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Review

Relation between soil organic matter and soil physical properties

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Soil physics is the study of soil physical properties and processes. It is applied to management and prediction under natural and managed ecosystems. Soil physics deals with the dynamics of physical soil components and their phases as solid, liquids, and gases. The change of soil climate and the incorporation of aboveground C inputs within the soil lead to no unique effect on biodegradation rates, because of their strong interaction with the regional climate and the soil physical properties. The periodical perturbation of soil structure by tools and the subsequent drying–rewetting cycles may be the major factor increasing organic matter decomposition rates by exposing the organic matter that is physically protected in microaggregates to biodegradation. This paper reviews the assessed effects of tillage on organic matter, the scale, extent and mechanisms of physical protection of organic matter in soils.

Keywords: Land use; Organic matter decomposition; Soil C

INTRODUCTION

Soil organic matter (SOM) is the organic matter component of soil, consisting of plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by soil organisms. SOM exerts numerous positive effects on soil physical and chemical properties, as well as the soil's capacity to provide regulatory ecosystem services. Particularly, the presence of SOM is regarded as being critical for soil function and soil quality.

The positive impacts of SOM result from a number of complex, interactive edaphic factors; a non-exhaustive list of SOM's effects on soil functioning includes improvements related to soil structure, aggregation, water retention, soil biodiversity, absorption and retention of pollutants, buffering capacity, and the cycling and storage of plant nutrients. SOM increases soil fertility by providing cation exchange sites and acting as reserve of plant nutrients, especially nitrogen (N), phosphorus (P), and sulfur (S),

along with micronutrients, which are slowly released upon SOM mineralization. As such, there is a significant correlation between SOM content and soil fertility.

The effect of tillage on soil organic matter (SOM) dynamics has been investigated for its consequence on soil structural stability, soil erosion, nutrient availability, nutrient loss and pollution (Doran et al., 1996). More recently, the sudden change in the global carbon cycle raised the question of carbon (C) storage in soils (Houghton, 1995). Policy makers asked if tillage, by incorporating plant material into the soil (thus moving the plant material outside the atmosphere) could store atmospheric C. More precisely, the question was asked, whether some tillage practices may contribute to reduce the emission of greenhouse gases. Tilling the soil has two main goals, on one hand the control of soil structure for seedbed preparation, on the other the incorporation of organic residues and weeds into the soil. The location of

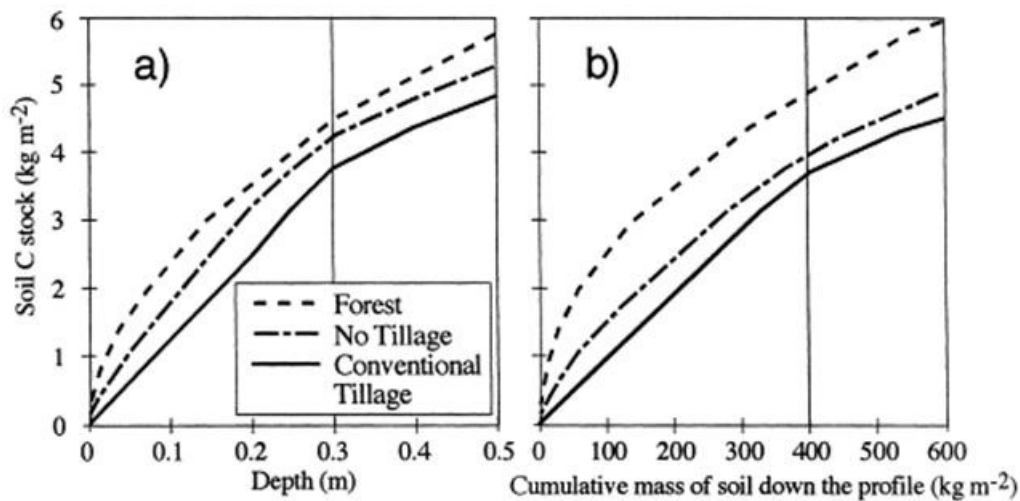


Figure 1. Carbon stocks in French silt loam under three management practices, presented versus depth (a) and versus cumulative mass of soil down the depth (b). The figure illustrates the necessity to take into account bulk density variation and appropriate depth for the comparison of carbon stocks.

SOM is thus modified by tillage at different scales: from that of the profile (cm to dm) to that of the soil structural units (μm to cm). The location of organic matter within the soil structural units has been demonstrated to control SOM dynamics (Oades, 1995; Skene *et al.*, 1996; Angers and Chenu, 1997; Feller and Beare, 1998). A process described as the “physical protection” of SOM from biodegradation is generally invoked. In this paper we focus on tillage-related changes in SOM through changes in physical protection. We first review the assessed changes of SOM dynamics with tillage, then the mechanisms of physical protection, and discuss their contribution to the changes in SOM dynamics, compared with other effects induced by tillage.

