.

Zaher, H., Sabir, M., Benjelloun, H., Paul-Igor, H., 2020. Effect of forest land use change on carbohydrates, physical soil quality and carbon stocks in Moroccan cedar area. J. Environ. Manag. 254, 109544. https://doi.org/10.1016/j.jenvman.2019.109544.

Zakari, S., Liu, W., Wu, J., Singh, A.K., Jiang, X., Yang, B., Zhu, X., 2020. Decay and erosion-related transport of sulfur compounds in soils of rubber based agroforestry. J. Environ. Manag. 274, 111200. https://doi.org/10.1016/j.jenvman.2020.111200.

Zhang, C., Tian, H., Liu, J., Wang, S., Liu, M., Pan, S., Shi, X., 2005. Pools and distributions of soil phosphorus in China. Global Biogeochem. Cycles 19 (1). https://doi.org/10.1029/2004GB002296.

Zhang, J.Q., Corlett, R.T., Zhai, D., 2019. After the rubber boom: good news and bad news for biodiversity in Xishuangbanna, Yunnan, China. Reg. Environ. Change 19 (6), 1713–1724. https://doi.org/10.1007/s10113-019-01509-4.

Zhang, K.Y., 1988. The climatic dividing line between SW and SE monsoons and their differences in climatology and ecology in Yunnan province of China in Climates, geoecology and agriculture in South China. Climatolo. Notes 38, 197–207. http://ir.xtbg.org.cn/handle/353005/4885.

Zhang, Z.S., Song, X.L., Lu, X.G., Xue, Z.S., 2013. Ecological stoichiometry of carbon, nitrogen, and phosphorus in estuarine wetland soils: influences of vegetation coverage, plant communities, geomorphology, and seawalls. J. Soils Sediments 13 (6), 1043–1051. https://doi.org/10.1007/s11368-013-0693-3.

Zhu, H., Cao, M., Hu, H., 2008. Geological history, flora, and vegetation of Xishuangbanna, southern yunnan, China [J]. Biotropica 38 (3), 310–317. https://doi.org/10.1111/j.1744-7429.2006.00147.x.

Zhu, X., Liu, W., Chen, H., Deng, Y., Chen, C., Zeng, H., 2019. Effects of forest transition on litterfall, standing litter and related nutrient returns: implications for forest management in tropical China. Geoderma 333, 123–134. https://doi.org/10.1016/j.geoderma.2018.07.023.

Zhu, X., Liu, W., Jiang, X., Wang, P., Li, W., 2018. Effects of land-use change on runoff and sediment yield: implications for soil conservation and forest management in Xishuangbanna, Southwest China. Land Degrad. Dev. 29 (9), 2962–2974. https://doi.org/10.1002/ldr.3068.

Zhu, Y., Xu, J., Li, Q., Mortimer, P.E., 2014. Investigation of rubber seed yield in Xishuangbanna and estimation of rubber seed oil based biodiesel potential in Southeast Asia. Energy 69, 837–842. https://doi.org/10.1016/j.energy.2014.03.079.