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Soil quality assessment of different *Hevea brasiliensis* plantations in tropical China

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ABSTRACT

Land degradation is a global problem caused by improper agricultural practices. In tropical China, the rubber (*Hevea brasiliensis*) plantations are predominantly practiced on forest-cleared lands, considering their sustainable land management potential compared to annual cropping. However, all rubber plantations may not have similar land management capacity. Soil quality index (SQI) can reveal the overall soil status with a single score, which is an efficient tool to evaluate the soil quality of each category of rubber plantations. We investigated 23 soil physical and chemical parameters of three categories of rubber plantations and a primary rainforest, and derived SQI based on these parameters. Soil samples were collected from a rubber monoculture (RM), a rubber–*Camellia sinensis* agroforestry (RT), a rubber–*Dracaena cochinchinensis* agroforestry (RD), and a primary rainforest (RF). The results showed that the SQI value of the RM decreased by 15.50% compared to the RF, with a significant degree of soil nutrient loss (18.90%). This indicates that monocultural rubber cultivation is causing land degradation to some extent. However, the SQI was significantly enhanced by rubber-based agroforestry practices (25.30% by RT and 33.10% by RD) compared to the RM, suggesting that polyculture practices are suitable to recover the soil quality in degraded agricultural lands. Moreover, the chemical parameters contributed more to the SQI than did the physical parameters, indicating that nutrient management is important in soil quality recovery. Overall, our results suggest that agroforestry should be preferred over monoculture in the rubber plantations for sustainable land management in tropical China.

1. Introduction

Land degradation is growing in many parts of the world as a consequence of increasing population and agricultural activities (Zhang et al., 2011). Globally, about 23% area is affected by some form of land degradation, with a rate of 5–10 million ha per year (Barbier and Hochard, 2016; Muñoz-Rojas, 2018). The degradation of land not only deteriorates the ecology of the area through soil erosion, desertification, and salinization but also threatens the economic and cultural development (Lal, 2001). Therefore, sustainable land management is of global urgency since 25% of the world population depends directly on degraded land (Zhang et al., 2011), mainly in tropical and subtropical areas of developing countries.

In tropical China (a biodiversity hotspot), the rubber (*Hevea brasiliensis* (Willd. ex A. Juss.) Muell. Arg.) plantations were preferably practiced on forest-cleared lands since the 1950s (Xu et al., 2014). The cultivation of tree crops is believed to be more sustainable because they require fewer management practices (e.g., pesticides application and

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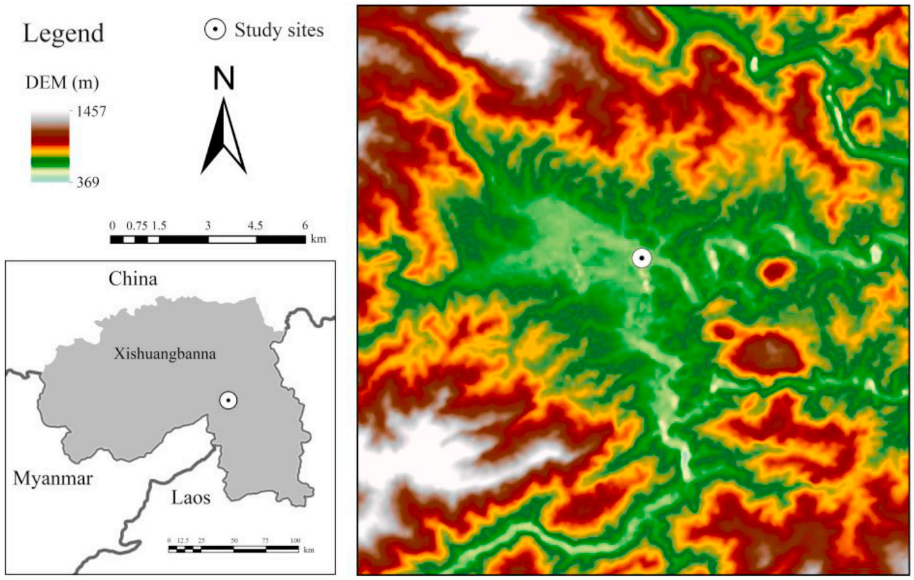

Fig. 1. Location of the study site (21°55′39″N, 101°15′55″E) in Xishuangbanna, southwest China.

fertilization) compared to annual crops (e.g., rice, wheat and corn). Also, the cultivation of tree crops provides higher economic benefits in the present region than those received from annual cropping. Therefore, the rubber plantations are expanding to meet the financial requirement and compensate for the latex demand, and it is now the main land use in tropical China. For example, in Xishuangbanna, a tropical region located in south western China, the rubber plantations cover almost 22.14% of the landscape, while only 3.60% of the tropical rainforests remain (Warren-Thomas et al., 2015). However, rubber cultivation may not be as sustainable as we believe. For instance, the conversion of tropical rainforest to rubber monoculture has led to numerous soil-related environmental issues, such as structure deterioration (Liu et al., 2015), severe erosion (Zhu et al., 2018), and the acceleration of acidification (Liu et al., 2019). Therefore, the suitable management of rubber plantations is urgently needed to restore soil quality and ensure sustainable land use.

For reducing the deteriorating impacts of rubber monoculture, rubber-based agroforestry or polyculture cropping has been established. These rubber-based agroforestry practices aimed for sustainable development, with better agriculture productivity and reduce ecosystem deterioration (e.g., control soil erosion and maintain soil quality). Therefore, some native species with economic or medicinal values have been introduced as intercrops into rubber plantations. Chen et al. (2019) found that the soil total porosity, initial moisture, and hydraulic conductivity can be improved by intercropping with *Theobroma cacao* L. and *Flemingia macrophylla* (Willd.) Prain, while Li et al. (2020) found that SOC and TN stocks can be enhanced by intercropping with *Coffea liberica* Bull ex Hiern and *Camellia sinensis* (L.) O. Ktze. The introduction of *F. macrophylla* and *Dracaena cochinchinensis* (Lour.) S. C. Chen into rubber plantations increases the formation of macroaggregates, and enhances the organic carbon and nitrogen accumulation within aggregates (Chen et al., 2017). Moreover, intercropping *C. sinensis* and *T. cacao* can increase the water use efficiency in rubber plantations by improving soil water infiltration and preferential flow (Wu et al., 2016; Zhu et al., 2019a).

Soil quality index (SQI) is an effective tool to evaluate the soil quality that unravels the overall soil status with a single score. SQI involves the assessment of soil physical, chemical, and biological variables that contribute to ecosystem functions and services (Bünemann et al., 2018; Vasu et al., 2020; Aravindh et al., 2020). Many approaches have been used to derive the SQI (Mukherjee and Lal, 2014). It is noteworthy that the applicability of SQI data is limited to a specific environment and management condition (Hemati et al., 2020). However, most of the previous research conducted in rubber plantations studied only several variables of soil physical (e.g., aggregate stability and hydrological property) and chemical properties (e.g., soil pH, carbon and nitrogen) on spatial and temporal scales (Zhu et al., 2019; Lungmuana et al., 2019; Jiang et al., 2020). To date, no study has computed an SQI for rubber plantations in the tropical region, which limits our understanding of soil ecology in this agroecosystem (Armenise et al., 2013).

This study aimed to investigate the soil physical and chemical properties in different categories of rubber plantations, and produce SQI basing on these soil properties as a tool to guide land management in tropical China. The *C. sinensis* and the *D. cochinchinensis* provide valuable beverage and medicinal materials, respectively, and these two species are widely intercropping with rubber trees in tropical China. Thus, we selected three rubber plantation sites, including a rubber monoculture, two rubber-based agroforestry practices (rubber–*C. sinensis* and rubber–*D. cochinchinensis*), and an undisturbed tropical rainforest site. The SQI was estimated from 23 soil physical and chemical parameters (including soil texture, bulk density, water holding capacities, porosities, aggregate stability, SOC, and total N, P, K, Ca, Mg contents) according to the method proposed by Andrews et al. (2002) and Bastida et al. (2006). We hypothesized that the monocultural rubber cultivation on tropical soils would lead to severe soil quality loss while the agroforestry practices would enhance the soil quality of rubber plantations.

