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Soil organic matter as sole indicator of soil degradation

S.E. Obalum · G.U. Chibuike · S. Peth · Y. Ouyang

Abstract Soil organic matter (SOM) is known to play vital roles in the maintenance and improvement of many soil properties and processes. These roles, which largely influence soil functions, are a pool of specific contributions of different components of SOM. The soil functions, in turn, normally define the level of soil degradation, viewed as quantifiable temporal changes in a soil that impairs its quality. This paper aims at providing a generalized assessment of the current state of knowledge on the usefulness of SOM in monitoring soil degradation, based on its influence on the physical, chemical and biological properties and processes of soils. Emphasis is placed particularly on the effect of SOM on soil structure and availability of plant nutrients. Although these properties are discussed separately, the soil system is of dynamic and interactive nature, and changes in one property will likely affect other soil properties as well. Thus, functions of SOM almost always affect various soil properties and processes and engage in multiple reactions. In view of its role in soil aggregation and erosion control, in availability of plant nutrients and in ameliorating other forms of soil degradation than erosion, SOM has proven to be an important indicator of soil degradation. It has been suggested, however, that rather than the absolute amount, temporal change and potential amount of SOM be considered in its use as indicator of soil degradation, and that SOM may not be an all-purpose indicator. Whilst SOM remains a candidate without substitute as long as a one-parameter indicator of soil degradation is needed, narrowing down to the use of its labile and microbial components could be more appropriate, since early detection is important in the control and management of soil degradation.

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Introduction

One of the greatest problems facing agricultural soils of the world is land/soil degradation. It can be triggered by many physical, chemical, biological and ecological processes that lower the quality and potential productivity of the soil (Tefera et al. 2002; Anikwe 2006; Lal 2015). Soil degradation thus poses a threat to ecological balance and environment safety. In sub-Saharan Africa, it is a major cause of reduced agronomic productivity

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(Obalum et al. 2012a). About 85% of land degradation worldwide is due to soil erosion and related problems (Mbagwu 2003a); hence, the two terms are often used interchangeably. The importance of soil erosion is underlined by the fact that it is often not easily detected. Such difficulty in detecting soil erosion seems to wreck more havoc than soil erosion itself, as the detection may not be timely enough to allow meaningful reclamation programmes at a justifiable cost. Considering that soil erosion is the most widespread form of soil degradation (Mbagwu 2003a; Biancalani et al. 2012), the agro-environmental cost of soil degradation speaks for itself.

Soil degradation, no doubt, is counterproductive. The selection of an appropriate indicator of soil quality under different conditions could help in tackling the problem of soil degradation. This is, however, a difficult task under field conditions where exceptionally large numbers of soil properties interact in the naturally complex soil to determine its quality. It becomes more difficult when considering the many different functions of the soil each with its own specific requirements of soil properties for optimum results. In essence, no single soil property can serve as indicator of soil quality (Acton and Gregorich 1995; USDA-NRCS 1996; Öztaş 2002). Hence, a set of soil properties is normally used as indicators of soil degradation (Doran and Parkin 1996). It is most appropriate to select soil properties that vary with the particular function for which soil quality is being evaluated (Anikwe 2006).

Although a multifactorial concept, soil degradation still needs a single sensitive soil property to serve as its indicator. Such an indicator can provide first-hand information on the extent of major soil degradative processes that represent others. Soil organic matter (SOM) seems promising in this regard due to its all-embracing influence on the physical, chemical and biological properties of soils (Krull et al. 2004), which makes it very sensitive to management, among other attributes. According to Plank (2001), there have been several attempts to use traditional test for SOM or some of its fractions as an index of soil productivity or sustainability. Many authors (Mbagwu et al. 1991; Piccolo et al. 1996; Piccolo and Mbagwu 1997; Spaccini et al. 2002; Obalum et al. 2011) have recognized that depletion of SOM, following decomposition and mineralization processes hastened by intensive agricultural activities, is the main cause of high rate of soil degradation. Mbagwu (2003b) indicates that the consistently weak soil structure in many parts of the arid and semi-arid

Mediterranean regions is caused by their low SOM levels, which result from intensive mechanized cultivation in the regions. He argues that organic substances could be useful in assessing structural stability of the soils, because these substances influence stability more than any other stabilizing agent does. Generally, SOM level and stratification ratio are often cited as major indicators of soil quality (Franzluebbers 2002; Krull et al. 2004; Blum et al. 2014).

Changes in SOM level could be manifested in a number of ways, all of which relate to soil degradation. For instance, SOM loss reduces soil fertility and capacity to produce crops (Acton and Gregorich 1995; Plank 2001), since SOM is a rich source of mineralizable nitrogen (N), phosphorus (P), sulphur (S) and other nutrients (Baldock and Broos 2011). Consequences of drastic changes in SOM status could also be well defined by measuring changes in biological parameters and structural stability (Gal et al. 2006). Weak soil structure and susceptibility to erosion, loss of plant nutrients and decline in soil biodiversity, to mention but a few, are all forms of soil degradation common in agricultural soils. As long as there remains no generally acceptable single soil property to monitor these phenomena, achieving sustainability especially in low-input agriculture would remain a mirage. With the recognition of the important role of SOM in the maintenance of sustainable agricultural systems (Fernandes et al. 1997), our success or failure in the use and management of soils seems to revolve around our ability to manage SOM. The failure-success paradigm here would therefore depend on SOM concentration maintained in the soil at any point in time, which would give a clue as to the level of degradation of the soil. This paper is, therefore, aimed at exploring the potentialities of SOM for this onerous task of serving as an all-purpose indicator of soil degradation, with emphasis on soil erosion and plant nutrient depletion.

An overview of soil degradation

Land degradation or soil degradation has been defined in many, at times discipline-oriented, ways; and this causes confusion, misunderstanding and misinterpretation. The distinction between land and soil is clear in that land encompasses soil, suggesting that land degradation is a broader concept than soil degradation. However, because any effect on the soil usually spreads to

other land components (landscape, vegetation, water and climate) in agricultural areas, soil degradation and land degradation are often used interchangeably. Soil degradation is "the temporary or permanent lowering of the productive capacity of soil caused by overgrazing, deforestation, inappropriate agricultural practices, over exploitation of fuel wood leading to desertification and other man-induced activities" (Mbagwu 2003a:15). Although soil degradation advances rapidly under anthropogenic activity but slowly under natural processes (Acton and Gregorich 1995), both situations reduce land quality which results in the loss of actual or potential productivity of the land (Eswaran et al. 2001).

