Post 30-years effects of conversion of sand dunes to rice ecosystem on soil organic matter, biological indices and micronutrient fractions in south-western Punjab, India

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Abstract

In south-western Punjab, we investigated how the soil organic C (TOC) pool changed over time in undisturbed sand dunes and adjacent field, along with its labile and non-labile fractions, their relationships to micronutrient fractions, and the biological characteristics of the soil (India). To explore the effects of land use changed from sand dunes to intensive rice-wheat cropping system (RCWS) the soil samples were collected from both the undisturbed sand dunes and the adjacent fields converted to farmland about 30 years ago. The results showed that in comparison to RCWS soils, the TOC pools under sand dunes remained lower by ~2.5 g C kg⁻¹ (about 86.2%) but, their active C pools (Fract. 1 + Fract. 2) were significantly better (by about 0.67 g kg⁻¹; 40.4%). Similarly, the passive C pool (Fract. 3 + Fract. 4) was significantly higher in RCWS soils by ~3.1 g kg⁻¹ (about 239%), compared with the sand dunes. The passive C comprised ~81.7% of TOC pool in RWCS soils that was only ~44.0% in sand dunes. Following land-use change, the RWCS fields gradually increased the TOC stocks by ~75% as compared to sand dunes. In RWCS, the soil alkaline-phosphatase (Alk-P) and dehydrogenase (DHA) activities was significantly higher by 4.6 to 6.7 times than the sand dunes. Besides, Micro-nutrients cations (e.g. Zn, Cu, Fe and Mn) and their transformations were also higher by ~1.7, 1.8, 2.9 and 5.2-times, respectively in RWCS than sand dunes. Likewise, total-Zn, total-Fe and total-Mn fraction in RWCS soils was ~77.3%, ~90.4% and ~64.4% higher than their respective contents in sand dunes. After 30 years of continuous RWCS system, the soil quality index increased from 0.23 to 0.97. These results clearly showed that due to increased soil microbial activities, the RWCS have significant potential to sustain C and build-up of micro-nutrients cations hence, underpins the clear signs of soil sustainability in south-western region of the Indian Punjab.

1 INTRODUCTION

Globally, climate change has greatly affected crop production potential, carbon sequestration, micronutrient availability, and biological processes in degraded landscapes (Kumar and Das, 2014; Sharma et al., 2021). Although degraded soils are themselves highly vulnerable ecosystems to climate change (Henry et al., 2007), they have considerable potential to contribute significantly to increased C sequestration through the implementation of rehabilitation and C balancing (Basu, 2014; Singh and Benbi, 2018a). Degraded landscapes are thought to have tremendous potential for C accumulation compared to greenhouse gas (GHG) emissions in response to anthropogenic activities; therefore, climate-resilient ecosystems are more responsive to C cycling (Franzluebbers and Doraiswamy, 2007; Brahma et al., 2017). A well-managed agroecosystem can either mitigate or ameliorate climatic impacts (Trabucco et al., 2008), e.g., by conserving soil and water resources (García et al., 2010) and enhancing ecosystem services (Evans et al., 2013) by sequestering atmospheric GHGs in degraded soils (Amichev et al., 2008). Land-use change has both positive and negative impacts on the C sequestration capacity of different ecosystems in response to spatiotemporal climate variability (Singh and Benbi, 2018a; Bhatt et al., 2021). In particular, when land-use change occurs from degraded landscapes to cropland ecosystems, it often leads to rapid restoration of native vegetation with a significant change in
plant cover to increase above and belowground biomass production of arable crops (Novara et al., 2017; Yu et al., 2018), resulting in a significant change in soil microbial composition (Zhao et al., 2005; Romero-Díaz et al., 2015). The positive effects of land use change on the ecosystem are increased biodiversity, reduced soil erosion, increased ability of soil to retain moisture, and increased total soil organic carbon (TOC) (Lasanta et al., 2015; Gabarrón et al., 2015). After land use change, colonization by natural vegetation begins, leading to widespread changes in soil properties (Bonet and Pausas, 2004; Raiesi, 2012). Changes in soil properties occur due to suspension of tillage, colonization by vegetation and succession, changes in the physical and biological environment of the soil, and changes in soil quality parameters (Raiesi, 2012; Sharma et al., 2022a). The increase in the organic C pool observed after land use changes is mainly due to the increased input of organic matter and resistance to litter decomposition (Gabarrón et al., 2015).

