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Conservation Agriculture and Soil Carbon Sequestration: Between Myth and Farmer Reality

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Improving food security, environmental preservation and enhancing livelihood should be the main targets of the innovators of today's farming systems. Conservation agriculture (CA), based on minimum tillage, crop residue retention, and crop rotations, has been proposed as an alternative system combining benefits for the farmer with advantages for the society. This paper reviews the potential impact of CA on C sequestration by synthesizing the knowledge of carbon and nitrogen cycling in agriculture; summarizing the influence of tillage, residue management, and crop rotation on soil organic carbon stocks; and compiling the existing case study information. To evaluate the C sequestration capacity of farming practices, their influence on emissions from farming activities should be considered together with their influence on soil C stocks. The largest contribution of CA to reducing emissions from farming activities is made by the reduction of tillage operations. The soil C case study results are not conclusive. In 7 of the 78 cases withheld, the soil C stock was lower in zero compared to conventional tillage, in 40 cases it was higher, and in 31 of the cases there was no significant difference. The mechanisms that govern the balance between increased or no sequestration after conversion to zero tillage are not clear, although some factors that play a role can be distinguished, e.g., root development and rhizodeposits, baseline soil C content, bulk density and porosity, climate, landscape position, and erosion/deposition history. Altering crop rotation can influence soil C stocks by changing quantity and quality of organic matter input. More research is needed, especially in the tropical areas where good quantitative information is lacking. However, even if C sequestration is questionable in some areas and cropping systems, CA remains an important technology that improves soil processes, controls soil erosion and reduces production cost.

Keywords C-credits, zero tillage, conservation tillage, C-cycling, food production

I. INTRODUCTION

A. Conservation Agriculture

Human efforts to produce ever greater amounts of food leave their mark on our environment. Soil degradation in all its forms is not a prelude to mass starvation, as analysts once feared. Nevertheless, it is eroding crop yields and contributing to malnourishment in many corners of the globe (*Science*, 11 June 2004, p. 1617). Despite the availability of improved varieties with increased yield potential, the potential increase in production is not attained because of poor crop system management (Reynolds and Tuberosa, 2008). Persistent use of conventional farming practices based on extensive tillage, and especially when combined with in situ burning of crop residues, have magnified soil erosion losses and the soil resource base has been steadily degraded (Montgomery, 2007). Another direct consequence of farmers' persistent use of traditional production practices is rapidly increasing production costs associated with the inefficient use of inputs whose costs continue to rise.

Farmers concerned about the environmental sustainability of their crop production systems combined with ever-increasing production costs have begun to adopt and adapt improved systems management practices that lead towards the ultimate vision of sustainable conservation agriculture solutions. The name conservation agriculture has been used to distinguish this more sustainable agriculture from the narrowly defined "conservation tillage" (Wall, 2006). Conservation tillage is a widely used term to characterize the development of new crop production technologies that are normally associated with some degree of tillage reductions, for both pre-plant as well as in-season mechanical weed control operations that may result in some level of crop residue retention on the soil surface. The definition of conservation tillage does not specify any particular optimum level of tillage, but it does stipulate that the residue coverage on the soil surface should be at least 30% (Jarecki and Lal, 2003). Conservation agriculture, however, removes the emphasis from the tillage component and addresses an enhanced concept of the complete agricultural system. It combines the following basic principles:

1. Reduction in tillage: The objective is the application of zero tillage or controlled tillage seeding systems that normally do not disturb more than 20–25% of the soil surface (including strip till or permanent raised bed planting systems, with only superficial reshaping in the furrows between the raised beds as needed before planting of each succeeding crop);
2. Retention of adequate levels of crop residues on the soil surface: The objective is the retention of sufficient residue on the soil surface to protect the soil from water/wind erosion, water run-off and evaporation to improve water productivity and to enhance soil physical, chemical, and biological properties associated with long-term sustainable productivity; and
3. Use of crop rotations: The objective is to employ economically viable, diversified crop rotations to help moderate/mitigate possible weed, disease, and pest problems.

These conservation agriculture principles seem to be applicable to a wide range of crop production systems from low-yielding, dry rain-fed to high-yielding irrigated conditions. Obviously, specific and compatible management components (weed control tactics, nutrient management strategies and appropriately-scaled implements) will need to be identified through adaptive research with active farmer involvement to facilitate farmer adoption of appropriate conservation agriculture-based technologies for contrasting agro-climatic/production systems. Applying conservation agriculture essentially means altering literally generations of traditional farming practices and implement use. In fact, the change in mind-set not only by farmers, but also by scientists, extension agents, private sector partners, and policy makers may be the most difficult aspect associated with the development, transfer, and farmer adoption of appropriate conservation agriculture technologies. As such, the movement towards conservation agriculture–based technologies normally comprises a sequence of stepwise changes in cropping system management to improve productivity and sustainability. The principles of marked tillage reductions are initially applied in combination with the retention of sufficient amounts of crop residue on the soil surface, with the assumption that appropriate crop rotations can be included or maintained to achieve an integrated, sustainable production system.

Conservation agriculture has been promoted as an agricultural practice that increases agricultural sustainability, concomitant with a potential to mitigating greenhouse gas emissions (Cole *et al.*, 1997; Paustian *et al.*, 1997; Schlesinger, 1999). There are, however, contrasting reports on the potential of conservation agriculture practices for C sequestration (e.g., Angers *et al.*, 1997; Blanco-Canqui and Lal, 2008). This paper summarizes the current existing knowledge of conservation agriculture and C sequestration and its underlying mechanisms in the frame of the constant striving towards sustainable and economical agricultural production systems for small-and large-scale farmers in the world.

B. Carbon and Nitrogen Cycling

1. *The Global Carbon and Nitrogen Cycle*

The global carbon cycle is constituted by a short-term biochemical cycle superimposed on a long-term geochemical cycle. Annually, anthropogenic activities distort both cycles by emitting 8.6 Pg C, which is absorbed by the atmosphere (3.3 Pg C), the oceans (2.2 Pg C), and other sinks. In the last 150 years, CO_2 emissions to the atmosphere have increased by 31% (circa 270 ± 30 Pg C). This has been mainly explained by C extraction from low turnover sinks (i.e., coal, gas, and oil sinks) (Lal, 2007). The soil C pool comprises two components: the soil organic carbon (SOC) pool and the soil inorganic carbon pool (SIC). Agricultural activities affect mainly the SOC pool, which constitutes a potential source of greenhouse gasses with an estimated current carbon content in the 1 m top layer 2.0 times greater than the atmospheric pool (Lal, 2007). In addition, soil C degradation leads to important soil quality losses and poses a threat for both agricultural production systems and food security (Lal, 2004). Ensuring net removal of carbon dioxide from the atmosphere into soils (soil carbon sequestration) favors the sustainability of farming systems (Reicosky, 2008).