Several mechanisms initiate soil degradation. Such mechanisms or processes of soil degradation can be grouped into three: physical, chemical and biological (Mbagwu 2003a), although a fourth category, ecological processes, is increasingly recognized (Lal 2015). The most important physical process is a decline in soil structure, as this leads to crusting, compaction and eventually erosion. Other physical processes include desertification, anaerobiosis, lowering of water table and water imbalance (Eswaran et al. 2001; Lal 2015). Processes that affect soil chemical properties are nutrient depletion and imbalance, acidification, contamination/ pollution, salinization and alkalization. Biological processes correspond to depletion of SOM, decline in soil microbial biomass carbon and reduction in soil biodiversity while ecological processes include disruption in elemental cycling and decline in carbon sink capacity (Lal 2015). Eswaran et al. (2001) summarize factors that influence the severity of soil degradation as follows: (i) biophysical, involving land use and management practises; (ii) socioeconomic, such as land tenure systems and (iii) political, such as the availability of government incentives.

Quantifying and rating soil degradation

It is often said that there is no zero degradation. This implies that every soil is "degraded" relative to its "full" productive potential. So, what is important is the status of degradation which depends, among other attributes, on the level of vulnerability of the soil to degradation and/or of a soil property to a degradation process. Eswaran et al. (2001) regard lands as varying from highly resistant or stable to extremely sensitive to degradation, depending on their inherent characteristics and climate. They argue that resistant lands do not

necessarily resist change; rather, they are in a stable steady-state condition with the new environment. This is never the case with the fragile ones which, under stress, degrade to a new steady state. The altered state is unfavourable to plant growth just as the soil is less capable of performing environmental regulatory functions. Resistant or stable soils can be regarded as not degraded, since they perform their normal functions; they only need to be managed well to avoid falling below this acceptable state. According to Mbagwu (2003a), a soil is degraded if its productivity falls below the economic threshold even under favourable weather conditions or with judicious input. He also opines that soil degradation is not always an absolute concept as long as degraded fragile soils respond to management practises to allow for any land use or, even if they fail to respond to management, can be put to some other uses.

Although land degradation has been a major global issue since the twentieth century and has received a lot of international attention in the twenty-first century, there exists a large variation in the available statistics on its extent and rate, due to differences in definitions, terminology, approaches and areas included in the assessment (Eswaran et al. 2001). Available data show that in dry areas of the world, degraded lands amount to 3.6 billion ha out of 5.2 billion ha in these areas (Dregne and Chou 1994). The data in Table 1 show that the global extent of land degradation by all processes is about 1.9 billion ha.

Soil degradation thus poses a threat to world food security and environmental quality. Eswaran et al. (2001) remark that the Agenda 21 of 1992 of the United Nations has only succeeded in diluting the problem of land degradation as the increased awareness campaign against it has produced little or no impact on members of the global community who can be an asset in reversing the ugly trend. People of the world need to be motivated and equipped to live up to their responsibility of stewardship onto the land. Places with the highest risks of soil degradation are in the tropics, because of the increasing agricultural intensification in these places to combat inadequate nutrition and satisfy the increasing population within the limited reserves of arable land (Craswell and Lefroy 2001).

The concept of soil degradation has become relevant in the context of the numerous soil functions of value to humans, gradual and progressive decrease in global land mass available for agricultural production and the need to adopt a positive approach to achieve sustainable

Table 1 Estimates of the global extent (in million km^2) of land degradation

Type	Light	Moderate	Strong + extreme	Total
Water erosion	3.43	5.27	2.24	10.94
Wind erosion	2.69	2.54	0.26	5.49
Chemical degradation	0.93	1.03	0.43	2.39
Physical degradation	0.44	0.27	0.12	0.83
Total	7.49	9.11	3.05	19.65

Source: Oldeman (1994); Eswaran et al. (2001)

management of this non-renewable resource. Developing technologies for reversing land degradation trends is therefore important in order to minimize the economic and environmental impact associated with land degradation (Eswaran et al. 2001). The soil science community has a task in this regard—to explain soil degradation in action-provoking terms to the laymen, especially the subsistent low-resource farmers of the world, and to standardize and prioritize indicators of soil degradation for alternative land uses.

Soil degradation from soil quality perspective: common indicators

Though monitoring soil quality is a relatively recent activity, Gal et al. (2006) highlighted the importance of this exercise. Soil quality is the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality and promote plant and animal health (Soil Science Society of America 2001). From an agronomic standpoint, soil quality or health is the soil's fitness to support crop growth without resulting in soil degradation or constituting harm to the environment (Acton and Gregorich 1995). Different perceptions exist for the term soil quality, and this depends on the particular soil and its intended use (Anikwe 2006), just as with soil degradation. However, a high quality soil is one that is able to adequately support plant growth and other biological activity as well as serve as an environmental buffer for hazardous compounds (Krull et al. 2004).

As with soil degradation, soil quality indicators are physical, chemical and biological properties that can be assessed to monitor changes in the soil (USDA-NRCS 1996). Soil quality strongly depends on degree of soil degradation, land use and management practises (Öztaş 2002). Also, evaluation of soil quality and soil degradation is based on same principle of measuring these

indicators and selecting a baseline or standard set of optimum values for comparison. Suffice it to say that soil quality and soil degradation are related thus (i) both are assessed by virtually the same indicators, though with opposing interpretations; (ii) the former is assessed to determine the extent of the latter so as to improve the soil conditions.

Since soil is the most vital natural, non-renewable resource, indexing and predicting its loss/degradation is a key task (Gal et al. 2006). Such indicators are measurable parameters used to estimate the level of soil degradation. There is no sharp demarcation among the indicators; they interact with and influence one another. An analogy of the nature of soil quality indicators is given by Acton and Gregorich (1995:15): "Human health is a composite picture of the condition of the body's various parts and functions. We assess human health by looking at many factors (including physical function, mental capacity, and emotional well-being) and by forming an overall sense of how the body and mind are working. In the same way, soil quality is a composite picture of the state of the soil's many physical, chemical and biological properties, and of the processes that interact to determine this quality. Furthermore as human health varies from person to person, soil health varies among soils".

