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Changes in soil labile and recalcitrant carbon pools after land-use change in a semi-arid agro-pastoral ecotone in Central Asia

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A B S T R A C T

Soil organic matter is a heterogeneous and complex entity that consists of a diverse range of compounds. However, the responses of soil organic carbon (SOC) fractions with different biochemical stabilities to land-use change are inadequately addressed. In this study, soil samples (0–60 cm) were collected with a 10 cm interval from grassland (GS), cropland (CS), woodland (WS), and orchard (OS) using paired-site approach in a typical semi-arid agro-pastoral ecotone in the Ili River Valley, Central Asia, to: (1) clarify the vertical changes in soil labile (LPI and LPII) and recalcitrant C pools (RP) after GS conversion to CS and CS conversion to WS and OS; and to (2) evaluate the impact of land-use change on SOC stability. The results indicated that LPI stocks in topsoil (0–30 cm) and subsoil (30–60 cm) showed opposite responses to land-use change. In contrast, LPII and RP stocks in both soil layers significantly decreased after conversion of GS to CS, and significantly increased after CS afforestation. These results demonstrated that RP in subsoils could also be altered by land-use change. In general, conversion from GS to CS decreased the recalcitrance index of SOC (RI_{SOC}), which increased after conversion from CS to WS. The results implied that cultivation decreased the stability of SOC, causing the depletion of SOC stock, whereas CS conversion to WS enhanced the stability of SOC, promoting SOC sequestration. The negative correlations between RI_{SOC}, pH, and electrical conductivity ($EC_{1:5}$) suggested that soil pH and salinity were potential indicators reflecting the biochemical recalcitrance of SOC. Since both soil pH and $EC_{1:5}$ showed decreasing trends after conversion from CS to WS and OS, the results suggested that afforestation on CS contributed to mitigate soil salinization while promoting SOC sequestration in this semi-arid agro-pastoral ecotone.

1. Introduction

As the largest terrestrial reservoir of carbon (C), soils contain more than 70% of the terrestrial organic C (Parras-Alcántara et al., 2015). The amount is greater than the combined organic C mass stored in living biomass and the atmosphere (Köchy et al., 2015). Therefore, soil organic C (SOC) plays a significant role in the global C cycle. In general, depletion of SOC pool leads to an increase in atmospheric carbon dioxide (CO_2) concentration and a decrease in soil quality, which are key driving forces of global warming and land degradation, respectively (Lal, 2010; Schuur et al., 2015). In contrast, SOC sequestration is a promising way to mitigate global warming and achieve sustainable agriculture (Stockmann et al., 2013). A better understanding of SOC dynamic and its influencing factors is thus necessary for better C management to ease the simultaneous crises of climate change and food shortage (Lal, 2010; Stockmann et al., 2013).

Land-use change can induce changes in plant species and land management practices, both of which are crucial factors determining the balance between gains and losses of soil organic matter (Muñoz-Rojas et al., 2015; Liu et al., 2017). Hence, SOC pool can be considerably influenced by land-use change (Deng et al., 2014; Yu et al., 2017). Globally, SOC loss caused by land-use change has varied from 45 to 114 Pg C (79.5 Pg C on average, 1 Pg = 10^{15} g) during 1870–2014 (de Moraes Sá et al., 2017), mainly due to conversion from natural lands to croplands (Guo and Gifford, 2002). By contrast, afforestation on croplands often contributes to SOC sequestration (Qiu et al., 2015; Liu et al., 2018b). Although the effect of land-use change on total SOC has been well documented (Bae and Ryu, 2015; Muñoz-Rojas et al.,

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2015; Liu et al., 2018a), information on the SOC fractions with different biochemical stabilities in response to land-use change is still limited. In general, soil organic matter is a heterogeneous and complex entity that consists of a diverse range of compounds with varying functions and bioavailabilities (Rovira and Vallejo, 2007; Paul, 2016). According to acid hydrolysis technique, SOC can be divided into two labile C pools (LPI and LPII) and one recalcitrant C pool (RP) (Rovira and Vallejo, 2002). LPI mainly consists of non-cellulosic polysaccharides, which originate from either plant or soil microorganisms. The main component of LPII is plant-derived cellulose (Rovira and Vallejo, 2002; Ding et al., 2012). Such compounds have high bioavailability and are sensitive to environmental changes (Ding et al., 2012). Consequently, the labile C fraction is usually identified as an early indicator of SOC dynamic after land-use change (Sheng et al., 2015; da Silva Oliveira et al., 2017). In contrast, RP plays an important role in determining long-term SOC sequestration because the compounds (e.g. lignins) in this pool are highly resistant to biodegradation (Rovira and Vallejo, 2002, Ding et al., 2012). The degree of biochemical recalcitrance of SOC can be changed by the alteration in the size of labile and recalcitrant C pools, and then influences the longevity of SOC sequestration (Rovira and Vallejo, 2007; Tang and Li, 2013). Therefore, assessing the responses of soil labile and recalcitrant C fractions to land-use change can provide valuable information for better understanding the role of land-use change in the global C budget (Yu et al., 2017).

Up to now, much attention has been paid to SOC dynamic in topsoil ($\leq$30 cm), mainly due to the fact that topsoil stores a large quantity of SOC that can be easily affected by external disturbances (Jiang et al., 2014; Wang et al., 2014). SOC in subsoil ($>$ 30 cm) receives less attention because this part of SOC is often assumed to be old, stable, inert, and insensitive to environmental changes (Lorenz and Lal, 2005; Rumpel and Kögel-Knabner, 2011). Nevertheless, there is a growing evidence that SOC in subsoil can be considerably affected by environmental changes such as fresh C inputs, fertilizer application, and warming (Fontaine et al., 2007; Xu et al., 2010; Srinivasarao et al., 2014). Recent studies reported that SOC pool in both topsoil and subsoil could be altered by land-use change, and the effects were different between the two soil layers (Mobley et al., 2015; Liu et al., 2018a). However, studies regarding the impacts of land-use change on fractions and biochemical recalcitrance of SOC in subsoil remain scarce.