The soil organic matter influenced by land use change and associated cropping practices has great potential to minimize greenhouse gas emissions and enrich C accumulation (Sharma et al., 2022b). Soil organic matter has a great influence on soil and plant productivity (De Paepe and Alvarez, 2013) and is therefore considered a critical component of the biogeochemical C cycle and the biological environment (Manlay et al., 2007). A change in plant cover, organic matter quality/quantity, soil physical structure, and physicochemical and microbial environments following agricultural intensification has a significant impact on the restoration of soil physical, chemical, and biological processes (Raiesi, 2012; Shang et al., 2014), which are early indicators of the effects of land-use change and/or restoration processes on in stream grasslands and other natural ecosystems (Wang et al., 2011). Cropland ecosystems act as C sinks by capturing and storing significant amounts of atmospheric CO2 in the soil (Singh and Benbi, 2018a,b; Sharma et al., 2022a,b). The importance of managing such systems through appropriate agroecological C sequestration approaches is increasingly documented as a robust approach to enhance soil C sequestration (Toensmeier, 2016; Singh and Benbi, 2022). The labile fractions of soil organic matter are important indicators for evaluating soil quality and its response to changes caused by crop production and soil management (Melero et al., 2009; Sharma et al., 2022a). Besides, several studies have been conducted to establish the relationships between edapho-climatic conditions and land use system with soil organic C sequestration (Jobbagy and Jackson, 2000; Franzluebbers and Follett, 2005). Obviously, rice-wheat cropping system (RWCS) is the predominant annual crop rotation in the Indo-Gangetic Plains, covering about 13.6 million hectares (Mha) in Bangladesh, India, Nepal, and Pakistan, and about 13 Mha in China (Timsina and Connor, 2001).

In the pre-Green Revolution period of the mid-1960s, cereal production was low (< 80 million tons (Mt) year-¹), which increased to 275 Mt year-¹ due to the use of chemical fertilizers in Indian agriculture (https://www.faidelhi.org/statistics/statistical-database,). Changes in crop production and land use before and after the Green Revolution resulted in higher crop productivity due to the intensification of RWCS, leading to greater resilience and sustainability in food production (Yadvinder Singh et al., 2004; Sharma et al., 2022a). Likewise after agricultural mechanization, the uncultivated land that remained in the form of sand dunes in southwestern Punjab (India) was brought for cultivation. Successively, continuous RWCS cultivation caused either yield decline and/or stagnation, loss of soil organic matter, micronutrient deficiencies, and alteration of soil physical, chemical, and biological properties (Bhatt et al., 2016; Singh and Benbi, 2016; Sharma et al., 2021, 2022a,b). In addition, RWCS has led to depletion of the water table (Bhatt et al., 2021). The effects of RWCS on soil health due to altered soil organic matter quality have also been reported by Benbi et al. (2015), Sharma et al. (2020), and Singh and Benbi (2021). Recently, C pools with variable liability have been studied in relation to crop management in different agricultural and ecoregions, including temperate, tropical and subtropical environments (Nair et al., 2009; Benbi et al., 2012; Benbi et al., 2015). However, there are few studies that have compared soil organic matter content and micronutrient transformation with land use change from uncultivated land (sand dunes) to intensive RWCS cultivation over time.

We hypothesized that conversion from uncultivated land to RWCS would help understand the underlying mechanisms responsible for the change in the C pool associated with altered substrate availability due to increased plant-mediated C inputs. Land use conversion from sand dunes to RWCS would help in increase of C pool in soils (Singh and Benbi, 2022). However, regular use of agrochemicals, synthetic fertilizers, and
micronutrients not only increases plant biomass and plant-mediated C, but also impose significant impacts on soil micronutrient transformation (Sharma et al., 2020; Sharma et al., 2022b). Land use conversion from sand dunes to intensive RWCS would help to quantify the C sequestration potential of degraded landscapes and to implement C-intensive strategies for sustainable crop production. Therefore, the objective of the present study was to quantify the changes in C pools and micronutrient fractions in RWCS soils converted from 30-year-old sand dunes in the alluvial plains of southwestern Punjab, India.