2. Materials and methods

2.1. Study site

This study was carried out in Xishuangbanna (21°55′39″ N, 101°15′55″ E), a tropical region located in SW China (Fig. 1). This region experiences alternate rainy (May to October) and dry seasons (November to April). The mean annual precipitation ranges from 1500 to 1800 mm (approximately 87% of the total rainfall occurs during the rainy season), and the mean annual temperature varies between 24 and 29 °C. The soils are laterites (Oxisols) developed from arenaceous shale sediments. This study selected three rubber plantations sites comprising a rubber monoculture (RM), two rubber-based agroforestry (rubber–*C. sinensis* (RT) and rubber–*D. cochinchinensis* (RD)), and a primary rainforest (RF) site. All study sites are located at similar elevation (750 m) and slope (30°), and the aerial distance between sites is less than 1 km.

Table 1
The studied soil properties under different land uses and soil depths. Data were presented as means ± standard deviation (n = 4).

Properties	Depth (cm)	RM	RT	RD	RF
BD (g cm^{-3})	0–5	1.46 ± 0.02Aa	1.24 ± 0.03Ab	1.21 ± 0.02Ab	1.11 ± 0.08Bc
	5–15	1.39 ± 0.03Aa	1.27 ± 0.06Ab	1.25 ± 0.03Bb	1.27 ± 0.05Ab
	15–30	1.37 ± 0.05Ba	1.29 ± 0.05Ab	1.29 ± 0.01Cb	1.32 ± 0.05Ab
SM (%)	0–5	26.81 ± 1.06Ab	34.83 ± 2.42Aa	31.30 ± 2.28Aab	36.36 ± 5.74Aa
	5–15	27.47 ± 1.25Ab	33.05 ± 1.42Aa	32.26 ± 0.46Aa	28.39 ± 4.41Bab
	15–30	26.76 ± 1.24Ab	32.35 ± 0.54Aa	33.32 ± 1.00Aa	26.56 ± 2.57Bb
Sand (%)	0–5	35.26 ± 0.18Aa	25.27 ± 2.50Bb	27.05 ± 1.93Ab	26.24 ± 1.11Bb
	5–15	28.39 ± 1.68Bb	22.31 ± 1.22Bc	21.52 ± 1.42Bc	30.79 ± 0.89Aa
	15–30	24.91 ± 1.03Bb	47.31 ± 3.33Aa	16.82 ± 0.94Cc	24.97 ± 0.43Bb
Silt + Clay (%)	0–5	64.74 ± 0.23Cb	74.73 ± 2.50Ba	72.95 ± 1.93Ca	73.76 ± 1.11Ba
	5–15	71.61 ± 2.06Bb	77.69 ± 1.22Aa	78.48 ± 1.42Ba	69.21 ± 0.89Cb
	15–30	75.09 ± 1.27Ab	52.69 ± 3.33Cc	83.18 ± 0.94Aa	75.03 ± 0.43Ab
pH	0–5	5.23 ± 0.15Aa	5.43 ± 0.06Aa	5.23 ± 0.06Ba	5.30 ± 0.5Aa
	5–15	4.93 ± 0.06Bb	5.23 ± 0.06Ba	5.33 ± 0.06Aa	4.97 ± 0.06Bb
	15–30	5.27 ± 0.06Ab	5.40 ± 0.10Aa	5.40 ± 0.00Aa	4.77 ± 0.06Cb
EC (μs m^{-1})	0–5	176.67 ± 10.60Ab	173.00 ± 5.57Ab	131.67 ± 2.52Ac	241.67 ± 7.77Aa
	5–15	140.67 ± 3.06Ba	122.67 ± 7.32Ba	115.67 ± 1.52Ba	122.00 ± 20.2Ba
	15–30	150.33 ± 7.37Ba	121.00 ± 3.61Bb	95.33 ± 5.03Cc	91.00 ± 3.00Cc
C/N ratio	0–5	8.67 ± 0.27Aa	9.06 ± 0.07Aab	9.09 ± 0.18Aa	9.34 ± 0.25Aa
	5–15	8.03 ± 0.16Bb	8.35 ± 0.09Cb	8.91 ± 0.01Aa	8.72 ± 0.18Ba
	15–30	8.48 ± 0.09Ba	8.50 ± 0.04Bb	8.35 ± 0.09Bc	8.84 ± 0.05Ba

RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. BD: bulk density, SM: soil moisture, EC: electrical conductivity. For each land use, values with different upper-case letters indicate significant differences among soil depths; for each soil depth, values with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

The rubber trees at each plantation site were 26-year-old and planted in double rows with a stem density of 3 × 4.5 m. A wide gap of 18 m separated each set of double rows. The rubber trees were 20.1 m tall with an average stem diameter of 24.5 cm. The two kinds of intercropping agroforestry systems (about 10 years old) were grown in the 18-m gaps between the rubber trees. In the RT, the *C. sinensis* trees reached 2 m on average, and were planted in seven rows 2 m apart and separated by a 0.5 m gap within the rows. In the RD, the mean height of *D. cochinchinensis* trees was 2.5 m, planted in five rows at 1.5 m distance, and separated by a 2.5 m gap within the rows. The same fertilizer containing N, P, and K had been applied with the same dose (0.1 kg N per tree per year) in March and August in all rubber plantation sites for the last 20 years (Li et al., 2012). All rubber trees were tapped from March to November and produce approximately 1500 kg ha^{-1} of latex annually. The wild understory vegetation (shrubs and herbs) is regularly cleared in each plantation using herbicide. However, the undisturbed tropical primary rainforest site was dominated by species of *Terminalia myriocarpa* Muell. Arg., *Pometia tomentosa* (Bl.) Teysm. et Binn., and *Barringtonia macrostachya* (Jack) Kurz, and encompasses an old-growth vegetation system with a stand density of more than 100 trees ha^{-1}.

2.2. Experimental design and soil sampling

The field experiments were conducted in the 18 m-wide gaps in the rubber plantation sites (RM, RT, and RD), and randomly conducted on the slope of the RF in November 2015. The soil samples were collected in four replicates at depths of 0–5, 5–15, and 15–30 cm after removing litter horizons at each site. Each soil sample was a mix of eight subsamples, which we collected in an "S" shape pattern from every experimental field (sample weight 2 kg). In the laboratory, the root, stones, and litter debris were carefully removed from soils, and each soil sample were air-dried at a constant temperature of 25 ± 2 °C. Each sample was divided into two parts: one part was used to determine particle size distribution and chemical properties, while the other was used to assess soil aggregate stability.

Moreover, three soil cores (per site and depth, 48 samples in total) were randomly collected using steel cylinders (70.00 mm inner diameter, 52.00 mm height, and 200 cm^3 volume), and transported to the laboratory to measure the related soil physical properties.