Assessment of changes in SOM status associated with tillage

Difficulties in comparing SOM stocks and status

Change in SOM with land use has been generally assessed by comparing SOM stocks in differentiated systems. Such comparison encounters several difficulties. The most important are due more to soil volume comparison than to C determination. We illustrate the importance of soil volume comparison in the case of French silt-loamy soils under three land uses (Figure. 1). The average bulk density in the top 30 cm of soil has been reported to vary there from 1.08 Mg m^{-3} in forest soils to 1.35 Mg m^{-3} in mouldboard-ploughed tilled soils, and to 1.48 Mg m^{-3} in long term zero-tillage systems where soil compaction by machinery is not compensated by decompacting tools (Balesdent *et al.*, 1990; Elzein and Balesdent, 1995). As a consequence, the C balance of different land-use systems requires the comparison of

different soil volumes. Moreover, mouldboard ploughing incorporates SOM by mechanical action plus further SOM translocation to layers deeper than the deepest ploughing event. This depth (the base of the Ap layer) has been recorded to reach up to 35 cm, so that C storage comparisons have to take this volume into account. An additional difficulty is that in virgin and zero-tillage systems, a substantial part of SOM lies above the mineral soil as litter, or even in the soil as dead tree stumps. The amount of litter has to be estimated by surface weighing and C determination. Studies that neglect such effects lead to biases in various directions. Neglecting the difference in bulk density or the litter generally underestimates native-SOM storage. Comparing volumes which are too shallow generally underestimates ploughed-SOM storage (Figure. 1). Considering other investigations such as that of SOM nature, location, or biological properties, it seems also necessary to take into account comparable soil volumes.

Cultivation of native soils

Cultivating a virgin soil has been reported extensively to cause a drop in SOM concentration. Burke *et al.* (1989) in the US Grassland, and Brown and Lugo (1990) in the tropical forest zone reported average C losses ranging from 10 to 55% of the native C. The variance was explained by climate and texture with the maximum losses under the wettest climate and in coarse-textured soils. Changes in the C input to the soil cannot account for all the observed decrease, since cultivated crops often return more C than virgin vegetation, despite the removal of harvested C. Soil erosion contributes to the drop in many places, but not in all. It is then necessary to invoke an increase in the rate of C biodegradation. Moreover, most kinetic and chronosequence studies showed that the

Table 1

Compared values of total C content, total N content and mineralizable N for the soil surface layer of no-tilled and conventionally-tilled soils of varying types and management histories

Reference	Soil texture ^a	Before tillage treatments	Under tillage treatments ^b	Layer depth LD1 ^c	NT/CT ratio						Meth ^e		
					Total CLD1 ^d	Total N LD1 ^d	Mineralizable N						
							Ld1 ^d	Ld3 ^d	Total	Ld ^d			
Doran (1980)	SiL	Bluegrass sod ^f	>50	Corn	9	0±30	1.55	1.48	1.52	0.94	0.86	A	
	CL	Corn		Corn	9	0±30	1.07	1.13	1.45	1.00	0.75	A	
	SiL	Bromegrass sod ^f	>50	Corn	5	0±30	1.20	1.08	1.13	0.95	0.92	A	
	SiCL	Soybeans ^g		Corn	3	0±30	1.10	1.00	1.19	0.93	0.97	A	
	L	Native sod		Wheat-fallow	9	0±30	1.50	1.45	1.58	0.97	1.07	A	
	SiL	Wheat-fallow		Wheat-fallow	10	0±30	1.25	1.12	1.34	1.12	0.97	A	
	SiL	Wheat-fallow		Wheat-fallow	9	0±30	1.10	1.12	1.25	0.81	0.83	A	
Carter and Rennie (1982)	SiL	Cultivation (CT)		Wheat-fallow	16	0±4	1.00	0.97	1.61	1.10		B	
	SiL	Cultivation (CT)		Wheat-fallow	16	0±4	1.00	0.97	2.00	0.75	1.42 ^j	C	
	CL	Cultivation (CT)		Wheat-fallow	12	0±10	1.02	0.85	1.48	0.58	0.99 ^j	C	
	CL	Cultivation (CT)		Continuous wheat	4	0±10	0.83	0.87	1.63	0.84	1.29 ^j	C	
	L	Cultivation (CT)		Wheat-soil seed	2	0±10	0.84	0.87	1.00	1.04	1.00 ^j	C	
Rice et al. (1988)	SiL	Bluegrass pasture ^e	>50	Corn and winter cereals	16	0±15		1.4	2.30	1.07	1.72	C	
Wood and Edwards (1992)	Fine SaL	Row crop product	>50	Crop rotations	10	0±20	1.77	1.84	1.92	1.11	0.5	1.44	C
Cabrera (1993)	Fine SaL					0±10	1.71	1.74	1.47			1.47	C
Franzluebbers et al. (1994)	SaCL			Sorghum ^h , winter rye ⁱ	13	0±15	1.82	1.77	1.32	0.75		0.95	C
Beare et al. (1994)	SiCL			Crop rotations	10	0±20			2	1	1	1.25 ^j	C
McCarthy et al. (1995)	SiL			Corn	10	0±15		1.27	1.75			1.75	D

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^a Si: silt; L: loam; C: clay; Sa: sand.

^b Tillage treatments are NT: no-till (except Ref. (Wood and Edwards (1992)): reduced tillage), and CT: conventional till (moldboard plowing).