The soil properties or attributes that are commonly used to estimate soil degradation, therefore, are those that are sensitive and reliable for obtaining changes in physical, chemical and biological properties. Anikwe (2006) outlined them under the following categories:

1. *Physical indicators*: these attributes are connected with the physical arrangement of soil particles and pores. Examples include soil texture, bulk density, aggregate stability, porosity, topsoil depth, soil compaction, sealing and crusting.

2. *Chemical indicators*: depending on the soil quality being evaluated, these properties include soil pH, electrical conductivity (salinity), SOM concentration, nutrient content, cation exchange capacity (CEC), presence of pollutants, etc.

3. *Biological indicators*: these indicators are very sensitive to changes in soil conditions and are therefore used for short-term evaluations. They include presence of earthworms, microbial biomass, enzyme activity, respiration and decomposition rates.

4. *Visible indicators*: these are observable attributes which occur as a result of changes in soil physical, chemical and biological properties. They include plant growth response, presence of weed species, runoff and changes in soil colour and exposure of subsoil.

It is practically impossible to measure all the above soil properties. Doran and Parkin (1996) propose a minimum dataset (MDS), which is the smallest set of soil properties needed to assess soil quality. Quantitative indicators of soil quality in the MDS include soil texture; rooting depth; bulk density; infiltration; water retention characteristics; SOM concentration; electrical conductivity; extractable N, P and K; microbial biomass; and soil respiration. The working principle of MDS is beyond the scope of this paper.

The place of SOM in the assessment of level of soil degradation

The SOM performs several important functions in the soil which can be summarized into physical, chemical and biological functions. These roles of SOM occur simultaneously with dynamic interactions (Fig. 1). Therefore, it is difficult to separate one from the other; discussing them in isolation is merely conceptual. The interactions among the roles of SOM may thus be viewed as defining soil degradation status. Because of its effect on these functions, SOM is often chosen as a suitable indicator of soil quality and agronomic suitability (Doran and Safely 1997; Reeves 1997), such that enhancing its pool is essential to restoring degraded soils, advancing food security and improving the environment (Lal 2009).

First, SOM, unlike other possible indicators of soil degradation, affects soil aggregate formation and stability. Being a widely accepted and used physical indicator

of erodibility (Nciizah and Wakindiki 2015), aggregate stability is a basic physical indicator of soil quality. From a wider perspective, SOM is related to the general soil condition in that it defines not only soil stability and erosion extent but also soil fertility (Anikwe 2006). Soil resilience is governed by the adequate performance of the soil's physical, chemical and biological functions which in turn are, to a large extent, determined by the quantity and quality of SOM (Krull et al. 2004). Because of the relativity of the concept of soil degradation with respect to SOM status, however, it is reasonable to consider two factors: the temporal change in SOM and the difference between the actual and potential SOM in a given soil (Biancalani et al. 2012).

SOM and soil erosion: modification of soil structure

Soil erosion results in lowering of soil fertility and corresponding reduction in food and fibre production (Lal and Stewart 1990; Pimentel 2006). Also, transportation and deposition of the detached soil often lead to sedimentation, water pollution and siltation of watercourses which have wider environmental implications (Myers 1993; Pimentel 2006).

Erosion-induced soil degradation is initiated as a result of a decline in soil structure (compaction, crusting and loss of rooting depth and plant available water) as well as a decline in soil organic carbon and fertility (Lal 1998). The SOM modifies soil physical properties, especially soil structure (Tisdall and Oades 1982; Six et al. 2000; Bossuyt et al. 2001); thus, it plays a key role in the control of soil erosion (Conforti et al. 2013). A structurally stable soil is less likely to erode because soil structure controls several biophysical processes such as water retention, soil aeration as well as activities of soil organisms (van Veen and Kuikman 1990; Fageria 2012). Generally, formation and stabilization of soil aggregates are influenced by both inorganic and organic factors such as the presence of Fe–Al oxides, clay content and the type and amount of SOM (Lynch and Bragg 1985; Six et al. 2000; Bossuyt et al. 2001).

The SOM contributes to aggregate formation through binding effect of products of microbial synthesis as well as through direct binding of soil aggregates by fungal hyphae and plant roots (Tisdall and Oades 1979, 1982; Tisdall 1991; Martens and Frankenberger 1992; Bearden and Petersen 2000). These highlighted roles of SOM may not always imply that, as a rule of thumb, as SOM concentration increases structural integrity of

Fig. 1 Schematic representation of the relationships among the major physical, chemical and biological functions of soil organic matter (Source: Krull et al. 2004)
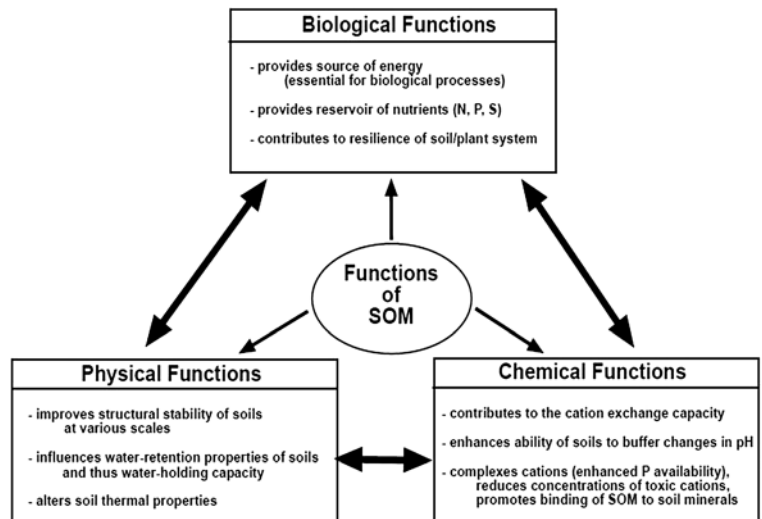
the soil increases and/or soil degradation decreases. Janzen et al. (1992) indicate that the relationship between soil quality indicators (especially SOM) and desirable soil attributes does not always conform to a simple linear and positive form and concluded that 'bigger is not necessarily better'. The SOM thus acts as an aggregating agent under certain conditions and as a dispersing agent under some other conditions (Oades 1984; Mbagwu and Bazzoffi 1988; Goldberg et al. 1990; Arias et al. 1996). There is need to explore some of these conditions and the governing processes in order to guide SOM management for optimum benefit.