Globally, more than 27% of SOC stores in arid and semi-arid regions, which cover about one third of the whole land surface and support over 40% of the human population (Saco et al., 2007; Han et al., 2016). Empirical evidence has indicated that ecosystems in these regions are important sinks for atmospheric CO_2 (Lal, 2004; Ahlström et al., 2015; Li et al., 2015). However, drastic land-use changes in arid and semi-arid regions have formed a large area of agro-pastoral ecotone, where SOC pool experiences frequent changes (Chuluun and Ojima, 2002; Liu et al., 2018a). For example, Sommer and de Pauw (2011) estimated that historically, conversion from natural lands to agricultural land-uses has induced a reduction of 3.9% in total SOC stock (0–30 cm) in Central Asia, which is located in the heart of the Eurasian continent and plays an important role in trading C credit (Hamidov et al., 2016). Unfortunately, the impacts of land-use change on fractions and stability of SOC in this area remain unclear. In this study, an investigation was carried out in a typical semi-arid agro-pastoral ecotone in the Ili River Valley, Central Asia, to clarify the effects of land-use change (grassland conversion to cropland and cropland conversion to woodland/orchard) on labile and recalcitrant C fractions and biochemical recalcitrance of SOC along the 0–60 cm soil profile. The following hypotheses were tested: (1) concentrations of all SOC fractions and biochemical recalcitrance of SOC decrease after grassland conversion to cropland, and increase after cropland afforestation; (2) recalcitrant C fraction in subsoil can be also influenced by land-use change. The results would be helpful for policy makers to promote SOC sequestration and restore soil quality in arid and semi-arid regions.

2. Materials and methods

2.1. Study area

The study area is situated in the Ili River Valley, Central Asia (43°24–′43°25′N, 82°49′–82°50′E, and 800–857 m a.s.l.), with a total area of approximately 50 ha. The area has a temperate semi-arid continental climate with an average annual temperature of 9.4 °C. The average annual precipitation, potential evaporation, and relative humidity are 433 mm, 1228 mm, and 65%, respectively (Liu et al., 2017). The soils are Mollisols with a silt loam texture (clay, 9%; silt, 61%; sand, 30%) (Soil Survey Staff, 2014), developing from loess-like materials. This soil type typically has a dark-colored A horizon with high organic matter content, which results from the long-term addition of organic materials derived from grass roots (Liu et al., 2012). The original land-use type in this area is grassland (GS), which was fenced by local farmers for intensive sheep grazing. The dominant plant species in GS are Syrian rue (*Peganum harmala* L.) and *Ceratocarpus arenarius* L.. In 1980, a part of this GS was transformed to cropland (CS), a portion of which was converted to woodland (WS) and orchard (OS) under the "Green for Grain" project in 2004. Maize (*Zea mays* L.) and wheat (*Triticum aestivum* L.) are major crops that being cultivated in CS. The above ground biomass of crops are removed during harvest, after which soils are plowed to a depth of 30–35 cm. The planted tree species in WS and OS are poplar (*Populus alba* L.) and apricot (*Prunus armeniaca* L.), respectively (Liu et al., 2018a). Inorganic fertilizers are applied annually in CS, WS, and OS, and manure is only applied in CS and OS every 3–5 years.

2.2. Soil sampling and analysis

The detailed sampling procedure is available in Liu et al. (2018a). In brief, four replicate plots (10 × 10 m) were selected in each land-use type in April 2017. In each plot, soils were randomly collected from six subplots. The maximum sampling depth and sampling interval were 60 cm and 10 cm, respectively. Soil samples collected in the same layer were thoroughly mixed to get a composite soil sample in each plot. Since the A horizon in the study area is about 30 cm, the 0–30 cm and 30–60 cm soil layers were defined as topsoil and subsoil, respectively. Soil samples were air-dried and crushed to determine soil properties after transporting to the laboratory. A pH meter (SevenEasy, Mettler-Toledo, Switzerland) and an electrical conductivity meter (DDSJ-308A, Rex, China) were used for determining soil pH and electrical conductivity ($EC_{1:5}$) at a 1:5 (*w*:*v*) soil–water ratio, respectively (Liu et al., 2018a). Soil bulk density (BD) was measured using the volumetric ring method (Zhang et al., 2014). The two-step acid hydrolysis approach was used for separating different SOC fractions (Rovira and Vallejo, 2002). Using this approach, the SOC pool was divided into LPI, LPII, and RP. Since the major compounds of RP are lignins, fats, waxes, resins, suberins, and humic substances, all of which are highly resistant to biodegradation (Rovira and Vallejo, 2002; Ding et al., 2012), a high proportion of RP in total SOC thus indicates that SOC has a high degree of biochemical recalcitrance (Rovira and Vallejo, 2007). The concentrations of LPI and LPII were measured using a TOC analyzer (model 1030, OI Analytical, USA) (Liu et al. 2017). The H_2SO_4-$K_2Cr_2O_7$ oxidation method was used for the determination of RP concentration (Jiang et al., 2014; Liu et al. 2017). Total SOC concentration was obtained by adding the concentrations of LPI, LPII, and RP together (Rovira and Vallejo, 2002).

2.3. Calculation

The equivalent soil mass method proposed by Ellert and Bettany (1995) was used to calculate the stocks of total SOC and each SOC fraction in this study. The following equation was used to obtain the recalcitrance index of SOC (RI_{SOC}) (Rovira and Vallejo, 2002):

$$RI_{SOC} = (Con_{RP}/Con_{total\ SOC}) \times 100 \quad (1)$$

where RI_{SOC} (%) is the recalcitrance index of SOC; Con_{RP} (g kg^{-1}) and $Con_{total\ SOC}$ (g kg^{-1}) are the concentration of RP and total SOC, respectively.

2.4. Statistical analysis

The differences in soil properties including BD, pH, $EC_{1:5}$, concentrations and stocks of total SOC and each SOC fraction, and RI_{SOC} among land-use types or soil layers were examined using one-way analysis of variance (ANOVA), followed by the least significant difference (LSD) test. Two-way ANOVA was used to evaluate the effects of land-use type, soil depth, as well as their interactions on soil properties. The differences in stock of each SOC fraction and RI_{SOC} between the previous and transformed land-use types in topsoil and subsoil were tested using independent *t* test. The relationships between LPI, LPII, RP, and total SOC and those between soil pH, $EC_{1:5}$, and RI_{SOC} were examined using linear regression analysis. All data were checked for normality and homogeneity of variance, and if necessary, were transformed using the log-transformation (base 10). All the statistical analyses were performed using SPSS software, version 22.0 (SPSS Inc., USA). Figures were drawn using OriginPro software, version 9.0 (Originlab Inc., USA).