The global C and N cycles are connected. The N on earth consists mainly of a pool of nitrogen gas in the atmosphere and a pool of N that cycles among biota and soils in the form of nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+) (Delwiche, 1970; Vitousek *et al.*, 1997). Human activity has doubled the N transfer from the atmosphere to biologically available pools (mainly through industrial N fixation) with associated increased emission, transport, reaction and deposition of trace nitrogen gases

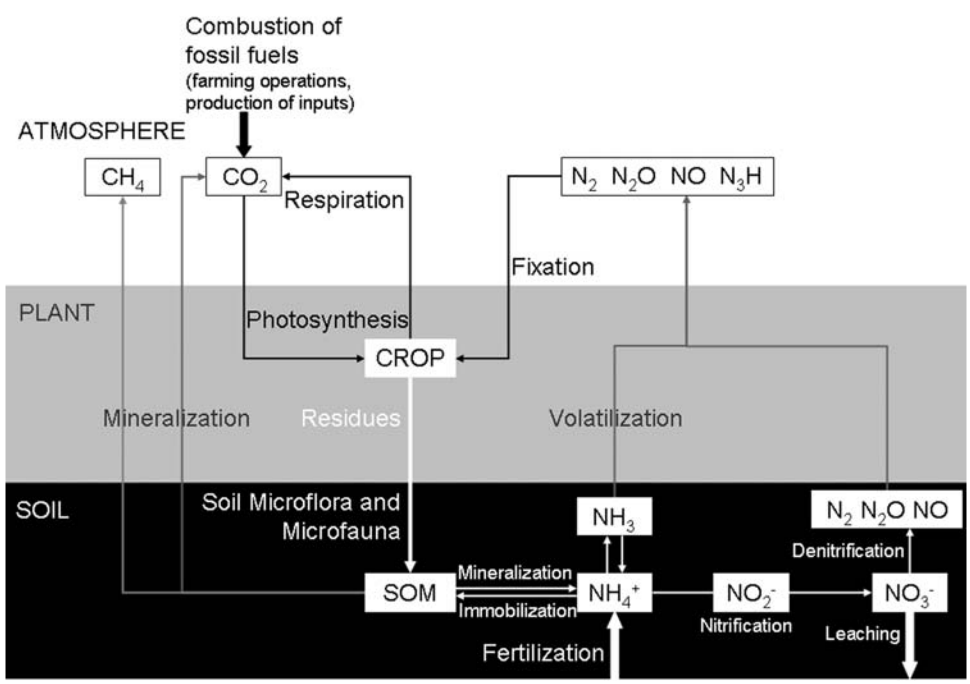

FIG. 1. Carbon and nitrogen cycle in agricultural ecosystems.

such as nitrous oxide (N_2O), nitric oxide (NO), and ammonia (NH_3) (Lokupitiya and Paustian, 2006).

Decomposition of soil organic matter releases CO_2 into the atmosphere and ammonium (NH_4^+) in the soil and, when not taken up by microorganisms, they are oxidized under aerobic conditions to nitrate (NO_3^-) (Fig. 1). During that process or nitrification, N_2O is formed, contributing to greenhouse gas emissions. When the oxygen status of a soil changes, nitrification is inhibited, and NO_3^- is reduced to nitrite (NO_2^-), nitric oxide (NO), nitrous oxide (N_2O), and dinitrogen (N_2). Production of NO is low under anaerobic conditions as it is immediately reduced, but the ratio of N_2O and N_2 produced is highly variable. The ratio of the nitrogen gases formed is largely controlled by pH, O_2, nitrate concentrations and the status of the reduction enzymes (Firestone *et al.*, 1980). In soil, nitrification and denitrification can occur at the same time and there are even microorganisms that can oxidize NH_4^+/NO_2^- and reduce NO_2^-/NO_3^- at the same time (Wrage *et al.*, 2001). Denitrification normally contributes more to emissions of N_2O than nitrification (Bateman and Baggs, 2005). The contribution to global warming of the most important biological greenhouse gases are respectively 70%, 23%, and 7% for carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (IPCC, 2001).

2. *Carbon and Nitrogen Cycling in Agricultural Systems*

Carbon uptake in crops occurs through photosynthesis and enters the soil as a residue of above- or below-ground biomass. The dead organic matter is colonized by a variety of soil organisms, which derive energy for growth from the oxidative decomposition of complex organic molecules. During decomposition, about half of the C is mineralized and released as CO_2 (White, 2006). Marland *et al.* (2003) distinguished four sources of CO_2 emissions in agricultural systems: (i) plant respiration; (ii) the oxidation of organic carbon in soils and crop residues; (iii) the use of fossil fuels in agricultural machinery such as tractors, harvesters, and irrigation equipment; and (iv) the use of fossil fuels in the production of agricultural inputs such as fertilizers and pesticides (Fig. 1). Soils can also be producers of CH_4, e.g., in wetlands or rice cultivation (Fig. 1).

The C and N cycles are linked through the reservoirs in crop and soil organic matter. Nitrogen can enter the soil from the atmosphere through dry and wet N deposition, fertilizers/manures and N fixation, while processes like ammonia (NH_3) volatilization and emissions of denitrification products (N_2, N_2O, NO) diminish the N content of the soil system. Mineral N in the soil can also be depleted through the uptake of nitrogen by the crop, whereas the return to the soil of the nonharvested crop will add to the organic nitrogen pool (Vlek *et al.*, 1981). The processes of immobilization and mineralization are continuously causing changes in the mineral N reserves of the soil. Organically bound N can be converted microbiologically into inorganic mineral forms (mineralization), leading first to formation of ammonium (NH_4^+) and possibly ending up in NO_3^- (nitrification) (Van Cleemput and Boeckx, 2002). Where excessive wetting prevails, mineral nitrogen (particularly NO_3^-) may leach beyond the reach of crop roots (Fig. 1).