2 MATERIALS AND METHODS

2.1 Brief description of study region

The study was confined at different locations of Faridkot district of Punjab in north-western India and lie within latitudes of 29°54′N-34°54′N, longitude of 74°15′E-75°25′E and altitude of 199 m to 204 m asml, and has total geographical area of 141900 ha covering 171 villages. The net sown area comprised 90.3% of total geographical area elevated at 204.3 m asml. The climate of the region is classified as subtropical, with semi-arid, extremely hot summers and bitterly cold winters. During monsoon season (June to September), the study region has high humidity and cloudiness. The average annual rainfall of study region is 420 mm in 24 days which is unevenly distributed. The south-west monsoon contributes nearly 78% of the total annual rainfall. The maximum mean air temperature during the growing season (June-October) ranged from 26 to 45°C, while minimum ranged from 14 to 32°C but, the mean of minimum and maximum temperature for crop season was 25 and 34°C, respectively.

2.2 Sampling sites

The soil samples from 0-15 cm were collected with core sampler (7.3 cm inner diameter) from 15 sites under RWCS and 10 sites under sand dunes in May-June, 2021. At each site, three pseudo-replications were established covering an area of 0.4 ha, and therefore, a total of 75 samples (25 sites x 3 replications) were collected. One portion of each sample was sieved through 2 mm sieve for analyzing various physio-biological properties such as pH$_{1:2}$, electrical conductivity (E.C.$_{1:2}$), particle size distribution (sand, silt and clay), calcium carbonate (CaCO$_3$; Puri, 1950), total soil organic C (TOC; Walkley and Black, 1934), available-P (Olsen et al., 1954) and available-K (Mervin and Peech,1950). Dehydrogenase (DHA) and alkaline phosphatase (Alk-P) activity was determined by following standard procedures (Tabatabai, 1982; Tabatabai and Bremner, 1969). Soil moisture constants at field capacity (FC), available water content (AWC) and permanent wilting point (PWP) were determined by pressure plate apparatus (Klute and Dirksen,1986). Total content of Zn, Fe, Mn and Cu was determined by digesting the soil samples with a mixture of nitric acid and hydrochloric acid (HNO$_3$: HClO$_4$, 3:1; w/v). The diethylene triamine pentaacetic acid (DTPA) extractable Zn, Fe, Mn and Cu were determined using a mixture of 0.005 M DTPA + 0.01M CaCl$_2$ + 0.1M TEA buffer (pH=7.3) following Lindsay and Norvel (1978), followed by determining the concentration of micro-nutrient cations on AAS (AAS FS 240 Model). The total micro-nutrient’s content was partitioned as water soluble + exchangeable (WSEx-), specifically adsorbed (SpAd-), manganese oxide bound (MnOx-), amorphous oxide bound (AmOx-), crystalline oxide bound (CrOx-), organic matter (OM-) and residual fraction (Res-) were determined by pre-standardized sequential extraction procedure (Singh et al., 2013; Singh et al., 2014; Sharma et al., 2021).

2.3 Soil quality index

For calculation of soil quality index, the first step is to select the minimum data set (MDS). According to Andrews et al. (2002), only those soil parameters that demonstrated significant treatment differences were chosen to make up a representative minimum data set (MDS). Through PCA, significant variables were selected for the subsequent stage of MDS creation. (Sharma et al., 2005a). It was considered that the variables with high factor loading and principal components with high eigen values better characterized system features. Therefore, only PCs with eigenvalues less than one were included (Brejda et al., 2000), as well as those that explained at least 5% of the variation in the data (Wander and Bollero, 1999). Only heavily weighted factors were kept for MDS within each PC. After the selection of PCs the highest factor loading from each PC was selected and in case of more than one parameter in one PC, multivariate correlation
Coefficient analysis was performed and then the factor with the highest sum of correlation coefficient was retained from each PC and other parameters were eliminated from the minimum dataset (Andrews et al., 2002). Following the selection of the MDS indicators, a linear scoring approach was used to transform each observation of each MDS indicator. (Andrews et al., 2002). Indicator scoring was done on the basis of soil function as good or bad. For soil quality indicator ‘more is better’, each observation was divided by the highest observed value so that the highest value get a score of 1. For ‘less is better’ indicator, the lowest observed value (in the numerator) was divided by each observation (in the denominator) such that the lowest observed value received a score of 1. The PCA results were used to weight the MDS variables for each observation after transformation. Each PC explained a certain amount (%) of the variation in the total data set. This percentage, divided by the total percentage of variation explained by all PCs with eigenvectors >1, provided the weighted factor for variables chosen under a given PC. We then summed up the weighted MDS variables scores for each observation using the following equation:

\[ \text{SQI} = \sum_{i=1}^{n} w_i s_i \]

Where, \( s_i \) is the score for the subscripted variable and \( w_i \) is the weighing factor derived from the PCA.

2.4 Statistical analysis

The statistical analysis of data on soil organic C pools, biological indices and micro-nutrients’ fractions were carried out using analysis of variance (ANOVA) technique in randomized block design (RBD) using SPSS Inc., Chicago, U.S.A. software. The significantly values for different land uses representing unique letters was used at \( p < 0.05 \).

3 RESULTS

3.1 Basic soil properties of different land-use/cover

Soils under the studied land uses differed significantly (\( p < 0.05 \)), except for pH and available K (Table 1). The EC of soils under RWCS was significantly higher (\( \sim 52\% \)) compared to sand dunes. Available P content was 3.7 times higher in RWCS soils over sand dunes. Figure 1 showed large differences in DTPA-extractable micronutrients in both the RWCS and sand dunes soils. DTPA-extractable micronutrients Zn, Cu, Fe, and Mn in RWCS were increased by 1.7, 1.8, 2.9, and 5.2 times respectively, compared to their content in sand dunes. Conversely, soil bulk density of the sand dune soils was considerably higher than RWCS. Figure 2 clearly illustrated the comparative moisture properties of soils under RWCS and sand dunes. Volumetric moisture content (\( \theta \)) at different soil potential (\( \psi_m \)) was significantly higher in RWCS soils compared to sand dunes. At field capacity (\( \psi_m=0.33 \text{ bar} \)), \( \theta \) was 71.8% higher in RWCS soils, but at permanent wilting point (\( \psi_m=0.33 \text{ bar} \)), \( \theta \) was 7.9 times higher in RWCS soils than the sand dunes.

3.2 Total organic C and its fractions of varying oxidizability

The RWCS significantly increased the TOC pool by \( \sim 2.5 \text{ g kg}^{-1} \) (\( \sim 86.2\% \)), compared with sand dunes (Table 2) contrarily, Fract. 1 and Fract. 2 were significantly higher in sand dunes, whilst the Fract. 3 and Fract. 4 were significantly higher in RWCS soils. The active C pool (Fract. 1+ Fract. 2) in RWCS soils significantly lower by \( \sim 0.67 \text{ g kg}^{-1} \) (\( \sim 40.4\% \)) than the sand dunes. Conversely, the passive C pool (Fract. 3 + Fract. 4) was significantly higher in RWCS soils by \( \sim 3.1 \text{ g kg}^{-1} \) (\( \sim 239\% \)), compared with the sand dunes. As a proportion of TOC, Fract. 1 comprised \( \sim 7.3 \text{ and 23.8\%} \), respectively for RWCS and sand dune soils, was significantly higher in sand dunes. Fract. 2 comprised \( \sim 11.0 \text{ and 32.2\%} \), respectively in RWCS and sand dune soils. The passive C pool comprised \( \sim 81.7\% \) of TOC pool in RWCS soils, that was only \( \sim 44.0\% \) in sand dunes. The lability index was significantly higher in sand dunes than the RWCS soils, which resulted in C management index (CMI) of 53.5%.