3. Results

3.1. Vertical characteristics of soil BD, pH, and $EC_{1:5}$

Soil BD of all land-use types and soil pH of GS, CS, and OS generally fluctuated along the soil profile, whereas soil pH of WS significantly increased with increases in soil depth ($P < 0.05$). The general ranges of soil BD and pH across the four land-use types were 1.25–1.63 g cm^{-3} and 8.42–9.41, respectively. In most soil layers, changes in soil BD and pH after GS conversion to CS and CS conversion to OS were not significant ($P > 0.05$). In contrast, BD significantly increased and pH significantly decreased after conversion of CS to WS in most soil layers ($P < 0.05$) (Fig. 1a and b). As presented in Fig. 1c, increases in soil $EC_{1:5}$ with soil depth were detected for all land-use types, leading to significantly higher $EC_{1:5}$ in the 50–60 cm soil layer than those in the 0–10 cm soil layer ($P < 0.05$). In the 0–30 cm and 50–60 cm soil layers, $EC_{1:5}$ significantly increased by 49.0–82.6% after conversion from GS to CS, and significantly decreased by 41.9–67.4% after CS conversion to WS. However, land-use change had little impact on $EC_{1:5}$ in the 30–50 cm soil layers. According to the FAO classification used for soil salinity assessment (Abrol et al., 1988), it was observed that the salinity level of topsoil in GS and WS was non saline, and topsoil in CS and OS had a low salinity. For subsoil, a mild salinity was found in GS, CS, and OS, whereas a low salinity was observed in WS.

As shown in Table 1, land-use type had significant effects on soil BD and $EC_{1:5}$ ($P < 0.001$) but had little effect on soil pH ($P > 0.05$). Soil pH and $EC_{1:5}$ were significantly affected by soil depth ($P < 0.05$), which had little effect on soil BD ($P > 0.05$). By comparison, the interactions of land-use type and soil depth significantly affected soil BD, pH, and $EC_{1:5}$ ($P < 0.001$).

Table 1
Two-way ANOVA results of the effects of land-use type and soil depth on soil properties.

Property	Land-use type		Soil depth		Land-use type × Soil depth	
	F	*P*	*F*	*P*	*F*	*P*
BD	14.510	< 0.001	1.985	0.093	4.060	< 0.001
pH	2.372	0.076	2.981	0.016	3.457	< 0.001
$EC_{1:5}$	9.983	< 0.001	15.140	< 0.001	7.915	< 0.001
Total SOC	17.364	< 0.001	28.550	< 0.001	21.317	< 0.001
LPI	0.765	0.517	2.902	0.018	3.793	< 0.001
LPII	13.659	< 0.001	31.737	< 0.001	23.940	< 0.001
RP	20.372	< 0.001	24.342	< 0.001	17.204	< 0.001
RI_{SOC}	16.550	< 0.001	3.162	0.011	3.470	< 0.001
$S_{total\ SOC}$	22.412	< 0.001	39.723	< 0.001	26.858	< 0.001
S_{LPI}	1.353	0.263	13.753	< 0.001	8.217	< 0.001
S_{LPII}	17.686	< 0.001	41.540	< 0.001	29.404	< 0.001
S_{RP}	25.217	< 0.001	32.454	< 0.001	21.487	< 0.001

Note: BD indicate bulk density; $EC_{1:5}$ indicate electrical conductivity; Total SOC, LPI, LPII, and RP indicate concentration of total soil organic carbon, labile carbon pool I, labile carbon pool II, and recalcitrant carbon pool, respectively; RI_{SOC} indicate recalcitrance index of soil organic carbon; $S_{Total\ SOC}$, S_{LPI}, S_{LPII}, and S_{RP} indicate stock of total soil organic carbon, labile carbon pool I, labile carbon pool II, and recalcitrant carbon pool, respectively.

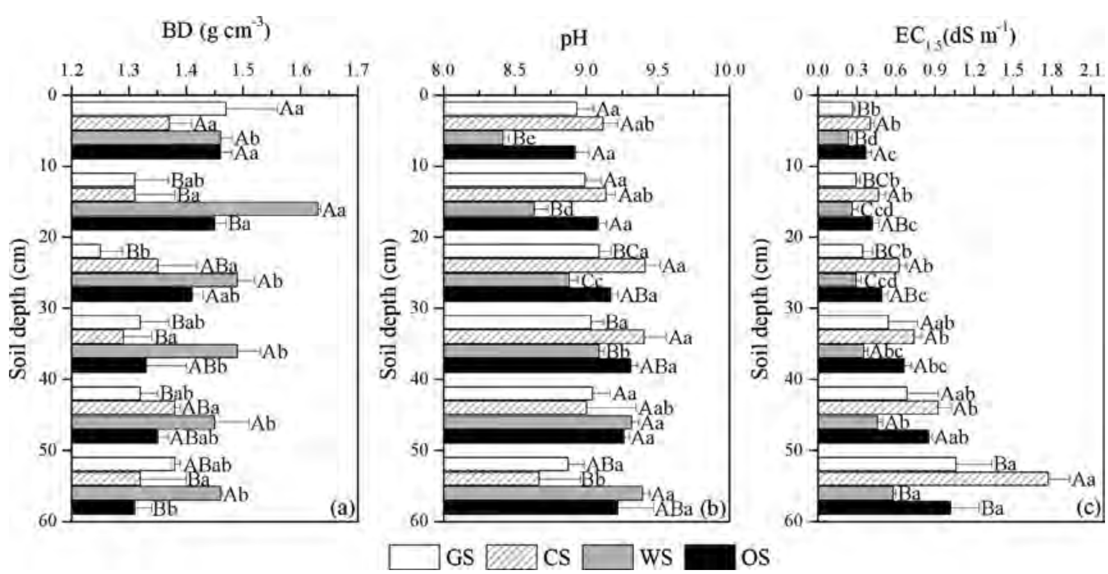

Fig. 1. Profile characteristics of soil bulk density (BD) (a), pH (b), and electrical conductivity ($EC_{1:5}$) (c) in each land-use type. Error bars indicate standard errors of the means (n = 3 for BD and 4 for other soil properties). GS, CS, WS, and OS indicate grassland, cropland, woodland, and orchard, respectively. Different capital letters indicate significant differences among soil layers in a specific land-use type; different lowercase letters indicate significant differences among land-use types in a specific soil layer ($P < 0.05$).