^c Ld 1, 2 and 3 are sublayers of increasing depths within each given soil layer; Total Ld: total layer depth.

^d Methods for mineralizable N determination. A: autoclaving in 0.01M CaCl₂; B: extraction in 0.5M NaHCO₃; C: laboratory incubations; D: ¹⁵N isotope-dilution technique.

^e *Poa pratensis* L.

^f *Bromus* sp.

^g *Glycine max* L.

^h *Sorghum vulgare* Pers.

ⁱ *Secale cereale* L.

^j Values for total soil surface layer calculated from data given in the reference for sublayers by assuming similar bulk densities among sublayers

decline of C concentration was very rapid in the first years following forest clearing or grassland conversion (Boiffin and Fleury, 1974; Tiessen and Stewart, 1983; Arrouays *et al.*, 1995). The SOM that disappears in a few years can be defined as a labile pool. Such a large labile C pool could not exist under steady-state condition in the native site: C necessarily had a slower decay rate before cultivation. The kinetic studies thus introduced the concept of SOM pools that are protected from degradation under the virgin vegetation and de-protected by cultivation (Van Veen and Paul, 1981). This deprotection apparently affects all SOM pools, with a rate depending on the intrinsic lifetime of each pool (Balesdent *et al.*, 1998).

Tillage may be one among other causes for increased biodegradation rates in native soils. Crop production is accompanied also by modifications of soil climate, nature of the carbon sources, nutrient availability to microorganisms, microbial and fauna diversity and, in many cases, by soil pH increase. An increase in the latter may enhance bacterial activity of acid soils and modify SOM adsorption and solubility. Long term tillage/no-tillage experiments are thus the most appropriate experiments to investigate the specific effect of tillage on SOM.

Cultivation under different tillage practices

SOM stocks

Kern and Johnson (1993) have synthesised seventeen studies comparing organic C under different tillage practices in the USA. The duration of the differentiation ranged from 3 to 44 years, and the tillage practice preceding the experimentation was generally conventional tillage. Kern and Johnson (1993) concluded that the C level was on the average higher under no-tillage (NT) than under conventional tillage (CT). The estimated difference (0.7–1.8 kg C m⁻²) was linearly related to the C level under CT. The difference was observed mainly in the top 8 cm, and no difference was observed below 15 cm. The conclusions of most similar studies are in agreement with these results (Table 1). Nitrogen stocks show similar trends but with smaller differences than those concerning C (Table 1). Staley *et al.* (1988) nevertheless pointed out the difficulty of predicting changes induced by tillage practices when considering data acquired through time-point comparisons, rather short or irregular periods of time under tillage practices, or in different soil types. As for other soil

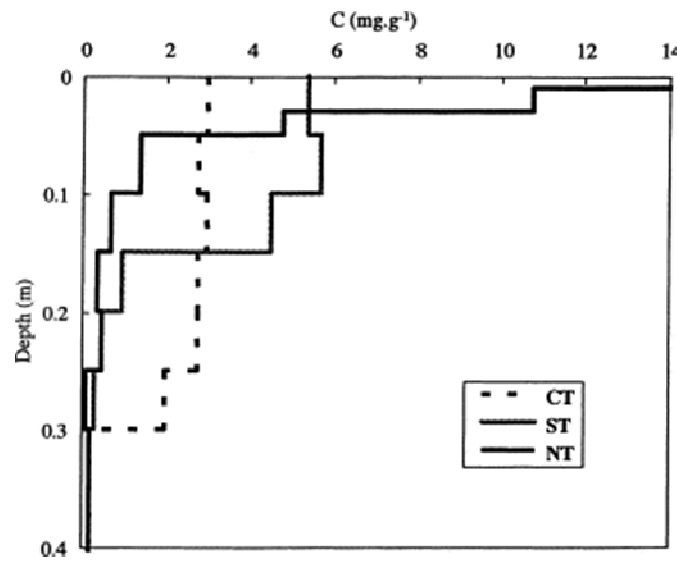


Figure 2. Depth profiles of the new organic carbon incorporated in the soil under three tillage practices, in the 17 years-old experiment of Boigneville (France). New carbon from the 17 successive crops (maize) was measured using natural ^{13}C abundances

parameters, the short term response of SOM dynamics to changes in soil management will depend on the equilibrium conditions of the system to which these practices are applied.

SOM location

While total SOM concentration is only weakly changed by tillage practices, SOM location is dramatically affected. The most obvious effects concern the organic profile. We illustrate this in the 17 years-long experiment of Boigneville (Ile-de-France, France), where three tillage practices were compared on a soil previously tilled for a long period. In that experiment the C derived from the crop was distinguished from older SOM using the natural ^{13}C labelling of the crop, i.e., *Zea mays* L. (Balesdent et al., 1990). In Figure. 2, we report the profiles of the organic C that is derived from the 17 successive crops, under three tillage practices. Whereas in the NT treatment more than 50% of the new C was found in the first 4 cm, and only 20% below 25 cm, C was homogeneously spread in the ploughed layer of the tilled system. According to the differences in the location of new C sources, differences are to be expected in the location of the biological fluxes such as plant material biodegradation, soil respiration or nitrogen immobilization. An accumulation of biological activity and biomass at the surface under NT practices has been reported in numerous studies (Doran, 1980; Lynch and Panting, 1980; Carter and Rennie, 1982; Doran, 1987). As most SOM is concentrated on the surface, the environment of biodegradation is that of the top few cm. The location of SOM can be affected by tillage at other scales also. In the tillage experiment of Boigneville (Figure.