3.3 Enzymatic activity in soils under different land-use/cover
The Alk-P activity varied between 9.6 and 41.4 µg p-nitrophenol g⁻¹ hr⁻¹ in RWCS soils, and 0.4 and 16.3 µg p-nitrophenol g⁻¹ hr⁻¹ in sand dunes (Figure 3). Mean Alk-P activity of 20.5±3.0 µg p-nitrophenol g⁻¹ hr⁻¹ in RWCS soils was significantly higher by ~4.6-times, compared with 4.5±1.7 µg p-nitrophenol g⁻¹ hr⁻¹ in sand dunes (Table 2). DHA activity in RWCS soils varied widely; 6.3-11.7 µg TPF g⁻¹ hr⁻¹ as compared to 0.2-2.7 µg TPF g⁻¹ hr⁻¹ in sand dunes. Mean DHA activity in RWCS soils was higher by ~6.7 times than the sand dunes. C fractions of varying oxidizability exhibited a significant increase with increased soil enzymatic activity viz. Alk-P and DHA in soils under contrasting land-use/cover (Figure 4). These relationships could best be described by linear functions (R²=0.534*, p <0.05 to 0.966**, p <0.01) (Table 3).

### 3.4 Total organic C stocks in soils under different land-use/cover

Total organic C stocks varied between 11.7 and 15.5 Mg C ha⁻¹ in RWCS soils, as compared to 5.7 and 11.5 Mg C ha⁻¹ in sand dunes (Figure 4). The mean TOC stocks in RWCS soils of 12.6±0.38 Mg C ha⁻¹ were significantly higher by ~75.1% as compared to 7.2±0.52 Mg C ha⁻¹ in sand dunes. TOC stocks increased linearly with increase in passive C pool in soils (Figure 5). The significant (p <0.01) relationship between two variables could best be described by straight line equations (Eq. 1-2).

TOC stocks (Mg C ha⁻¹) = 2.3223 (passive C pool; g kg⁻¹) + 2.3862, R² = 0.978**, p <0.01 (RWCS) (1)

TOC stocks (Mg C ha⁻¹) = 4.9828 (passive C pool; g kg⁻¹) + 0.7207, R² = 0.993**, p <0.01 (Sand dunes) (2)

### 3.5 Total micro-nutrient’s content and fractions

Total-Zn content in RWCS soils was ~77.3% higher, compared with sand dunes (Table 4). Except for WSEx-Zn, all other fractions of variable solubility were significantly higher in RWCS soils than sand dunes. The WSEx-Zn was the smallest (~1.6-2.7% of total-Zn), while the Res-Zn was the largest (~55-57% of total-Zn) fraction of total-Zn in soils under different land-use/cover. Amongst the oxide bound fractions, CrOx-Zn was the largest (~19.6-24.4% of total-Zn) and MnOx-Zn was the smallest fraction of total-Zn (~2.8-3.1%), whilst the AmOx-Zn fraction in-between (~9.3-11.4% of total-Zn). The WSEx-Cu was significantly lower in RWCS soils, compared with the sand dunes (Table 4). The SpAd-Cu, however, did not differ significantly amongst the compared land-use/cover. All other Cu fractions, except AmOx-Cu were significantly higher in RWCS soils, compared with sand dunes. Unlike Zn where Res-Zn was the largest fraction, the CrOx-Cu was the largest fraction (~34.2-39.1%) of total-Cu content in compared land-use/cover. Amongst the oxide bound-Cu fractions, the relative occurrence followed the order; MnOx-Cu < AmOx-Cu < CrOx-Cu in compared land-use/cover. The OM-Cu fraction was ~44.5% higher in RWCS soils, compared with sand dunes. Similar to Zn, the WSEx-Cu was the smallest fraction comprising ~1.3 and 2.9% of total-Cu content in RWCS and sand dunes, respectively. Total-Fe content in RWCS soils was ~90.4% higher, compared with sand dunes. (Table 4). All Fe fractions of variable solubility were significantly higher in RWCS soils than the sand dunes. Unlike, Zn and Cu, the MnOx-Fe was the smallest (~0.96-1.19% of total-Fe), while the CrOx-Fe was the largest (~26.9-29.6% of total-Fe) in soils under different land-use/cover. However, WSEx-Fe comprised ~3.7-4.5% of total-Fe in soils. Amongst the oxide bound fractions, CrOx-Zn was the largest followed by AmOx-Fe and the lowest by MnOx-Fe. The Res-Fe comprised ~25.3% of total-Fe content in soils under RWCS and sand dunes. The OM-Mn was the smallest, while the Res-Mn was the largest fraction of total-Mn in soils under RWCS and sand dunes (Table 4). Although, Om-Mn was the smallest fraction, yet was significantly higher by ~69.1% in RWCS soils, compared with the sand dunes. Amongst the oxide bound fractions, the CrOx-Mn was the smallest, AmOx-Mn was the largest, whilst the MnOx-Mn in-between. The total-Mn fraction was significantly higher by ~64.4% in RWCS soils, compared with the sand dunes.