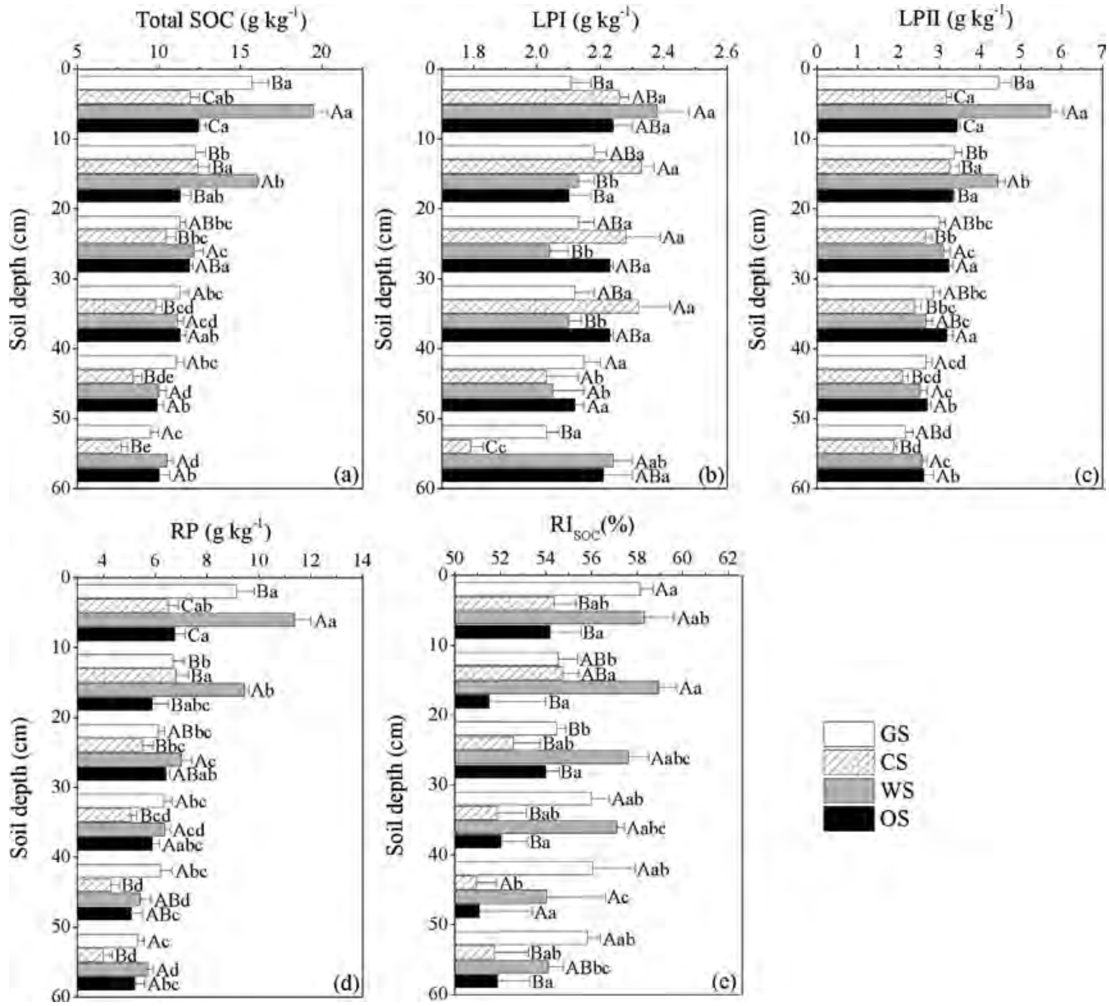

Fig. 2. Profile characteristics of concentrations of total soil organic carbon (SOC) (a), labile carbon pool I (b), labile carbon pool II (LPII) (c), and recalcitrant carbon pool (RP) (d) as well as recalcitrance index of soil organic carbon (RI$_{SOC}$) (e) in each land-use type. Error bars indicate standard errors of the means (n = 4). GS, CS, WS, and OS indicate grassland, cropland, woodland, and orchard, respectively. Different capital letters indicate significant differences among soil layers in a specific land-use type; different lowercase letters indicate significant differences among land-use types in a specific soil layer ($P < 0.05$).

3.2. Vertical characteristics of total SOC, LPI, LPII, and RP concentrations and RI$_{SOC}$

Total SOC concentrations of each land-use type generally showed decreasing trends with increases in soil depth. For different land-use types, total SOC concentrations in the 0–10 cm soil layer were 24.7–85.9% higher than those in the 50–60 soil layer, and the differences between the two soil layers were significant ($P < 0.05$). Conversion from GS to CS significantly decreased total SOC concentrations by 18.6–23.9% in the 0–10 cm and 30–60 cm soil layers ($P < 0.05$). By comparison, total SOC concentrations in all soil layers significantly increased after CS conversion to WS ($P < 0.05$). Significant increases in total SOC concentration were only observed in the 30–60 cm soil layers after land-use change of CS to OS ($P < 0.05$), and the increase rates were all lower than those after conversion from CS to WS (Fig. 2a).

As presented in Fig. 2b, LPI concentrations fluctuated with increases in soil depth for all land-use types. After GS conversion to CS, significant changes in LPI concentrations were only detected in the 0–10 cm (7.1%) and 50–60 cm soil layers (−11.8%) ($P < 0.05$). Conversion of CS to WS significantly decreased LPI concentrations by 8.6–10.5% in the 10–40 cm soil layers, and significantly increased LPI concentration by 25.1% in the 50–60 cm soil layer ($P < 0.05$). Comparatively, the differences in LPI concentrations between CS and OS were only significant in the 10–20 cm and 50–60 cm soil layers ($P < 0.05$). LPII and RP concentrations generally decreased with increases in soil depth for all land-use types. LPII concentrations in CS were 3.3–28.8% lower compared with those in GS, but the differences were only significant in the 0–10 cm and 40–50 cm soil layers ($P < 0.05$). By contrast, RP concentrations in most soil layers were significantly lower in CS than those in GS ($P < 0.05$). After CS conversion to WS, LPII and RP concentrations increased by 12.6–80.8% and 25.9–75.2%, respectively. In most soil layers, the increases were significant ($P < 0.05$). Although LPII concentrations in all soil layers showed increasing trends after CS conversion to OS, the increases were only significant in the 20–60 cm soil layers ($P < 0.05$). By comparison, significant increases in RP concentrations after land-use change of CS to OS were only found in the 30–40 cm and 50–60 cm soil layers ($P < 0.05$) (Fig. 2c and d).

In the 0–60 cm soil profile, RI$_{SOC}$ in GS, CS, WS, and OS varied within 54.5–58.2%, 51.0–54.7%, 54.0–59.0%, and 51.1–54.2%, respectively. After GS conversion to CS, RI$_{SOC}$ in the 0–10 cm, 30–40 cm, and 50–60 cm soil layers significantly decreased by 6.6–7.3% ($P < 0.05$), whereas no significant change in RI$_{SOC}$ was detected in other soil layers ($P > 0.05$). RI$_{SOC}$ increased by 4.6–10.1% after conversion from CS to WS. However, the increases were only significant in the 0–10 and 20–40 cm soil layers ($P < 0.05$). By contrast, RI$_{SOC}$ showed no response to conversion from CS to OS in all soil layers