2), Puget et al. (1996) separated particulate organic carbon (POC) coarser than $50\ \mu\text{m}$ in two different locations: outside stable aggregates or occluded within stable aggregates. The results (Figure. 3) showed significant differences between tillage treatments. The distribution of POC within aggregate size classes was similar, but more free POC was found in the untilled than in the tilled layer (37% of POC vs. 26% of POC). This result suggests a rapid incorporation of organic C within aggregates by the tillage itself, which favoured the contact between plant-derived material and the soil mineral matrix.

SOM decay rates

Evidence for increased decay rates of SOM due to tillage was derived from the previous experiment of Boigneville. In Figure. 4 we report the concentration of the organic C, that is derived from that initially present in the topsoil. Initial SOM concentration ($3.6\ \text{kg C m}^{-2}$) was the same in all plots and its vertical distribution was homogeneous, due to tillage preceding the experiment. The amount of original SOM mineralized in 17 years was doubled in the conventional tillage ($0.95\ \text{kg C m}^{-2}$) compared with that in no till ($0.45\ \text{kg C m}^{-2}$). In the superficial tillage (to circa 12 cm), the result was intermediate, and the profile showed that the mineralization was as in CT in the 0–10 cm layer and as in NT in the 20–30 cm layer (Figure. 4). We considered this increased mineralization to be a consequence of release of previously physically protected SOM. In this site, the soil had been tilled for a long time before the experiment (with a weaker intensity than at present), so that the protected SOM could only represent a fraction of the initial C. It was therefore concluded that the

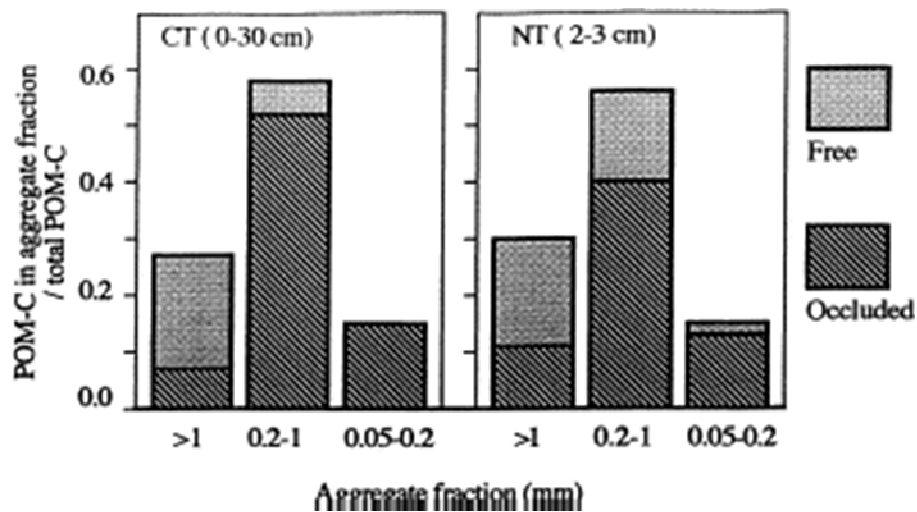


Figure 3. Comparison of the distribution of particulate organic carbon (POC) coarser than 50 μm in aggregate fractions of samples from conventionally-tilled (CT) and no-till (NT) treatments. Free and occluded POC are separated by densimetric fractionation and refer to POC located within or out of the aggregates. (after Puget et al., 1996).

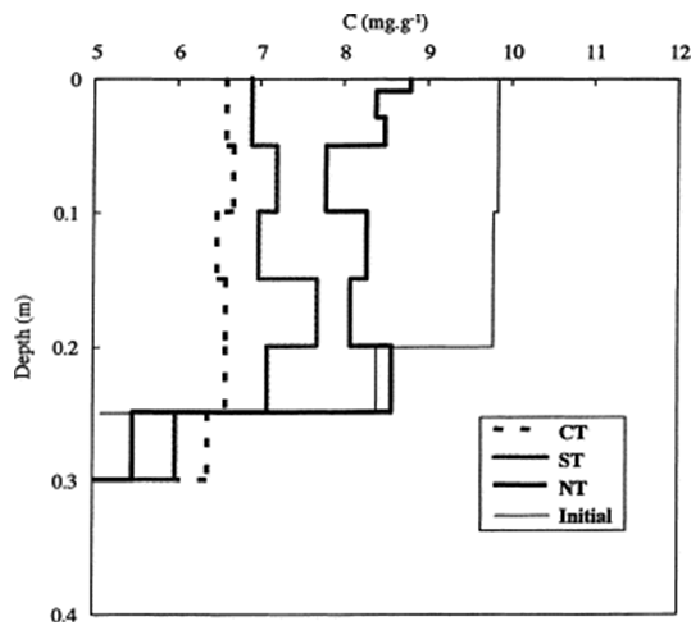


Figure 4. Depth-profiles of the organic carbon remaining from the initial carbon under three tillage practices, in the 17 years-old experiment of Boigneville (France) (redrawn after Balesdent et al., 1990).

decay rate of SOM under CT was much more than twice that of protected SOM.