Is SOM an agent of aggregation or dispersion?

Aggregation is the formation and stabilization of micro- and macroaggregates. At the level of microaggregates (< 0.25 mm), SOM could have either aggregating or dispersing effect on soils (Oades 1984). The SOM can act as an aggregating or dispersing agent or have no effect depending on its composition in the soil and/or the relative contributions of such other aggregate-stabilizing agents as Fe and Al oxides (Goldberg et al. 1990; Arias et al. 1996; Mbagwu and Schwertmann 2006). Mbagwu et al. (1993) noted that the low stability of tropical soils is due not only to their high sand and silt contents but also to the rapid decline in SOM once forests are cleared and continuously cultivated.

Soil aggregate stability is influenced by a number of factors including soil texture, clay mineralogy, SOM level, type and content of cations, contents of Fe–Al oxyhydroxides and $CaCO_3$, with multiple interactions

among these properties capable of modifying their individual influence (Le Bissonnais 1996). The two most important are SOM (Idowu 2003), especially in low-sodium soils (Elliott 1986; Golchin et al. 1995) and Fe–Al oxides mainly in tropical and lateritic soils (Römkens et al. 1977; Mbagwu and Schwertmann 2006). In the Mediterranean region, all of SOM, Fe oxides, $CaCO_3$, SiO_2 and clay minerals are important (Mbagwu 2003b). The SOM, besides acting as a bonding agent between mineral soil particles (Tisdall and Oades 1982), may make the soil hydrophobic to reduce wetting rate and slaking (Sullivan 2004). Soils respond to organic SOM sources in accordance to texture and other physical properties (Mbagwu 1989), implying that SOM-based assessment of soil degradation with aggregation as index must consider the underlying influence of particularly soil texture.

The contribution of SOM to aggregate stability is usually diminished in soils high in Fe–Al oxides (Oades 1984). In texturally contrasting, weakly structured soils, Gerard (1987) found aggregation of particles >0.05 mm to increase linearly with rate of organic residue and increased overall aggregation at low rates of the residue, though that depended on antecedent moisture content. Similarly, in three texturally contrasting soils in north-central Italy whose native stability decreased with decreasing clay and silica/alumina ratio, microaggregate stability was linearly related to cattle dung rate but disaggregation set in at 2 and 6% rates in the most and second most stable soils, respectively (Mbagwu and Bazzoffi 1988; Mbagwu 1989). Still on Italian soils, Mbagwu et al. (1991) reported that addition

of pig and cattle slurry and sewage sludge increased soil macro- and microaggregate stability, with highest and lowest increases in least and most stable soils, respectively.

In southeastern Nigeria, Igwe et al. (1995) observed more distinct effect of SOM on water-stable aggregates (WSA) >0.50 mm and mean-weight diameter (MWD) of aggregates within than between soil profiles and concluded that the aggregate-stabilizing effect of SOM was soil-specific. The SOM and carbonates had no effect on clay aggregation probably due to their low concentrations in the soils, whereas the role of Fe_2O_3 and Al_2O_3 was outstanding (Igwe et al. 1995). Similar observation was made on clay dispersion for a wide range of Nigerian soils by Mbagwu and Schwertmann (2006) and Igwe et al. (2009) who also reported Al oxide as exerting stronger influence than Fe oxide. Conversely, in southeastern Nigeria, SOM but not Fe oxide was found to affect clay dispersion (Opara 2009). There are other reports of several relationships portraying SOM as either aggregating or dispersing agent particularly in tropical soils (Igwe and Obalum 2013).

In two soils differing in SOM, Lado et al. (2004) reported higher aggregate slaking and clay dispersion (in <2 and 2–4 mm aggregates) in the lower- than the higher-SOM soil during wetting-leaching process; however, SOM did not affect aggregate stability when other stabilizing agents exerted strong influence. From Ethiopia, it was reported that neither SOM nor total soil carbohydrates could explain the variability in aggregate stability (MWD) of both cultivated and forested soils, suggesting that SOM components other than carbohydrates may explain aggregate stability (Spaccini et al. 2001, 2002). Spaccini et al. (2002) also noted that the Vertisols were better aggregated and contained more SOM than the Andisols or Alfisols.

The effects of humic substances and other SOM fractions on aggregate stability may help to appreciate the overall role of SOM in aggregation. The carbohydrate (polysaccharide) component of SOM normally has short-term effect (Piccolo and Mbagwu 1999), whereas humified component has long-lasting effect on stability of especially soil microaggregates (Tisdall and Oades 1982; Chaney and Swift 1986; Piccolo and Mbagwu 1990). Specifically, Piccolo and Mbagwu (1990) reported that microaggregate stability showed stronger positive correlation with humic acid than with total SOM and carbohydrate content. Humic substances thus enhance aggregate stability (Chaney and Swift 1986; Piccolo and Mbagwu 1997); carbohydrates contribute to stabilization when adsorbed onto humic substances and hence protected from microbial metabolism (Piccolo and Mbagwu 1999). The aggregate-stabilizing effect of humic substances is usually pronounced in fragile soils compared to stable soils (Mbagwu and Piccolo 1989; Mbagwu et al. 1993; Piccolo and Mbagwu 1994) and rarely the reverse (Piccolo et al. 1996).

At microaggregation level, however, Mbagwu et al. (1993) and Piccolo and Mbagwu (1994), who worked respectively with tropical and Mediterranean soils of contrasting clay mineralogy, noted some inconsistencies on the effects of applied humic acids on soil stability. In these studies, microaggregate stability increased (by flocculation of clay particles) steadily with increasing humic acid rate till the maximum in soils that contained expanding types of clay minerals, whereas it decreased (by dispersion of clay in water) in those without these clay minerals. The differential effects of organic materials on aggregate stability in the literature may thus be related to the amount and quality of humic acids they contain (Mbagwu et al. 1993). See Visser and Caillier (1988) and Mbagwu and Piccolo (1989) for an insight on mechanisms involved in soil aggregate destabilization and stabilization, respectively by humic acids.