### 3.6 Soil Quality Index

The relationship between the eigenvalue and PC in the form of scree plot is given in fig. The PCA and communalities up to PC 2 (which have eigenvalue greater than 1) to evaluate SQI are given in table 6. Two PCs have been extracted which had eigen value greater than land theses four PCs can explain 98.4 per cent of the total variances. PC1 having eigenvalue of 16.79 explained about 88.3 per cent of variance, which
includes enzyme activities such as dehydrogenase activity with positive factor loading (0.977), alkaline phosphatase (0.971), recalcitrant carbon pool (0.973) and micronutrient copper content (0.966). PC2 explained about 10.06 per cent of variance and eigenvalue of 1.913, PC2 included pH with positive factor loading, i.e. 0.994, potassium (0.876), very labile C pool (-0.972) and active C pools (0.872). Communalities indicate relative importance of each soil attribute in terms of its contribution to all the extracted PCs (Brejda et al., 2000). All the soil attributes have contributed to the betterment of soil quality which was explained on the basis of communalities.

The highly weighted variables under two PCs were subjected to Pearson correlation separately. The variables with the highest correlation total were thought to best represent the group. A correlation matrix for the highly weighted variables under different PCs was run separately. Among the four variables in PC1, dehydrogenase activity was chosen for the MDS because of its highest correlation sum (3.99). The other variables in PC1 i.e. copper (3.75), recalcitrant pool (3.86) alkaline phosphatase (3.89) were dropped as the sum of correlation was lower than DHA (Table 7). In PC2 active C pool was chosen for MDS due to its highest correlation as compared to other parameters (Table 8).

These selected soil attributes, viz., DHA and active C pool indicate the important role being played by them in enhancing the sustainability under the land uses. The remaining soil attributes were either less factor loading or not well correlated with each other, thus these were excluded.

The indicator score was then multiplied by the weighting factor derived from the PCA to obtain the ultimate index value for soil quality under different land uses. The weight of each PC on the basis of percent variance to total variance was 0.89 and 0.11 for DHA and active C pools and is presented in table 6. Conversion of sand dunes to rice wheat system resulted in improving the SQI from 0.23 to 0.97(Table 2). Further from fig it was revealed that under RW system, the contribution of DHA was 92.8 per cent, whereas active C pools contributed 7.1 per cent to the soil quality index (SQI). Whereas in case of sand dunes the corresponding contribution was 55.8 and 44.2 for DHA and active C pools, respectively (figure 8).