Tillage may affect also the decay of more resistant organic matter pools. In Sanborn Field (MO, USA), we compared the amount of old, stable, organic matter derived from the initial prairie in a 100 year-old experiment, which included two treatments: continuous wheat (*Triticum aestivum* L.), tilled yearly, and grass, tilled and receded every 4 years (Balesdent et al., 1988). The residual amount of old C in the top 20 cm was 6.5 mg g⁻¹ under wheat and 10.5 mg g⁻¹ under grass. This residual old SOM has at present a mean

residence time of more than 500 years, in both cases. Some erosion may have occurred in the early years of wheat cultivation, but it could not account for all the observed results. The differences in SOM mineralization were then ascribed to the de-protection of a fraction of the stable C pool.

Mineralizable nitrogen (N) and microbial biomass

The consequence of tillage on the N cycle has been the subject of numerous studies. Table 1 shows data from the

literature comparing total and mineralizable N under NT and CT, on various types of soils and throughout a wide range of past management (Table 1.). Mineralizable N in the topsoil (0 to 5–10 cm) is in all cases greater, up to two-fold, in the NT compared with that in the CT. On the whole, the trend at each soil depth is for more mineralizable N under NT compared with that under CT. Moreover, McCarty *et al.* (1995) found that whereas tillage practice, geographic location, fertilizer N rate, and depth of sampling had a marked influence on the amount of total N, biomass N and active N in soils, only tillage seemed to influence the distribution of soil N in the different pools. (The slope of total N vs. active N was 2.5 greater under NT compared with that under CT.)

Considering microbial biomass, Staley *et al.* (1988) studied a 0–20 years NT chronosequence settled on previously CT soils. They reported levels of biomass C in the 0–7.5 cm layer of NT of twofold those measured under CT after one year, followed by a gradual equilibrium to approximately 30% greater levels in NT compared with CT soils after 10 years. In the process of SOM turnover, the microbial biomass is considered to contribute to the labile pool of nutrients and C. Therefore, increased labile N under NT soils may be related to the general observation of a higher microbial biomass. Experiments with ^{13}C labelling helped to assess differences in the size of soil active N pool under different tillage practices (Fredrickson *et al.*, 1982; Kitur *et al.*, 1984; Carter and Rennie, 1984; Meisinger *et al.*, 1985; Haugen-Kozyra *et al.*, 1993). They showed a greater immobilization and a lower efficiency of use by the crop of fertilized N under NT. Differences in the efficiency of N fertilizer use are related to the differences in the isotope dilution of fertilizer N with soil organic N through mineralization–immobilization, with the extent of mixing probably depending on the size of soil active N pool. The lower efficiency of N fertilizer use by the crop under NT than under corresponding CT treatments could then reflect an overall larger active N pool in soil under NT compared with that in CT soils.

Following the conclusions of Germon *et al.* (1991) on reduced tillage effects on soil N dynamics, more mineralizable N under NT than under CT may be related to a greater labile N pool, but with a slower turnover rate. When NT practices are imposed on native soils, i.e., pastures converted to annual crops, the trend under NT would be towards a more efficient preservation of the initial active N pool in soil (Rice *et al.*, 1986). When NT is imposed on a soil previously under CT, the trend would be towards rebuilding of an active soil N pool which is continually mineralized under CT. This is illustrated by the findings of Staley *et al.* (1988) who found that the shift from CT to NT resulted after 1 year in a “stratified ecosystem under NT resembling pastures”.

Mechanisms responsible for tillage-induced changes in SOM status

Tillage is suspected to influence SOM dynamics through three major actions: the change of soil climate, the incorporation of SOM into the soil matrix, and finally the periodical disruption of the soil structure.

Changes of soil climate and physico–chemical environment due to tillage

Tillage affects soil climate, and is thus expected to directly affect microbial activity. The presence or absence of the mulch and the changes in soil compaction are mainly involved. No-till or conservation tillage have been reported to decrease topsoil temperature, by 0–3°C (Coote and Malcolm-McGovern, 1989), and to increase the water content of soil and thus reduce soil aeration. The combination of the two latter effects is expected to increase biological activity as long as soil is not saturated and to possibly reduce it when the water content is close to saturation (Mielke *et al.*, 1986). According to the Rothamsted C model parameters (Jenkinson *et al.*, 1992), decreasing the temperature by 2°C is estimated to multiply the decay rates by a factor of 0.8. Increasing soil moisture by 10% of pore space (e.g., by 10 mm water in the topsoil) may multiply decay rates by a factor of 1.8 under driest conditions, to 1.2 near the biological optimum. As a result, the balance of all effects would range from a nil difference to a higher decay rates under no-till. This will depend highly upon climate and soil physical properties. Under climates where dry and hot seasons coincide, the balance of both effects is expected to increase biological activity under no till, since a higher water content maintained in summer is the major contribution to a high annual biological activity. The seasonal pattern of soil respiration is thus also suspected to depend on tillage practices (Hendrix and Groffman, 1988). Furthermore, the higher evaporation in tilled soils leads to more drastic drying/rewetting cycles in the upper part of the soil, as compared with no-till.

