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## Soil organic carbon storage, ecosystem dynamics and climate change: current agro-environmental perspectives and future dimensions: A review

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### Abstract

Soil is the largest organic carbon (C) pool of terrestrial ecosystems, and C loss from soil accounts for a large proportion of land-atmosphere C exchange. Therefore, soil organic carbon (SOC) plays an important role in the carbon cycling of terrestrial ecosystems, deviations in SOC stocks are very important for the ecosystem. Globally, soil organic matter (SOM) comprises more than three times as much C as either the atmosphere or terrestrial vegetation. Yet it remains mainly unknown why some SOM persists for ages whereas other SOM decomposes readily and this limits our ability to predict how soils will respond to climate change. Soils form via multiple interactions of various forces, including climate, organisms, parent material, all acting over time. It takes centuries for a soil to form and many of the soils are still evolving subsequent changes due to various soil forming factors, mainly climate and vegetation, in last few decades. Climate is one of the main factors influencing the formation of soil with important implications for their advancement, use and management perspective with reference to soil structure, stability, water holding capacity, availability of nutrients and erosion. Further Indirect effects corresponds to changes in growth rates or water-use efficiencies, through sea-level rise, through climate-induced decrease or increase in vegetative cover or anthropogenic intervention. Assuming constant inputs of carbon to soils from vegetation, different estimate predict that expected changes in temperature, precipitation and evaporation cause significant change in organic matter turnover and CO<sub>2</sub> dynamics.

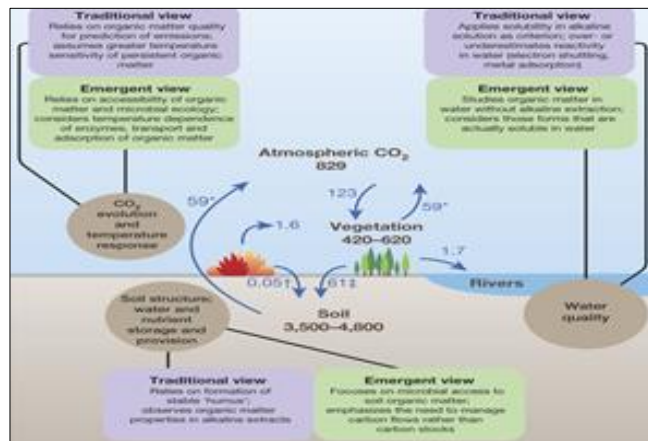
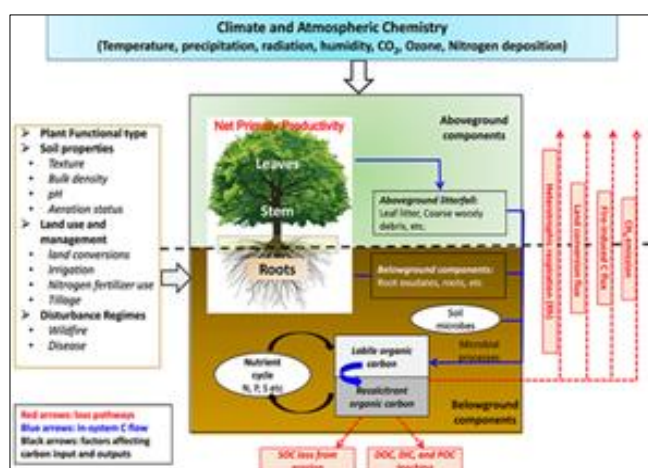
**Keywords:** Carbon accumulation, soil organic carbon, carbon storage, carbon sequestration

### Introduction

Increasing global interest regarding climate change, increased interest in carbon (C) sequestration in agricultural soil. Soil is not only the foundation of crop production but also the key originator of C sequestration in geological ecosystems. As the largest C pool in the geological ecosystem, soil has high ecological value. A carbon dioxide (CO<sub>2</sub>) emission takes place from C sequestration in the geological ecosystem. Furthermore, SOC and CO<sub>2</sub> can be mutually converted. If soil ecosystem gets demolished, then decomposition rate of organic C in the soil gets accelerated, greenhouse gas emissions increases, and the greenhouse effect gets intensified, which eventually results in global warming (Forte *et al.*, 2017; Krauss *et al.*, 2017) [22, 37]. In addition, soil aggregate stability and SOC are important indicators of soil quality and environmental sustainability in agro-ecosystems. Firstly, the decomposition and transformation of SOC were affected by aggregation construction (Zhao *et al.*, 2018) [83]. It was reported that stable aggregates, physically prevent quick decay of SOC (Sun *et al.*, 2018) [71]. Secondly, SOC was considered as main binding agent contributing to aggregate stability (Bhattacharyya *et al.*, 2011; Naresh *et al.*, 2017) [52]. Furthermore, the SOC content in macro-aggregates was an important index of soil aggregate stability and C loss, influenced by numerous management methods (Sheehy *et al.*, 2015) [65]. For example, the quantity and stabilization mechanisms of SOC, as related to soil aggregates, are influenced by tillage practices (Crittenden *et al.*, 2015) [14].

Numerous studies revealed that unreasonable tillage over the long period destroy the soil ecosystem, including its soil aggregate stability, porosity and nutrients, causing a chain of changes in the physical and chemical characteristics of soil (Crittenden *et al.*, 2015; Buchi *et al.*, 2017; Naresh *et al.*, 2018) [14, 8, 51]. Conservation tillage, such as the adoption of a no-tillage regime, can increase soil, macro-aggregates formation and stability and offer a good protective

effect for SOC (Kumar *et al.*, 2018; Dai *et al.*, 2015; Naresh *et al.*, 2018) [38, 15, 51], as compared with conventional tillage systems. Increased productivity would generally lead to greater inputs of C to soil, thus increasing organics. SOC stocks are altered by biotic activities of plants (the main source of C through litter and root systems), microorganisms (fungi and bacteria) and ecosystem engineers (earthworms, termites, ants). In the interim, abiotic processes related to the soil-physical structure, porosity and mineral fraction also modify these stocks. By acting on both biotic and abiotic mechanisms, land use and management practices (choice of plant species and density, plant residue exports, amendments, fertilisation, tillage, etc.) drive soil spatiotemporal organic inputs and organic matter sensitivity to mineralisation. A brief review of SOC storage, its dynamics and with climate change and future implications is summarized below;



### Soil organic carbon and climate change

Climate change is both a cause and an effect of ecosystem dynamics. Along with anthropogenic dispersion, climate change is the main driver of change in the geographical distribution of both beneficial and harmful species of crops, livestock, harvested wild species, pests, predators and pathogens. And the capacity of ecosystems to adapt to climate change depends on the diversity of species they currently support. Climate change is also a consequence of the way which biological resources are converted into useful goods and services, and especially of the way in which forests are converted into croplands. The production of biological resources for foods, fuels and fibres, and the conversion of forests for agriculture both directly affect emissions of several greenhouse gases (GHGs). Changes in stocks of biomass also affect the volume of sequestered C. It follows that options for

the mitigation of climate change include the management of both GHG emissions from productive processes and C sequestration, while options for adaptation to climate change centre include the management of ecosystem resilience. Understanding soil biogeochemistry is essential to the stewardship of ecosystem services provided by soils, such as soil fertility (for food, fibre and fuel production), water quality, resistance to erosion and climate mitigation through reduced feedbacks to climate change. Soils store at least three times as much C (in SOM) as is found in either the atmosphere or in living plants [Fischlin *et al.*, 2007] [21]. The major pool of organic C is sensitive to changes in climate or local environment, but how and on what timescale will it respond to such changes? The feedbacks between soil organic C and climate are not fully understood, so we are not fully able to answer this questions [Heimann and Reichstein, 2008; von Lutzow and Kogel-Knabner, 2009] [29, 78], but we can explore those SOC cycling. We can not only simulate feedbacks between climate change and ecosystems, but also evaluate management options and analyse C sequestration and ecosystem dynamics strategies. These management practices, however, rest on some assumptions that have been challenged and even disproved by recent research arising from new isotopic, spectroscopic and molecular marker techniques and long-term field experiments. Here we describe how recent evidence has led to a framework for understanding SOM cycling, and we highlight new approaches that could lead us to a new generation of soil C storage, which could better reflect observations and inform predictions and policies.