Mechanisms involved in soil aggregation/dispersion by SOM

The SOM seems to have two opposing effects on aggregation. On one hand, it repels water to decrease wettability of aggregates (Bartoli et al. 1992) or increases friction between particles (Zhang and Horn 2001), thereby decreasing the tendency of macroaggregates to disintegrate in water. Mineral particles are also bound by organic polymers or physically enmeshed by fine roots or fungi (Chenu and Guerif 1991; Dorioz et al. 1993; Chenu et al. 2000), and this increases their internal cohesion. The binding effect is normally due to increased menisci (Zhang and Horn 2001). On the other hand, SOM coats the soil clay, thereby increasing the negative charge of the fine particles and favouring their dispersion (Arias et al. 1996).

Hydrophobicity occurs in soil due to humic acids (Giovannini et al. 1983). Generally, hydrophobic or water-repellent materials enhance aggregate stability (Capriel et al. 1990). As earlier noted, humic substances

are somewhat resistant to change. Piccolo et al. (2005) ascribed the stability of accrued SOM to selective preservation of hydrophobic compounds following deforestation and arable cropping. Therefore, cases of organic substances imparting hydrophobicity to clays (Chenu et al. 2000) are supposedly a reflection of their high content of resistant, hydrophobic humic substances. In their conceptual model of soil aggregation, Tisdall and Oades (1982) note that aggregation by SOM seems to be favoured by the presence of metal ions that can act as bridges between the anionic groups of polymeric SOM and the negatively charged clay surfaces. Organic wastes improve aggregation due to their constituent fractions which stabilize soil aggregates by forming strong complexes with polyvalent metals (Fe, Al and Mn) adsorbed on the edges of clay crystals (Mbagwu et al. 1991). Retention of humic acids increases in the presence of Fe and Al, though their association with Fe has smaller effect than with Al on soil flocculation and aggregation (Arias et al. 1996).

Structural breakdown is caused mainly by clay dispersion which in turn is influenced by a number of soil properties. Clay dispersion in water is influenced by SOM and Fe–Al oxides (Igwe and Obalum 2013), as well as by high content of monovalent cations resulting from excessive application of animal waste (Krull et al. 2004). This dispersing effect of negatively charged organic materials seems to contradict the well-known aggregate stabilizing effect of SOM. An increase in the ratio of fulvic to humic acid, by rendering organic matter soluble, results in clay dispersion (Oades 1984). Chandra and De (1982) postulated that the formation of large quantities of organic acids from CO_2 evolved during manure decomposition reduced soil pH (below neutral) and bacterial population, and that this caused a decrease in soil aggregation.

On the basis of their temporal persistence, Tisdall and Oades (1982) classified organic binding agents of soil aggregates into: (i) transient or temporary such as polysaccharides, roots, fungal hyphae, bacterial cells and algae all of which stabilize macroaggregates and (ii) persistent aromatic humic materials associated with polyvalent metal cations and polymers strongly adsorbed to clays, responsible for the integrity of microaggregates. The difference in persistence of binding agents explains the common observation that, upon cultivation, macroaggregates are generally less stable than microaggregates (Oades 1984; Elliott 1986; Cambardella and Elliott 1993; Ashagrie et al. 2007). This may be surprising considering that macroaggregates contain more SOM than do

microaggregates (Cambardella and Elliott 1993; Puget et al. 1995; Spaccini et al. 2001). However, larger SOM and associated stability of macroaggregates is due to polysaccharide by-products of fine roots and fungal hyphae in young (< 6 or 23 years) plant-derived SOM (Puget et al. 1995). The polysaccharides and materials released by microbial activities in such incompletely decomposed SOM probably favours microaggregation more than macroaggregation (Igwe and Nwokocha 2006).

The labile SOM fraction, comprising mostly carbohydrates, is active in aggregate formation (Kay and Angers 1999). Under tillage, carbohydrates associated with clay are relatively stable, but 27–43% of the total polysaccharides are in the light fraction and this is prone to rapid loss (Sims 1990). The formation and binding of WSA by polysaccharides and their exudates is mediated by microbial activity (Sims 1990), and this is the first stage in aggregate formation before stabilization by humic materials (Haynes and Swift 1990). Humic acids improve soil physical properties due to their high contents of both hydrophilic and hydrophobic SOM (Piccolo and Mbagwu 1997), as well as to the forming of clay-humic complexes bridged by polyvalent cations adsorbed onto clay surfaces (Krull et al. 2004).

It is clear that the labile SOM fractions are complemented by the recalcitrant ones in the entire aggregation process in soils. The use of such fractions as humic and fluvic acids rather than whole SOM as indicators of soil quality has been suggested (Martinez-Salgado et al. 2010; Guimaraes et al. 2013). In a long-term fertilization trial under Chinese monsoon climate, Shujie et al. (2009) who found soil aggregate formation to be driven by products of microbial synthesis serving as binding agents also presented data which highlight the role of particularly fluvic acid. The data show that the association of aggregate fractions >2 and 2–1 mm with SOM was slightly weaker than with fluvic acid and that these aggregate fractions had very weak associations with humic acid and humin. There are yet indications that SOM may not, but its fractions may be affected by soil and agronomic management practises (Abril et al. 2013), and that soils showing higher SOM may not necessarily be the ones with higher microbial activity (Kuwano et al. 2014), suggesting further that the true value of SOM as indicator of soil degradation lies not on its whole content in the soil but on its fractionation.

Preference of clay-metal-clay complexes to clay-metal-SOM complexes (metal here refers to polyvalent

metal) in clayey soils could explain why some sources of SOM usually reduce aggregate stability in clayey soils but improve it in sandy soils (Mbagwu et al. 1991). Not just the amount, type of clay is also important, since clay mineralogy influences the response of soils (in terms of aggregation) to addition of SOM sources (Mbagwu et al. 1993).

Estimating the extent of soil erosion from SOM level

After particle size distribution of soils, SOM ranks next as an indicator of soil erodibility (Wischmeier and Mannering 1969), the proneness of soil to erosion. The SOM is without doubt the most important factor in the formation of good soil structure, which helps to reduce the erosion coefficient of soils (Chandra and De 1982). The Universal Soil Loss Equation (USLE) and other erosion prediction models developed from it show the importance of SOM in soil erosion prediction. For instance, the USLE shows that covering a soil with 30% organic matter reduces about 72% of soil erosion (Renard et al. 1991).