Another obvious change is the incorporation of aboveground C inputs within the soil. When compared with their location in the top cm under no till, their average depths may currently be 0.15 m in the case of mouldboard ploughing. The climate of biodegradation is changed accordingly and the diurnal and seasonal variations are much weaker. Furthermore, the contact of the SOM with the soil solutes, such as inorganic N, is stronger and may enhance biodegradation (Parker, 1962), but the contact of SOM with the clay matrix may be increased too, which may conversely reduce biodegradation. These processes are still poorly quantified. It is nevertheless expected that no

Table 2
Comparison of the effect of aggregate crushing on the C and N mineralization in different soil types

Reference	Soil texture ^a	C or SOM/2 content (%)	Sampling depth (cm)	Soil management	Incubation (days)	Initial size of aggregates (mm)	Crushed to (mm)	C mineralization crushed/uncrushed	N mineralization crushed/uncrushed
Craswell and Waring (1972)	71% clay	1.91	0–10	Virgin	4	<2	<0.15	1.23	1.82
		1.16		Cultivated		<2	<0.15	1.34	2.00
	60% clay	2.68		Virgin		<2	<0.15	1.14	1.78
Elliott (1986)		0.98		Cultivated		<2	<0.15	1.25	1.64
	S	2.35		Virgin	20	>0.3	<0.3	1.04	1.38
		1.82		Cultivated		>0.3	<0.3	1.19	1.41
Gupta and Germida (1988)	SaL	3.86	0–15	Virgin		>0.25	<0.25	1.14	1.43
		2.04		Cultivated		>0.25	<0.25	1.05	1.18
	2% clay	1.7	0–20	Virgin	70	<8	<1	0.75	0.82
Hassink (1997)	2.4% clay	3.8				<8	<1	0.8	0.74
	23% clay	1.6				<8	<1	1.16	1.66
	24% clay	4.6				<8	<1	1.65	1.21
	51% clay	5.3				<8	<1	2.31	3.49
	54% clay	4.9				<8	<1	1.13	3.40
Beare et al. (1994)	SaCL		0–5	Tilled	20	>2	<0.25	1.06	1.07
						0.25–2	<0.25	1.12	1.04
			0–5	No-till		>2	<0.25	1.29	1.55
						0.25–2	<0.25	1.29	1.35
			5–15	Tilled		>2	<0.25	1.14	1.08
						0.25–2	<0.25	1.26	1.09
			5–15	No-till		>2	<0.25	1.20	1.10
						0.25–2	<0.25	1.26	1.09
Powlson (1980)	LC	2.95	0–30	Virgin	10	<6.35	<1 (wet, mortar)	2.49	1.61
	25% clay	1.07		Cultivated		<6.35	<1 (wet, mortar)	3.48	3.00
Sorensen (1983)	6% clay	0.8		Cultivated	10		<0.15 (mortar)	4.30	
	12 % clay	1.3					<0.15 (mortar)	6.90	
	16% clay	2.0					<0.15 (mortar)	10.80	
	34% lay	2.4					<0.15 (mortar)	6.40	

^a S: silty, Sa: sandy, C: clay, L: loam.

unique trend would be derived from these differences of SOM location and of biodegradation climate, due to their strong interaction with the regional climate, the soil physical properties, and the nature of microbial populations. These changes in the climatic environment cannot be solely responsible for the systematically lower SOM storage in tilled soils.

Soil structure control over SOM decomposition

The rates of decay of SOM compounds in the soil matrix are at least one order of magnitude less than the rates of decay in the absence of soil (Oades, 1995). The concept of physical protection was then introduced to account for any limitation of biodegradation of SOM due to its location in the soil matrix, i.e., to its interaction with mineral surfaces or to its location within aggregates. Physical protection is generally evidenced by the flushes of SOM mineralization observed upon soil aggregates disruption. The ability of different levels of soil structure to protect SOM against decomposition has been generally assessed. Information was mainly inferred from studies with controlled destruction of the soils structure, fractionation of different structural units or microscopic observations.

Information deduced from destructive treatments

The breaking of macroaggregates to microaggregates (<150–300 µm diameters) generally causes an increase in C or N mineralization (Table 2). The increase was generally larger in cultivated soils rather than in virgin ones (Craswell and Waring, 1972; Powlson, 1980; Elliott, 1986). Within cultivated soils the increase was weak or even insignificant in tilled soils, whereas it was larger in untilled ones (Beare et al., 1994). The pools of SOM physically protected in macroaggregates were in this case estimated by Beare et al. (1994) to account for 10% of C standings stocks in the tilled soil and for 19% in its no-tilled counterpart. As apparent in Table 2, it may be concluded that macroaggregates offer some protection against biodegradation and that their “protective capacity” increases with SOM and clay content, and absence of tillage (Table 2). These parameters relate to some extent to the frequency at which aggregates are liable to be destroyed in situ conditions, by tillage operations or in relation to their intrinsic water stability, which depends on their SOM and clay content.