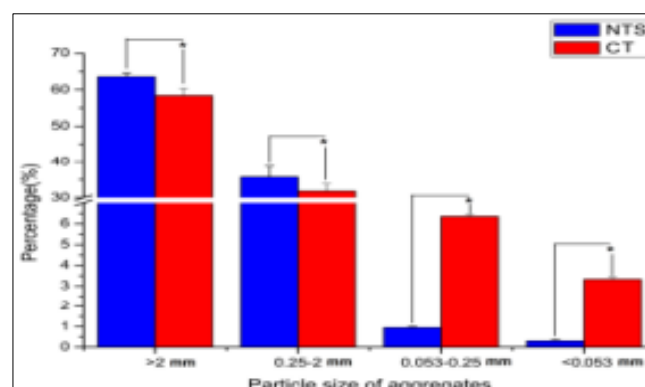


Fig 1(a): Effect of tillage treatments on soil aggregates size distributions

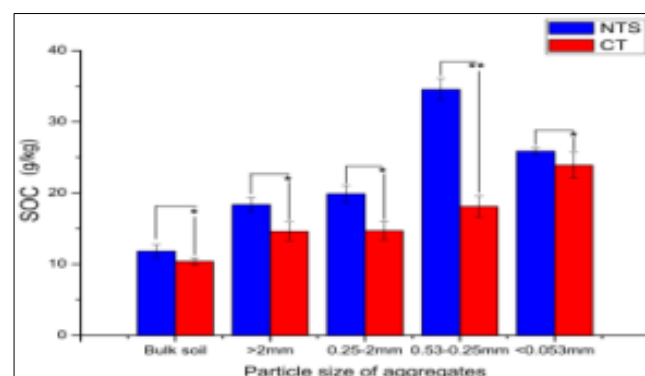
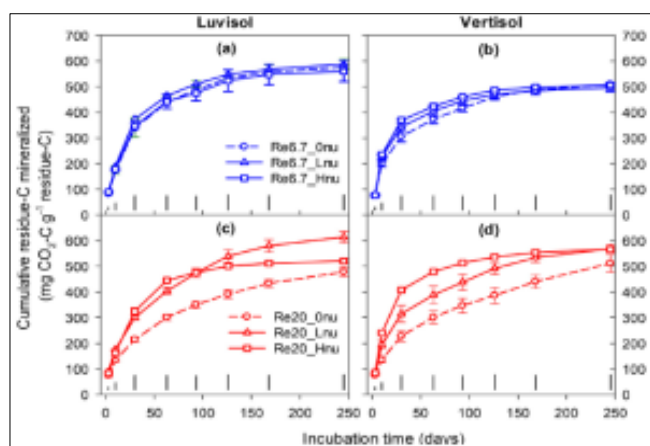


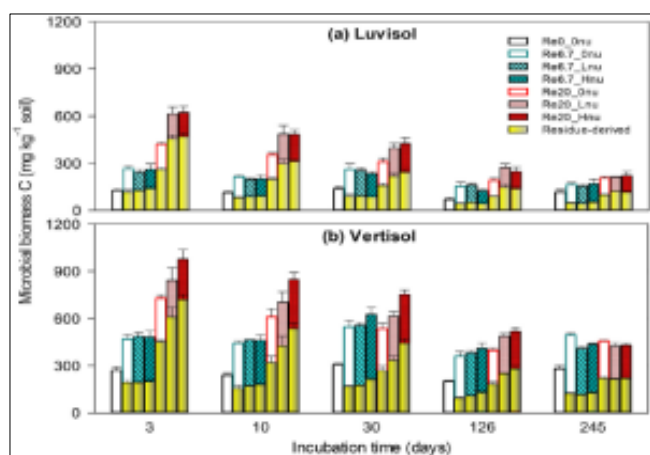
Fig 1(b): The SOC (soil organic C content) in bulk soil and its aggregations under no tillage with crop straw returning (NTS) and conventional tillage without the crop straw (CT) treatments

Fang *et al.* (2018) [20] reported that over the 245-day incubation, the cumulative total C mineralized ranged between 280 and 5840 mg kg<sup>-1</sup> soil, which increased with the

increasing levels of residue input,  $Re\ 0 < Re\ 6.7 < Re\ 20$ , in the Luvisol and Vertisol (Fig.2a). On average cumulative total C mineralized was 4–20% higher in the Luvisol than Vertisol. In the Luvisol with high residue (Re 20), the high-nutrient supply slowed the C mineralization rapidly, that is, the amount of cumulative total C mineralized changed from  $0nu < Lnu \leq Hnu$  to the order of  $0nu < Hnu < Lnu$  after day 90. In the Vertisol, the amount of cumulative total C mineralized was in the order of  $0nu < Lnu \leq Hnu$  in the Re20 treatment and the difference across the three levels of nutrient supply decreased over time. However, the total C mineralization over the 245-day incubation period, the proportions of  $CO_2$ -C derived from residue ranged from 32.0 to 96.8% in the lower residue input and 43.3–98.7% in the higher residue input (Fig. 2b). In the Luvisol, with the high-residue input, cumulative residue- C mineralized ( $mg\ CO_2\text{-C}\ g^{-1}\ residue\text{-C}$ ) was in the order of  $0nu < Lnu \leq Hnu$  over the first 90 days, which turned to the order of  $0nu < Hnu < Lnu$  afterwards. In the Vertisol, with the high-residue input, the amount of cumulative residue-C mineralized was in the order of  $0nu < Lnu < Hnu$  over the first 126 days, and, afterwards, there was no difference in residue-C mineralized between the different levels of nutrient supply.



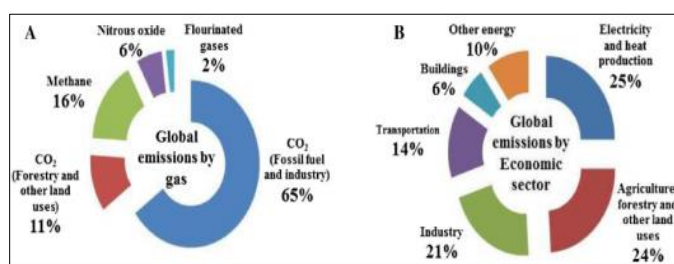
**Fig 2(a):** Cumulative total C mineralized ( $mg\ CO_2\text{-C}\ kg^{-1}\ soil$ ) in the Luvisol (a) and Vertisol (b) over the 245-day incubation period. Abbreviations: Re0=no residue input; Re 6.7=residue input at  $6.7\ kg^{-1}\ soil$ ; Re 20=residue input at  $20\ kg^{-1}\ soil$ ; 0nu=without nutrient input; Lnu=low nutrient input; Hnu=high nutrient input



**Fig 2(b):** Cumulative residue C (C) mineralized ( $mg\ CO_2\text{-C}\ g^{-1}\ residue\text{-C}$ ) in the Luvisol (a, c) and Vertisol (b, d) over the 245-day incubation period.