Generally, SOM concentration is correlated positively with soil aggregate stability (Chaney and Swift 1984; Haynes and Swift 1990; Angers and Carter 1996; Chenu et al. 2000; Gal et al. 2006) and, hence, negatively with erodibility (Lickacz and Penny 2001). Many authors (e.g. Chaney and Swift 1984; Ekwue 1990; Carter 1992; Jastrow 1996) have defined algorithm that related aggregate stability to SOM in different soils. Many others have shown decline in SOM and the attendant reduction in aggregate stability to be either the cause (Tisdall and Oades 1982; Elliott 1986; Guerra 1994; Le Bissonnais and Arrouays 1997) or the effect of soil erosion (Olson et al. 2016). Therefore, measurements of SOM and aggregate stability could serve as indicators of structural degradation and soil erosion and as guide to appropriate management practises for enhancing soil productivity and/or sustainability (Le Bissonnais and Arrouays 1997; Ashagrie et al. 2007; Nciizah and Wakindiki 2015).

The severity of soil erosion by water is intricately linked to SOM concentration (Conforti et al. 2013), and so could be estimated from SOM by calibration involving matching possible range of SOM values with established erosion severity classes. However, the use of such a tool to estimate soil erosion extent requires certain precautionary measures: (i) it should be site-specific to take care of inherent stabilities of soils; (ii)

SOM determination should be standardized to allow for comparison of locations; (iii) the soil erosion severity classes should be established in such a way as to accommodate all possible erosion hazards, alongside their corresponding quantitative information on soil loss. A simple illustration of an imaginary case where SOM rarely exceeds 5% and topsoil depth is 20 cm is given (Table 2).

The SOM-mediated structural stability confers a lot of other attributes to the soil. Yet, there are many attributes of soils which, though not related to soil structure, are mediated by SOM. All these attributes which portray the functions of SOM are measurable, and useful inference could be drawn from them as regards the extent of soil degradation. As additional information about the numerous roles of SOM in assessing soil quality particularly its effect on soil structure and related attributes including some of those discussed in the subsequent sections of this paper, findings from selected studies are summarized in Table 3.

Notably, regulated increases in SOM are useful not only to boost aggregate stability, but also for good soil and crop management. Such increases are directly related to reduced soil bulk density; enhanced aeration, infiltration and sub-surface drainage; improved water-holding capacity in sandy soils; minimized crusting of fine-textured soils; ease of penetration; improved workability; increased cation/anion exchange and nutrient reserves; enhanced microbial activity and better plant nutrition (USDA-NRCS 1996; Esu 1999; Lal 2002).

Table 2 An imaginary case of potential extent of soil erosion as determined by SOM level

SOM (%)	Erosion severity class	Corresponding amount of topsoil loss expected (cm)
4.6–5.0	10	2
4.1–4.5	9	4
3.6–4.0	8	6
3.1–3.5	7	8
2.6–3.0	6	10
2.1–2.5	5	12
1.6–2.0	4	14
1.1–1.5	3	16
0.6–1.0	2	18
0.1–0.5	1	20

Table 3 Further selected examples of the relationships between indices of soil degradation and soil organic matter (SOM)

Location/study condition	Soil depth zone (cm)	Index of soil degradation	Degree of dependence on SOM (or SOM component)[a]	Reference
Iowa, USA; long-term tillage trial on loamy soil	0–15	Volumetric moisture content	$R^2 = 0.49$	Al-Kaisi et al. (2014)
Heilongjiang Province, China; long-term fertilization trial on soil formed on diluvial loess-like clay	0–20	Aggregate fraction >2 mm; Aggregate fraction 2–1 mm	$R^2 = 0.34$ (microbial activity); $R^2 = 0.52$ (microbial activity)	Shujie et al. (2009)
Savanna zone of Ghana; 2-year field study on fallow management systems on sandy clay loam soil	0–20	Soil colloidal stability indexed by dispersion ratio	$R^2 = 0.90$ and 0.58 in natural and legume fallows, respectively	Dowuona et al. (2011)
Savanna zone of Ghana; 2-year field study on fallow management systems on sandy clay loam soil	0–20	Soil colloidal stability indexed by water-dispersible silt	$R^2 = 0.79$	Dowuona et al. (2011)
Eastern South Dakota; tillage trial on silty clay loam soil	0–24	Water stable aggregation	$R^2 = 0.86$ (fPOM/SOM ratio and carbon–nitrogen ratio of humic acid)	Pikul et al. (2009)
Nsukka, Southeastern Nigeria; field experiment on organic and inorganic amendments on sandy loam soil	0–30	Soil total porosity; Soil available water capacity	$R^2 = 0.86$; $R^2 = 0.92$	Obi and Ebo (1995)
Quebec, Canada; long-term soil quality monitoring on loamy boreal forest soils	0–100 (per 5 cm)	Soil bulk density	$R^2 = 0.81$	Perie and Quimet (2008)
Hormozgan Province with arid climate in Southeastern Iran; stability study of sandy and loamy calcareous soils	0–150 (horizons sampled)	Soil colloidal stability indexed by water-dispersible clay	$R^2 = 0.49$	Moghimi et al. (2012)
Franklin County, Ohio; on-farm cover cropping study on silty clay loam soil	0–60	Soil structural stability indexed by % water-stable aggregates	$R^2 = 0.60$nr	Mukherjee and Lal (2015)
Kwadaso Kumasi in Semi-deciduous Forest Zone of Ghana; field trial on soil amendments and cropping systems on sandy loam Acrisol	0–15	Effective cation exchange capacity (ECEC) of the soil	$R^2 = 0.71$, 0.40 and 0.64; $R^2 = 0.95$, 0.55 and 0.96 (microbial biomass carbon) in 1st, 2nd and 3rd seasons, respectively	Logah et al. (2010)
Gazipur, Dhaka Division, Bangladesh; soil quality response to land use on loamy reddish brown Ultisols	0–15	Soil bulk density	$R^2 = 0.49$; $R^2 = 0.86$, 0.81 and 0.79 (fluvic acid, labile SOM and TMB, respectively)	Islam and Weil (2000)
Gazipur, Dhaka Division, Bangladesh; soil quality response to land use on loamy reddish brown Ultisols	0–15	Soil aggregate stability	$R^2 = 0.55$; $R^2 = 0.77$, 0.66 and 0.55 (fluvic acid, labile SOM and total SMB, respectively)	Islam and Weil (2000)
Gazipur, Dhaka Division, Bangladesh; soil quality response to land use on loamy reddish brown Ultisols	0–15	SOM and its components	Land use affected not SOM but fluvic acid, labile SOM and most indices of microbial biomass carbon	Islam and Weil (2000)
Putra and Buloh areas of Malaysia; soil properties and aggregate stability assessment of loamy/clayey soils	0–15	Soil aggregate stability indexed by water-dispersible silt + clay	SOM contributed not but phenolic fluvic acid & carboxylic humic acid contributed to aggregate stability; Model with adjusted R^2 of 0.88	Sung (2012)
Trinidad, West Indies; incubation study on soils of variable clay contents	0–10	Soil structural stability indexed by % water-stable aggregates	Increase in stability with increase in SOM particularly in low-clay soils	Wuddivira and Camps-Roach (2007)