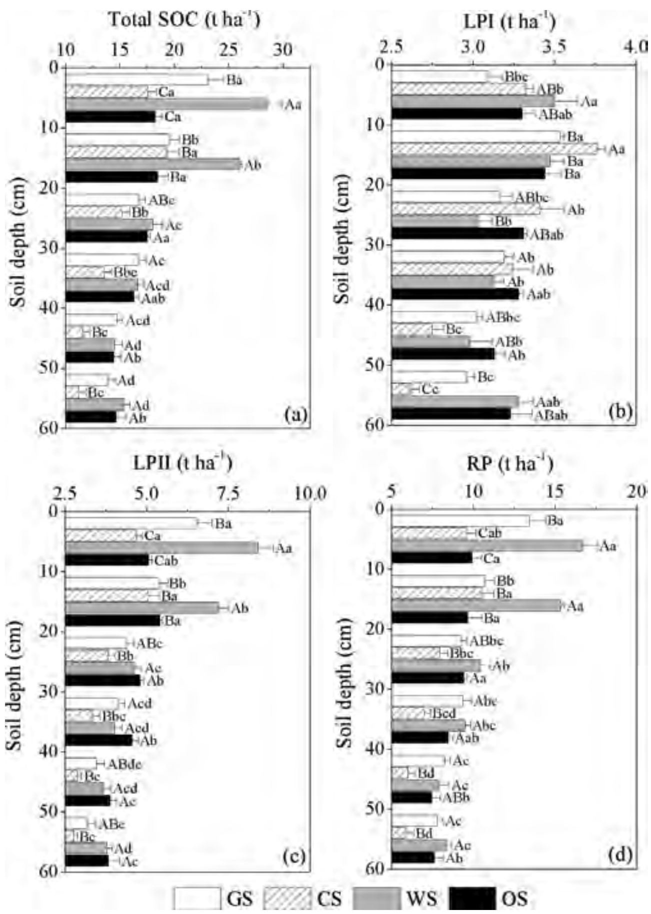

Fig. 3. Stocks of total soil organic carbon (SOC) (a), labile carbon pool I (b), labile carbon pool II (LPII) (c), and recalcitrant carbon pool (RP) (d) in each land-use type. Error bars indicate standard errors of the means (n = 4). GS, CS, WS, and OS indicate grassland, cropland, woodland, and orchard, respectively. Different capital letters indicate significant differences among soil layers in a specific land-use type; different lowercase letters indicate significant differences among land-use types in a specific soil layer ($P < 0.05$).

($P > 0.05$) (Fig. 2e).

Results of two-way ANOVA showed that land-use type, soil depth, as well as their interactions had significant effects on total SOC concentration, LPII concentration, RP concentration, and RI_{SOC} ($P < 0.05$). Similar to soil pH, LPI concentration was only influenced by soil depth and the interactions of land-use type and soil depth ($P < 0.05$) (Table 1).

3.3. Changes in total SOC, LPI, LPII, and RP stocks after land-use change

Total SOC stocks in GS were 1.3–23.9% higher than those in CS, but the differences were only significant in the 0–10 cm and 30–60 cm soil layers ($P < 0.05$). Significant increases in total SOC stocks were observed for all soil layers after CS conversion to WS ($P < 0.05$). The highest increase rate (62.9%) was found in the 0–10 cm soil layer. By comparison, CS conversion to OS only significantly increased total SOC stocks in the 20–60 cm soil layers ($P < 0.05$), and the increase rates were all lower than those after CS conversion to WS (Fig. 3a).

After GS conversion to CS, LPI stocks in the 0–40 cm soil layers increased by 1.6–8.0%, while those in the 40–60 cm soil layers decreasing by 8.8–11.5%. However, such changes were only significant in the 10–20 cm and 50–60 cm soil layers ($P < 0.05$) (Fig. 3b). By contrast, LPII and RP stocks in all soil layers showed decreasing trends after GS conversion to CS (Fig. 3c and d). The reduction rates varied within 5.7–28.6% for LPII and 1.9–28.9% for RP, respectively. LPI stocks in the 0–10 cm and 40–60 cm soil layers increased by 5.4–24.9% after CS conversion to WS, but the increase was only significant in the 50–60 cm soil layer ($P < 0.05$). In contrast, LPI stocks in the 10–30 cm soil layers significantly decreased by 7.9–11.4% after conversion of CS to WS ($P < 0.05$). Significant increases in LPII and RP stocks were detected in all soil layers after conversion from CS to WS ($P < 0.05$). After CS conversion to OS, little change was observed for LPI stock throughout the soil profile ($P > 0.05$), whereas LPII stocks in the 20–60 cm soil layers and RP stocks in the 20–40 cm and 50–60 cm soil layers showed significant increases ($P < 0.05$).

As presented in Table 1, land-use type, soil depth, as well as their interactions significantly affected stocks of total SOC, LPII, and RP ($P < 0.001$). By contrast, LPI stock was not affected by land-use type ($P > 0.05$) but by soil depth and the interactions of land-use type and soil depth ($P < 0.05$) (Table 1).

3.4. Differences in changes in LPI, LPII, and RP stocks and RI_{SOC} between topsoil and subsoil after land-use change

After dividing soil layers into topsoil and subsoil, it was observed that LPI stocks in topsoil and subsoil showed opposite responses to land-use changes. However, the changes were not significant in most cases ($P > 0.05$). By comparison, LPII stocks in both topsoil and subsoil significantly decreased after GS conversion to CS, and significantly increased after CS conversion to WS and OS ($P < 0.05$). Although RP stocks in both topsoil and subsoil decreased after land-use change of GS to CS and increased after land-use change of CS to OS, the changes were only significant in subsoil ($P < 0.05$). Significant increases in RP stock were detected in both topsoil and subsoil after conversion from CS to WS ($P < 0.01$). Similar to LPII stock, RI_{SOC} in both soil layers significantly decreased after GS conversion to CS, and significantly increased after CS conversion to WS ($P < 0.05$). Nevertheless, conversion of CS to OS did not affect RI_{SOC} in either soil layers ($P > 0.05$) (Table 2).

3.5. Relationships between concentrations of total SOC, LPI, LPII, and RP

As shown in Fig. 4, significantly positive relationships were detected between concentrations of total SOC, LPI, LPII, and RP ($P < 0.001$). The highest correlation between total SOC and SOC fractions was

Table 2
Changes in stocks of labile carbon pool I (LPI), labile carbon pool II (LPII), and recalcitrant carbon pool (RP) as well as recalcitrance index of soil organic carbon (RI_{SOC}) in topsoil (0–30 cm) and subsoil (30–60 cm) after land-use change.