Nitrogen dynamics in agricultural systems are very much influenced by the large quantities added as nitrogen fertilizers. Since N supply to soils increases productivity and biomass accumulation in the short-term increased nitrogen input levels have been perceived as a strategy to favor soil C sequestration (Batjes, 1996). However, N application as fertilizers implies CO_2 emission costs. For example, 1 kg of N fertilizer leads to the emission of 0.86–1.3 kg of CO_2 in production, packaging, transport and application (Lal, 2004). Additionally, increases in soil organic matter might accelerate N dynamics and thus emissions of N_2O, a known greenhouse gas (Butterbach-Bahl *et al.*, 2004). In summary, nitrogen affects the net greenhouse gas balance in four ways: (i) CO_2 is released from the energy and fossil fuel intensive production of nitrogen fertilizer; (ii) crop yield changes as a function of the nitrogen application rate; (iii) the application rate and CO_2 emissions associated with the energy-intensive production of agricultural lime depend on the rate of nitrogen fertilization, since increased use of N fertilizer can cause a decline in soil PH; and (iv) N_2O emissions vary with tillage practice and as a function of the nitrogen application rate (Marland *et al.*, 2003).

Carbon levels in soil are determined by the balance of inputs, as crop residues and organic amendments, and C losses through organic matter decomposition (Paustian *et al.*, 1997). Upon cultivation of previously untilled soils, this balance is disrupted and generally 20% to 40% of the soil C is lost, most of it within the first few years following initial cultivation (Davidson and Ackerman, 1993; Murty *et al.*, 2002). Afterwards, the rate of decrease levels off, and some decades later a new, management-dependent soil humus level is attained (Sauerbeck *et al.*, 2001; Fig. 2).

Management to build up SOC requires increasing the C input, decreasing decomposition, or both (Paustian *et al.*, 1997). Decomposition may be slowed by altering tillage practices or including crops with slowly decomposing residue in the rotation. The C input may be increased by intensifying crop rotations, including perennial forages and reducing bare fallow, by reducing tillage and retaining crop residues, and by optimizing agronomic inputs such as fertilizer, irrigation, pesticides, and liming. Following an improvement in agricultural management practices, soil organic carbon will gradually approach a new steady state that depends on the new suite of practices (Marland *et al.*, 2003, Fig. 2). Estimates of the time necessary to reach the new steady state range from 20–40 years (West and Marland, 2002) to 50–100 years (Sauerbeck, 2001). It is important to remember that the use of agricultural inputs such as fertilizer, irrigation, pesticides and liming carry a 'hidden' carbon cost, so any effort to estimate the effect of changing tillage practice on the net flux of CO_2 to the atmosphere should consider both the C sequestered in soil and the emissions from fossil-fuel use in the affected system (West and Marland, 2002).

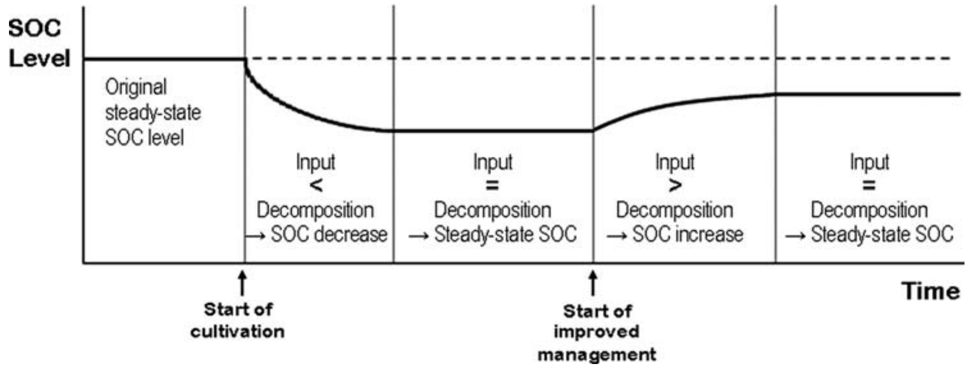

FIG. 2. Long-term soil organic carbon level changes depending on carbon input and decomposition in agricultural ecosystems.

II. MANAGING SOIL CARBON: CONVENTIONAL VERSUS CONSERVATION AGRICULTURE

A. Microbial Carbon Decomposition and Immobilization

The increase in soil organic carbon in cropping systems depends on the input and characteristics of organic material added to the soil and its decomposition by microorganisms. Organic matter is the principle C substrate for soil microorganisms. Upon mineralization, some of the C in the organic material is used for microbial growth and maintenance, while the rest is respired as CO_2 and returns to the atmosphere (Lavelle *et al.*, 1993). Decomposition is thus essentially a biological process. Factors regulating decomposition have been divided along a hierarchical categorization with four levels: (i) climatic factors (determining soil temperature and moisture regimes); (ii) physical properties of soils; (iii) chemical constraints relating to soil biota resources; and, (iv) biological regulation through interactions between macro- and microorganisms (Lavelle *et al.*, 1993). Research on the distribution of carbon decomposition within the framework of the soil pore system suggests that plant material decomposed most rapidly in soils with a relatively large volume of pores with neck diameters c. 15–60 μm (Strong *et al.*, 2004). This might be due to particularly rapid rates of decomposition near the gas–water interface because conditions favor organisms' motility, nutrient or toxin diffusion, and oxygen supply. Organic matter in large air-filled pores might decompose more slowly than in the intermediate-sized pores due to decreased organism motility, decreased diffusion of solutes, and decreased intimacy of contact between organic matter and soil minerals, whereas carbon that enters the smallest pores is protected against biodegradation (Strong *et al.*, 2004).

As decomposition proceeds, resource quality changes: the substrates which assimilate readily are rapidly metabolized whereas resistant compounds (e.g., tannin-protein complexes, lignin substances and resistant humic compounds) tend to accumulate (Lavelle *et al.*, 1993). Nutrient element deficiency at any stage of decomposition may limit microbial activity and thereby block nutrient release (which occurs when the C: nutrient ratio of the decomposing resource is too high compared to that of the microorganisms) (Swift *et al.*, 1979; Lavelle *et al.*, 1993). The lacking nutrient is most often N. The remaining organic matter after mineralization constitutes the 'passive' SOM pool or biochemically protected fraction (Parton *et al.*, 1987).

B. Soil Aggregation: Boundaries for Decomposition

Soils are not physically uniform. Carbon immobilization is strongly related to micro-site availability in relation to soil microorganisms. Due to low microbial mobility, the role of these organisms is greatly influenced by the compartmentalization of substrate and microbial biomass by macro- and microaggregates and thus the physical structure has a direct impact on the C decomposition/immobilization processes. Aggregation is a dynamic process that depends on various agents such as soil fauna, roots, inorganic binding agents and environmental variables (Six *et al.*, 2004).