2.3. Laboratory analysis

Several soil physical properties were measured using the steel cylinders. Firstly, the weight of empty steel cylinders (W_{ESC}) was recorded before taking the samples. After collecting the core soil samples, the weight of the steel cylinders with fresh soils (W_{SCF}) was measured. The samples were then placed in distilled water and saturated via the porous base, ensuring that the water almost reached the cylinder surface but did not enter from the top. After ponding for 24 h, the weights of the saturated samples (W_{SAT}) were measured. The samples were then placed on a layer of dry sand and weighed after draining by gravity for 2 h and 5 days (W_{2h} and W_{5d}). Finally, the samples were oven-dried at 105 °C for 24 h to weight the steel cylinders with dry soil (W_{SCD}). The bulk density (BD), soil moisture (SM), saturated water capacity (SWC), capillary holding capacity (CHC), field capacity (FC), noncapillary porosity (NP), capillary porosity (CP), and total porosity (TP) were calculated using the following formulas:

$$BD\ (g\ cm^{-3}) = \frac{W_{SCD}(g)\ - W_{ESC}(g)}{200\ (cm^3)} \tag{1}$$

$$SM(\%) = \frac{W_{SCF}(g)\ - W_{SCD}(g)}{W_{SCD}(g)\ - W_{ESC}(g)} \times 100\% \tag{2}$$

$$SWC(\%) = \frac{W_{SAT}(g) - W_{SCD}(g)}{W_{SCD}(g) - W_{ESC}(g)} \times 100\% \tag{3}$$

$$CHC(\%) = \frac{W_{2h}(g) - W_{SCD}(g)}{W_{SCD}(g) - W_{ESC}(g)} \times 100\% \tag{4}$$

$$FC(\%) = \frac{W_{5d}(g) - W_{SCD}(g)}{W_{SCD}(g) - W_{ESC}(g)} \times 100\% \tag{5}$$

$$CP(\%) = \frac{BD(gcm^{-3}) \times CHC(\%)}{\rho_{water}(g\ cm^{-3})} \tag{6}$$

$$TP\left(\%\right) = \left(1 - \frac{BD\ (g\ cm^{-3})}{2.65\ (g\ cm^{-3})}\right) \times 100\% \tag{7}$$

$$NP(\%) = TP - CP \tag{8}$$

The analysis of soil aggregate stability was performed using a modified Yoder type apparatus (Yoder, 1936). Air-dried soil samples (100 g) were placed on the top of a set of sieves with six 200 mm diameter mesh apertures of 5, 2, 1, 0.5, 0.25, and 0.053 mm from top to bottom. Each sample was prewetted with deionized water for 10 min,

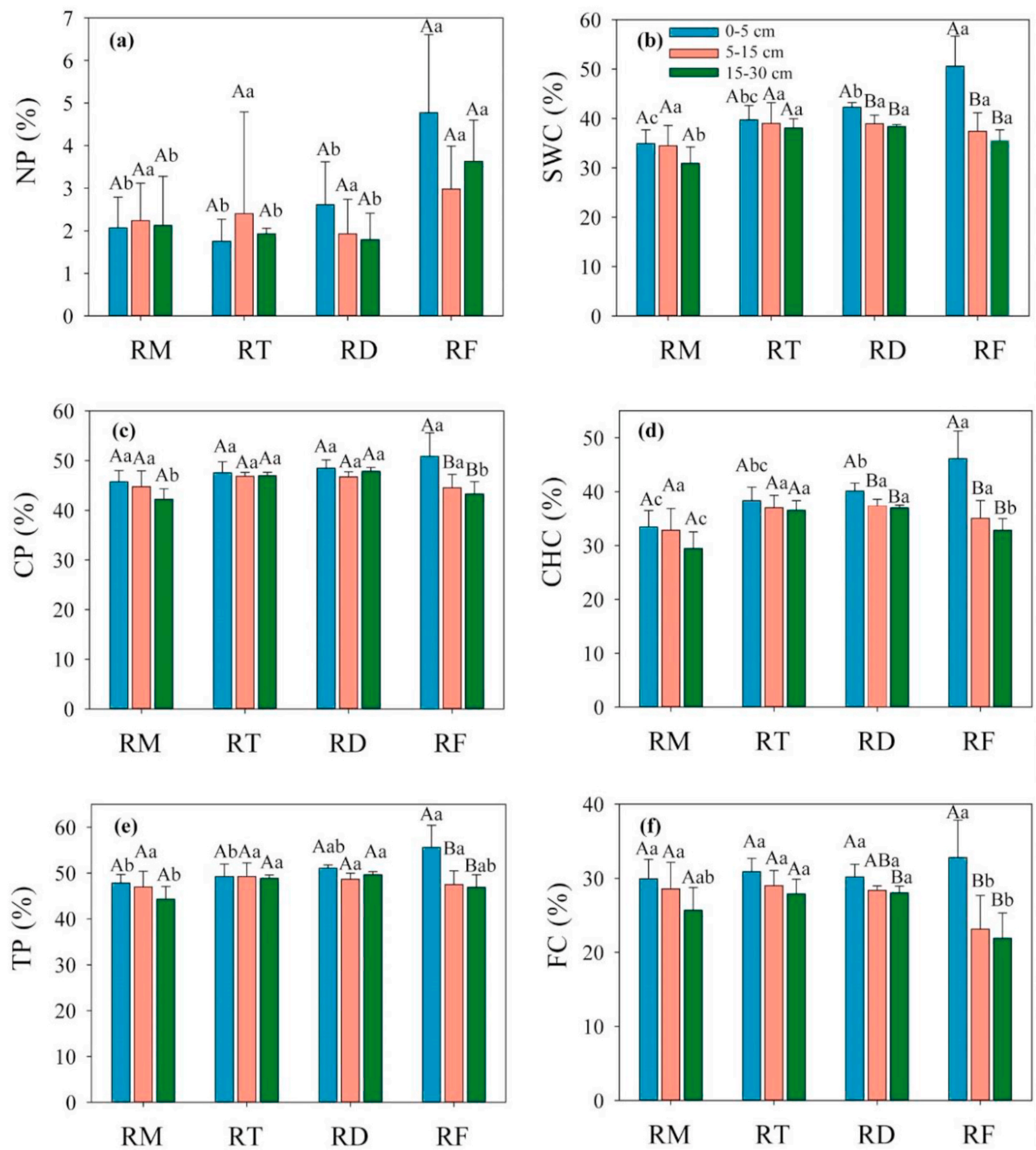

Fig. 2. Soil noncapillary porosity (NP), saturated water capacity (SWC), capillary porosity (CP), capillary holding capacity (CHC), total porosity (TP) and field capacity (FC) among different land uses. RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. Results are given as means ± standard deviation (n = 4). For each land use, bars with different upper-case letters indicate significant differences among soil depths; for each soil depth, bars with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

then the set of sieves were shaken up and down vertically (amplitude about 30 mm) at 30 rpm for 5 min with the samples submerged in water. The soil remaining on each sieve was collected and oven-dried at 60 °C for 24 h. Then, each size of soil aggregate was weighed for the calculation of the water-stable macroaggregate (WSMA) and mean weight diameter (MWD). Later, the aggregate samples were transported to the laboratory to determine the soil organic carbon (SOC) and total nitrogen (N) within the water-stable macroaggregate (WSMAC, WSMAN).

The WSMA was calculated using the following formula:

$$WSMA\ (\%) = \frac{Mr}{Mt} \times 100 \quad (9)$$

where M_r is the mass of water-stable aggregates > 0.25 mm (g), and M_t is the total mass of the wet sieved soil (g).

The MWD of water-stable aggregates was calculated using the following formula (Pinheiro et al., 2004):

$$MWD\ (mm) = \sum_{i=1}^{n} x_i y_i \quad (10)$$

where x_i is the mean diameter of each size of aggregates separated by sieving (mm), y_i is the percentage of the weight of aggregates in that size range to the total dry weight of soil, and n is the aggregate class.

The particle size distribution (silt, sand, and clay fraction) was measured using the pipette method (van Reeuwijk, 2002) after the soils were dispersed with sodium hexametaphosphate. Soil pH and electrical conductivity (EC) were measured in a supernatant water suspension (1:2.5) using a pH meter and a conductivity meter, respectively. The SOC and total N contents were analysed using the dry combustion method with an NC-2500 Elemental Analyser (Carlo Erba, Milan, Italy). The soil total phosphorus (P), potassium (K), calcium (Ca), and magnesium (Mg) contents were analysed using an inductively coupled plasma atomic emission spectrometer (ICP-AES; iCAP6300; Thermo Fisher Scientific, USA) after acid digestion and hydrofluoric acid

Table 2
The general characteristic of saturated water capacity and nutrient stocks along 0–30 cm depth under different land uses. Data were presented as means ± standard deviation (n = 4).