4 DISCUSSION

4.1 Carbon fractions and soil enzymatic activity under different land-use/cover

Soil management and cropping practices, e.g., chemical fertilizers, tillage, and cropping systems, have different effects on the production of labile C substrates, resulting in an increased organic C pool and biological properties with different effects on micronutrient fractions (Xie et al., 2019; Sahoo et al., 2019). The balance between the rate and extent of photosynthate deposition and the rate of C mineralization by microorganisms affects the suitability of soils for C sequestration (Mathieu et al., 2015). In addition, root biomass is more resistant to microbial decomposition than surface soil wastes, and root C and rhizo deposition has a long turnover period (Rasse et al., 2005), which is an indispensable mechanism to support soil health and food security by maintaining ecosystem sustainability (Gregorich et al., 1994; Anantha et al., 2018). The total organic C pool includes all C pools with variable oxidizability (active + passive C pool), which plays a key role in improving other soil properties (Paddhushan et al., 2016; Singh and Benbi, 2018a,b). In this study, it was observed that conversion of cultivation from sand dunes to RWCS improved C pools, labile C stocks, and biological environment in the form of oxidizable soil organic matter (Campbell et al., 1999), in addition to slow shifting of above and below ground plant biomass and additional release of water-soluble organic C (Mandal et al., 2012; Naik et al., 2017). The higher amount of root exudates (e.g., lignolytic and cellulolytic) benefits the very labile and unstable C pool in soils (Bhattacharyya et al., 2007). Yagi et al. (2005) reported that freshly added soil organic matter promotes microbiological activity, which is thought to be responsible for the release of labile C fractions after decomposition. The higher concentration of recalcitrant C pool is attributed to the greater accumulation possible through biochemical recalcitrance of organic C compounds supplied via soil organic matter or as plant-mediated C inputs (Yan et al., 2013). The prevailing saturated soil environment for ~90 days per year causes slow decomposition of organic matter and therefore reduces the accumulation of non-labile organic C components (Lopez-Capel et al., 2008). Alternating wetting and drying makes the organic C fractions more resistant to microbial degradation (Tian et al., 1992). The larger
labile C pools observed in diversified practices under intensive cultivation have been attributed to increased accumulation of reactive C and enhanced microbial activity (Blair et al., 1995; Tripathi et al., 2014). In any agroecosystem, the loss of labile C has greater significance than the loss of nonlabile C fractions because microbiological activity and associated enzymatic activities are linearly linked to labile C fractions and the eventual cycling of essential plant nutrients, which affects the physical and chemical protection of organic C factors (Whitbread, 1995).

The abundance and activity of soil microorganisms are important indicators of soil quality (Li et al., 2021), and microbiological processes regulate the decomposition of organic matter, the release of important plant nutrients in the soil, and thus help maintain plant productivity. The increased activity of enzymes enables microorganisms to extract structural components and substrates from the diverse biomolecules in the soil, providing energy for biogeochemical cycling in the soil ecosystem (Lopes et al., 2021). Such association is considered as a key link between soil microorganisms and essential plant nutrients (Li et al., 2009; Xu et al., 2018). Soil physicochemical properties influence the biological environment of soil microorganisms and therefore directly and indirectly affect soil enzymatic activities (Zhang et al., 2011; Kivlin and Treseder, 2014), soil organic C turnover, and soil nutrient release (Li et al., 2009; Weintraub et al., 2013). In the present study, there were significant differences in soil enzyme activity in soils converted to RWCS compared to undisturbed sand dunes. These differences were attributed to differences in vegetation type/cover, abundant root growth, and a nutrient-rich environment under managed field conditions (Wallenius et al., 2011), which greatly favored the proliferation of microflora for increased production of enzymes responsible for decomposition of soil organic matter due to increased biological activity and improvement of soil properties (Gajda and Martyniuk, 2005; Allison et al., 2007).

4.2 Total micro-nutrient’s content and fractions

In the present study, the different soluble fractions of micronutrients were significantly higher in the RWCS soils than in the sand dunes. The different cropping systems and associated soil management and crop production practices result in a redistribution of micronutrients from an unavailable form to a readily and potentially available form (Brunetto et al., 2014; Couto et al., 2016). Micronutrients have different transport mechanisms in the root system (White and Zasoski, 1999), which can alter biological activity and soil properties (Chen et al., 1999; Lambers et al., 2009). Plant and root biomass additions play an important role in controlling the availability of micronutrient cations in the soil (Saha and Mandal, 2000; Chami et al., 2013) because of their effect on altering the labile and nonlabile C pool due to changes in soil organic matter complexity. Rapidly degradable plant biomass components added to the soil efficiently dissolve the crystalline fractions of micronutrient cations, restoring their solubility and availability within a soil-plant system, as the WSEx (or labile organic fractions) enriched in amino-, carboxyl-, and phenol-type functional groups have higher chelating ability (Stevenson, 1991; Fuente et al., 2011). Exogenous C input in the form of aboveground biomass, organic fertilizers, and plant-mediated root biomass and root exudates provides labile C compounds that contribute to the enhancement of soil microbiological activity (Singh and Beubi, 2018a,b), which influence the accessibility of soil micronutrients (Saikia et al., 2019; Sharma et al., 2021). Soil organic matter binds micronutrient cations as organometallic complexes to enhance their availability in the soil rhizosphere (Rengel et al., 1999), thereby preventing the formation of insoluble forms such as carbonate- and oxide-bound micronutrient fractions (Schulin et al., 2009). The micronutrient cations combine with the organic molecules to form organic-metallic complexes as chelates, and the soluble chelates increase the mineralization of the micronutrient and make it less susceptible to adsorption, fixation, and precipitation reactions (Schulin et al., 2009; Sharma et al., 2014). These chelates are synthesized by plant roots and released into the soil rhizosphere to enhance micronutrient availability (Brady and Weil, 2002), confirming these results with a significantly higher pool of total and other fractions of micronutrient cations in soils under RWCS. Formation of organometallic complexes through ligand exchange, mineralization, and solubilization from organic sources increased soil Zn concentrations (Umesh et al., 2013).

