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Comparative study of soil aggregation, carbon and microbial dynamics under agro-forestry, grasslands, orchard and cultivated land use systems

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Abstract

Land use has a profound impact on soil aggregate stability, carbon (C) sequestration and soil microbial activities. This study tried to explore the dynamics of soil C pools, aggregation, aggregate occluded C, microbial biomass and enzymatic activities within soil aggregates under four types of land uses *viz.* cultivated land, orchard, social/agro-forestry and grasslands. Outcomes indicated the maximum presence of total C in undisturbed soils of natural ecosystems like social/agro-forestry system (6.35 g kg^{-1}), followed by orchard (4.05 g kg^{-1}), grassland (4.03 g kg^{-1}) and cultivated lands (3.07 g kg^{-1}). Nevertheless, the proportion of water extractable C (WEC) + hot water extractable C (HWEC) per unit total C in soil was highest in tilled arable lands (0.128) and orchard (0.113) followed by social/agro-forestry (0.104) and grasslands (0.048). Although, land use showed no significant impact on soil microaggregates, macroaggregate quantity followed the order: social/agro-forestry (523.8 g kg^{-1} of total aggregate amount)> grassland (469.9 g kg^{-1})> orchard (454.1 g kg^{-1})> cultivated lands (332.7 g kg^{-1}). Macroaggregate associated C was also found highest in soils of social/agro-forestry (61.5 g C $kg^{-1}$of macroaggregate), followed by grasslands (50.8 g kg^{-1}), orchard (49.5 g kg^{-1}) and cultivated lands (37.1 g kg^{-1}). Analysis of microbial biomass C, present within the water stable aggregates, showed the same trend of macroaggregate quantity with land use change, while it was not so for the soil enzymatic activities. Analysis of both the dehydrogenase and β-glucosidase enzymes showed their presence in the following order: cultivated soils> orchard> social/agro-forestry> grasslands. It was postulated that these enzymatic activities were controlled by the proportionate presence of labile C (WEC and HWEC) in soils.

Keywords: Aggregate occluded C, Enzymatic activities; Land use, Microbial biomass C, Soil aggregates

Introduction

Soil is the largest global terrestrial carbon (C) sink (Deb *et al.*, 2015). Carbon sequestration in soil largely depends on the aggregation as C is believed to be protected from microbial mineralization and enzymatic degradation within soil aggregates (Bronick and Lal, 2005). Earlier studies have indicated that soil aggregates can be of different sizes, which determines their C sequestration capacity (Deb *et al.*, 2018). In soil, attachment of organic molecule with clay and cations results in the formation of microaggregates, while several microaggregates join together by chelating agents to form macroaggregates (Deb *et al.*, 2019). Consequently, microaggregate occluded C gets two-fold protection and are found to be older and less labile than the macroaggregate associated C (Six *et al.*, 2000; Lichter *et al.*, 2008). The small pore size of microaggregates restricts the free microbial movements and activity, which are further reduced by the water-filled pores and reduced oxygen diffusion within these microaggregates (Bronick and Lal, 2005; Govaerts *et al.*, 2009).

Land use and land cover influences soil aggregation, while soil properties also affect sustainability of different land forms (Singh *et al.*, 2011, Deb *et al.*, 2015). Devi *et al.* (2013) and Bangroo *et al.* (2013) indicated the impact of natural forests, plantation and agro-forestry on soil properties and on C sequestration. Studies have indicated that macroaggregate associated C is more vulnerable to changes in land use than the C present in microaggregates (Puget *et al.*, 2000). Saha *et al.* (2012)

found higher soil aggregate occluded C in grasslands and forestlands than cultivated soils. The less aggregate protected C in surface soils of arable lands was due to tillage operations, which destroy soil aggregates and thus increase C mineralization (Six *et al*., 1999). Destruction of macroaggregates by tillage operation results in the predominance of microaggregates in agricultural soils, while the abundance of macroaggregates was found more in grasslands and forest soils (John *et al*., 2005).