Srivastava *et al.* (2016) [67] revealed that C mineralization from the integrated residue–nutrient management indicated

that nutrient availability was not a limiting factor of mineralization of residue-C under a certain residue-input threshold. The lack of effects of nutrients on residue-C mineralization at the low-residue input was likely due to the following reasons. Firstly, according to the nutrient stoichiometry of residue and SOM, up to  $\sim 7\text{--}58\ mg\ N$ ,  $\sim 3.0\text{--}37.3\ mg\ P$  and  $\sim 0.3\text{--}6.4\ mg\ S\ kg^{-1}\ soil$  may be released following mineralization of residue-C and native SOC over time (Fig.3). Further, the low-molecular-weight-organic acids in soil produced during residue decomposition (Küsel and Drake, 1998) [39] may have also expedited the release of mineral-bound SOM through dissolution of protective mineral phases, with potential to enhance nutrient release from SOM (Sarker *et al.*, 2018) [63]. Additionally, microorganisms can directly utilize significant amounts of organic-N compounds from plant residue or microbial products (Geisseler *et al.*, 2010) [24]. Consequently, nutrients released or obtained from the residue and soil reserves were sufficient at the low-residue input to support microbial growth and activity.



**Fig 3:** Major greenhouse gases (a) and their economic sector – wise (b) distribution [adapted from IPCC, 2014; USEPA 2014] [31, 76]

Srivastava *et al.* (2016) [67] also found that the various pools and fluxes of C on the Earth are depicted in (Fig. 4a). SOC plays a major regulatory role in biogeochemical cycles and biosphere functioning due to its complex interaction with soil physical, chemical and biological factors and thereby its effect on the soil multi-functionality. SOC plays a central role in defining the soil quality and agro-ecosystem productivity (Lal 2003) [41]. It is identified that an increase in SOC may help in alleviating the soil degradation and, thus, ensures sustainable food production for the growing world population (Swift, 2001) [72]. Therefore, there has been an additional interest in the role of SOC as a potential sink for atmospheric  $CO_2$  (Post and Kwon, 2000) [57]. On the other hand, Trumbore, (1997) [74] reported that the turnover rate of labile C pool ranges from several weeks to months or years, whereas recalcitrant, non-labile pools have a turnover rate of centuries to millennia (Fig 4b). Labile C pool comprises easily oxidizable forms of SOC such as microbial biomass C (MBC), water-soluble C (WSC), particulate organic C (POC), which responds more rapidly to the agro-management practices (Purakayastha *et al.*, 2008) [59]. The decomposition of labile C by soil microorganisms helps to stabilize the soil aggregates, releases nutrients by mineralization and provides food for soil microbial activity. Thus, it can serve as the sensitive indicator of soil quality (Purakayastha *et al.*, 2008) [59]. Additionally, it plays an important role in improving soil quality and thus can minimize the negative environmental impacts. The amount of these active fractions and their proportion to total SOM are the good indicators of soil health. Therefore, the identification of highly sensitive SOC fractions may help to elucidate changes and trajectories in the SOC pool at early stages of changes in land use and management (Yang *et al.*, 2009) [82].

Active SOC pool plays a very different role than passive does. As labile (active) and non-labile (stable) SOC play differential roles in SOM dynamics and nutrient cycling, the pool size and turnover time of these two fractions may be important to assess and evaluate the soil management practices for the monitoring of SOC (Srivastava *et al.*, 2016) [67]. Organic amendments have been proposed as a means to increase the soil C storage (Powlson *et al.*, 2012) [58]. It may occur directly from the C inputs in the form of organic amendments and indirectly due to increased plant production due to amendment (Ryals and Silver 2013) [62]. It has also been found related to the enhanced soil water-holding capacity, decreased bulk density and improved soil fertility (Lynch *et al.*, 2005) [46]. It influences the soil microbial biomass, activity and community (Jimenez *et al.*, 2002) [34]. Soil microbial biomass and activity has been found higher in the organic than conventional management system (Tu *et al.*, 2006) [75]. Moreover, integrated use of fertilizers has been reported to accumulate higher SOC and its fractions as compared to sole chemical fertilization in some studies (Lal, 2003) [41].

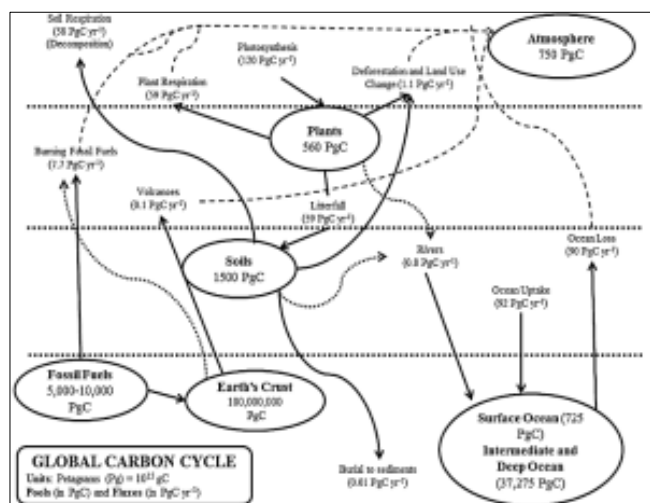


Fig 4(a): Global C cycle depicting various pools and fluxes of C [adapted from Houghton, 2007]

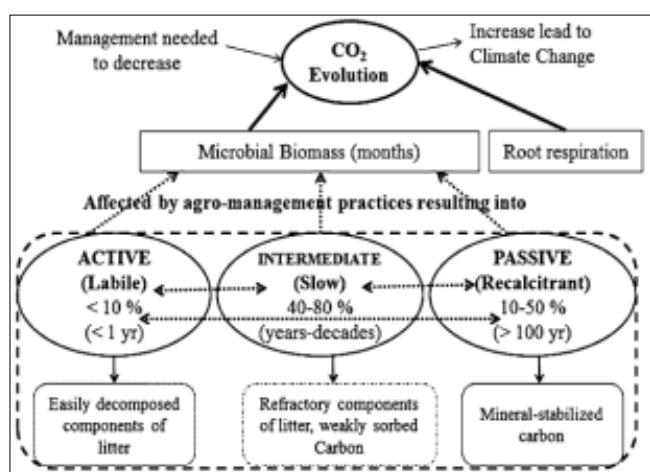


Fig 4(b): Characteristics of different soil organic C pools and soil CO<sub>2</sub> efflux (from Trumbore 1997) [74]

and resources with the global ecosystem within which it is placed. The global ecosystem provides energy and resources to the economy (source functions), and absorbs, stores or recycles the energy and waste produced by the economy (sink functions). The global ecosystem has solar energy as an input and waste heat as an output; other than that, it is a closed system. In current models of economic growth, as the economic system grows within the global ecosystem it requires more resources and energy and generates more waste making it more difficult for the global ecosystem to perform its source and sink functions (Fig 5a). In parallel, some activities within the economic system affects the ecosystem's ability to perform its source and sink functions, both positively (e.g. technology) or negatively (e.g. pollution or destruction of ecosystems). The fixed size and closed nature of the planetary ecosystem imposes a limit to the resources and energy that can be sourced from the ecosystem and also imposes a limit to the amount of waste it can absorb, store or process. In sum, the economy cannot expand beyond the confines of ecological limits.

Regional-level boundaries as well as globally aggregated PBs have now been developed for biosphere integrity (earlier "biodiversity loss"), biogeochemical flows, land system change, and freshwater use. Steffen *et al.* (2015) [68] observed that three of the planetary boundary [PBs (climate change, stratospheric ozone depletion, and ocean acidification)] remains essentially unchanged from the earlier analysis. At present, only one regional boundary can be established for atmospheric aerosol loading. Although we cannot identify a single PB for novel entities they are included in the PB framework, given their potential to change the state of the Earth system (ES). Two of the PBs—climate change and biosphere integrity—are recognized as "core" PBs based on their fundamental importance for the ES. The climate system is a manifestation of the amount, distribution, and net balance of energy at Earth's surface; the biosphere regulates material and energy flows in the ES and increases its resilience to abrupt and gradual change. Anthropogenic perturbation levels of four of the ES processes/features exceed the proposed PB (Fig.5b).