1. *Macroaggregates and Microaggregates-within-Macroaggregates*

Macroaggregates are gradually bound together by temporary (i.e., fungal hyphae and roots) and transient binding agents (i.e., microbial- and plant-derived polysaccharides) as the decomposition of soil organic matter takes place (Six *et al.*, 2004). Temporary binding agents gradually decompose into fragments (particulate organic matter or POM) which, coated with bacterial and fungi mucilages, become encrusted with clays resulting in the inception of microaggregates-within-macroaggregates (Oades, 1984). This has been reflected in labeled C studies where initial C accumulation in soil macroaggregates as coarse ($> 250\mu$m) intra-aggregate particulate organic carbon (iPOM C) is redistributed into microaggregates as fine (53–250 μm) iPOM C (Angers *et al.*, 1997; Six *et al.*, 2000a) while macroaggregates break down. It is occluded iPOM C in soil microaggregates which constitute the main mechanism for long-term soil C sequestration in agricultural soils (Six *et al.*, 2004). Microaggregates-within-macroaggregates constitute relatively stable and secluded habitats for microorganisms, when compared to microaggregate outer surfaces or macroaggregates as a whole (Mummey *et al.*, 2004). Low levels of microbial activity are explained by inaccessibility due to pore size exclusion and related to water-filled porosity (Killham *et al.*, 1993) as well as reduced oxygen diffusion into these fractions (Sexstone *et al.*, 1995; Sollins *et al.*, 1996).

Macroaggregate turnover under low level disturbance conditions in conservation agriculture is slow enough to allow fine iPOM C to be predominantly stabilized in free and intra-microaggregates (Six *et al.*, 1999a; Six *et al.*, 2000b), while conventional systems alter this process. Tillage disrupts macroaggregates exposing coarse iPOM C to microbial attack (Beare *et al.*, 1994) and preventing its incorporation into microaggregates as fine iPOM C (Six *et al.*, 2000a). Increased macroaggregate turnover leads to a loss of C-rich macroaggregates and a decrease of microaggregate formation (which would be mainly constituted by C-depleted microaggregates) (Elliott, 1986; Six *et al.*, 2000a; Six *et al.*, 2000b). In fact, free iPOM C contents (i.e., not protected by macroaggregates) are mostly explained by input rates, and differences are generally not significant between conventional and conservation agriculture systems (Six *et al.*, 1999; Denef *et al.*, 2007). SOC differences found between these two systems are mostly explained by iPOM C protected in microaggregates due to drastic differences caused by disturbance regimes. Denef *et al.* (2004) found that the microaggregate-associated C fraction accounted for up to 90% of the total SOC differences in widely varying soil types (Mollisol, Alfisol, Oxisol) and environments (i.e., in the United States and Brazil). The observed difference in C in the microaggregates-within-macroaggregates of the soils of the different management practices are not necesarily accompanied by a difference in the proportion of water-stable microaggregates-within-macroaggregates (Lichter *et al.*, 2008; Six *et al.*, 2002b; Del Galdo *et al.*, 2003; Denef *et al.*, 2004. Denef *et al.*, 2007). The original conceptual idea of a tight relationship between the amount of microaggregates-within-macroaggregates and C stabilization within the microaggregates-within-macroaggregates as influenced by management (Six *et al.*, 2000a) is not applicable for all soil types and/or management systems (Denef *et al.*, 2007). Enhanced C stabilization within the microaggregate-within-macroaggregate fraction is related to the dynamic 'behavior' rather than the 'amount' of the microaggregates (and the macroaggregates that protect them). In other words, similar to the importance of both amount and turnover of macroaggregates for C sequestration upon reduced physical disturbance (Six *et al.*, 1999), the differences in C concentration within the fraction of microaggregates occluded in the macroaggregates among management systems can be linked to the differences in amount and stability, as well as the turnover of the microaggregates-within-macroaggregates (Denef *et al.*, 2007). The slower turnover of microaggregates-within-macroaggregates in zero tillage allows greater protection of coarse POM and greater stabilization of mineral-bound C decomposition products in the microaggregates-within-macroaggregates (Denef *et al.*, 2007).

2. *The Influence of Soil Macrofauna on Aggregation*

Soil macrofaunal activity has been found to contribute to both macro- and microaggregate formation (Pulleman *et al.*, 2005). While litter transformers (mesofauna, large arthropods, and part of the soil macrofauna) have a limited impact on soil structure, ecosystem engineers (earthworms, ants, and termites) ingest a mixture of organic matter and mineral soil favoring residue incorporation into the soil and thus contribute to aggregation levels (Edwards, 1998). Mutualistic relationships with the microflora within their gut enable these organisms to undertake partial decomposition of complex substrates (Kladivko, 2001). In the case of earthworms, casting promotes the creation of stable organic-mineral complexes with reduced decomposition rates (although it is characterized by enhanced mineralization during the first few hours to days) and favors soil stability if allowed to dry or age (Shipitalo and Protz, 1988; Marinissen and Dexter, 1990). Soil macrofauna also plays an important role in microaggregate formation. During gut transit, organic materials are intimately mixed and become encrusted with mucus to create nuclei for microaggregate inception (Shipitalo and Protz, 1988; Lobry de Bryun and Conacher, 1990; Barois *et al.*, 1993; Holt and Lepage, 2000; Bignell and Holt, 2002; Six *et al.*, 2004). However, in conventional systems, there is a direct impact (physical abrasion by tillage and absence of residue cover) and indirect impact (habitat destruction by tillage) on soil macrofauna leading to absent or very low populations. This explains why soil macrofauna populations are generally greater under conservation agriculture when compared to conventional systems (Folgarait, 1998; Chan, 2001; Kladivko, 2001). Generally, responses to tillage are scale-dependent and result in the dominance of soil biota with high reproduction rates and fast colonization capacities (*r* strategists such as mesofauna vs. macrofauna or bacteria vs. fungi) and thus increased mineralization versus humification (Wardle, 1995), as well as the predominance of physicogenic soil aggregates with low stability (Six *et al.*, 2004). The greater biological complexity under conservation agriculture implies that macrofauna partly regulates decomposition by microbial biomass, and favors biogenic aggregate formations, new C inputs are partly protected and stored in soils.

C. The Importance of Full Carbon Cycle Analysis

C sequestration in soil, C storage in crop residues and CO_2 emissions from farming activities should be considered together (Wang and Dalal, 2006, Fig. 1) to evaluate the atmospheric CO_2 mitigation capacity of different farming practices. To include farming activities, estimates must be made of energy use and C emissions for primary fuels, electricity, fertilizers, lime, pesticides, irrigation, seed production, and farm machinery (West and Marland, 2002).