Properties	RM	RT	RD	RF
SWC (%)	33.45 ± 2.10b	38.94 ± 2.93a	39.87 ± 2.09a	41.17 ± 8.60a
SOC (Mg ha^{-1})	57.91 ± 3.92c	68.85 ± 1.48b	76.21 ± 2.00a	68.01 ± 1.82b
N (Mg ha^{-1})	6.74 ± 0.26d	7.95 ± 0.18b	8.63 ± 0.30a	7.52 ± 0.14c
K (Mg ha^{-1})	44.34 ± 0.62a	37.12 ± 0.53b	44.28 ± 0.66a	24.67 ± 0.26c
P (Mg ha^{-1})	1.42 ± 0.01d	1.72 ± 0.02c	2.24 ± 0.06a	1.87 ± 0.01b
Mg (Mg ha^{-1})	4.45 ± 0.06d	5.14 ± 0.13c	7.35 ± 0.10b	9.92 ± 0.15a
Ca (Mg ha^{-1})	5.48 ± 0.05d	5.52 ± 0.60c	8.10 ± 0.11a	6.77 ± 0.66b

RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. SWC: saturated water capacity, SOC: soil organic carbon, N: total nitrogen, P: total phosphorus, K: total potassium, Ca: total calcium, Mg: total magnesium. Values with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

decomposition (Chapman and Pratt, 1961). The nutrient stock was calculated using the following formula:

$$Stock\ \left(Mg\ ha^{-1}\right) = N \times BD \times D \times 0.1 \tag{11}$$

where Stock is the nutrient stock, N represents the nutrient content (g kg^{-1}), BD is the soil bulk density (g cm^{-3}), D is the soil sampling depth (cm).

2.4. Soil quality index assessment

The principal component analysis (PCA) was performed on the 23 pre-transformed chemical and physical soil parameters we measured for data reduction. The principal components (PC) with an eigenvalue >1 were kept (Kaiser, 1960), and its eigenvalue represented the relative contribution of a PC to the total variance. In each PC, highly weighted parameters with absolute values within 10% of the highest factor loading were chosen for further analysis (Andrews et al., 2002). The Pearson correlation analysis was employed to determine the redundant parameters under each PC. Generally, within each PC, the highly weighted parameters that did not show a significant correlation with each other were selected as minimum data set (MDS) indicators (Sinha et al., 2009). Non-linear scoring functions were used to transform the MDS indicators into scores (S) ranging from 0 to 1 (Bastida et al., 2006). Two kinds of sigmoidal curves were used to obtain the most suitable shape for each proposed MDS indicator. The "more is better curve" for indicators that positively influence soil quality by higher values, and the "less is better curve" for indicators that negatively influence soil quality by higher values (Armenise et al., 2013).

$$S = \frac{a}{1 + \left(\frac{x}{x_0}\right)^b} \tag{12}$$

where S is the MDS indicator score, a is the maximum value reached by the function (in the present case, $a = 1$), x is the observed value of the indicator, x_0 is the mean value of the indicator among different land uses, and b is the slope value of the equation ($b = -2.5$ for the "more is better curve", $b = 2.5$ for the "less is better curve").

A weight was assigned to each MDS indicator based on the PCA outcomes, which equalled the percentage of the variance (%) explained in the total dataset under a given PC. Then, the scores and weights of the MDS indicators were combined into an overall SQI (Zhang et al., 2011). The higher SQI value indicates better soil quality or a superior performance of soil functions.

$$SQI = \sum_{i=1}^{n} W_i S_i \tag{13}$$

where W_i is the weight of an MDS indicator and S_i is the score of an MDS indicator.

2.5. Statistical analysis

The effects of land use and soil depth on the soil physical and chemical properties were determined by a general linear model test and one-way analysis of variance (ANOVA). The differences among the means were compared by Duncan's multiple range test ($p < 0.05$). The relationships between WSMAN and SQI were determined by linear regression. All statistical analyses were carried out using IBM SPSS 19.0 and R software 3.5.2.

3. Results

3.1. Soil physical properties

The RT and RD soils were classified as having a clay texture, with approximately 25% sand, 30% silt, and 45% clay, whereas RM and RF soils had a loamy clay texture with approximately 30% sand, 35% silt, and 35% clay (Table s1). The soil BD was significantly higher in the RM than in the other treatments at each sampled depth ($p < 0.05$, Table 1). The SM and soil porosities (TP, CP, and NP) did not show much variance with the soil depths in agroecosystems (RM, RT and RD) (Fig. 2a, c and e). While in the RF, the SM, TP and CP were significantly higher at 0–5 cm depth compared to those at 5–15 cm and 15–30 cm depths ($p < 0.05$). The SM was significantly lower in the RM when compared with the other treatments at each soil depth ($p < 0.05$). The soil porosities were in the decreasing order of RF ≥ RD ≥ RT ≥ RM at 0–5 cm depth, while no significant difference occurred between the treatments at 5–15 cm depth. The soil water holding capacities (SWC, CHC and FC) decreased with increasing soil depth, though the changes were not significant in the RM and RT (Fig. 2b, d and f). At 0–5 cm depth, the SWC and CHC were in the order of RF > RD ≥ RT ≥ RM. However, the agroforestry practices (RT and RD) showed better soil water holding capacities than did the other treatments in deeper soil (5–30 cm), especially at 15–30 cm depth. Along the whole 30 cm soil layer, the RM had a significantly lower SWC level than other land uses ($p < 0.05$, Table 2).

The WSMA and MWD of the surface soil (0–5 cm) were lower than the deeper soil (5–30 cm) in the RM, while were higher than the deeper soil in other land uses. The WSMA was in the decreasing order of RF ≥ RD > RT > RM at 0–5 cm depth. Among the four treatments, the WSMA was highest in the RD at 5–15 cm and 15–30 cm depths (Fig. 3a). The MWD was in the decreasing order of RD ≥ RF > RT ≥ RM at 0–5 cm depth, while it was in the decreasing order of RD > RT ≥ RM > RF at 5–15 cm and 15–30 cm depths (Fig. 3b).

3.2. Soil chemical properties

Soils in this region were generally acidic, with pH values ranging from 4.77 to 5.43 (Table 1). The soil EC ranged from 91.00 μs cm^{-1} to 241.67 μs cm^{-1} and showed a downward trend with increasing soil depth in all land use treatments. The C/N ratio ranged from 8.03 to 9.34 among the four treatments, and the highest values were found in the surface soil layer (0–5 cm). The SOC was significantly affected by both land use and soil depth ($p < 0.05$, Table s2). A decreasing trend was found in SOC with increasing soil depth (Fig. 3c). Among the four treatments, the RM had the lowest SOC content irrespective of each soil depth and the lowest SOC stock along the 0–30 cm soil layer (Table 2). However, among the four treatments, RF had the highest SOC level at 0–5 cm depth, while the agroforestry treatments had the highest SOC levels at 5–15 cm and 15–30 cm depths. Coincidentally, the soil total N