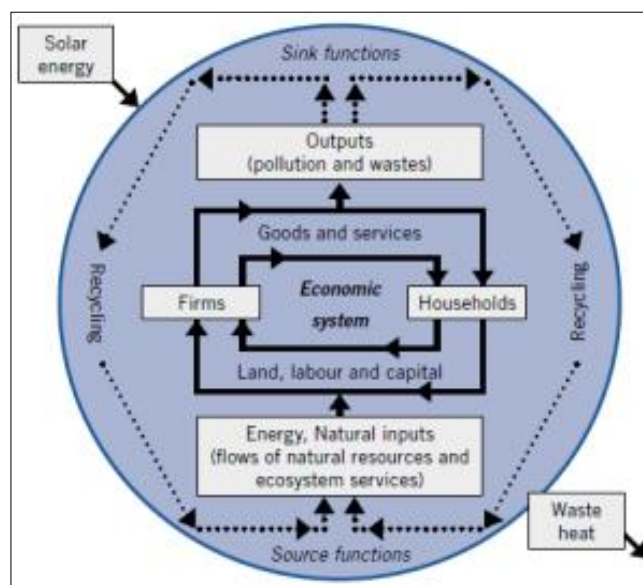
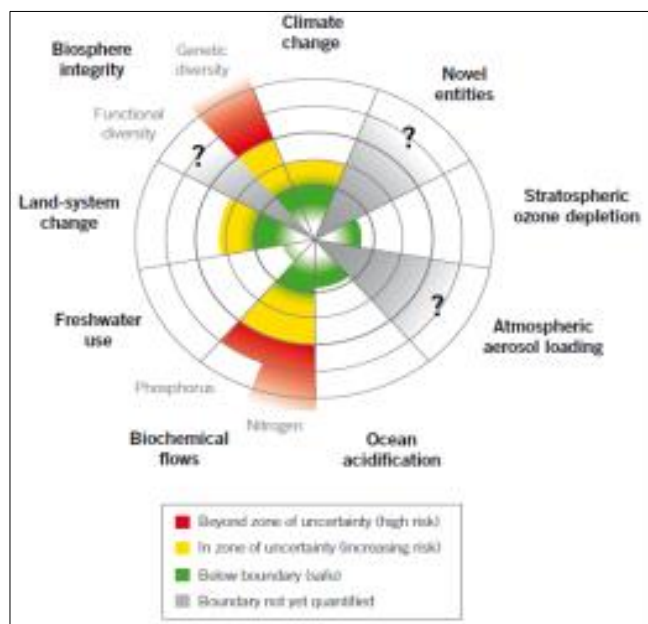


Fig 5(a): The economy as a subsystem of the global ecosystem [Source: Harris and Roach, 2017] [27]

### Ecosystem dynamics and climate change

In a study, Harris and Roach, (2017) [27] reported that the economic system is an open system as it exchanges energy



**Fig 5(b):** Current status of the control variables for seven planetary boundaries [Source: Steffen *et al.*, 2015]<sup>[68]</sup>

Changed climatic conditions, crop plants suffer from adverse environmental conditions such as high temperature (or temperature changes from freezing to scorching), drought (Water stress), variable light conditions that affect photomorphogenetic responses, and nutrient deprivation in soil, which directly influence the growth, morphology, physiology, and developmental aspects of plants (Aamir *et al.*, 2019)<sup>[1]</sup>. The most detrimental effects of the increased CO<sub>2</sub> level in the atmosphere on plants are the altered photosynthetic rate and disturbed metabolism (Philippot *et al.*, 2013)<sup>[56]</sup>, which ultimately cause changed physiological processes in plants. Furthermore, the distribution of assimilated C to decomposers is an important component of the ecosystem's function. The altered physiological mechanism due to climate change disturbs this partition of assimilated C to microbial entities associated with rhizospheric soils and, therefore, affects the relationships between plants and microbes.

It is also conceivable that the rising temperature, change in rainfall duration, and changes in relative humidity have not only affected agricultural productivity but also have a significant impact on the severity of plant diseases (Chakraborty and Newton, 2011)<sup>[11]</sup>. Agricultural productivity is largely determined by the presence of pathogens and the status of plant diseases in any environment. In changing environments, the condition of occurring diseases in crop plants is boosted due to a change in distribution pattern, an evolution of the new races and path types, and epidemic development (Yanez-Lopez *et al.*, 2012)<sup>[81]</sup>. Microbial populations with short life cycles adapt the re-occurring changes in the environment with faster reproductive processes and dispersal mechanisms (Coakley *et al.*, 1999)<sup>[13]</sup> and become more complicated in physiological attributes in a stressed environment developed through changed climatic events (Sturrock *et al.*, 2011)<sup>[70]</sup>. The interactions between soil microbes and plants assist in the regulation and maintenance of ecosystem properties (Classen *et al.*, 2015)<sup>[12]</sup>. In a changed environment, the interaction networks between species are altered and, therefore, change the dynamical aspects of ecosystem properties.

The direct effect of climate change on microbial activities, response mechanisms, and their functional profile could be interpreted from the relative abundances and diversity of microbial communities in soil. Furthermore, the differential behaviour could be explained based on their different growth rates, temperature sensitivity, and other physiological attributes (Briones *et al.*, 2014; Delgado-Baquerizo *et al.*, 2014)<sup>[7, 17]</sup>. One of the most important contributions of the soil microbiota is the decomposition of organic matter and an increased warming effect that definitely increases microbial activity for organic matter decomposition. Explicitly, the increased decomposition would generate a large amount of GHGs and, therefore, have increased efflux of CO<sub>2</sub> to the atmosphere and export of the dissolved organic C by the process of hydrologic leaching (Davidson and Janssens, 2006)<sup>[16]</sup>. Under the effect of changed environment, microbial activities are influenced as reflected from observed changes in functions like microbial enzymic activity, soil respiration, and decomposition of litter. However, the exact mechanism through which these changes occurred is not known. Further, it has been reported that whole soil, aggregate functional responses result from the individual activities of a diverse community of soil microbes may involve different mechanism working simultaneously to create the observed function. The most common response mechanism includes microbial physiology, evolution, community composition, and feedbacks (Keitt *et al.*, 2016)<sup>[36]</sup> as microbial traits that correlate physiological attributes with environmental performances and fitness of microbial species that lead to sorting of species and compositional change over gradients. This could be interpreted as mean annual air temperatures, and mean annual precipitation could be positively correlated with the rate of soil respiration (Raich and Schlesinger, 1992)<sup>[60]</sup>.

However, the temperature dependence and strong response of soil respiration rates over productivity (Schimel *et al.*, 1994)<sup>[64]</sup> may result in a net transfer of C from land to the atmosphere, which would generate positive feedback on climate change. The indirect feedback of climate change on the microbial system affects the potential abilities and functioning of microbes through their impact on plant growth and composition of vegetation. The indirect effects of climate change to the microbial system are regulated through different feedback loops that incorporate plant-microbe interactions, microbe-microbe interactions, soil mineralization events, plant chemistry, and plant composition, and the most likely shifting in other ecosystem interactions that mediate the other functions of the ecosystem (Adler *et al.*, 2012; Steinauer *et al.*, 2015)<sup>[1, 69]</sup>. The results of increased C flux from vegetation to the soil and the microbial biomass is unpredictable due to its dependence on multiple factors such as the status of the soil health and properties, soil-food web interactions, and other ecosystem functions. However, the most accurate outcome of such C transfer is the loss of C from the soil through microbial respiration and its dissolution in water bodies due to stimulation of microbial activities and enhanced mineralization of soil organic C. Furthermore, other possible results include stimulation of microbial biomass and immobilization of soil nitrogen, thereby delimiting the N availability to plants, creating negative feedback that constrains future increases in plant growth and C transfer to the soil (Diaz *et al.*, 1993)<sup>[18]</sup>.