The largest contribution of conservation agriculture to reducing the CO_2 emissions associated with farming activities is made by the reduction of tillage operations. Erenstein and Laxmi (2008) compared studies in rice–wheat systems in the Indo-

Gangetic Plains and found seasonal savings in diesel for land preparation with zero tillage to be in the range of 15–60 l ha^{-1}, with an average of 36 l ha^{-1} or 81% saving across the studies, equivalent to a reduction in CO_2 emission of 93 kg CO_2 ha^{-1} per year. Similar values were obtained in a wheat–fallow system in semi-arid subtropical Queensland, Australia, where practicing zero tillage reduced fossil fuel emissions from machinery operation by 2.2 Mg CO_2 ha^{-1} over 33 years or 67 kg CO_2 ha^{-1} per year (four to five tillage operations with a chisel plough to 10 cm during fallow each year were replaced by one herbicide spray) (Wang and Dalal, 2006). West and Marland (2002) reported estimates for C emissions from agricultural machinery, averaged over corn, soybean, and wheat crops in the United States of 69.0, 42.2, and 23.3 kg C ha^{-1} per year for conventional tillage, reduced tillage, and zero tillage respectively. While enhanced C sequestration will continue for a finite time, the reduction in net CO_2 flux to the atmosphere, caused by the reduced fossil-fuel use, can continue indefinitely, as long as the alternative practice is continued and could more than offset the amount of C sequestered in the soil in the long term (West and Marland, 2002).

Conservation agriculture can also reduce CO_2 emissions by saving irrigation water. Irrigation contributes to CO_2 emissions because energy is used to pump irrigation water and, when dissolved, calcium (Ca) precipitates in the soil, forming $CaCO_3$ and releasing CO_2 to the atmosphere (Schlesinger, 2000). In rice–wheat systems in the Indo-Gangetic Plains, zero tillage is reported to save irrigation water in the range of 20–35% in the wheat crop compared to conventional tillage, reducing water usage by approximately 1 million l ha^{-1} (Mehla *et al.*, 2000; Gupta *et al.*, 2002; Hobbs and Gupta, 2003). The savings arise because with zero tillage wheat can be sown just after the rice harvest, making use of the residual moisture for wheat germination, potentially saving a pre-sowing irrigation, and because irrigation water advances faster in untilled soil than in tilled soil (Erenstein and Laxmi, 2008). Harman *et al.* (1998) report the elimination of the pre-sowing irrigation in a zero tillage system, resulting in water savings of 25% compared to conventional tillage systems for corn and sorghum in the Texas High Plains.

Herbicide use increases in U.S. corn production systems when switching from conventional tillage with the moldboard plow to zero tillage (Lin *et al.*, 1995), but in the full C cycle analysis for U.S. farming systems, the increase in herbicide use was offset by far by the reduction in fossil fuel for tillage operations (West and Marland, 2002). Based on U.S. average crop inputs, zero tillage emitted less CO_2 from agricultural operations than did conventional tillage, with 137 and 168 kg C ha^{-1} per year respectively, including the C emissions associated with the manufacture, transportation, and application of fertilizers, agricultural lime, and seeds (West and Marland, 2002).

D. The Influence on Soil Organic Carbon Stocks

Soil organic carbon (SOC) stocks can be measured directly with soil samples or can be inferred via soil CO_2 emissions. When measuring SOC in soil samples, several factors have to be taken into account. Bulk density can be affected by tillage practice. Several authors report higher bulk density values for zero-tillage than for conventional tillage in the top soil (Gál *et al.*, 2007; Thomas *et al.*, 2007; Yang and Wander, 1999). If bulk density increases after conversion from conventional tillage to zero tillage, there will be a relative drop in the soil surface in zero tillage. Thus, if samples are taken to the same depth within the plow layer, more mass of soil will be taken from the zero tillage soil. This could increase the mass of SOC in the zero tillage and could widen the difference between the two systems if there is significant SOC beneath the maximum depth of sampling (VandenBygaart and Angers, 2006). Ellert and Bettany (1995) suggested basing calculations of SOC stocks on an equivalent soil mass rather than on genetic horizons or fixed sampling depths in order to account for differences in bulk density. Consideration of equivalent mass in calculating SOC storage is, however, only critical when the entire topsoil is not sampled and there are significant amounts of SOC situated beneath the lowest sampling depth. If standard depth measurements are made in which the entire topsoil is contained within the profile sampled, and there is little SOC below, any changes in bulk density affecting SOC storage calculations are accounted for (VandenBygaart and Angers, 2006).

Tillage practice can also influence the distribution of SOC in the profile with higher soil organic matter (SOM) content in surface layers with zero tillage than with conventional tillage, but a higher content of SOC in the deeper layers where residue is incorporated through tillage (Jantalia *et al.*, 2007; Gál *et al.*, 2007; Thomas *et al.*, 2007; Dolan *et al.*, 2006; Yang and Wander, 1999; Angers *et al.*, 1997). Consequently, rates of SOC storage under zero tillage compared with conventional tillage can be overstated if the entire plow depth is not considered (VandenBygaart and Angers, 2006).

Baker *et al.* (2007) state that not just the entire plow depth, but the entire soil profile should be sampled in order to account for possible differences in root distribution and rhizodeposition between management practices. This was confirmed in the study of Gál *et al.* (2007) determining SOC stocks in a silty clay loam in Indiana. Calculations based on SOC content increases solely from the upper 30 cm in zero tillage would have suggested that zero tillage resulted in 23 t ha^{-1} more SOC than with conventional tillage (moldboard plowing to 20–25 cm). However, calculations based on SOC to the full 1.0 m depth suggested that zero tillage resulted in an overall gain of just 10 t ha^{-1}. However, Doran *et al.* (1998) found, more than 20 years after the start of cultivation on Kastanozems in Nebraska, that measurement of SOC losses to a depth of 122 cm in different tillage systems closely approximated losses measured to 30 cm depth, indicating that under the cropping, tillage, and climatic conditions of that study, soil C changes were adequately monitored by sampling to a depth of 30 cm.