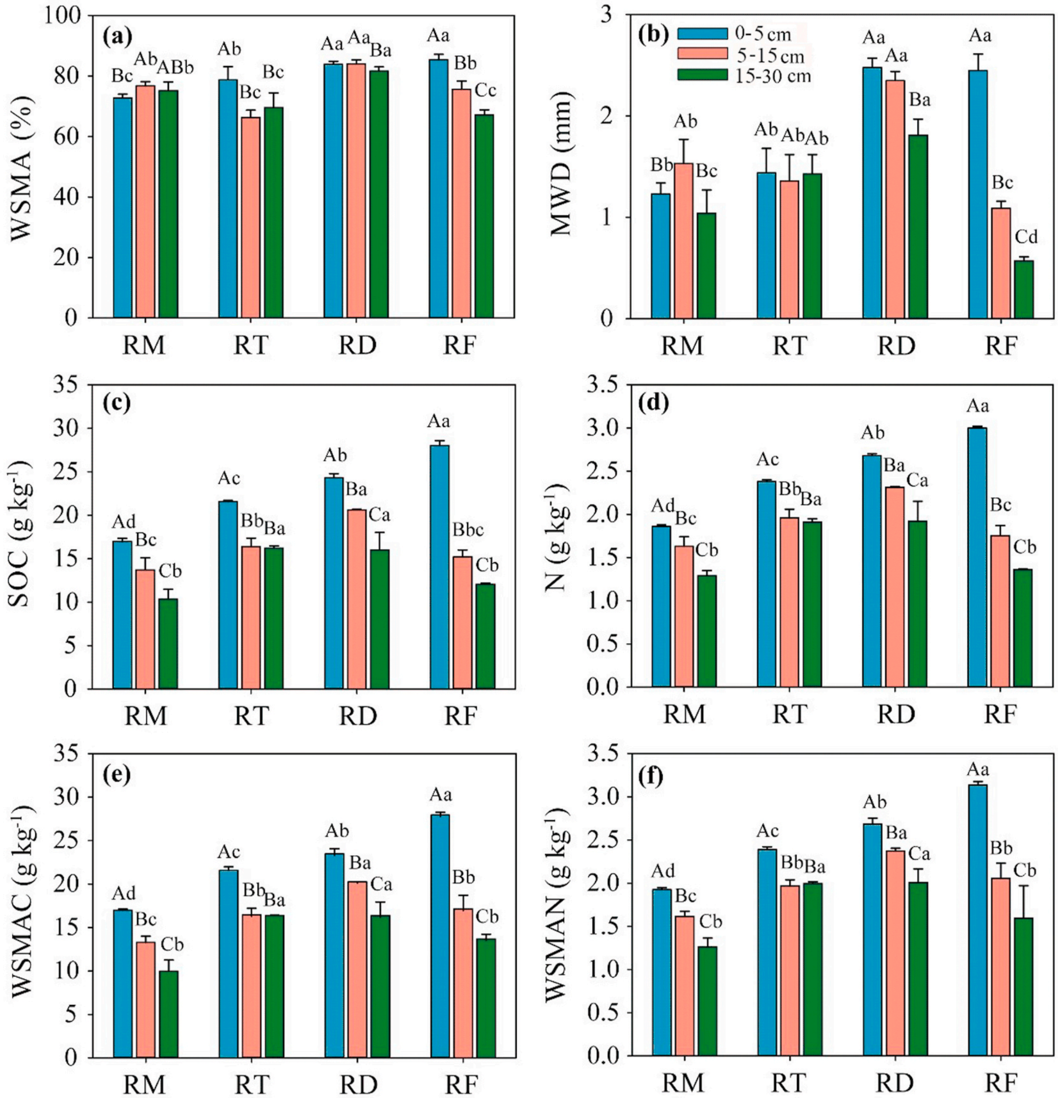

Fig. 3. Soil water-stable macroaggregate (WSMA), mean weight diameter (MWD), soil organic carbon (SOC), total nitrogen (N), water-stable macroaggregate carbon (WSMAC) and water-stable macroaggregate nitrogen (WSMAN) among different land uses. RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. Results are given as means ± standard deviation (n = 4). For each land use, bars with different upper-case letters indicate significant differences among soil depths; for each soil depth, bars with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

content, WSMAC, and WSMAN followed the same trend across the soil depths and treatments observed for SOC (Fig. 3c, d, e and f). The concentration and stock of total K, P, Ca and Mg varied significantly among different treatments ($p < 0.05$, Fig. 4 and Table 2). At each measured depth and along the 30 cm soil layer, the K was in the order of RD > RM ≥ RT > RF, the P was in the order of RD > RF > RT > RM, and the Mg was in the order of RF > RD > RT > RM. At each measured depth, the Ca concentration was found significantly higher in the RD than in the RM ($p < 0.05$). The Ca stock along the 0–30 cm soil layer was in the decreasing order of RD > RF > RT > RM.

3.3. Soil quality index

The first four PCs were kept representing the original variability of the whole data set, which cumulatively explained more than 90% of the total variation. Nine highly weighted parameters were found in PC1 (WSMAN, CHC, N, etc.), one was found in PC2 (K), two were found in PC3 (Sand and Silt + Clay), and one was found PC4 (EC), respectively (Table 3). However, the highly weighted parameters under PC1 were all found to be significantly correlated (Fig. 5). With consideration of the loadings, we selected WSMAN as the MDS indicators under PC1. Likewise, parameter Sand was chosen to be the MDS indicator under PC3. Eventually, one physical parameter (Sand) and three chemical parameters (WSMAN, K and EC) were included as MDS indicators. Simultaneously, the weight of PC1(0.653), PC2 (0.164), PC3 (0.125) and PC4 (0.058) was assigned to WSMAN, K, Sand and EC, respectively (Table 3). The "more is better curve" was chosen for the scoring function because all of the MDS indicators positively influence the soil quality. Consequently, the SQI was computed as follows:

$$SQI = \frac{0.588S_{WSMAN} + 0.148S_K + 0.112S_{Sand} + 0.053S_{EC}}{0.900} = 0.653S_{WSMAN} + 0.164S_K + 0.125S_{Sand} + 0.058S_{EC} \quad (14)$$

Among different treatments, the scores of Sand and EC did not show significant variation, while the score of K varied significantly with the order of RD > RM > RT > RF ($p < 0.05$, Fig. 6). And the score of WSMAN was significantly lower in the RM than the other treatments ($p < 0.05$). Irrespective of treatments and soil depths, the chemical parameters shared a higher proportion of the SQI than the physical parameter

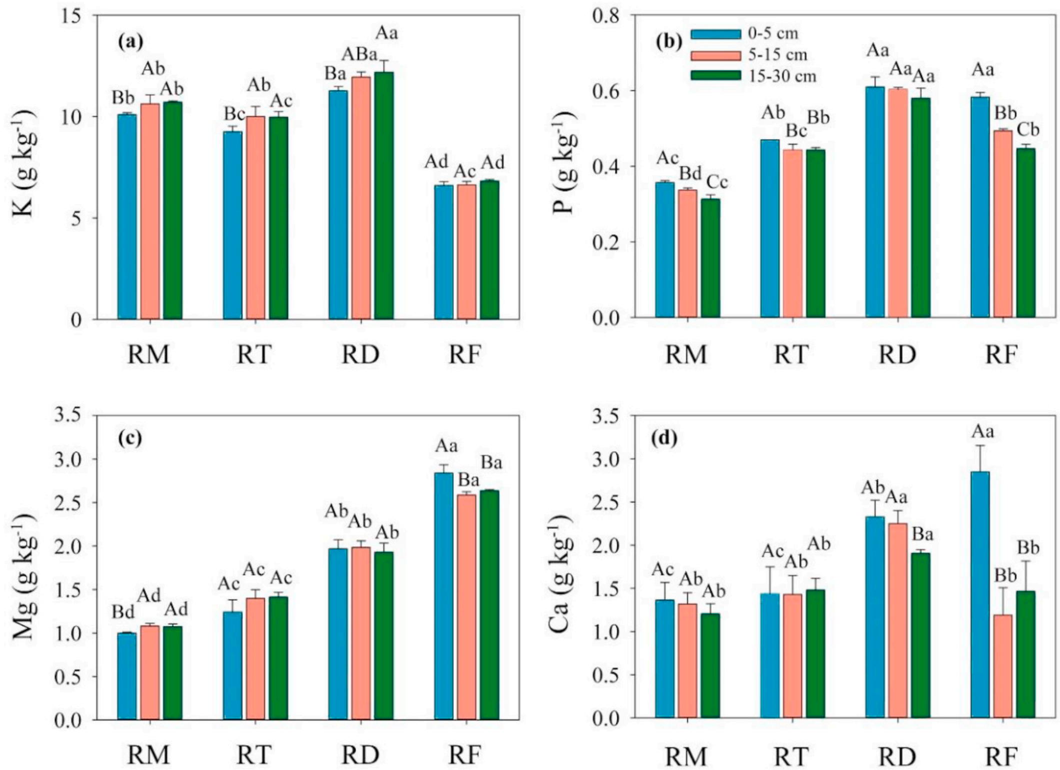

Fig. 4. Soil total potassium (K), total phosphorus (P), total magnesium (Mg), total calcium (Ca) among different uses. RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. Results are given as means ± standard deviation (n = 4). For each land use, bars with different upper-case letters indicate significant differences among soil depths; for each soil depth, bars with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

(Fig. 7). Among the four treatments examined, the SQI value ranged from 0.394 to 0.610. In the case of surface soil layer (0–5 cm), the SQIs were 0.422, 0.473, 0.538 and 0.568 for RM, RT, RD and RF, respectively (Fig. 7a). However, in the deeper soil (5–30 cm), the SQIs of agroforestry practices were higher than the RF. The decreasing order of SQIs were RD (0.583), RF (0.496), RT (0.492) and RM (0.441) at 5–15 cm depth, while the order were RT (0.610), RD (0.553), RF (0.424) and RM (0.394) at 15–30 cm depth (Fig. 7b and c). The average SQIs along the 30 cm soil were in the order of RD (0.558) ≥ RT (0.525) ≥ RF (0.496) ≥ RM (0.419), with a significantly higher values in agroforestry treatments compared to RM ($p < 0.05$, Fig. 7d).