The change in soil aggregation dynamics with land use change also influences the accessibility of aggregate occluded C to microbes (Deb *et al*., 2018). Although researches have revealed the impact of land use on soil C chemistry, very few have focused on the changes in aggregate occluded C and associated microbial dynamics. Therefore, the objectives of this study were to evaluate the impact of land use change on surface soil aggregation and aggregate occluded C, and to identify the alteration of microbial activities in macro and microaggregate fractions as a result of land use changes.

Materials and Methods

Study area: The study was conducted at the University Farm of Uttar Banga Krishi Viswavidyalaya (26° 23' N and 89° 23' E), Cooch Behar district of West Bengal, India. The bio-climate of the area is humid sub-tropical (Bhattacharyya *et al*., 2008). The area receives >3000 mm annual rainfall, while the mean annual maximum and minimum temperatures are 28.8°C and 19.3°C, respectively. Four common land use practices were selected for this study *viz*. cultivated land, orchard, social forest/agro-forestry and grassland (Table 1). All these land-use practices were located within a similar physiographic area, with the maximum distance of 500 m among them.

Soil analysis: Composite soil sampling (0-20 cm) and their analysis was done from the above mentioned 4 different land use systems during the year 2014-15. The soil samples were air-dried, ground and passed-through 2 mm sieve for the physico-chemical analysis. All the visible root pieces were removed. Soil pH was measured using a digital glass electrode pH meter (Systronics, model 335). Soil cation exchange capacity (CEC) was measured using 1.0 *N* ammonium acetate at pH 7.0 (Deb *et al*., 2016). Soil texture was measured by the international pipette method with an error of ±1% clay (Gee and Or, 2002).

Determination of soil C pools: Soil total C was estimated using a Vario EL III elemental analyser (Elementar, Germany) (Deb *et al*., 2016). For water extractable C (WEC), each sample was mixed in deionized water in a 50 ml centrifuge tube at a ratio of 1:10 followed by 30 minutes extraction at 20°C and centrifugation at 3000 rpm for 20 minutes (Ghani *et al*., 2003). After centrifugation, the supernatant was filtered through cellulose nitrate membrane filter paper (0.45 μm pore) to get the WEC. After removing the WEC, 30 ml of distilled water was added in each of the soil samples. The mixture was then shaken on a vortex shaker for 10 sec and was kept in a water-bath at 80°C for at least 16 hours. Subsequently, the solution was centrifuged for 20 minutes at 3000 rpm and finally filtered through cellulose nitrate membrane filter paper to get hot water extractable C (HWEC) (Ghani *et al*., 2003). The estimation of WEC and HWEC were done using Vario EL III elemental analyser (Elementar, Germany).

Determination of soil aggregation and aggregate occluded C: For aggregate analysis, 50 g air-dried soil sub-samples of 2.0-5.0 mm size were used. Yoder's wet sieving apparatus was used with nested sets of 6 sieves with the opening of 2.0, 1.0, 0.5, 0.25, 0.1 and 0.05 mm (Kemper and Rosenau, 1986). Before sieving, the soils were kept on 2.0 mm sieve and slaked by submerging it in deionized water. These soils were then physically fractionated by vertically moving the set of sieves up-and down within water for 30 minutes, 25 times per minute with 4 cm amplitude (Deb *et al*., 2018). Subsequently, the soil sub-samples were separated into water stable macro (> 0.25 mm) and microaggregates (0.25 -0.05 mm) (Liu *et al*., 2015). A soil sub-sample was passed through 0.05 mm sieve to get the 'silt + clay' fraction (< 0.05 mm). The sand content in each of the aggregate size classes was rectified by repeating the same experi-

Table 1. Details of land use practices under the study

Land use	Vegetation/ cropping systems
Cultivated land	Jute-rice-fallow, jute-maize-potatoFallow-maize-wheat, brinjal-pumpkin-cabbage
Orchard	Mango, litchi, guava, safeda
Social/agro-forestry	Sissoo, teak, gamar and rubber
Grassland	*Cyperus sp., Polygonum sp., Cynodondactylon, Panicum sp.*