Climate adversities like the elevated levels of atmospheric CO<sub>2</sub>, the rise in global temperature, drought has affected the ecology and physiology of both plants and microbes. Since

plants distribute some the assimilated C to feed microbial populations associated with them, the interruption in the C assimilation pathway under the effect of climate change would definitely influence plant-microbial interactions (Jin *et al.*, 2009) [35]. It has been reported that microbial diversity and abundances are highly susceptible to climate change events (Maestre *et al.*, 2015) [48]. The beneficial microbes associated with plants have a large impact on host cell physiology and protect their host from disease and various abiotic stress factors. The reproductive physiology of the host plant under the warming effect has been found to be changed by early leafing and flowering time in the growing season (Wolkovich *et al.*, 2012) [80]. This has resulted into an alteration in the functional traits of plants (Verheijen *et al.*, 2016) [77] and therefore affected the multiple properties of the ecosystem (Butler *et al.*, 2017) [10]. It has been reported that microbial interactions with plants result in a plethora of local and systemic responses that improve the metabolic capabilities of plants to tolerate harsh environmental conditions (Nguyen *et al.*, 2016) [53]. Moreover, the interactions of plants with beneficial microbes cause metabolic reprogramming those results in the accumulation of defence-related compounds to counteract the effects of harsh environmental condition.

The primary spatial product of the study is the soil organic C map which demonstrates the distribution of the current soil organic C stocks (Fig. 6a). Gottschalk *et al.*, (2012) [25] reported that the decomposition usually increases by warmer temperatures, but can also be slowed by decreased soil moisture. Underlying the global trend of increasing SOC under future climate is a complex pattern of regional SOC change. A typical result of geostatistical interpolation is a map of predictions and prediction error, which is an estimate of prediction uncertainty (Hengl *et al.*, 2004) [30]. The prediction uncertainty map is shown in (Fig. 6b). Rusu (2013) [61] stated that, in terrestrial environments, increasing temperature determines an increase in the amount of natural atmospheric CO<sub>2</sub>, which would significantly boost photosynthesis, and enhance metabolism as well as increase the amount of vegetation biomass.



Fig 6(a): Soil organic C prediction map

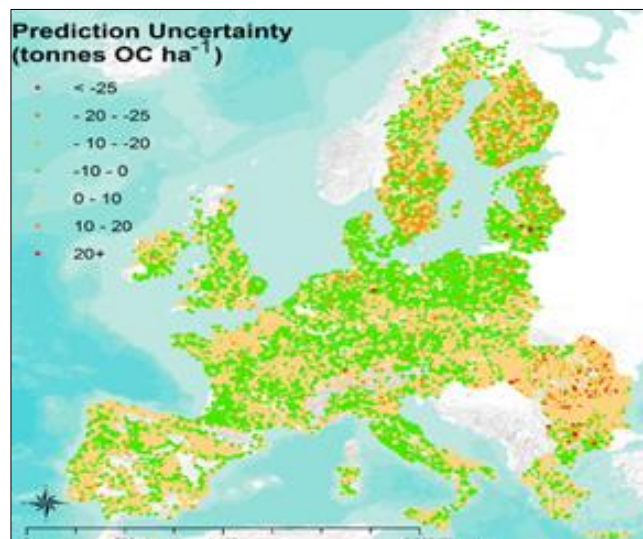


Fig 6(b): Prediction uncertainty map

The integrated impact of climate change is expected to generally increase crop yields (with winter wheat, sunflower and sugar beet) as a result of the combined effects of CO<sub>2</sub> fertilisation, radiation use efficiency and longer growing seasons which mostly applies to species with the C<sub>3</sub> photosynthetic pathway (Pathak *et al.*, 2012; Mihra and Rakshit, 2008) [55, 49] and not necessarily to species with the C<sub>4</sub> pathway. Elevated CO<sub>2</sub> increases the size and dry weight of most C<sub>3</sub> plants and plant components (Fig. 7a). Relatively more photo-assimilate is partitioned into structural components (Stems and petioles) during vegetative development in order to support the light-harvesting apparatus (Leaves). Increased yields were expected for sunflower might whereas smaller increases in yield or possible decreases in yield for potatoes, oilseed rape and high quality horticultural crops was expected when grown under water stressed light textured soils. Increases in grass yields are also generally expected. Both climatic warming and rising CO<sub>2</sub> levels in the atmosphere enhances tree growth in the short term (Fig. 7b).

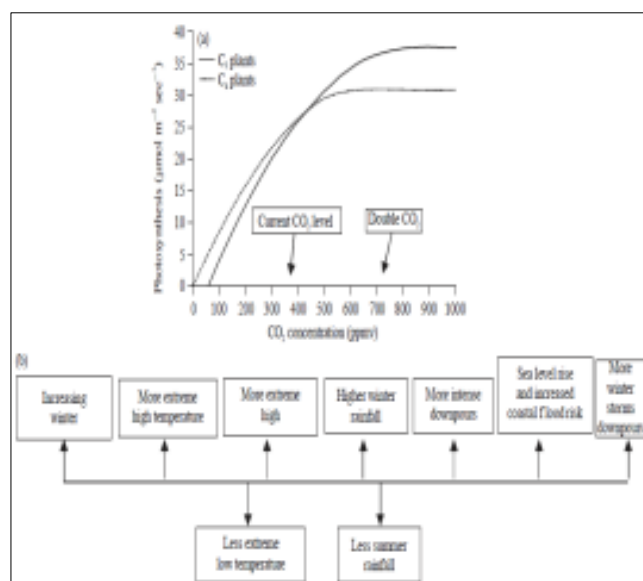
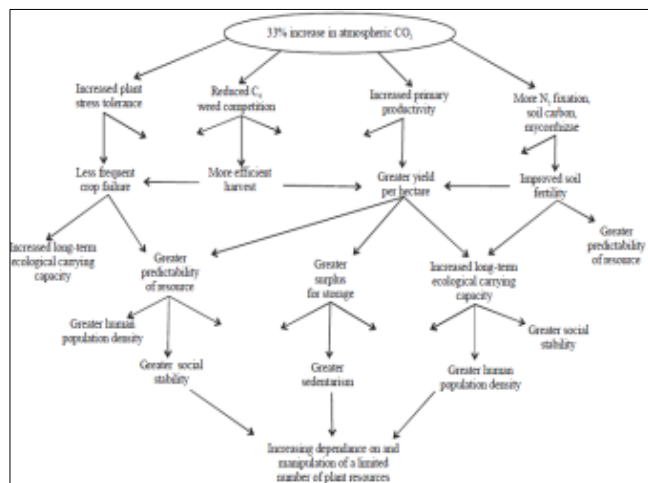


Fig 7(a): Schematic effect of CO<sub>2</sub> concentrations of C<sub>3</sub> and C<sub>4</sub> plants [Source: Wolfe and Erickson, 1993] [79].