4. Discussion

4.1. Soil physical quality affected by land use change

The present study found a significant effect of land use on soil physical quality irrespective of assessed soil depth (Table s2). These changes in soil physical properties can be ascribed to multiple factors, such as community composition, plant traits, and human activities (Duffera et al., 2007; Xiong et al., 2008). The soil BD was significantly lower in the agroforestry treatments than in the rubber monoculture ($p < 0.05$, Table 1), which was comparable to those found by the previous researchers with a different set of soil variables (Li et al., 2012; Jiang et al., 2017). These results likely occurred due to severe soil compaction in the RM, caused by intensive tramping during the harvesting of agricultural products and herbicide application. However, such kind of soil adversity can be mitigated or reduced by agroforestry practices. As reflected in our results, the soil TP was higher in the RT and RD than the RM (Fig. 2e), which was likely attributed to higher soil animals' activity and abundant plant roots in agroforestry treatments (Shaxson and Barber, 2003). Moreover, the high TP values in the agroforestry treatments were mainly attributed to the increased CP (Fig. 2c and e), which improves the soil capability to hold moisture and thus maintain plant growth (Shi et al., 2005). On the contrary, the high TP in the RF was mainly attributed to the increased NP (Fig. 2a and e), which allows the infiltration of rainfall into the soil and reduces flooding (Shi et al., 2005). In our study, the SM and soil water holding capacities were generally lower in the RM (Table 1, Fig. 2b, d and f). Ayutthaya et al. (2011) demonstrated that large xylem vessels of rubber trees could lead to greater water consumption in the monocultural rubber plantations than the rainforest. In addition, the monocultural structure leads to higher exposure of the surface to solar radiation, thereby, higher direct evapotranspiration in monocultural rubber plantations than the rainforest and the agroforestry (Tan et al., 2011). Therefore, the subsoil water depletes during the dry season in the monocultural rubber plantations, which further contributes to environmental problems such as decreased groundwater, low streamflow, and higher desiccation (Qiu, 2009; Guardiola-Claramonte et al., 2010). However, the multi-layered canopy of agroforestry is conducive to more fog interception, a considerable water source for the plants during the dry season in Xishuangbanna (Xu et al., 2014). Moreover, intercrops can facilitate a cooler environment and reduce soil water loss from evaporation. Therefore, agroforestry practices generally enhance the soil hydrological balance.

Although all study sites were characterized by having a similar soil texture, significantly higher silt + clay distribution (especially the clay proportion) occurred in the agroforestry treatments than in the RM ($p < 0.05$, Table 1 and s1). Since splash and water erosion preferably remove the fine soil particles (Tuo et al., 2018), the higher distribution of silt + clay in agroforestry could be due to the less fine soil particles loss from erosion. The narrow canopy complexity, high throughfall kinetic energy, and scarce ground litter in monocultural rubber plantations can lead to intensive splash erosion (Liu et al., 2015, 2016). While a multistrata canopy in agroforestry can significantly reduce the splash

Table 3
Principal component analysis (PCA) output of the studied soil properties.

Principal component	1	2	3	4
Eigenvalue	13.524	3.395	2.579	1.209
Variance %	58.800	14.759	11.215	5.258
Cumulative %	58.800	73.559	84.774	90.031
BD (g cm^{-3})	−0.857	0.239	0.148	0.186
SM (%)	0.828	0.241	−0.010	−0.167
SWC (%)	**0.965**	−0.182	0.086	−0.030
CHC (%)	**0.979**	−0.060	0.079	−0.080
FC (%)	0.698	0.548	0.285	0.303
NP (%)	0.442	−0.811	0.075	0.244
CP (%)	**0.930**	0.205	0.110	−0.108
TP (%)	**0.951**	−0.085	0.119	−0.015
Sand (%)	−0.171	−0.025	**0.885**	−0.367
Silt + Clay (%)	0.171	0.026	**−0.885**	0.367
pH	0.460	0.722	0.135	−0.109
EC (μs m^{-1})	0.534	−0.012	0.511	**0.651**
WSMA (%)	0.699	0.234	−0.318	0.196
MWD (mm)	0.839	0.354	−0.155	−0.029
SOC (g kg^{-1})	**0.969**	0.068	0.124	0.053
N (g kg^{-1})	**0.974**	0.127	0.096	0.013
WSMAC (g kg^{-1})	**0.969**	−0.054	0.119	0.007
WSMAN (g kg^{-1})	**0.980**	−0.036	0.091	−0.053
C/N ratio	0.728	−0.344	0.073	0.072
P (g kg^{-1})	0.798	−0.063	−0.418	−0.396
K (g kg^{-1})	−0.114	**0.893**	−0.318	−0.120
Ca (g kg^{-1})	**0.900**	−0.019	−0.200	0.007
Mg (g kg^{-1})	0.485	−0.766	−0.302	−0.223

BD: bulk density, SM: soil moisture, SWC: saturated water capacity, CHC: capillary holding capacity, FC: field capacity, NP: noncapillary porosity, CP: capillary porosity, TP: total porosity, WSMA: water-stable macroaggregate, MWD: mean weight diameter; EC: electrical conductivity, SOC: soil organic carbon, N: total nitrogen, WSMAC: water-stable macroaggregate carbon, WSMAN: water-stable macroaggregate nitrogen, P: total phosphorus, K: total potassium, Ca: total calcium, Mg: total magnesium. Bold-faced numbers are the values for high loading parameters under each PC, bold-underlined numbers are the values for MDS indicators.

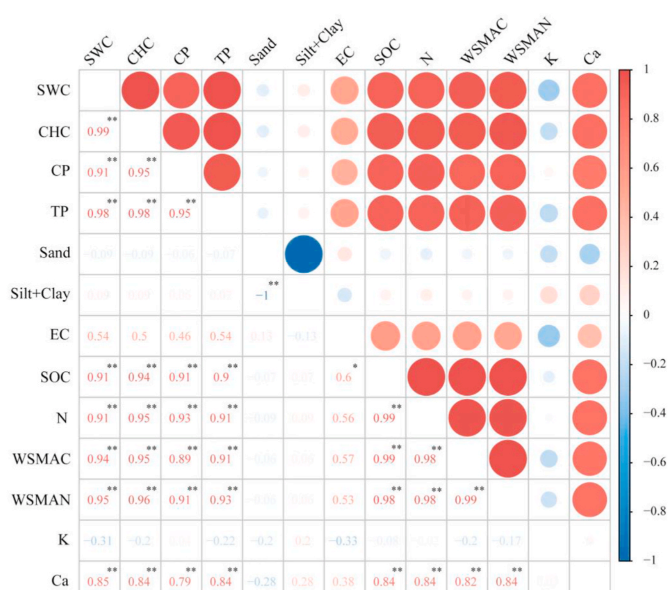

Fig. 5. Pearson's correlation coefficients between highly weighted parameters. SWC: saturated water capacity, CHC: capillary holding capacity, CP: capillary porosity, TP: total porosity, EC: electrical conductivity, SOC: soil organic carbon, N: total nitrogen, WSMAC: water-stable macroaggregate carbon, WSMAN: water-stable macroaggregate nitrogen, K: total potassium, Ca: total calcium. *Correlation is significant at the $p < 0.05$ level; **Correlation is significant at the $p < 0.01$ level.