Information obtained from soil microscopic observations

Observations of the soil with electron microscopy techniques (scanning and transmission electron microscopies) have provided visual evidence of potential sites for physical protection. Fine organic particles or amorphous SOM may be located in pores too small to be entered by microbes, or held within the sub-units of clay particles; microorganisms or very fine plant fragments may be coated by clay particles (Kilbertus *et al.*, 1979; Kilbertus, 1980; Foster, 1981; Chenu, 1995). It is also apparent in such observations that microorganisms and organic components are physically separated by the arrangement of soil minerals.

Several levels of soil organization then offer some degree of physical protection against biodegradation. The bulk of results tends to demonstrate that microaggregates offer more effective physical protection than macroaggregates in relation to their respective physical stability, larger for microaggregates than for macroaggregates.

The mechanisms of physical protection

Several mechanisms have been invoked to explain the reduced decay rates in soil aggregates.

Sorption of SOM to solid surfaces

In vitro studies have demonstrated that organic compounds adsorbed to mineral surfaces may not be utilizable by microorganisms in the case of substrates directly absorbed in the cell (Dashman and Stotsky, 1986; Ogram *et al.*, 1985) as well as in the case of polymeric substrates requiring the action of extracellular enzymes (Cortez, 1977; Olness and Clapp, 1972). Reduced availability of SOM due to adsorption on minerals is thus assumed to take place in soils although its importance is not assessed. It is very unlikely that physical treatments such as tillage operation will affect the availability of adsorbed species. Conversely, wetting events in soil are likely to cause desorption of organics and thus de-protect them.

Sequestration in small pores

The access of substrates to microorganisms may be limited because they are located in pores inaccessible to microorganisms or their enzymes or because their diffusion to the cells is very slow. Evidence of the physical sequestration of organics in very small pores presumably inaccessible to microorganisms comes from microscopic observations (Kilbertus, 1980; Foster, 1981). On the other hand, experimental studies based on model systems have shown that the extent and kinetics of biodegradation were controlled by the ability of the substrate to diffuse to the cell

(Adu and Oades, 1978; Barlett and Doner, 1988; Scow and Alexander, 1992). In the soil matrix, diffusion is limited by the pores geometry and by the continuity and tortuosity of diffusion pathways. In tilled soils the periodic perturbation of the soil structural units on one hand may favour the accessibility of organic substrates to microorganisms as compared with no-till soils.

Control of microbial turnover by predators

Drying–rewetting events or the breaking of aggregates by grinding treatments are lethal to the microorganisms. About 25–80% of the biomass was killed by fine grinding (Powlson, 1980; Sorensen, 1983) and 15–30% by drying–rewetting treatments (Sorensen, 1983; Bottner, 1985; Van Veen *et al.*, 1985; Cortez, 1989; Van Gestel *et al.*, 1991). However, the contribution of the dead biomass itself to the mineralization flush was found to be minor (Powlson, 1980; Van Gestel *et al.*, 1991).

Soil structure may also indirectly control decomposition processes through predation of microorganisms by the microfauna. Grazing of microbes by nematodes and protozoa stimulates microbial growth (Rutherford and Juma, 1991) and can markedly increase C and N mineralization (Elliott *et al.*, 1980; Kuikman *et al.*, 1990; Rutherford and Juma, 1991). For example Rutherford and Juma (1991) reported 8–18% increases in C mineralization when protozoa were added to sterile soils previously inoculated with bacteria. Bacteria may occupy micropores (e.g., <3 µm) especially micropores between clay platelets (Heijnen *et al.*, 1993) and thereby become protected against predation by protozoa and nematodes because the latter are restricted to larger pores (Heijnen *et al.*, 1988; Postma and Van Veen, 1990). In no-till soils micropores generally represent a larger proportion of the porosity. It can be then hypothesized that the grazing pressure will be less in no-tilled soils as compared with tilled ones.

O₂ limitation

The mineralization of organic matter (i.e. CO₂ evolution) closely depends on the air/water balance in soils. Aerobic microbial activity increases with the soil water content until a value at which water restricts the diffusion and availability of oxygen. CO₂ production may then be decreased to 40% of its value at optimal aeration and water conditions (Linn and Doran, 1984). The occurrence of anaerobic zones within soil structural units may then result in slower rates of decomposition of SOM in these locations. Anaerobic zones were evidenced in the centre of centimetric soil aggregates when the aggregate porosity was water-saturated (Sextone *et al.*, 1985; Sierra *et al.*, 1995). Higher soil water contents and larger bulk densities, resulting of a larger proportion of the pore space being water-filled may result in less aerobic conditions in no-till soils as compared with tilled ones (Linn and Doran, 1984).