-ment after dispersing with sodium hexametaphosphate (Deb *et al*., 2018). The water stable aggregate (WSA) present in each of the size fractions was calculated as:

$$\mathrm{WSA} = \frac{\text{weight of nondispersed aggregates} - \text{weight of dispersed soils}}{\text{total weight of aggregates}}$$

In addition, parameters like aggregate ratio (AR) and % aggregate stability (%AS) were also calculated as:

$$\mathrm{AR} = \frac{\%\text{ of WSA} > 0.25\ mm}{\%\text{ of WSA} < 0.25\ mm}$$

$$\%\ \mathrm{AS} = \frac{\%\text{ of soil particle} > 0.25\ mm - \%\ primary\ particle > 0.25\ mm}{\%\text{ primary particle} < 0.25\ mm}$$

Soil collected in each of the sieves was dried at 65°C and analysed in the Vario EL III elemental analyser (Elementar, Germany) for determination of total C associated with each aggregate size (Deb *et al*., 2018).

***Determination of soil aggregate associated microbial dynamics*:** Soil microbial biomass C (C_{mic}) and enzymatic activities were estimated for the aggregates of each size fractions. The modified dry-sieving method was used to separate soil aggregates with minimum disturbances to the microbial population and their activities (Wang *et al*., 2015). Freshly collected 50 g soil sub-samples of field-moist condition, sieved between 2.0-5.0 mm, were used for this purpose to further curb the impact of mechanical stress on soil microbes (Dorodnikov *et al*., 2009). In amplitude of 1 cm, a set of nested sieves of 2.0, 0.25 and 0.05 mm were vertically shaken for 2 minutes. As a result, the soil sub-samples were fractionated into macro (> 0.25 mm) and microaggregates (0.25 -0.05 mm) (Liu *et al*., 2015). These aggregate fractions were further used for the analysis of microbial parameters.

Chloroform fumigation technique was used to quantify C_{mic} (Vance *et al*., 1987). The magnitude of the difference of soil C between the fumigated and non-fumigated soil samples was converted to C_{mic} as per Voroney and Paul (1984). β-glucosidase activity was measured by treating soil samples with toluene, modified universal buffer (at pH 6.0) and p-nitrophenyl ß-D-glucoside solution and then incubating at 37°C. The solution was then filtered after adding $CaCl_2$ solution and tris buffer (at pH 12.0). The absorbance of the p-nitrophenyl (pNP) in the filtrate was measured at 440 nm (Dick *et al*., 1996, Lebrun *et al*., 2012) using a spectrophotometer (Shimadzu UV-1800). To assay dehydrogenase enzyme activity, soil samples were incubated with 2, 3, 5-triphenyltetrazolium chloride substrate at 37°C. The concentration of triphenylformazan (TPF) was measured colorimetrically at 485 nm (Tabatabai, 1994; Wu *et al*., 2013). While standard curves were prepared for each of the enzymes separately to convert the absorbance values to μg substrate g^{-1} soil h^{-1}, control was performed to eliminate all the variables except the tested one.

Results and Discussion

Soil analysis indicated the acidic nature of the soils (pH: 5.41). Soil texture, mainly the clay content was important as it controls C sequestration capacity (Deb *et al*., 2016). This study found, however, no significant changes in soil clay content across different land uses. The soil CEC varied from 4.99 to 6.85 cmol (P^+) kg^{-1}.