Conversion type	LPI stock (t ha^{-1})		LPII stock (t ha^{-1})		RP stock (t ha^{-1})		RI_{SOC} (%)	
	Topsoil	Subsoil	Topsoil	Subsoil	Topsoil	Subsoil	Topsoil	Subsoil
GS to CS	0.72*	−0.56	−2.75*	−1.78*	−5.36	−6.55**	−1.9*	−4.4***
CS to WS	−0.51	0.77	6.70***	2.47**	14.45***	6.89**	4.4***	3.6**
CS to OS	−0.45	1.02*	1.74*	3.27**	0.96	4.69*	−0.7	0.1

Note: GS, CS, WS, and OS indicate grassland, cropland, woodland, and orchard, respectively.
*$P < 0.05$; **$P < 0.01$; ***$P < 0.001$.

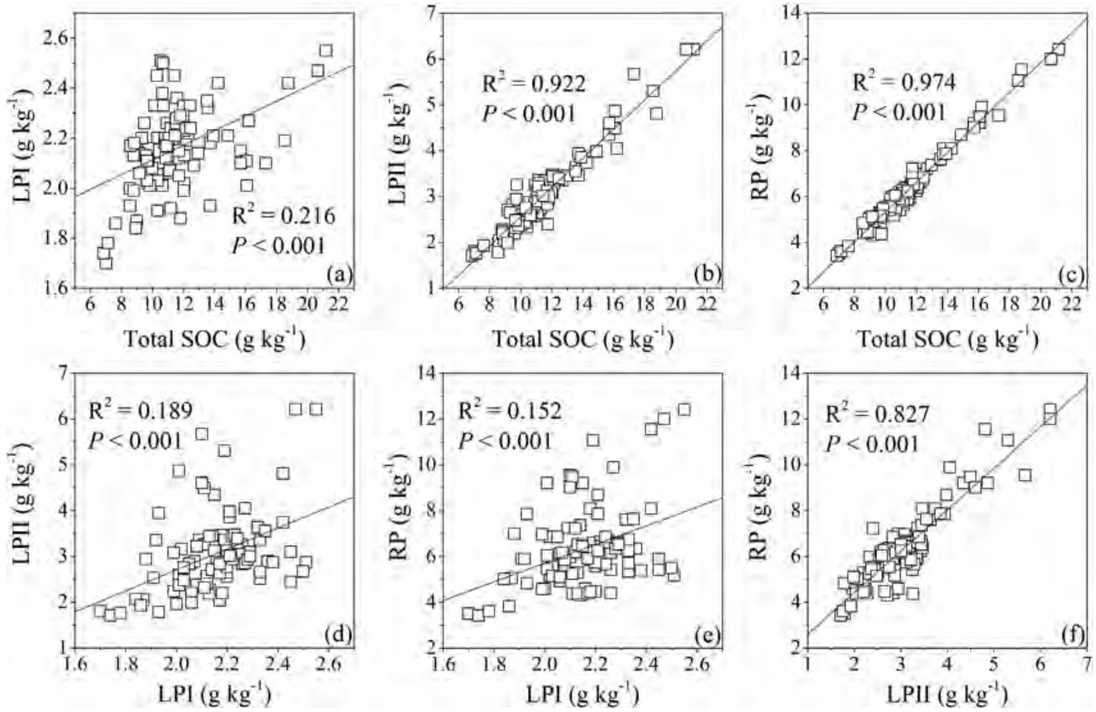

Fig. 4. Relationships between concentrations of total soil organic carbon (SOC), labile carbon pool I (LPI), labile carbon pool II (LPII), and recalcitrant carbon pool (RP).

detected between total SOC and RP ($R^2 = 0.97$, $P < 0.001$), and the lowest correlation was found between total SOC and LPI ($R^2 = 0.22$, $P < 0.001$). The highest correlation between different SOC fractions was observed between LPII and RP ($R^2 = 0.827$, $P < 0.001$), which was considerably higher than those between LPI and LPII/RP ($R^2 = 0.152$–0.189, $P < 0.001$).

3.6. Relationships between soil RI_{SOC}, pH, and $EC_{1:5}$

Significantly negative relationships were observed between RI_{SOC} and pH in both topsoil and subsoil ($P < 0.05$). However, the correlation between RI_{SOC} and pH in subsoil was weak ($R^2 = 0.092$, $P < 0.05$) (Fig. 5a). Although RI_{SOC} significantly correlated to $EC_{1:5}$ in topsoil ($R^2 = 0.339$, $P < 0.001$), the correlation between them was not significant in subsoil ($R^2 = 0.016$, $P > 0.05$) (Fig. 5b).

4. Discussion

4.1. Relationships between total SOC and SOC fractions

Soil organic matter contains a variety of compounds with different stabilities and turnover rates (Rovira and Vallejo, 2007; Paul, 2016). LPI mainly consists of non-cellulosic polysaccharides, which originate from either plant or soil microorganisms. Plant-derived cellulose is the main component of LPII. The major components of RP are lignins, fats, waxes, resins, suberins, and humic substances, all of which are highly resistant to biodegradation (Rovira and Vallejo, 2002; Ding et al., 2012). Since transfers of compounds between different SOC fractions can occur (e.g. degradation of lignocellulose) (Jurado et al., 2015), close relationships may exist between different SOC fractions. For instance, Ding et al. (2012) observed that correlations between LPI, LPII,

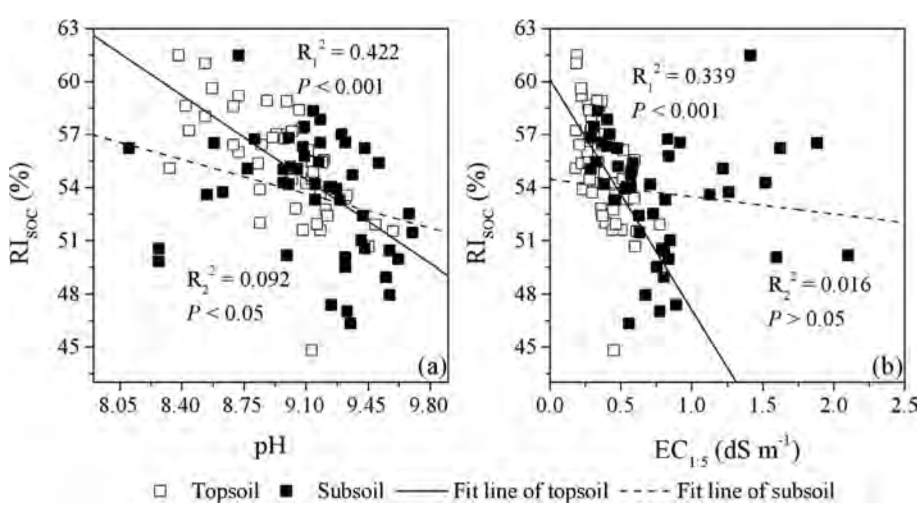

Fig. 5. Relationships between recalcitrance index of soil organic carbon (RI_{SOC}), pH, and electrical conductivity ($EC_{1:5}$) in topsoil and subsoil. R_1^2 and R_2^2 are the coefficient of determinations of topsoil and subsoil, respectively.