Changes in soil C can, in principle, be inferred from continuous measurement of net ecosystem CO_2 exchange between the land surface and the atmosphere provided other C additions or losses (e.g., harvested grain) are properly credited (Baker *et al.*, 2007). Measurements of CO_2 emissions have been confined mainly to the period after tillage events. Reicosky (1997) reported that short-term (55 hours) fluxes after tillage strongly differ between moldboard plowed soils (2–29 g CO_2 m^{-2} h^{-1}) and conservation agriculture treatments (0.7 g CO_2 m^{-2} h^{-1}). Differences strongly decreased in 55 hours but still implied a greater cumulative emission in the short-term (19 days) in conventional systems (0.9 kg CO_2 m^{-2} during 19 days) as compared to conservation agriculture (0.2 kg CO_2 m^{-2}). Chatskikh and Olesen (2007) recorded a 25% decrease in emissions in conservation agriculture during a period of 91 days when compared to conventionally managed plots. However, at seasonal level, Hendrix *et al.* (1988) reported that carbon dioxide fluxes were slightly greater under conservation agriculture. From a long-term experiment in northern France, Oorts *et al.* (2007) reported significantly larger cumulative emissions for zero tillage (4064 ± 138 kg CO_2-C ha^{-1} over a 331-day period) than for conventional tillage (3160 ± 269 kg CO_2-C ha^{-1} over 331 days). Nouchi and Yonemura (2008) did not find significant differences in the annual soil respiration amount between conventional and zero tillage in soil in Japan. Different decomposition rates are explained by a large variation in microbial biomass abundance and composition, moisture and temperature fluctuations, as well as quality and quantity of organic C substrates. Further research is needed to determine how conservation agriculture influences the net ecosystem CO_2 exchange on the long term.

In order to understand better the influence of the different components comprising conservation agriculture (reduced tillage, crop residue retention and crop rotation) on SOC stocks, an extensive literature search has been done. It is summarized in Tables 1 and 2 and discussed below. Some of the already existing reviews on the influence of agriculture and management on C sequestration made by West and Post (2002), Jarecki and Lal (2003), VandenBygaart *et al.* (2003) and Blanco-Canqui and Lal (2008) were used as a basis and completed through further literature search. It should be noted that based on the above-mentioned constraints, we only considered those results that came from measurements done to at least 30 cm deep after at least 5 years of continuous practice.

In general, it is striking that, for the developing world and the more tropical and subtropical areas, information is lacking on the influence of tillage, residue management and crop rotation on C storage. We found limited information on the influence of management changes on C stocks in the depleted soils of Africa, Central America, etc. Some information exists, but most of it was rejected for inclusion in the tables as it did not meet our criteria of reliability (depth of sampling, etc.). It is, of course, clear that most information is coming from areas where reduced tillage and conservation agriculture have been implemented already for a substantial time; however, in order to really quantify and understand underlying processes for C sequestration, serious knowledge gaps have to be bridged.

1. *The Influence of Tillage Practice on Soil Organic Carbon Stocks*

West and Post (2002) concluded from a global database of 67 long-term experiments that SOC levels under zero tillage were significantly different from SOC levels under conventional and reduced tillage, while SOC levels under conventional and reduced tillage were not significantly different from each other. On the contrary, Alvarez (2005) found no differences in SOC between reduced (chisel, disc, and sweep tillage) and zero tillage, whereas conventional tillage (moldboard plow, disc plow) was associated with less SOC in his compilation of data from 161 sites with contrasting tillage systems (at least whole tillage depth sampled). Consequently the influence of reduced tillage on SOC stocks still seems undecided (Table 1). West and Post (2002) concluded that a move from conventional tillage systems to zero tillage (both with residue retention) can sequester on average 48 ± 13 g C m^{-2} yr^{-1}. Alvarez (2005) found that the accumulation of SOC under reduced and zero tillage was an S-shape time-dependent process, which reached a steady state after 25–30 years. Averaging out SOC differences in all of the 161 experiments under reduced and zero tillage, there was an increase of 2.1 t C ha^{-1} over plowing. However, when only those cases that had apparently reached equilibrium were included (all zero tillage vs. conventional tillage comparisons from temperate regions), mean SOC increased by approximately 12 t C ha^{-1}.

Results are not always pointing in the same direction. West and Post (2002), for example, found that moving to zero tillage in wheat–fallow rotations showed no significant increase in SOC and, therefore, may not be a recommended practice for sequestering C. Conversely, Alvarez (2005) reported in his compilation study that soils from wheat–fallow ($n = 13$) under reduced and zero tillage had a mean SOC content that was 2.6 t C ha^{-1} higher than under conventional tillage, an increase similar to that for the other rotations. There is no consensus between the studies in wheat–fallow systems reported in Table 1 about the effect of a conversion to zero tillage on SOC stocks. Doran *et al.* (1998) report a positive effect of zero tillage on SOC stocks, whereas Halvorson *et al.* (2002) and Thomas *et al.* (2007) did not find a significant difference between zero and conventional tillage, and Black and Tanaka (1997) even reported a negative effect from a conversion to zero tillage (Table 1). In other crop rotations, some research indicates no difference or reduced carbon sequestration in zero tillage (Table 1). Dolan *et al.* (2006) reported that the summation of soil SOC over depth to 50 cm did not vary among tillage treatments. Yang and Wander (1999) found no overall (0–90 cm) increase in SOC storage in zero tillage compared to moldboard plowing in Illinois. Angers *et al.* (1997) evaluated SOC storage in the cool, humid climates of eastern Canada, and concluded that there tended to be a lower SOC at or below