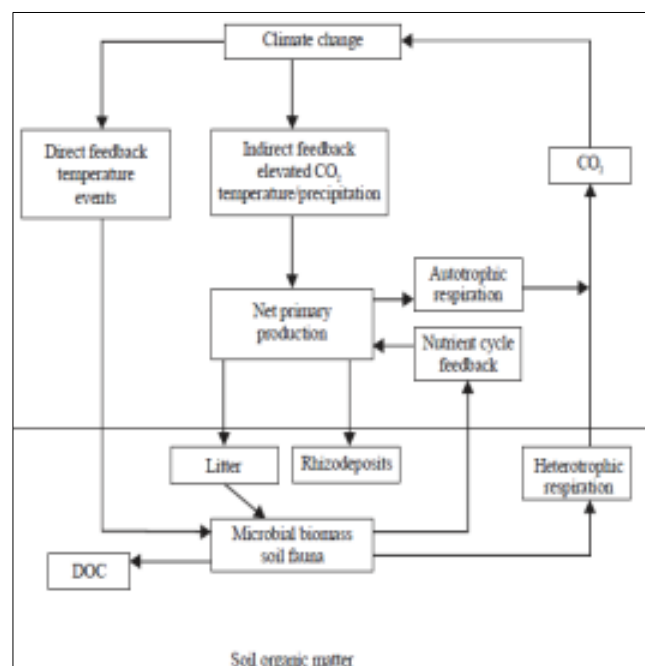


**Fig 7(b):** Possible linkages between increases in atmospheric CO<sub>2</sub> from 200-270 mol per mol

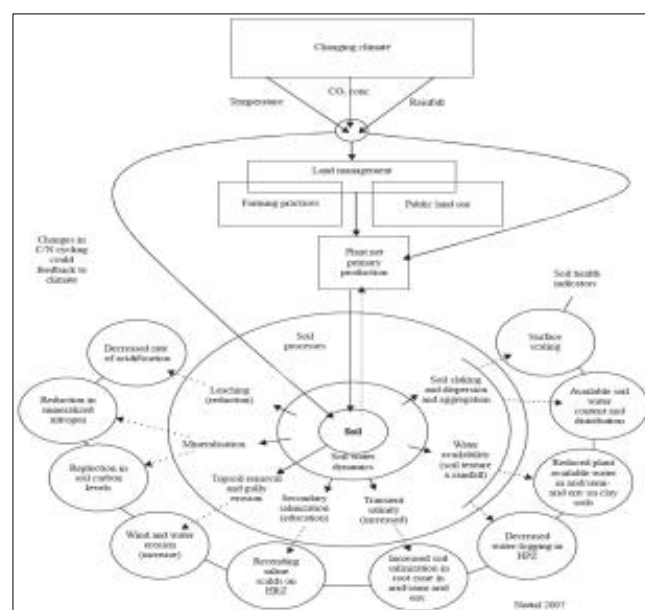
The impacts of climate change in the coastal lowland acid sulphate soils (Bush *et al.* 2010)<sup>[9]</sup>. The anticipated impacts of climate change are warmer conditions, an increasing proportion of rainfall to occur from heavy falls, increasing occurrence of drought in many regions, increasing frequency of intense tropical cyclones, rising sea levels and frequency of extreme high seas (e.g., storm surges). All of these predicted impacts have direct relevance to coastal acid sulphate soils landscapes, through either exacerbating sulphide oxidation by drought, re-instating reductive geochemical processes or changing the export and mobilisation of contaminants. The interaction of specific land management factors such as man-made drainage also had significant role in how the predicted impacts of climate change affect these landscapes. Understanding the potential impacts of climate change for coastal lowland acid sulphate soils is particularly important, given the utility of these areas for agriculture and urban communities, their unique capacity to cause extreme environmental degradation and their sensitivity to climatic factors such as temperature and hydrology and susceptibility to sea-level inundation.

Climate change may have stronger or weaker, permanent or periodical, favourable or unfavourable, harmful (sometimes catastrophic), primary (direct) or secondary (indirect) impact on soil processes. Among these processes soil moisture regime plays a distinguished role. It determines the water supply of plants, influences the air and heat regimes, biological activity and plant nutrient status of soil. In most cases it determines the agro-ecological potential, the biomass production of various natural and agro-ecosystems and the hazard of soil and/or water pollution (Fig. 8a). Organic matter is vital because it supports many soil processes that are associated with fertility and physical stability of soil across the various ecosystem services. In particular organic matter provides an energy source for microbes structurally stabilizes soil particles, stores and supplied plant essential nutrients such as nitrogen, phosphorus and sulphur and provides cation/anion exchange for retention of ions and nutrients. C within the terrestrial biosphere can also behave as either a source or sink for atmospheric CO<sub>2</sub> depending on land management, thus potentially mitigating or accelerating the greenhouse effect (Lal, 2004)<sup>[42]</sup>. Cycling of soil organic C is also strongly influenced by moisture and temperature, two factors which are predicted to change under global warming. Overall, and climate change shift the equilibrium, both directly and indirectly of numerous soil processes. These

include C and nitrogen cycling, acidification, risk of erosion, salinization, all of which have impact on soil health (Fig. 8b).



**Fig 8(a):** Relationships between soil moisture regime other soil ecological conditions and soil fertility



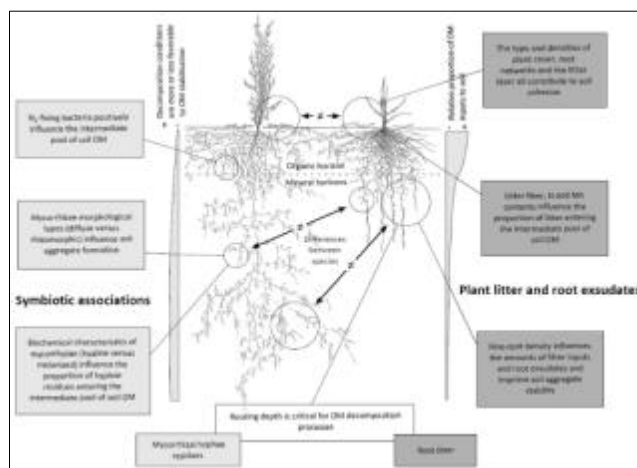
**Fig 8(b):** Schematic representation of the potential links between climate change and soil health

Freschet *et al.* (2013)<sup>[23]</sup> revealed that plants have a broad range of root systems and their influence on soil OM varies with the plant species and root functional traits (i.e. architecture, morphology, physiology, chemical composition and symbiotic associations, Fig. 9a). The contribution of belowground input to C storage occurs through the persistence of plant residues or via the stimulation of soil microbial activity and the increase of the contribution of microbial necrosis to the slow cycling soil OM pools (Lange *et al.*, 2015; Morriën *et al.*, 2017)<sup>[43, 50]</sup>. The architecture and rooting profile of species are thus critical traits that control the amount and location of C inputs in the soil profile. Lange *et al.* (2015)<sup>[43]</sup> also demonstrated that higher plant diversity increases rhizosphere C inputs.

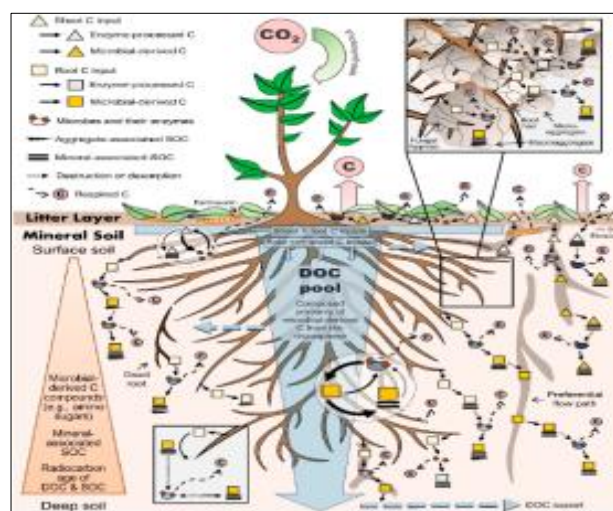
Predicting changes in the soil organic C stock through the understanding of mechanisms raises at least two crucial related issues that upscaling (from  $\mu\text{m}^3$  to  $\text{dm}^3$  and then to the plot, landscape and global scale) and validation (from the potential action of a mechanism to its quantitative expression in different soil and climatic contexts). Each of these approaches must then be validated on suitable datasets (Fig. 9b). Hassink (1997) [28] proposed that the proportion of the fine fraction ( $<20 \mu\text{m}$ ) of a soil implies an upper limit to its capacity to store stable C. This theoretical limit can be calculated ( $C_{\text{sat}}$ ) by particle-size measurements ( $C_{\text{sat}} = 4.09 + 0.37 \times (\text{clay} + \text{fine silt})$ ). This indicator has recently been used to draw the first map of the potential of organic C storage in the fine fraction in the 0–30 cm horizon (Angers *et al.*, 2011) [3].

deep SOC may be negligible compared to DOC transport (Braakhekke *et al.*, 2013) [6]. Because root- and microbial-derived C are input belowground, incorporation into the DOC pool may be more direct. Nonetheless, most belowground C inputs undergo stages of decomposition via repeated microbial processing, protection, and release into the DOC pool. Dimassi *et al.* (2014) [19] also show that the response of SOC to no tillage is dependent on climate, and in particular precipitation, with a greater response in drier conditions.