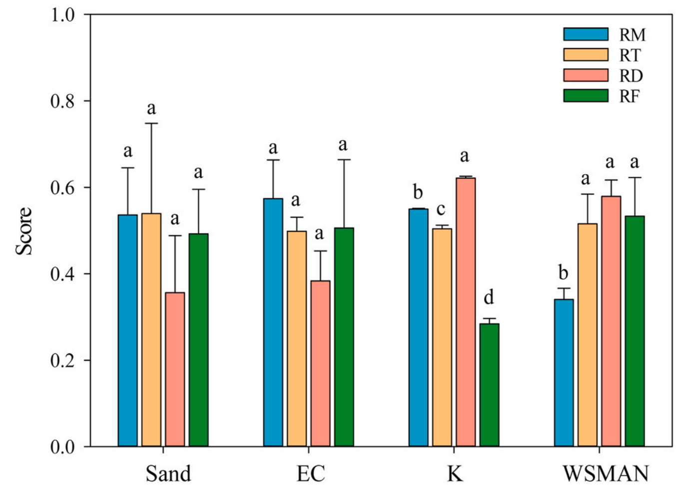

Fig. 6. Scores of MDS indicators for different uses. Results are given as means ± standard deviation (n = 3). RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. EC: electrical conductivity, K: total potassium, WSMAN: water-stable macroaggregate nitrogen. Bars with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

erosion compared to a uniform canopy (Liu et al., 2016). Similarly, throughfall kinetic energy decreases with increasing canopy complexity (or leaf area index) and is generally lower in agroforestry systems than in monocultural systems (Liu et al., 2018). The rubber trees' leaf litter has a high decomposition rate, low water affinity, and weak capacity to intercept rainfall, which further leads to substantial surface runoff (Zhu et al., 2018). In contrast, the thick and intact litter layer in agroforestry is an important soil protection agent that can abstract runoff (Liu et al., 2017).

Aggregate stability is an important soil property that influences various ecosystem functions, such as carbon sequestering, nutrient retention, and erosion control (Wang et al., 2001; Barthès and Roose, 2002). The aggregate stability was significantly higher in the agroforestry (particularly RD) than RM (Fig. 3a and b). These results are consistent with those found by Gupta et al. (2009) and Gama-Rodrigues et al. (2010) in *Populus deltoides* (Bartr. ex Marsh) and *T. cacao* plantations, respectively. Some factors, such as abundant fine roots, rich soil binding agents, thick litter layer, and low sub-canopy, could improve soil aggregation in the agroforestry. Fine roots in agroforestry generally promote macroaggregate formation through compressing soil particles and binding organic matter (Erktan et al., 2016). Soil binding agents, such as clay and root exudates, combine the microaggregates into macroaggregates (Morel et al., 1991; Erktan et al., 2016). Thick litter layer and sub-canopy act as ground mulch, preventing the macroaggregates' breakdown by raindrops (Zuazo et al., 2009; Gupta et al., 2009).

4.2. Soil chemical quality affected by land use change

It is widely recognized that converting primary forests into agricultural lands usually leads to rapid nutrient leaching of soils (Trumbore et al., 2015). The low C/N ratios in these study sites (ranging from 8.03 to 9.34) were likely attributed to the high decomposition rate under moist tropical climate (Callesen et al., 2007). A study conducted in the sub-tropical climate reported that the SOC and total N accumulation decreased by 48.20% and 54.10%, respectively, in a 40-year-old rubber plantation (Cheng et al., 2007). Our results are comparable to this study, with SOC and total N depleted by 25.72% and 21.67%, respectively, after 26 years of rubber monoculture cultivation. However, agroforestry practices significantly improved the SOC and total N contents of soil ($p < 0.05$). The SOC increased by 31.99% in RT and 48.43% in RD, the total

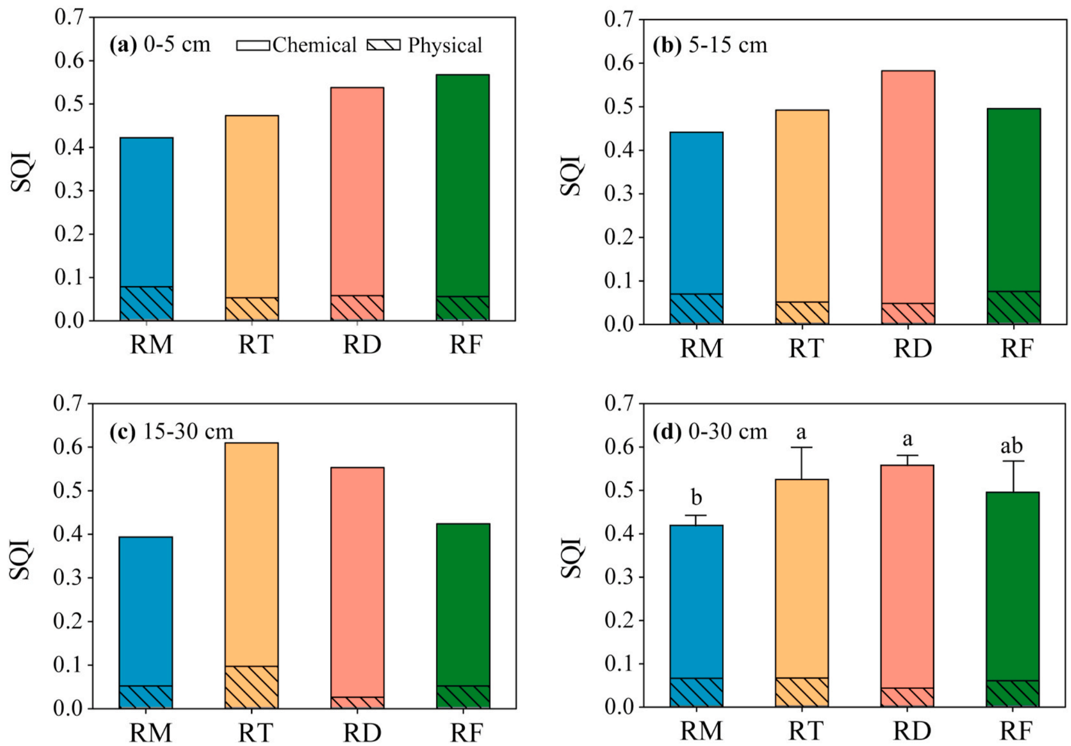

Fig. 7. Soil quality index (SQI) values for different land uses. RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. For each use, the lower density and upper smooth areas are the contribution of soil physical and chemical properties to the SQI, respectively. Results in (a), (b), and (c) are estimated SQI values, and results in (d) are given as means ± standard deviation (n = 3). Bars with different lower-case letters indicate significant differences among land uses ($p < 0.05$).

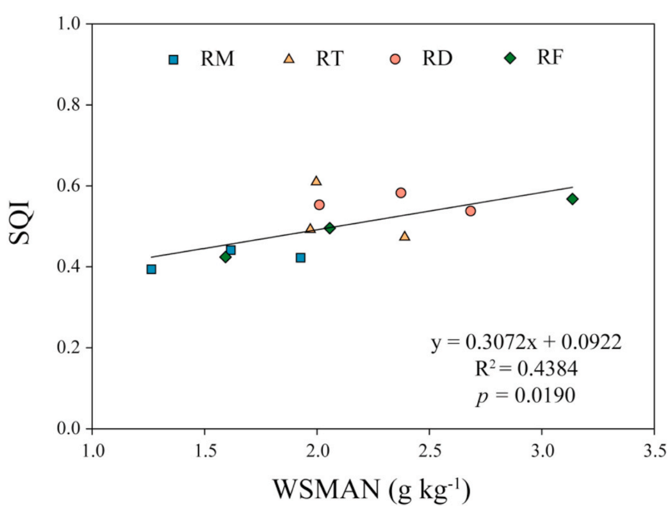

Fig. 8. The relationship between the water-stable macroaggregate nitrogen (WSMAN) and soil quality index (SQI) for the 0–30 cm layer of the four land uses. RM: rubber monoculture, RT: rubber–*Camellia sinensis* agroforestry, RD: rubber–*Dracaena cochinchinensis* agroforestry and RF: primary rainforest. Symbols represent the means of each soil depth (n = 3).