Relating the physical protection with the effect of tillage on soil structure

Tillage, by affecting the life-time and amount of aggregates wherein SOM is sequestered, is naturally suspected to influence the extent of physical protection. A decrease of the mean weight diameter of soil aggregates under tillage has been generally described (e.g. Beare et al., 1994, Puget et al., 1995). According to Tisdall and Oades (1982), macroaggregates coarser than 200 μm are more affected than microaggregates (2–200 μm) by management practices. The direct action of tilling tools and machinery probably do not result by itself in the most important deconstruction. Further changes are due to the action of water on the modified aggregate assembly. Such processes have been largely studied for their effect on seedbed degradation and hardsetting. Bresson and Moran (1995) experimentally investigated the change of seedbed structure by wetting and drying. Wetting caused a coalescence of the seedbed (100 mm thick) which was attributed to microcracking of coarse aggregates, partial slaking of aggregate margins and agglomeration of dust material. The rainfall action itself, by splash and soil material displacement, may lead to dramatic structural changes (Bresson and Boiffin, 1990). Microhorizons may be formed: a crust where soil particles could have been almost completely dispersed, and an underlying horizon of coalescent aggregates. The volume of these horizons may reach a few cm in one rain event. The fine displaced material can move into soil cracks and pores down to depths of a several cm. Tillage is thus suspected to indirectly increase aggregate disruption rates and probably also aggregate build-up rates (Dexter, 1988). Macroaggregates are obviously concerned, but microaggregates, which are the main sites of the physical protection are also concerned, even though the extent of the process is still poorly quantified. As a consequence of the periodical ploughing, the whole ploughed zone is progressively exposed to rainfall actions. The SOM included in microaggregates that is physically protected from biodegradation becomes accessible to biodegradation when released from the disrupted aggregates (Figure. 5). Under no till, due to the soil protection by the mulch and to the greater SOM content in the surface, the rate of disruption of the microaggregates rapidly decreases. The “physical protection” (of the soil) by the mulch thus contributes to the “physical protection” (of SOM) from biodegradation. Since the amount of SOM itself contributes to the stability of these aggregates (Monnier, 1965), SOM contributes to its self protection.

Modelling the effect of tillage on SOM dynamics through physical protection

Physical protection has been included in some models of SOM dynamics. For instance, Van Veen and Paul (1981)

introduced SOM pool in two states: physically protected and not physically protected (with a reduced life-time). The proportion of SOM which was protected was indicated by a “protection coefficient”, depending on land use. Molina et al. (1983) also used a slow HUMAD pool continuously formed, but periodically transferred to a more labile pool at cultivation events. Hassink (1997) attributed to a given soil a limited protective capacity and Hassink and Whitmore (1997) developed a model where the rate at which organic matter becomes protected depends on the degree to which the protective capacity is filled. Most of the current models of SOM dynamics simply affect the decay rates of SOM pools by an empirical parameter corresponding to land use. Further mechanistic modelling of the physical protection of SOM should ideally reproduce the process described by Golchin et al. (1994). It should take into account the forces that drive the dynamics of soil structural units wherein SOM is protected, among which the stabilisation of these structural units by the SOM itself. The resulting ‘self protection’ of SOM, i.e., decay rate constants depending on the amount of SOM itself, has a strong implication: physical protection would be involved in any change affecting SOM storage, e.g., SOM inputs. It also raises one mathematical difficulty: decay of SOM could not be described by first-order differential equations, and the parameters describing the process would become highly dependent on the scale of observation.

CONCLUSIONS

Tillage may influence C dynamics through various mechanisms. The first are the change of soil climate and the incorporation of aboveground C inputs within the soil. No unique trend for biodegradation rates can be derived from these differences of location, due to their strong interaction with the regional climate, the soil physical properties, and the nature of microbial populations. The second process is the periodical disruption of the soil structure. Two levels of organization are concerned. Macroaggregates and clods are partly disrupted by the tilling tools, especially by power harrows, in the top 0.1 m. The disrupting process is intensified in the seedbed upon drying–rewetting events. Microaggregates can be dispersed then and by further action of the rain, particularly in medium and coarse textured soils. A fraction of the SOM that is included in these microaggregates is physically protected from biodegradation and it becomes accessible to biodegradation when released from the aggregates. Physical protection and related deprotection by tillage appears to affect all classes of SOM life-times, from the year, and probably less, to the centuries. The deprotection obviously provokes a dramatic change of decay rates, indicating the high potential of biodegradation of protected SOM. Concerning total C storage, most of protected SOM has to be searched in the slow pool of small-sized SOM

(the HUM compartment of Jenkinson et al., 1992) found in microaggregates. As a consequence, the reconstruction of this pool is a very slow process, much slower than its destruction. When tilled and no-tilled soils are compared in terms of either C storage, C and N mineralization, the difference between treatments will thus be first dependent on the history of the system before change and on the duration after change. SOM dynamics is there in a transient state. The rate at which SOM dynamics changes is necessary linked to the life-time of soil structural units. For a better prediction of the consequences of tillage on SOM turnover, further quantitative investigation is needed, concerning the sites of SOM protection, its extent and the in situ rates of microaggregate build-up and disruption.

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