***Soil C pools*:** Comparison of the soils under different land uses indicated highest soil total C under the social/ agro-forestry systems (6.35 g kg^{-1}), followed by orchard (4.05 g kg^{-1}) and grasslands (4.03 g kg^{-1}), respectively. The lowest soil C status was observed in arable agricultural lands (3.07 g kg^{-1}). This trend was similar to the earlier studies (Chaturvedi *et al*., 2016; Soinne *et al*., 2016). Tillage operations possibly were the principle reason behind the rapid microbial oxidation of C and consequential lowest C status in cultivated soils (Blanco-Canqui and Lal, 2008). In the agricultural (0.25 g kg^{-1} Yr^{-1}) and orchard lands (0.14 g kg^{-1} Yr^{-1}), C was added in soils through manuring, while no such practices were common under social/agro-forestry and grasslands. Highest C addition (in terms of manure) was observed in the jute-maize-potato followed by fallow-maize-wheat cropping system. This was probably due to the presence of maize, a heavy feeder crop, in crop rotation (Verma and Minhas, 1987). On the contrary, C input in soils in terms of plant residue was found highest under social/agro-forestry (2.13 g kg^{-1} Yr^{-1}), followed by orchards (1.08 g kg^{-1} Yr^{-1}), grasslands (0.73 g kg^{-1} Yr^{-1}) and cultivated lands (0.71 g kg^{-1} Yr^{-1}). This was a due to continuous biomass input in the forest and orchard floors through leaf and litter fall and by root exudates and subsequent higher organic matter build-up (Berg, 2000; Wiesmeier *et al*., 2014).

Land use influences vegetation type, quantity and quality of C input in soil and thus soil labile C pool (Chantigny, 2003). To understand this labile C dynamics, soil WEC and HWEC analysis were done (Hedges, 2002; Callesen *et al*., 2003). These natural pools of soil C represent the labile fraction of C, readily available for soil biological processes (Uchida *et al*., 2012). For both these water-soluble fractions, the lowest amount of C was observed in grassland soils (WEC: 51.6 mg kg^{-1}, HWEC: 143.2 mg kg^{-1}). The highest amount of WEC was present in social/ agro-forestry soils (315.7 mg kg^{-1}), followed by soils of orchards (216.8 mg kg^{-1}) and arable lands (157.1 mg kg^{-1}). The trend of soil HWEC was also similar i.e. the highest amount of HWEC was in soils of social/agro-forestry systems (350.9 mg kg^{-1}), followed by soils of orchard (239.0 mg kg^{-1}) and agricultural lands (238.6 mg kg^{-1}). It indicated that the magnitude of soil total C predominantly controlled these water-soluble labile C pools (except grasslands) as found earlier by Jinbo *et al.* (2006). However, ratio of (WEC + HWEC) with soil total C (under different land uses) indicated that the relationship of total C and these water soluble C was not linear as found by Jinbo *et al.* (2006). The relative highest presence of soil labile C was found in cultivated lands (0.128) (Fig 1). Similar findings were reported earlier by Hu *et al.* (1997). While soils of orchard and social/agro-forestry systems had a lesser proportion of labile C (0.113 and 0.104, respectively), the lowest was observed under grassland soils (0.048). However, few earlier studies recorded a different trend *i.e.* higher WEC and HWEC in grassland soils in comparison to cultivated soils (Haynes, 2000; Chantigny, 2003). Possibly management practices, mainly quality of the added organic C by manuring was responsible for higher labile C in these cultivated soils (Hu *et al.*, 1997). The highest labile C (WEC + HWEC) to total C ratio in soils under jute-maize-potato followed by fallow-maize-wheat (Fig 1) supported it as these cropping systems had received the highest C as manure.