N increased by 30.82% in RT and 44.65% in RD, respectively (Fig. 3c and d). Cong et al. (2015) also found that agroforestry practices can enhance agricultural lands' potential to sequester C and N. Generally, the increase of organic material (e.g., litter, root exudates and animal residues), a specific dose of fertilizer application, and improvement of soil structure can increase the SOC and total N accumulations in agroforestry systems. The higher organic matter availability could benefit soil microbial functional diversity, including those related to the carbon and nitrogen cycle (Wang et al., 2020). Besides, N–P–K compound commercial fertilizer (0.1 kg N per tree per year) and organic fertilizer (0.08 kg C per tree per year) application is an additional medium of soil carbon and nitrogen income. Moreover, the SOC and total N increase is usually accompanied by soil structure improvement (e.g., decreased bulk density, increased water holding capacity and increased soil porosity) (Zhang et al., 2011). Hoorman et al. (2011) demonstrated that soils with lower bulk densities generally had better soil aeration and drainage, which can further reduce the nitrogen loss from denitrification as gas and from leaching as nitrate. The high intensity of rainfall (1500–1800 mm annually, and 87% occurs during the rainy season) can cause a large amount of runoff and soil loss in the present tropical region (Zakari et al., 2020), and is usually accompanied by soil carbon and nitrogen leaching. But, the fine roots in agroforestry can capture the erosion-related leached carbon and nitrogen from the deep soils (Zhu et al., 2019b), which could explain the larger increase of SOC and total N at 5–30 cm depth.

The simultaneous increase in SOC and total N proved the synergy between C and N sequestration (Cong et al., 2015). In other words, C sequestration can enhance N sequestration, and vice versa. We observed that WSMAC and WSMAN showed a similar pattern as SOC and total N among different land uses, which indicates a strong link between these nutrients and soil aggregation. It means a stable soil structure with high aggregation is essential for building and maintaining carbon and nitrogen pools (Bronick and Lal, 2005; O'Brien et al., 2013; Cong et al., 2015).

In a rubber plantation transformed from a natural rainforest, without adding fertilizers, the consumption periods were found to be only 825, 329, 94, and 65 years for P, N, K, and Mg, respectively (Cheng et al., 2007). We found lower total P, Ca, and Mg in the RM than the RF by 33.55%, 29.40%, and 60.97%, respectively (Fig. 4). However, the total P, K, Ca, and Mg in the agroforestry treatments (RT and RD) were on average increased by 55.45%, 2.77%, 39.33%, and 57.62%, respectively, compared to those in the RM. On the one hand, the multistrata canopy and constant ground cover in the agroforestry play essential roles in soil and water conservation, which can reduce the sediment-bound nutrient loss (Liu et al., 2016; Zhu et al., 2019b). On the other hand, the extensive root systems under agroforestry prevent nutrient leaching by driving soil aggregation and capturing the leached nutrients from the deep soils (Zhu et al., 2019b).

4.3. Implications of land management based on soil quality assessment

The SQI can reflect the soil degradation level only under the same conditions, such as climate and substrate (Bastida et al., 2006), which means the SQI in our study is specifically useful for the present tropical region. The SQI assessment is often conducted to understand the soil status from agricultural and environmental perspectives (Bastida et al., 2006; Zhang et al., 2011). Therefore, our SQI assessment was focused on soil structure and fertility. Our 23 physical and chemical parameters are sensitive to environmental stresses (Karlen et al., 1997) and represent various soil functions (Table s2), including structural stabilization, hydrological movement, and nutrient cycling. The four MDS indicators (WSMAN, K, Sand and EC) derived based on the PCA results (Table 3) were mostly similar to those found in previous researches (Erkossa et al., 2007; Bastida et al., 2008; Juhos et al., 2019), while WSMAN is uniquely identified in this study. It was demonstrated that total N is a good proxy for assessing soil quality in tropical regions (Ruiz et al., 2020; Kurmangozhinov et al., 2020). Our results suggest that soil aggregate stability coupled with total N (WSMAN) is a more discriminating and dominant indicator compared to other indicators (Table 3). Moreover, the linear regression revealed a positive correlation between WSMAN and SQI ($p < 0.05$, Fig. 8). Therefore, we strongly recommend adopting WSMAN for the evaluation of soil quality. Additionally, considering the biological processes are closely related to soil nutrient cycling, the biological parameters (e.g., microbial biomass C, basal respiration and soil enzymes) are suggested to include in further research.

Our results demonstrated the conversion of natural rainforest (RF) to RM lead to a decline of SQI by 15.50%, which was mainly caused by soil nutrients loss (18.90%) (Fig. 7d). This result implies that rubber monoculture has led to a certain degree of land degradation in the present region. We also found that the best soil physical quality appeared in the RF compared to the agroecosystems (RM, RT and RD), probably due to the absence of human intervention into the natural ecosystem. This result is consistent with the previous finding that conventional agroecosystems are incapable of maintaining the same soil physical quality level as natural ecosystems (Silva et al., 2011). However, the SQI was significantly enhanced by agroforestry practices (25.30% by RT, 33.10% by RD) compared to the RM ($p < 0.05$). The improvement in soil nutrients (K and WSMAN) is the major reason for higher SQIs in the agroforestry treatments. In our study, the chemical parameters contributed more to the SQI than the physical parameters (Fig. 7), suggesting the importance of nutrient management for better soil quality in agroecosystems.

As the agroforestry practices are increasingly viewed as environmental restorative prescriptions, rubber-based agroforestry practices have been advocated for decades in tropical China. However, without proper scientific guidance, the selection of intercropping species remains chaotic. The RD performed better in recovering soil quality than the RT (Fig. 7d), indicating *D. cochinchinensis* is more suitable than *C. sinensis* for intercropping with rubber trees. This is likely due to the strong environmental adaptability, high canopy closure, and deep roots of *D. cochinchinensis* (Mulyono et al., 2019).

5. Conclusion

The agroforestry practices significantly influenced the soil quality of rubber plantations in tropical China. The SQI was established for three categories of rubber plantations and a primary rainforest based on PCA analysis. One soil physical parameter (Sand) and three soil chemical parameters (WSMAN, K and EC) were selected as MDS indicators for computing SQI. The SQI value was 15.50% lower in the RM compared to the RF, indicating rubber monoculture cultivation in the present tropical region has led to a certain degree of land degradation. However, the SQI was significantly higher in the agroforestry than the RM, demonstrating that agroforestry practices are relatively suitable approaches for recovering the soil quality in degraded agricultural lands. Moreover, the chemical parameters contributed more to the SQI than the physical parameters, indicating the importance of nutrient management for restoring soil quality in agroecosystems. Considering the greater potential in soil quality restoration of RD (33.10%) versus RT (25.30%), the *D. cochinchinensis* intercropping could be preferred over *C. sinensis*. Overall, our findings provide a reference for the restoration of soil quality in the rubber-dominated agroecosystem and future agroforestry researches.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2021.112147.

Author credit

Xin Zou: Conceptualization, Methodology, Writing – original draft. Xiai Zhu: Editing. Peng Zhu: Review. Ashutosh Kumar Singh: Writing – review & editing. Sissou Zakari: Formal analysis. Bin Yang: Writing – Visualization. Chunfeng Chen: Project administration, Resources. Wenjie Liu: Funding acquisition; Supervision; Resources.

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