TABLE 1
The Influence of Tillage Practice on SOC stocks

Location	Textural Class	Crop	Duration (year)	Treatment	Depth (cm)	ΔSOC	References
Elora, Ontario, Canada	SiL	Various	20	ZT vs. CT	40	+	Yang and Kay (2001a)
Clinton, Ontario, Canada	SL	C-W-S	19	ZT vs. CT	30	+	Yang and Kay (2001b)
Clinton, Ontario, Canada	LS	C-W-S	19	ZT vs. CT	30	NS	Yang and Kay (2001b)
Clinton, Ontario, Canada	CL	C-W-S	19	ZT vs. CT	30	NS	Yang and Kay (2001b)
Ottawa, Ontario, Canada	SL	C	5	ZT vs CT	60	NS	Angers *et al.* (1997)
Ottawa, Ontario, Canada	SL	W	5	ZT vs CT	60	NS	Angers *et al.* (1997)
Harrow, Ontario, Canada	CL	C	11	ZT vs CT	60	NS	Angers *et al.* (1997)
Ontario, Canada	SiL	C	29	ZT vs CT	50	NS	Wanniarachchi *et al.* (1999)
Harrington, Québec, Canada	SL	W-B; B-S	8	ZT vs CT	60	NS	Angers *et al.* (1997)
La Pocatiere, Québec, Canada	C	B	6	ZT vs CT	60	NS	Angers *et al.* (1997)
Prince Edward Island, Canada	SL	S-B	16	ZT vs CT	60	NS	Carter (2005)
Mandan, North Dakota, USA	SiL	W-F	12	ZT vs CT	30.4	NS	Halvorson *et al.* (2002)
Mandan, North Dakota, USA	SiL	W-W-Sf	12	ZT vs CT	30.4	+	Halvorson *et al.* (2002)
Fargo, North Dakota, USA	C	S-B-Sf-B	10	ZT vs. CT	60	+	Deibert and Utter (1989)
Fargo, North Dakota, USA	C	S-B-Sf-B	10	ZT vs. RT	60	NS	Deibert and Utter (1989)
Fargo, North Dakota, USA	C	S-B-Sf-B	10	RT vs. CT	60	+	Deibert and Utter (1989)
Mandan, North Dakota, USA	SiL	W-F	7	ZT vs. CT	30	–	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	W-F	7	ZT vs. RT	30	–	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	W-F	7	RT vs. CT	30	–	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	W-W-Sf	7	ZT vs. CT	30	+	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	W-W-Sf	7	ZT vs. RT	30	+	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	W-W-Sf	7	RT vs. CT	30	+	Black and Tanaka (1997)
Waseca, Minnesota, USA	NA	C	6	ZT vs. CT	30	–	Mielke *et al.* (1986)
Waseca, Minnesota, USA	CL	C	14	ZT vs CT	45	+	Huggins *et al.* (2007)
Waseca, Minnesota, USA	CL	S	14	ZT vs CT	45	NS	Huggins *et al.* (2007)
Waseca, Minnesota, USA	CL	C-S	14	ZT vs CT	45	NS	Huggins *et al.* (2007)
Rosemount, Minnesota, USA	SiL	C-S	23	ZT vs CT	45	NS	Dolan *et al.* (2006)
Pendleton, Oregon, USA	SiL	W-F	44	RT vs. CT	30	+	Rasmussen and Smiley (1997), Rasmussen and Rohde (1988)
Sidney, Nebraska, USA	SiL	W-F	11	ZT vs. CT	30	+	Doran *et al.* (1998)
Sidney, Nebraska, USA	SiL	W-F	11	RT vs. CT	30	NS	Doran *et al.* (1998)
Sidney, Nebraska, USA	SiL	W-F	11	ZT vs. RT	30	+	Doran *et al.* (1998)
Manhatten, Kansas, USA	Si	S	11	ZT vs. CT	30	+	Havlin *et al.* (1990), Havlin and Kissel (1997)
Manhatten, Kansas, USA	Si	Sm-S	11	ZT vs. CT	30	+	Havlin *et al.* (1990), Havlin and Kissel (1997)
Manhatten, Kansas, USA	Si	Sm	11	ZT vs. CT	30	+	Havlin *et al.* (1990), Havlin and Kissel (1997)

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TABLE 1
The Influence of Tillage Practice on SOC stocks *(Continued)*

Location	Textural Class	Crop	Duration (year)	Treatment	Depth (cm)	ΔSOC	References
Manhatten, Kansas, USA	SiC	S	12	ZT vs. CT	30	–	Havlin *et al.* (1990)
Manhatten, Kansas, USA	SiC	Sm-S	12	ZT vs. CT	30	+	Havlin *et al.* (1990)
Manhatten, Kansas, USA	SiC	Sm	12	ZT vs. CT	30	+	Havlin *et al.* (1990)
Elwood, Illinois, USA	NA	C-S	6	ZT vs. CT	30	+	Mielke *et al.* (1986)
Urbana, Illinois, USA	SiL	C-S	9	ZT vs. CT	30	+	Yang and Wander (1999)
Urbana, Illinois, USA	SiL	C-S	9	RT vs. CT	30	NS	Yang and Wander (1999)
Urbana, Illinois, USA	SiL	C-S	9	ZT vs. RT	30	NS	Yang and Wander (1999)
DeKalb, north Illinois, USA	SiCL	C-S	17	ZT vs CT	30	NS	Yoo *et al.* (2006)
Monmouth, middle Illinois, USA	SiL	C-S	17	ZT vs CT	30	+	Yoo *et al.* (2006)
Perry, south Illinois, USA	SiL	C-S	17	ZT vs CT	30	+	Yoo *et al.* (2006)
West Lafayette, Indiana, USA	SiCL	C-C	28	ZT vs CT	100	+	Gál *et al.* (2007)
West Lafayette, Indiana, USA	SiCL	S-C	28	ZT vs CT	100	+	Gál *et al.* (2007)
South Charleston, Ohio, USA	LS	C	18	ZT vs. CT	30	+	Hunt *et al.* (1996)
South Charleston, Ohio, USA	LS	C	18	ZT vs. RT	30	+	Hunt *et al.* (1996)
South Charleston, Ohio, USA	LS	C	18	RT vs. CT	30	–	Hunt *et al.* (1996)
Wooster, Ohio, USA	SiL	C	19	ZT vs. CT	30	+	Dick *et al.* (1997)
Wooster, Ohio, USA	SiL	C-S	19	ZT vs. CT	30	+	Dick *et al.* (1997)
Wooster, Ohio, USA	SiL	C-S	19	ZT vs. RT	30	+	Dick *et al.* (1997)
Wooster, Ohio, USA	SiL	C-S	19	RT vs. CT	30	+	Dick *et al.* (1997)
Wooster, Ohio, USA	SiL	C-O-G	19	ZT vs. CT	30	+	Dick *et al.* (1997)
Hoytville, Ohio, USA	SiCL	C	18	ZT vs. CT	30	+	Dick (1983)
Hoytville, Ohio, USA	SiCL	C-S	18	ZT vs. CT	30	+	Dick (1983)
Hoytville, Ohio, USA	SiCL	C-O-G	18	ZT vs. CT	30	+	Dick (1983)
Fremont,Ohio, USA	SiCL	C-S	15	ZT vs CT	60	–	Blanco-Canqui and Lal (2008)
Jackson, Ohio, USA	SiL	C-S-A; C	12	ZT vs CT	60	–	Blanco-Canqui and Lal (2008)
Canal Fulton, Ohio, USA	L	C-S	30	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
Grove City, Pennsylvania, USA	SiL	C-S	10	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
Greenville, Pennsylvania, USA	SiL	C-S	8	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
Troy, Pennsylvania, USA	L	C	20	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
Lewisburg, Pennsylvania, USA	SiL	C-S	5	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
Lexington, Kentucky, USA	NA	C	20	ZT vs. CT	30	+	Blevins *et al.* (1983), Ismail *et al.* (1994)