Table 3 (continued)

Location/study condition	Soil depth zone (cm)	Index of soil degradation	Degree of dependence on SOM (or SOM component)[a]	Reference
Kaduna, Northern Guinea Savanna of Nigeria; soil aggregate stability and SOM under different forest tree species	0–25	Soil aggregate stability indexed by mean-weight diameter of soil aggregates	$R^2 = 0.22$ $R^2 = 0.85$ (chemically protected SOM) $R^2 = 0.74$ (biochemically protected SOM)	Lawal et al. (2012)
Inner Mongolia Province, China; grazing impact on loamy soils	0–10	Soil compaction indexed by tensile strength	Decrease in soil compaction with increase in SOM stabilization	Wiesmeier et al. (2012)
Sergipe, northeast Brazil; SOM and its components under four land use types on loamy sand soil	0–10 and 10–30	Land use-mediated differences in SOM and its components	Differences in SOM and fluvic acid but not in humic acid	Guimaraes et al. (2013)

SOM soil organic matter, *fPOM* fine (0.5–0.053 mm) particulate organic matter, *nr* negative relationship against the more commonly observed positive correlation, and the authors attributed this to the high clay content of the soil, *SMB* soil microbial biomass

[a] Where the relationship was between an index of soil degradation and SOM component, the component in question is enclosed in a parenthesis

SOM and plant nutrients: predicting nutrient availability from SOM

Organic matter releases many plant nutrients (especially N, P and S) as it is broken down in the soil (Bot and Benites 2005). Being a reservoir of plant nutrients, SOM measurements can lead to inferences on the quantity of nutrients present in a soil (Craswell and Lefroy 2001).

Nitrogen

The SOM contains considerable amount of organic N. Kapland and Estes (1985) argued that SOM contained about 5.6% of total N. Loss of SOM is often followed by a corresponding loss of N (Harms et al. 2004; Ashagrie et al. 2007). In Canada, it is known that a decrease in SOM of 2% releases about 2690 kg N per ha (Lickacz and Penny 2001). Generally, SOM is highly correlated with organic N, the relationship being SOM = 11.60 N (Plank 2001). It is also known that 93–97% of topsoil N is organic, while the rest is mostly non-exchangeable ammonium (Stevenson 1982; Fageria 2012). Thus, organic N is hardly denitrified or leached unlike inorganic N (nitrate). Positive relationships often exist between SOM and total N (Kapland and Estes 1985; Obalum et al. 2012b). The SOM thus contributes to mineralizable soil N pool, though this depends on the quantity and quality of SOM (Sahrawat 2004).

Phosphorus

In most surface soils, organic P constitutes 20–80% of total P (Lefroy et al. 1995). Reports about SOM effect on P sorption are conflicting. Although increase in SOM has been reported to have limited influence on P sorption (Borggaard et al. 2005) or even increase P sorption (Harter 1969; Debicka et al. 2016), a negative relationship between SOM and P sorption is a more common observation (Earl et al. 1979; Frossard et al. 1986; Violante and Gianfreda 1993; Strom et al. 2002; Hunt et al. 2007). This is because SOM decomposition releases organic acids which form complexes with Al and Fe that sorb P (Hue 1991; Fageria 2012). Overall, SOM effect on P sorption depends on P content of organic material added to the soil (Guppy et al. 2005), as well as on soil pH and initial soil P status (Debicka et al. 2016).

Sulphur

Freney (1986) observed a linear relationship between SOM and total S content of soils. This is because most S in especially non-calcareous and humid tropical soils is in the organic form (David et al. 1982; Lefroy et al. 1995). The SOM also improves S oxidation which is important in S-deficient and alkaline soils (Wainwright et al. 1986; Chibuike et al. 2012).

Trace elements

Such fractions of SOM as fulvic acid form complexes with trace elements such as Zn, Cu, Fe and Mn, and this increases the availability of these micronutrients for plant uptake in soils deficient in them (Stevenson 1991; Fageria 2012). By contrast, high-molecular-weight SOM fractions can reduce the availability of these elements especially when they occur in toxic amounts, by forming complexes with them to reduce their solubility (Stevenson 1991). Modification of soil pH in the presence of certain SOM also helps to immobilize the above elements and other toxic substances. Hence, organic amendments are increasingly used in remediation of soils polluted with heavy metals (Zhang et al. 2013; Gul et al. 2015).

Cation exchange capacity

The role of SOM in the cation exchange capacity (CEC) of soils is well documented (Drake and Motto 1982; Asadu and Akamigbo 1990; Asadu et al. 1997; Obalum et al. 2013; Asadu and Chibuike 2015). In highly weathered and acid soils, the contribution of SOM to CEC greatly exceeds that of particle size fractions (Krull et al. 2004). Turpault et al. (1996) found that the CEC of SOM in the surface layer of acid forest soils (pH, 3.4–4.7) represented 10–85% of the total CEC of the fine earth. They attributed the lower contributions to the low pH of the soils which resulted in the carboxyl groups of SOM being largely undissociated. Oorts et al. (2003) reported that SOM contributed 75–85% to the CEC of a Ferric Lixisol under different organic matter inputs in the form of litter from several trees in south-western Nigeria. They noted that differences in carbon content of the litter were responsible for the observed variations and that the biochemical composition of the litter did not affect the CEC of SOM.