In the present study, higher soil pH increases the negative pH-dependent charge density in the soil colloidal complex, resulting in a decrease in micronutrients available to plants (Nascimento et al., 2007). Complexation
of Zn with dissolved organic compounds increases the solubility and mobility of Zn in soils (Weng et al., 2002; Houben and Sonnet, 2012). As observed in the present study, the residual Zn contained in the mineral fraction comprised the native Zn pool and was the predominant fraction of the total Zn pool (Priyanka et al., 2017; Kumari et al., 2018). In the present study, Fe fractions with variable solubility were significantly higher in the RWCS soils than in the sand dunes because plant roots release more protons in Fe-deficient soils, which lowers soil pH and increases Fe solubility. Soil organic matter alters soil physicochemical properties and microbialological associations with Fe oxide particles, which are closely linked to soil organic matter (Colombo et al., 2014). The organic matter supplied by organic sources contributes to the amelioration of the amount of various fractions of micronutrient cations through a variety of mechanisms (Wang et al., 2012). The study area had low soil organic matter content and therefore could benefit from organic fertilizers to improve the availability of micronutrients, especially Mn, in these soils. The highest correlation and direct path coefficients between soil organic matter and available Mn in soil confirm these results (Okin et al., 2008; Li et al., 2011). It has been reported that the addition of plant-mediated C input through plant and root biomass increases the content of Cu fractions in soils (Sharma et al. 2014; Sharma et al. 2021), confirming these results.

4.3 Soil Quality Index

The Dehydrogenase activity was selected as indicator selected under PC1. Dehydrogenase activity is recognized as a useful indicator in evaluating the metabolic activity of soil microorganisms (Nannipieri, 1994). Dehydrogenase enzyme, an oxidoreductase, was considered to be a sensitive indicator of soil quality and a reliable biomarker reflecting changes in total microbial activity as a result of changes in different land use practices (Saikia et al., 2019). Several researchers shortlisted DHA as important indicator of soil quality (Masto et al., 2009; Nosrati et al., 2013). Under PC2 active pools of carbon was selected as parameter for soil quality indexing. All the pools of SOC (active and passive) were not considered as precise indicator of land use induced differences in soil properties, because different SOC fractions vary in their chemical and physical properties, and turnover rates (Jenkinson and Coleman, 1994). Different soil management, land use practices strongly influence the active carbon fractions as compared passive pools of carbon and could provide an early indication of changes in soil quality (Sharma et al., 2019).

The higher SQI under RWCS could be attributed due to the higher content of the contributing factors. Minimal vegetation, fewer plant residue recycling resulted in no organic matter building might have contributed to lower SQI under sand dunes. Chandel et al., 2018 and Singh et al., 2011 reported higher SQI under cultivated soils than under bare soils.

CONCLUSIONS

These results indicate that a change in land management and cropping practices following a land use change has a significant impact on soil organic matter quality. Land use change from naturally occurring sand dunes to intensive RWCS has agroecological significance as evidenced by increased organic C pool and altered soil micronutrient availability. High microbial activity in RWCS soils due to altered substrate availability reflects a low risk of nutrient loss and contribution to biodiversity, and is a clear indication of soil sustainability. The large differences in micronutrient fractions, soil organic C pools, and enzymatic activity observed in the present study indicate that land use conversion from low vegetation landscapes to RWCS has great potential for soil organic C retention. Nonetheless, cropping helps to improve soil micronutrient sustainability and increase crop productivity.

Conflict of Interest

Authors have no conflict of interest

REFERENCES


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