Soil aggregates and aggregate occluded C: Soil aggregation dynamics was observed by the quantitative presence of WSA in different aggregate size fractions (Deb *et al.*, 2019). These WSA in soil act as a protective cover to C, encapsulated within, from soil microbes (Saha *et al.*, 2011). The pore geometry of these aggregates restricts the physical movements of soil microbes (Kravchenko *et al.*, 2015; Deb *et al.*, 2019). No significant changes in the number of soil microaggregates were observed with changes in land uses (Fig 2a). However, land use affected soil macroaggregate amount. This might be due to the higher sensitivity of macroaggregates to land use change in comparison to microaggregates (Puget *et al.*, 2000). Highest macroaggregate formation was observed in soils under social/agro-forestry (523.8 g kg^{-1} of total aggregate amount), followed by grasslands (469.9 g kg^{-1}), orchards (454.1 g kg^{-1}) and lowest in agricultural soils (332.7 g kg^{-1}). The highest and lowest soil aggregate formation under social/agro-forestry and cultivated lands, respectively, was also supported by aggregate indices like AR (4.92: social/ agro-forestry; 3.07: cultivated lands) and % AS (16.0: social/ agro-forestry; 6.2: cultivated lands) (Fig 2b). This was possibly due to undisturbed nature of forestry and grassland soils but breaking down of soil macroaggregates in arable cultivated lands due to tillage operations (Ghosh and Mahanta, 2014; Singh *et al.*, 2019). While the high organic C, polysaccharides (as cementing agents) and network of fungal hyphae were responsible for the high macroaggregate formation in forest soils, the strong grip by grassroots was the possible reason of aggregation under grasslands (An *et al*, 2010). Although a good amount of C accumulation was observed in orchard soils, periodic tillage operations might have affected aggregation dynamics in these soils (Vanwalleghem *et al.*, 2011). Among the social/agro-forestry systems, highest soil macroaggregates were found under rubber (544.6 g kg^{-1}) and sissoo plantation (543.4 g kg^{-1}), while soils under teak plantation (496.2 g kg^{-1}) had lowest macroaggregates (Fig 2a). While the highest AR was found in soils of sissoo plantation (5.73), rubber soils had the highest % AS (18.5) (Fig 2b). In cultivated lands, jute-maize-potato cropping system resulted in lowest macroaggregate formation (312.6 g kg^{-1}) as well as lowest AR (2.44) and % AS (5.8). Similar to earlier studies, this study indicated no significant impact of tillage and other cultivation practices on soil microaggregates stability, possibly due to the lesser impact of tillage and other anthropogenic activities on strongly cemented microaggregates (Six *et al.*, 1999; Six *et al.*, 2004).

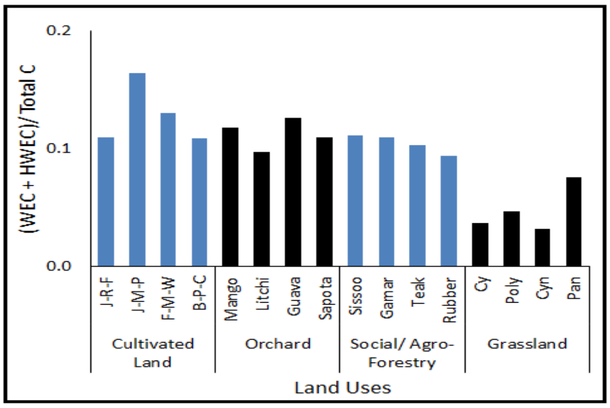

Fig 1. Ratio of water soluble C fraction (water extractable carbon (WEC) and hot water soluble carbon (HWEC) to soil total C. In X-axis, J-R-F: Jute-rice-fallow, J-M-P: Jute-maize-potato, F-M-W: Fallow-maize-wheat, B-P-C: Brinjal-pumpkin-cabbage under cultivated land and Cy: *Cyperus sp.*, Poly: *Polygonum sp.*, Cyn: *Cynodon dactylon*, Pan: *Panicum sp.* under fallow grassland