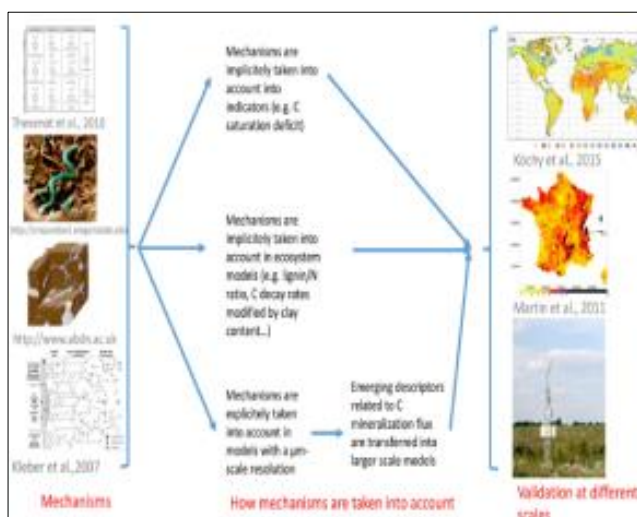
Changes in micro-environmental conditions induced by ploughing seem to be the main mechanisms controlling the effects of tillage on SOC dynamics (Fig. 10b). Autret *et al.* (2016) [4] reported that C inputs and SOC stocks have either been measured or modelled, increases in OC inputs due to alternative management were sufficient to explain the observed SOC stocks changes. Practices affect different components of the overall C balance. Fertilization (Ladha *et al.*, 2011) [40], liming (Paradelo *et al.*, 2015) [54] and irrigation (Zhou *et al.*, 2016) [84] increase primary production and thus increase inputs to soil, modify plant rooting (fertilization and irrigation), and accelerate C and N mineralization. Their net effect on SOC stocks is hence highly variable, and likely depends on local conditions.



**Fig 9(a):** Differences in functional traits and symbiotic associations between different plant species influence soil organic matter stabilisation.

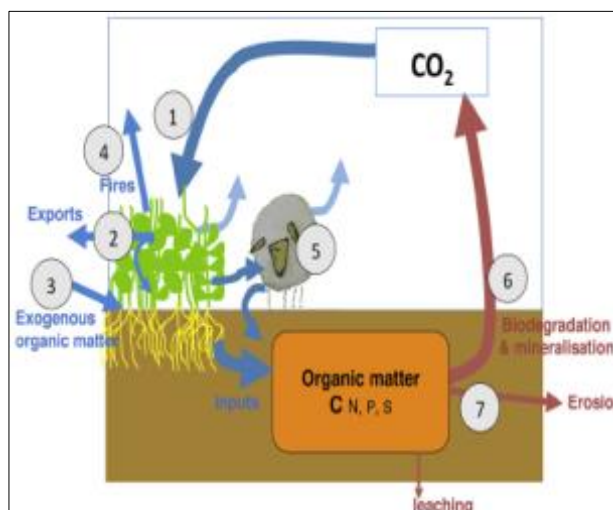


**Fig 10(a):** Proposed model for soil organic C (SOC) cycling showing root C (C) inputs as the primary source of both SOC and dissolved organic C (DOC) in most ecosystems



**Fig 9(b):** From the identification of stabilisation mechanisms to their effective consideration to improve the prediction of soil organic C stock evolution

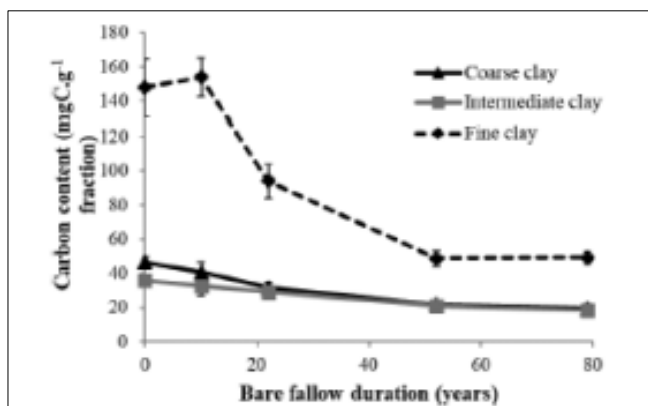
Gross and Harrison, (2019) [26] revealed that shoot-derived C is incorporated into the bulk SOC through the transport of DOC from the litter layer as well as through the mixing of particulate organic matter into superficial soil layers via soil fauna (i.e., bioturbation). Bioturbation can play an important role in SOC cycling in some ecosystems (Fig. 10a). However, the abundance and effect (particularly long-term) of bioturbation agents usually declines sharply with depth (Jiménez and Decaëns, 2000) [33] and their contributions to



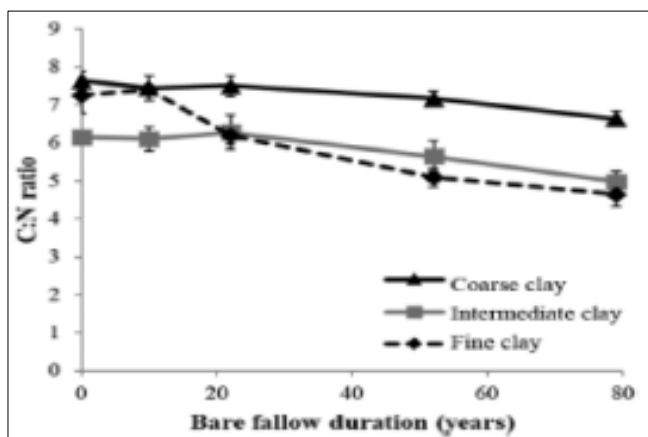
**Fig 10(b):** Levers associated with agricultural practices that may influence SOC stocks



Lutfalla *et al.* (2019) [45] observed that the texture of soil viz. sand and coarse silt fractions, respectively, but much higher in the fine silt and clay fractions (Fig.11a). The IC and CC sub-fractions displayed an initial OC content (35:9- 2:19 mg Cg<sup>-1</sup> fraction and 46:6-2:95 mg Cg<sup>-1</sup> fraction, respectively) 3 to 5 times lower than that of the FC fraction (147:8-16:9 mg Cg<sup>-1</sup> fraction).

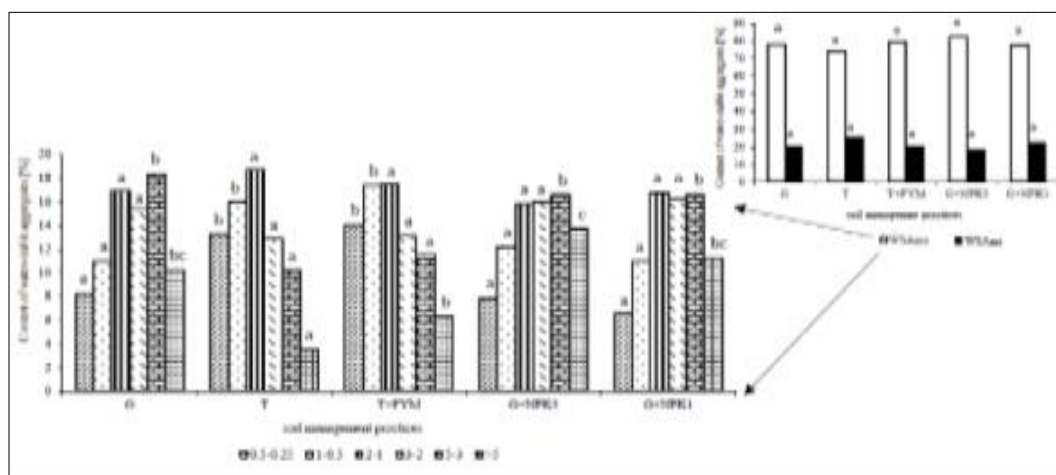


**Fig 11(a):** Evolution of carbon content (mg Cg<sup>-1</sup> fraction) with time in the three different clay sub-fractions