and RP were significant in croplands under soybean and maize rotation in Northeastern China. Similarly, significant correlations between different SOC fractions were also detected in the present study (Fig. 4), indicating that soil labile and recalcitrant C pools were tightly linked rather than isolated in this semi-arid agro-pastoral ecotone. However, since the correlations between LPI and the other two C pools were generally weak, more studies should be conducted in the future to better understand the relationship between soil labile and recalcitrant C pools in this area. Furthermore, significantly positive correlations were observed between total SOC and each SOC fraction, and the highest correlation was detected between total SOC and RP (Fig. 4). Considering the high concentration and biochemical stability of RP (Cheng et al., 2008; Belay-Tedla et al., 2009; Ding et al., 2012), the dynamic of RP was thus crucial for long-term SOC sequestration in the study area.

4.2. Effect of land-use change on total SOC stock

SOC is vital to maintain soil fertility and function because it play important roles in influencing physical, chemical, and biological properties of the soil (Bhogal et al., 2009; Moharana et al., 2012). In the present study, the results showed that GS conversion to CS significantly decreased total SOC stocks in most soil layers, indicating that the SOC pool was depleted by long-term agricultural activities. Hence, a decline in soil fertility might be caused after land-use change of GS to CS. The findings were in agreement with those of previous researches being conducted in arid and semi-arid regions (Sommer and de Pauw, 2011; Assefa et al., 2017). In GS, above ground litter and fine root led to a large quantity of C inputs. With high C concentrations, sheep waste could also contribute to maintain or increase SOC stock (Soussana et al., 2004). After conversion from GS to CS, tillage or other soil disturbances could induce soil erosion, disrupt soil aggregate, or improve soil aeration, which might accelerate the decomposition of soil organic matter (McLauchlan, 2006; Li et al., 2013). Additionally, above ground biomass of crops in CS were usually removed after harvest, leading to reductions in soil organic matter inputs (McLauchlan, 2006). Conversely, CS conversion to WS and OS considerably increased the level of total SOC. It was estimated that the average rate of SOC sequestration after afforestation of former cropland was approximately 1.10 t C ha^{-1} yr^{-1} in the top 40 cm soil layer in China (Shi and Cui, 2010). The rate was higher than that in OS (0.38 t C ha^{-1} yr^{-1}) but lower than that in WS (1.83 t C ha^{-1} yr^{-1}). Furthermore, the SOC sequestration rate in WS was also higher compared with those of afforested lands in other regions of Central Asia (Hbirkou et al., 2011; Khamzina et al., 2012). The findings indicated that afforestation on CS with poplar was effective in SOC sequestration in this semi-arid agro-pastoral ecotone. The increases in total SOC stock after CS conversion to WS were mainly due to the large quantity of litter produced by trees and understory plants, which not only increased the inputs of soil organic matter, but also reduced the losses of soil organic matter from external disturbances (Deng et al., 2014; Segura et al., 2016). By contrast, total SOC stocks in the 0–20 cm soil layers in OS were significantly lower than those in WS (Fig. 3a). The differences were possibly due to that the quantity of litter from trees and understory plants was higher in WS than that in OS (Pérez-Cruzado et al., 2012; Józefowska et al., 2017).

4.3. Effects of land-use change on SOC fractions

In most soil layers, conversion from GS to CS did not affect the stock of LPI, but significantly reduced the stocks of LPII and RP, which showed increasing trends after CS conversion to WS and OS. The results, which were in agreement with observations of previous studies (Cheng et al., 2008; Ding et al., 2012; Zhang et al., 2014), implied that soil RP could also be influenced by land-use change. For example, Zhang et al. (2014) found that conversion from natural lands to croplands increased RP stock in the 0–100 cm soil layer in an arid basin in Northwestern China. They speculated that the accumulation of RP after land-use change was mainly due to the application of fertilizer or the incorporation of straw, which enhanced the inputs of soil organic matter. In the present study, the main sources of soil organic matter were different among land-use types. The sources of soil organic matter in GS were mainly fine root, livestock waste, and above ground biomass. In CS and OS, soil organic matter was derived from organic fertilizer and a small quantity of plant litter. As mentioned above, above ground litter and fine root were the major sources of soil organic matter in WS. Consequently, the changes in soil organic matter sources after land-use change may have considerable impact on soil RP (Guo et al., 2016; Bordonal et al., 2017). Since labile plant constitutes can be easily utilized by soil microorganisms, Cotrufo et al. (2013) argued that litter with high quality was the dominant source of microbial products, which led to the formation of stable soil organic matter through strong chemical bonding to the mineral soil matrix. In this study, the increases in RP stocks were higher after conversion of CS to WS than those after conversion of CS to OS (Fig. 3 and Table 2), possibly because the litter of poplar had higher quality than that of apricot. In addition, previous studies pointed out that land-use change could alter the activities of soil microorganisms and enzymes, which played important roles in the decomposition of soil organic compounds (Tian et al., 2010; Ren et al., 2018). Hence, changes in SOC fractions after land-use change were likely attributed to the differences in quantity and quality of soil organic matter and soil biological properties among land-use types. Nevertheless, how soil microorganisms and enzymes influenced the dynamics of different SOC fractions was still unclear, which should be focused on in the future to better understand the effect of land-use change on the C cycle in arid and semi-arid regions. Considering the important role of inorganic fertilizer application in inorganic C sequestration in drylands (Mikhailova and Post, 2006), the effect of land-use change on soil inorganic C stock should also be paid closer attention to in future studies.

In this study, changes in stocks of both labile and recalcitrant C fractions after land-use change were detected in subsoil (Table 2). The results, which were in line with those of previous studies (Zhang et al., 2014; Sheng et al., 2015), implied that SOC fractions in subsoil could also be affected by land-use change, and the effects were different from those on SOC fractions in topsoil. Since the quantity and quality of soil organic matter and the activities of soil microorganisms and enzymes might be altered by land-use change, the formation and decomposition of SOC in subsoil could thus be influenced by land-use change (Rumpel and Kögel-Knabner, 2011). It was noteworthy that subsoil played a more important role in SOC sequestration than topsoil after conversion from CS to OS (Table 2). The reason was unclear but might be partly explained by the physical or biological transportation of soil organic matter from topsoil to subsoil (Rumpel and Kögel-Knabner, 2011). The results supported previous assertion that subsoil has tremendous potential for long-term SOC sequestration (Lorenz and Lal, 2005; Rumpel et al., 2012). Therefore, it is suggested that RP, particularly those in subsoil, should be taken into consideration when evaluating the impact of land-use change on SOC dynamic to better understand the role of land-use change in the global C cycle.