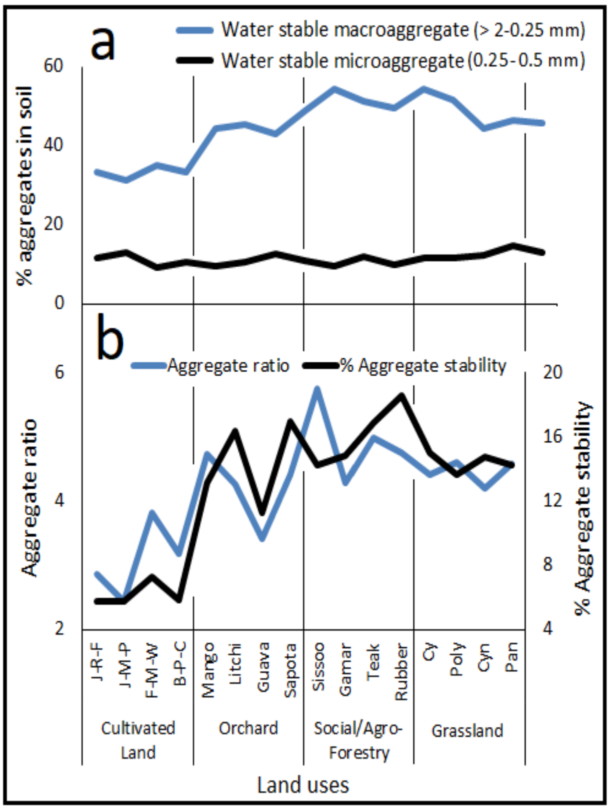

Fig 2. Frequency polygon of (a) water stable macro and microaggregates in soils and (b) aggregate indices under different land uses

Analysis of C occluded within per unit of aggregate indicated higher presence of C in soil macroaggregates (> 2-0.25 mm) (49.7 g kg^{1}) than microaggregate fractions (0.25-0.05 mm) (4.9 g kg^{1}) (Fig 3). These results were in agreement with the concept that microaggregates holds soil C within them very strongly resulting nearly inaccessible organic matter for soil microbes (Blanco-Canqui and Lal, 2008; Deb *et al*., 2019). Next, numbers of microaggregates join together to form macroaggregate, where transient organic matters (like microbial and plant-derived polysaccharides, fungal hyphae) act like glue (Six *et al*., 2000). Thus the amount of C within larger soil aggregates comprises microaggregate occluded C and the C which binds those microaggregates to macroaggregates (John *et al*., 2005).

Considering all the macroaggregate size fractions together, amount of C within per unit of soil aggregate was found highest under social/agro-forestry systems (61.5 g C kg^{1} of macroaggregate). This was followed by macroaggregate associated C of grasslands (50.8 g kg^{1}), orchard (49.5 g kg^{1}) and agricultural lands (37.1 g kg^{1}) (Fig 3). This order remained the same or altered slightly while considering macroaggregates of different size fractions individually. For example, this sequence remained same for > 2 mm, 2.0-1.0 mm and 0.5-0.25 mm size fractions. However, in the 1.0-0.5 mm size fraction, highest aggregate occluded C in social/agro-forestry soils were followed by soils of orchard and grasslands, respectively. The no-till condition (forestry and grasslands) or only periodic tillage (orchard) and high C input (as leaf and litter fall and through root exudates) in social/agro-forestry, orchard and grasslands was the possible reason behind the high aggregated C under these land uses (Wiesmeier *et al*., 2014). In case of grasslands, absence of any disturbing management practice and the continuous dense root system of grasses were also responsible for stable soil aggregate formation and higher occlusion of C within aggregates (Saha *et al*., 2012). In all these macroaggregate size fractions, soils of arable cultivated lands showed distinctly less aggregate associated C. It suggested that tillage operation not only altered the soil macroaggregates but also curbed C occlusion within aggregates of different sizes, as recorded earlier (John *et al*., 2005; Ghimire *et al*., 2012). No specific impact of land use was found on microaggregate occluded C fractions. In 0.25-0.1 mm size, highest C was found in unit amount of orchard soil aggregates whereas social/agro-forestry soils had maximum aggregated C (per unit) for 0.1-0.05 mm size fraction. For microaggregates, the quantitative difference of aggregated C with a change in land use was much less in comparison to soil macroaggregates. It clearly indicated that the impact of land use was more in soil macroaggregate associated C than the C occluded within soil microaggregates (John *et al*., 2005; Ghimire *et al*., 2012). The amount of C associated with silt + clay size fraction (< 0.05 mm) was similar for grasslands, orchard and social/agro-forestry systems. Thus it can be stated that land use affected soil macroaggregate formation and stability but had no significant impact on microaggregate dynamics. Soil macroaggregates and macroaggregate associated C followed the order: social/agro-forestry> grassland> orchard> cultivated lands.