**Fig 11(b):** Evolution of the carbon to nitrogen ratio (C: N) with time in the three different clay sub-fractions

Ma *et al.* (2016) [47] reported that the proportion of SMBC to TOC ranged from 1.02 to 4.49, indicating that TOC is relatively low, or due to sampling for the summer after spring harvest, when soil temperature is high, the microbial activity is relatively strong. The SMBC at all depths (0–90 cm) with a sharp decline in depth increased perhaps due to a higher microbial biomass and organic matter content. SMBC was significantly higher in PRB in the surface soil layer (0–10 cm) than in TT and FB, which showed that no-till and accumulation of crop residues enriches the topsoil with microbial biomass. Microbial biomass concentrations are controlled by the level of SOM and oxygen status. Tripathi *et al.* (2014) [73] observed that the significant positive correlations were observed between TOC and organic C fractions (POC and SMBC), illustrating a close relationship between TOC and POC and TOC and SMBC and that SOC is a major determinant of POC and SMBC. Liu *et al.* (2016) [44] also found that the averaged across soil depths (0–25 cm depth), MBC of the grassland (1624.1 mg kg<sup>-1</sup>) and forestland (839.1 mg kg<sup>-1</sup>) were 6.9 and 3.6 times more, respectively than those for arable land use (245.9 and 226.2 mg kg<sup>-1</sup> for no tillage (NT) and plow tillage (PT), respectively). Similarly, the MBN concentration was 4.1 and 2.5 times more in grassland (78.0 mg kg<sup>-1</sup>) and forest (50.0 mg kg<sup>-1</sup>) than in arable land (20.0 and 18.0 mg kg<sup>-1</sup> for NT and PT, respectively), in the 0–25 cm soil layer. The higher MBC and MBN concentrations under NT than that of PT could be attributed to several factors including higher moisture content, more soil aggregation, higher SOC and TN concentration, and minimum disturbance, which provide a steady source of SOC and TN to support microbial community near the soil surface. Simansky *et al.* (2017) [66] reported that the soil-management practices significantly influenced the soil organic C in water-stable aggregates (SOC in WSA). The content of SOC in WSA ma increased on average in the following order: T<G<G+NPK<sub>1</sub><G+NPK<sub>3</sub>< T+FYM. Intensive soil cultivation in the T treatment resulted in a statistically significant build-up of SOC in WSA ma at an average rate of 1.33, 1.18, 0.97, 1.22 and 0.76 gkg<sup>-1</sup>yr<sup>-1</sup> across the size fractions > 5 mm, 5–3 mm, 2–1 mm, 1–0.5 mm and 0.5–0.25 mm, respectively (Fig.12).

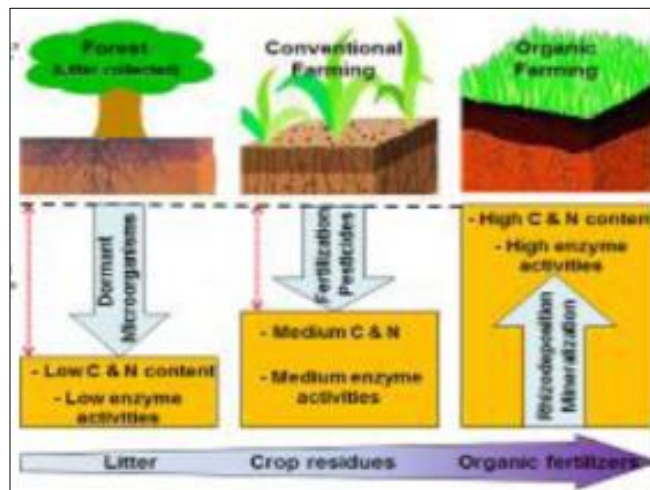


**Fig 12:** Water-stable aggregates contents under different soil-management practices [Source: Simansky *et al.*, 2017] [66]

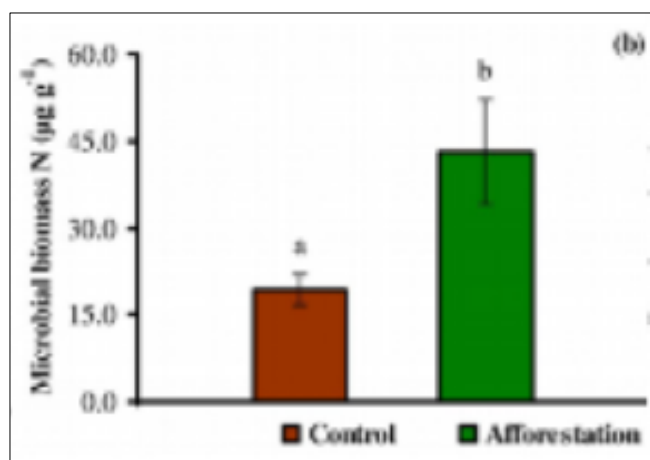
Ma *et al.* (2016) [47] reported that the differences in SMBC were limited to the surface layers (0–5 and 5–10 cm) in the PRB treatment [Fig.13a]. There was a significant reduction in SMBC content with depth in all treatments. SMBC in the PRB treatment increased by 19.8%, 26.2%, 10.3%, 27.7%,

10% and 9% at 0–5, 5–10, 10–20, 20–40, 40–60 and 60–90 cm depths, respectively, when compared with the TT treatment. The mean SMBC of the PRB treatment was 14% higher than that in the TT treatment. There were no significant differences in SMBC content between the three

treatments from 10 to 90 cm depth [Fig.13a]. Jiang *et al.* (2011)<sup>[32]</sup> observed that the highest levels of MBC were associated with the 1.0–2.0 mm aggregate size class (1025 and 805 mg C kg<sup>-1</sup> for RNT and CT, respectively) which may imply that RNT was the ideal enhancer of soil productivity for this subtropical rice ecosystem. It is interesting to note the sudden decrease of MBC values in 1–0.25 mm aggregates (511 and 353 mg C kg<sup>-1</sup> for RNT and CT, respectively) [Fig.13b].



**Fig 13(a):** Microbial biomass carbon content with depth under traditional tillage (TT), flat raised bed with controlled traffic and zero tillage (FB) and permanent raised bed (PRB) [Source: Ma *et al.*, 2016]<sup>[47]</sup>



**Fig 13(b):** Soil microbial biomass C associated with different sizes of aggregate under RNT and CT (RNT, combines ridge with no-tillage; CT, conventional tillage) [Source: Jiang *et al.*, 2011]<sup>[32]</sup>

## Conclusions

Climate Change poses serious interlinked challenges in times to come with reference to scale and scope, never anticipated in the last century. More or less the most important change in soils expected as a result of these changes would be a gradual improvement in fertility and physical conditions of soils change from one major soil-forming process to another in certain fragile tropical soil and changes in soil property due to pole ward retreat of the permafrost boundary. Again changes due to climate change are expected to be relatively well buffered by the mineral composition, the organic matter content or the structural stability of many soils. As a matter of fact, the impact of climate change on soil system should be monitored in different agro-ecological regions on regular basis. Climate change and land degradation are closely linked

issues and conservation farming has shown promise in minimizing land degradation. Hence, the potential of conservation agriculture in minimizing the impact of climate change needs thorough investigation. There is need for harmonization of data base on land degradation keeping in view the productivity and economic losses vis-à-vis climate change effects. Plants influence labile, intermediate and stable soil C pools. The effects of plants on soil OM stabilisation and protection seem to be mostly positive, although the balance between positive and negative effects (i.e. over mineralisation) differs according to interactions between plants and the soil abiotic and biotic conditions.

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