Factors such as soil type influence the role of SOM in the CEC of soils. For instance, SOM contribution to CEC can be greater in coarse-textured soils and in low-activity-clay soils compared to fine-textured soils and high-activity-clay soils (Weil and Magdoff 2004). Furthermore, SOM fractions differ in their roles, with the lower-molecular-weight fractions having higher CEC of about 500 cmol kg^{-1} compared with the higher-molecular-weight fractions with CEC of about 170 cmol kg^{-1} (Wolf and Snyder 2003; Krull et al. 2004).

Soil microbial biomass

The soil microbial biomass (SMB)—the living part of SOM excluding plant roots and soil animals—despite constituting a small proportion is a major component of SOM as it serves as both source and sink of nutrients (Smith and Paul 1990). The SMB regulates the supply of nutrients through the decomposition of plant and animal residues (transformation of carbon); thus, it provides the energy for cycling of major plant nutrients (Paul and Voroney 1980; Krull et al. 2004). About 1–5% of soil C and N is stored in soil microbes (Duxbury et al. 1989), and these elements are released into the soil when the microbes die.

Parameters of SMB, especially C and N, have been used as indicators of soil quality (Jordan et al. 1995; Trasar-Cepeda et al. 1998; Baaru et al. 2007; Franchini et al. 2007). Such indices as SMB, enzyme activity, basal respiration rate and microbial diversity are used to assess soil degradation not only because of their sensitivity to change but also of their role in nutrient cycling (Visser and Parkison 1992; Brookes 1995; Baaru et al. 2007; Martinez-Salgado et al. 2010). The SMB easily changes with variations in environmental conditions; hence, it can be used as an early indicator of changes in soil properties resulting from climate and land use change (Jordan et al. 1995; Trasar-Cepeda et al. 1998; Baaru et al. 2007).

Besides cycling of plant nutrients, SOM improves soil quality by regulating the population of pathogenic organisms. Adequate SOM level reduces the population of parasitic organisms by favouring the growth of saprophytic organisms (Fageria 2012). Certain soil microbes such as mycorrhizal fungi may enhance the availability of nutrients to plants, since the fungi increase the surface area of plant roots which enables plants to access nutrients from deeper layers. Other

mechanisms for greater plant uptake of nutrients particularly P in the presence of mycorrhiza have been summarized by Cardoso and Kuyper (2006). Furthermore, Griffiths et al. (2000) reported that soils with greater microbial diversity are more resilient and resistant to perturbation compared to those with lower diversity. However, identification of the composition of soil microbial communities is not necessary for a better quantification of nutrient dynamics in the soil; a more holistic and efficient approach is measurement of the different nutrient fluxes linking specific pools in a system (Nannipieri et al. 2003).

Ameliorative effect of SOM on other forms of soil degradation

The SOM can ameliorate other forms of soil degradation. Dumat et al. (1997) observed that the presence of SOM reduced the affinity of soil clays for caesium, though that depended on the concentration of caesium. Also, high SOM and associated microbial activity may help to counteract the side effects of persistent soil-applied herbicides (Plank 2001). A comparison of SOM profiles of soils stressed by environmental contaminants with those of closely matched reference soils was undertaken by Beyer (2001). He observed that stressed soils showed an abrupt decrease in SOM below a depth of 2.5 cm, whereas the profiles of the unstressed soils showed a continuous decrease in SOM from the uppermost mineral soil layer (0–2.5 cm) down to 15 cm. This was ascribed to low macrofaunal activities by those stressors, inhibiting the addition of organic matter into the soil profile. In a similar study, soils under serious environmental stress were compared with adjacent unaffected ones in the Netherlands (Kemmers and Van Delft 2006). The degraded sites had distinct horizon differentiation due to decreased bioturbation. They were characterized by poor drainage and low pH, nutrient status and microbial activity, as a result of which they sequestered poorly decomposed SOM.

Perspectives/conclusions

The SOM is a valued component of any sustainable production system. Considering its importance in the physical, chemical and biological functions of the soil, SOM is a necessary tool for assessing the level of degradation or otherwise of a soil. It would continue

to be relevant as long as the need still exists of a single parameter indicator of soil degradation. Many management practises can influence the long-term accretions or depletions of SOM (Lal 2009, 2015). It is the extent of such changes in SOM that determines the extent of soil degradation. The relationship between SOM and soil functions (especially aggregate stability) that define soil quality is, however, not always linear. The relation may vary with the method used to measure stability and other soil properties that influence structural behaviour. Increases in SOM level beyond a certain critical limit in some soils could sometimes be detrimental to structural stability and fertility status of the soil. Caution should, therefore, be exercised in adopting practises devoted to increasing SOM so as to avoid exceeding the limit.

Some good attributes make the choice of SOM as indicator of soil degradation superb. It is a soil property that makes sense to the grower as useful inference about its status in the soil could be made from mere visual observation. Also, its status could be inferred in the field from such an easily measurable soil property as infiltration rate. The SOM defines stability and fertility of soils. The severity of different forms of soil degradation other than erosion could also be estimated from SOM level or condition in the soil. The SOM is sensitive to management; it also indicates the direction of management once determined. The effects of management practises to improve SOM and the condition of degraded soils are almost predictable. The use of suitable management practises can enhance SOM and soil quality. These practises directly or indirectly add organic material to the soil to improve water and nutrient use efficiencies.

Although establishing the relationship between SOM and soil quality and agronomic productivity has been proposed (Lal 2009), shifting research focus from the use of whole SOM to the use of its labile or microbial component as indicator of soil quality may be a more sensitive approach to the study of soil degradation (Guimaraes et al. 2013). In this approach, SOM could still be viewed as indirect indicator, since soil microbes thrive on SOM. Both soil structural stability and biodiversity are manifestations of SOM level, and they influence each other. Therefore, the use of microbial activity does not imply a new indicator altogether. Since physicochemical parameters compared to biological parameters change slower in soils, the use of biological parameters for early detection of degradation could be more appropriate.

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