Soil aggregate associated microbial dynamics: For soil microbial analysis, aggregate fractions (as obtained by dry-sieving) were grouped into 2 distinct parts *viz.* macroaggregates (> 2.0-0.25 mm) and microaggregates (0.25-0.05 mm). A significantly higher amount of C_{mic} was found within soil macroaggregates in comparison to the microaggregate fractions (Fig 4). Analysis of soil enzymatic activities also indicated a similar trend *i.e.* the higher enzymatic activities within soil macroaggregates in comparison to microaggregates. This was possibly due to the different degree of physical protection by macro and microaggregates to soil C, which serves as the food and nutrient source for the microbes (Six *et al.*, 2004; Deb *et al.*, 2019). The loosely bound C within soil macroaggregates is easily accessible for soil microbes resulting in higher microbial biomass and activities within those aggregates (Six *et al.*, 2004). On the contrary, smaller pores and complex pore geometry of microaggregates resulted microbially inaccessible C (Kravchenko *et al.*, 2015). Partial or completely water-filled micropores and reduced oxygen diffusion might had created anaerobic condition in some microaggregates, resulting further decelerated oxidation of C residing within (Govaerts *et al.*, 2009).

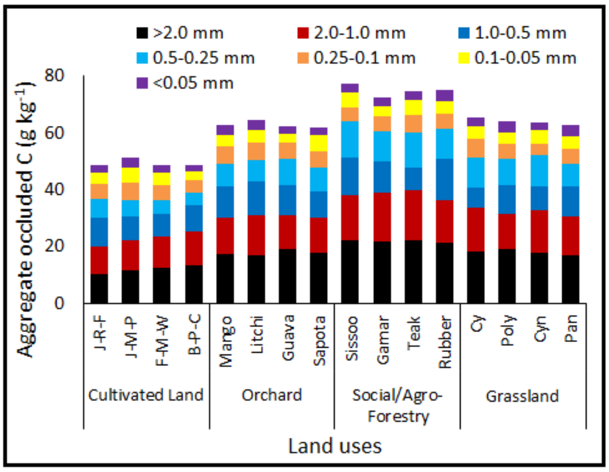

Fig 3. Distribution of soil C in different aggregate size fractions under diverse land uses

Impact of different land use systems on soil aggregate associated microbial dynamics was found to be very prominent (Fig 4). In soil, C_{mic} represents microbial population and biomass (Deb *et al.*, 2016). The C_{mic} was found highest in per unit soil macroaggregates under social/agro-forestry (0.68 mg kg^{-1}), followed by grassland soils (0.64 mg kg^{-1}). The C_{mic} status of arable cultivated soils, as well as orchard soils, was comparatively less. It indicated that C_{mic} was positively correlated with the total C content of macroaggregates under each of the land use classes.