4.4. Effect of land-use change on SOC stability

For different land-use types, the proportions of LPI, LPII, and RP in total SOC varied within 11.8–25.3%, 20.5–33.5%, and 44.8–61.5%, respectively, demonstrating that RP was the major part of SOC in the study area. Similarly, Ding et al. (2012) observed that the proportion of RP in total SOC varied from 54.0% to 59.3% in Northeastern China. Liu et al. (2017) found that the proportions of SOC fractions in total SOC in natural lands and croplands in Eastern Ili River Valley decreased in the following order: RP (49.4–66.3%) > LPI (18.7–39.2%) > LPII (8.9–16.2%). In general, a high RI_{SOC} implied a high degree of biochemical recalcitrance of SOC (Rovira and Vallejo, 2002; Dou et al., 2016). Therefore, the resistance of SOC pool to biodegradation often increases with increase in RI_{SOC} (Rovira and Vallejo, 2002). As shown in Table 2, RI_{SOC} in both topsoil and subsoil significantly decreased after GS conversion to CS, and significantly increased after conversion of CS to WS. The results suggested that agricultural activities decreased the stability of SOC, causing the depletion of SOC, whereas CS conversion to WS enhanced the stability of SOC, promoting SOC sequestration. Similarly, Shrestha et al. (2008) reported that both concentration and recalcitrance of SOC could be altered by land-use change in the Pokhare Khola watershed in Nepal. They attributed such changes to the differences in litter quality among land-use types. As mentioned above, the quality of soil organic matter in the study area might be altered by land-use change. Hence, the changes in biochemical recalcitrance of SOC after land-use change were likely due to the changed soil organic matter sources. However, Dou et al. (2016) found that RI_{SOC} in the 0–20 cm soil layer was significantly increased by agricultural activities, especially by the combined application of inorganic fertilizer and corn straw residue. They suggested that the increased RI_{SOC} was due to the high RP concentrations in the applied fertilizers and straws. Therefore, incorporating straw into soils may be another way to increase the stability of SOC in this semi-arid agro-pastoral ecotone aside from afforestation, which should be tested in the future.

Soil salinization is a serious environmental issue that often occurs after long-term irrigation in arid and semi-arid regions (Singh, 2015). It is reported that more than 47% of the irrigated lands in Central Asia are under the threat of soil salinization, which is harmful to crop growth and production (Singh, 2015; Hamidov et al., 2016). After GS conversion to CS, the degree of salinity in topsoil increased from no saline to low salinity (FAO classification) (Abrol et al., 1988), indicating that soil salinization was induced by long-term cultivation in this semi-arid agro-pastoral ecotone. Previous studies reported that soil salinization had adverse impacts on SOC sequestration. For instance, Setia et al. (2013) estimated that the historic loss of SOC from saline soils was 3.47 t C ha^{-1} on average on a global scale. They claimed that less C inputs to soils caused by the reduced plant productivity was the main reason for the decreased SOC stock in salt-affected soils. In tidal wetlands in Virginia, Morrissey et al. (2014) pointed out that salinity stimulated the activity of C-degrading extracellular enzymes, thus accelerating the decomposition of soil organic matter. In our earlier study (Liu et al. 2018a), the results showed that soil pH and $EC_{1:5}$ were negatively correlated to SOC concentration in the Eastern Ili River Valley. In the present study, significantly negative correlations were also detected between pH, EC, and RI_{SOC} (Fig. 5), suggesting that soil pH and salinity might be indicators of both quantity and stability of SOC in this area. In Northwestern Uzbekistan, Central Asia, Hbirkou et al. (2011) found that SOC stock, the stability of soil aggregate, and soil salinity were tightly linked. They further suggested that the reduction of soil salinity and sodicity might favor the recovery of soil structure, improve the productivity of plant, thus enhance the accumulation of SOC. High soil salinity may also result in flocculation of clay particles into aggregates, and then restrict substrate availability and SOC decomposition (Wong et al., 2010). Conversely, on wetting of salt-affected soils can cause the dispersion of soil aggregates, which accelerates SOC loss by increasing accessibility and availability of physically protected SOC (Oades, 1984). Consequently, soil salinization might influence the biochemical recalcitrance of SOC by affecting the stability of soil aggregates. Furthermore, empirical studies pointed out that soil salinity could either accelerate or inhibit the decomposition rate of plant litter mainly through affecting the activities of soil microorganisms (Lopes et al., 2011; Sun et al., 2016). Since litter turnover is crucial for the formation of soil organic matter, the influence of soil salinization on litter decomposition and the implication of such influence for SOC stability should also be focused on in future studies. As shown in Fig. 5, although the correlations between RI_{SOC}, pH, and $EC_{1:5}$ were significant in the present study, the correlations were generally weak, particularly in subsoil. Therefore, more sites should be considered in future studies to better understand the role of soil pH and salinity as indicators of the biochemical recalcitrance of SOC in this semi-arid agro-pastoral ecotone. Since both soil pH and $EC_{1:5}$ generally showed decreasing trends after CS conversion to WS, the results still suggest that afforestation on CS with poplar contributed to mitigate soil salinization while promoting SOC sequestration in this area.

5. Conclusion

Results of this study showed that the proportions of different SOC fractions in total SOC decreased in the order: RP > LPII > LPI, indicating that RP was the major form of SOC in this semi-arid agro-pastoral ecotone. The significant correlations between different SOC fractions suggested that labile and recalcitrant C were tightly linked. After GS conversion to CS, LPI stocks in topsoil and subsoil showed increasing and decreasing trend, respectively. In contrast, CS afforestation led to reductions of LPI stock in topsoil and increases of LPI stock in subsoil, respectively. The results implied that land-use change had opposite impacts on LPI in topsoil and subsoil. In contrast, LPII and RP stocks in both soil layers decreased after GS conversion to CS, and increased after CS afforestation, demonstrating that LPI was not the only SOC fraction that was influenced by land-use change. Therefore, it is suggested that the effect of land-use change on RP, especially those in subsoil, should be taken into account in future studies to better understand the role of land-use change in the global C cycle. The changes in RI_{SOC} after land-use change indicated that long-term cultivation decreased the stability of SOC, causing the depletion of SOC stock, whereas CS conversion to WS enhanced the stability of SOC, promoting SOC sequestration. The significantly negative correlations between RI_{SOC}, pH, and $EC_{1:5}$ implied that soil salinization might have adverse impact on SOC sequestration, which should be paid closer attention to in future studies.

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