Among the soil extracellular enzymes, dehydrogenase is involved in microbial oxidative activities and it is considered as a biomarker of soil conditions under different land uses (Caravaca *et al.*, 2002). Contrariwise, β-glucosidase is considered to be an indicator of microbial capacity to breakdown polysaccharides (Caldwell, 2005; Salazar *et al.*, 2011). Both of these enzymes are significantly related to soil C cycle. Some of the earlier studies indicated low activities of these enzymes in tilled-arable soils with low C status in comparison to high C soils under undisturbed natural land uses (Caravaca *et al.*, 2002; Nsabimana *et al.*, 2004). However, outcomes of this study indicated no correlation of these enzymes with soil total C or macroaggregate occluded C status. Highest dehydrogenase activities were observed in macroaggregates of arable soils (7.1 μg^{-1} TPF g^{-1} soil h^{-1}) followed by orchard soils (4.9 μg^{-1} TPF g^{-1} soil h^{-1}). Within cultivated lands, highest dehydrogenase activity was found under fallow-maize-wheat cropping system (10.8 μg^{-1} TPF g^{-1} soil h^{-1}). Macroaggregate associated β-glucosidase activities were found high in both cultivated and orchard soils (116.9 and 103.7 μg^{-1} pNP g^{-1} soil h^{-1}, respectively). Soils of social/agro-forestry and grasslands exhibited relatively low dehydrogenase as well as β-glucosidase activities. These trends were similar for the soil microaggregates. Although, these enzyme activities got increased with decrease in soil total and aggregate occluded C, positive correlations of these enzymes were observed with soil labile C pools especially with the presence of WEC and HWEC in per unit of soil total C. Earlier studies also observed similar trend (Rees and Parker, 2005; Salazar *et al.*, 2011). Possibly the reason was presence of the organic substrates predominantly in labile forms.

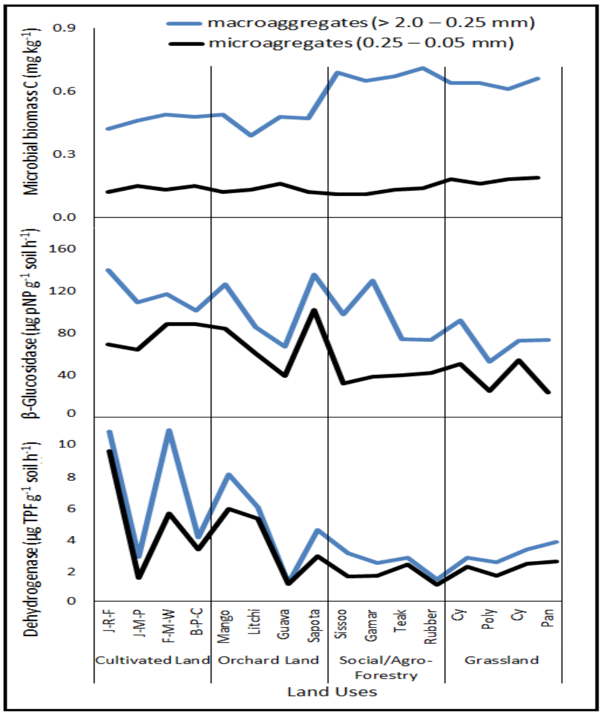

Fig 4. Frequency polygon of soil microbial biomass C and enzymatic activities in different soil aggregate size fractions

Therefore, a positive correlation of soil C_{mic} was found with aggregate occluded C. While enzymatic activities were positively influenced by the higher share of WEC and HWEC in unit amount of soil total C. In a nutshell, soil aggregate associated C_{mic} quantitatively followed the trend: social/agro-forestry> grassland> orchard = cultivated lands, whereas enzymatic activities followed: cultivated landse" orchard> social/agro-forestry> grassland.

Conclusion

This study very evidently indicated the extent of variation in soil C status, aggregation, aggregate associated C and microbial dynamics with the change in land use. Higher presence of soil macroaggregates and macroaggregated C were observed in undisturbed natural ecosystems than in arable tilled soils. Possibly occurrence of macroaggregated C also influenced the presence of soil C_{mic}, as the C is the food and energy source of all soil microbes (Deb *et al*., 2018). Interestingly, soil enzymatic activities were not inclined to the same direction. Rather a higher concentration of dehydrogenase and β-glucosidase were observed in cultivated and orchard soils. This study inferred that neither soil total C nor aggregated C, but the proportion of labile C (WEC and HWEC) in total C controlled the soil enzymatic activities. It indicated the complex and dynamic relationship of soil C and microbes and the difficulty to conclude about their any specific trend. Indeed, it requires further studies to understand the regulating capacity of C and microbes on soil enzyme dynamics under different land use ecologies.

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