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Georgetown, Kentucky, USA	SiL	SC-S-Pu; C-S-Veg	8	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
Glasgow, Kentucky, USA	SiL	C-S; C-S-T	10	ZT vs CT	60	NS	Blanco-Canqui and Lal (2008)
McKee, Kentucky, USA	SiL	C; T; W-R	15	ZT vs CT	60	–	Blanco-Canqui and Lal (2008)
Celaya, Mexico	C	W/C	5	ZT vs CT	30	+	Follett *et al.* (2005)
Celaya, Mexico	C	W/Bn	5	ZT vs CT	30	NS	Follett *et al.* (2005)
El Dorado do Sul, Brazil	SCL	O-C	9	ZT vs. CT	30	NS	Bayer *et al.* (2000)
El Dorado do Sul, Brazil	SCL	O/V-C/Cp	9	ZT vs. CT	30	NS	Bayer *et al.* (2000)
Londrina, Brazil	C	S-W; S-Cn	21	ZT vs CT	40	NS	Machado *et al.* (2003)
Ponta Grossa, Brazil	C	W-S; O-S;O-C	22	ZT vs CT	40	+	Sa *et al.* (2001)
Southwestern Paraná State, Brazil	C	C-Cc; S-Cc	19	ZT vs CT	40	NS	Calegari *et al.* (2008)
Rio Grande do Sul, Brazil	C	W/S	13	ZT vs CT	100	NS	Sisti *et al.* (2004)
Rio Grande do Sul, Brazil	C	W/S-V/C	13	ZT vs CT	100	+	Sisti *et al.* (2004)
Rio Grande do Sul, Brazil	C	W/S-O/S-V/C	13	ZT vs CT	100	+	Sisti *et al.* (2004)
Edinburgh, Scotland	CL/L	B	24	ZT vs. CT	30	+	Soane and Ball (1998), Ball *et al.* (1997)
Tänikon, Switserland	SL	W-C-W-Ca	19	ZT vs CT	40	NS	Hermle *et al.* (2008)
Central Spain	NA	B-V; B-Sf; B	11	ZT vs CT	30	+	Lopez-Fando and Pardo (2001)
Southern Queensland, Australia	CL	W/F	9	ZT vs CT	30	NS	Thomas *et al.* (2007)

Textural class: LS, Loamy sand; SiL, Silt loam; Si, Silt; L, Loam; SCL, Sandy clay loam; SiCL, Silty clay loam; CL, Clay loam; SiC, Silty Clay; C, Clay. **NA**, not available. **Crop**: B, barley (*Hordeum vulgare* L.); Bn, bean (*Phaseolus vulgaris* L.); C, corn or maize (*Zea mays* L.); Ca, canola (*Brassica* napus); Cc, cover crop; Cn, cotton (*Gossypium hirsutum* L.); F, fallow; G, grass; Leg, Legume; O, oats (*Avena* spp.); P, potato (*Solanum tuberosum* L.); Pu, Pumpkin (*Cucurbita maxima* Duch); R, Rye; S, soybean (*Glycine max* L. Merr.); Sb, Sugar beet (*Beta vulgaris* L.); SC, Sweet Corn; Sf, sunflower (*Helianthus annuus* L.); Sm, sorghum (*Sorghum bicolor* L.); T, Tobacco; V, vetch (*Vicia sativa* L.); Veg, vegetables; W, wheat (*Triticum aestivum* L.). Crops separated by '/' are planted in the same year. **Duration**: Duration of experiment (year). **Treatment**: ZT, zero tillage; RT, reduced tillage; CT, conventional tillage. **Depth**: Sampling depth (cm). Δ**SOC**: Difference in SOC stocks between treatments: +, positive; -, negative; NS, Not significant.

TABLE 2
The Influence of Crop Rotation on SOC Stocks

Location	Textural Class	Tillage	Duration (year)	Treatment	Depth (cm)	ΔSOC	References
Bow Island, Alberta, Canada	CL	CT	6	W vs.F-W	30	+	Bremer *et al.* (2002)
Bow Island, Alberta, Canada	CL	CT	6	W vs.F-W-W	30	NS	Bremer *et al.* (2002)
Bow Island, Alberta, Canada	CL	CT	6	F-W vs. F-W-W	30	NS	Bremer *et al.* (2002)
Lethbridge, Alberta, Canada	CL	CT	41	W vs.F-W	30	+	Bremer *et al.* (1994)
Lethbridge, Alberta, Canada	CL	CT	41	W vs.F-W	30	+	Janzen (1987), Janzen *et al.* (1997)
Lethbridge, Alberta, Canada	CL	CT	41	W vs.F-W-W	30	+	Janzen (1987), Janzen *et al.* (1997)
Lethbridge, Alberta, Canada	CL	CT, Fert. with manure	41	W vs.F-W-W	30	–	Janzen (1987), Janzen *et al.* (1997)
Lethbridge, Alberta, Canada	CL	CT	41	W vs. F-W-W-H-H-H	30	–	Janzen (1987), Janzen *et al.* (1997)
Woodslee, Ontario, Canada	CL	CT	35	C-O-A-A vs. C	70	+	Gregorich *et al.* (2001)
Elora, Ontario, Canada	SiL	CT; RT	20	C-Leg vs. C	40	+	Yang and Kay (2001a)
Mandan, North Dakota, USA	SiL	CT (low N)	7	W-W-Sf vs. W-F	30	+	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	CT (medium N)	7	W-W-Sf vs. W-F	30	–	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	CT (high N)	7	W-W-Sf vs. W-F	30	–	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	ZT (low N)	7	W-W-Sf vs. W-F	30	+	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	ZT (medium N)	7	W-W-Sf vs. W-F	30	+	Black and Tanaka (1997)
Mandan, North Dakota, USA	SiL	ZT (high N)	7	W-W-Sf vs. W-F	30	+	Black and Tanaka (1997)
North Dakota, USA	SiL	ZT	12	W-W-Sf vs. W-F	30.4	+	Halvorson *et al.* (2002)
North Dakota, USA	SiL	RT	12	W-W-Sf vs. W-F	30.4	+	Halvorson *et al.* (2002)
North Dakota, USA	SiL	CT	12	W-W-Sf vs. W-F	30.4	–	Halvorson *et al.* (2002)
Mead, Nebraska, USA	SiCL	CT	8	C-O-Sm-S vs. C	30	+	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	C-O-Sm-S vs. S	30	+	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	C-O-Sm-S vs. Sm	30	+	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT (0 N, 90 N)	8	C-S vs. C	30	+	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT (180 N)	8	C-S vs. C	30	–	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	C-S vs. S	30	–	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	C-S-Sm-O vs. C	30	+	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	C-S-Sm-O vs. S	30	–	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	C-S-Sm-O vs. Sm	30	–	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	Sm-S vs. S	30	–	Varvel (1994)
Mead, Nebraska, USA	SiCL	CT	8	Sm-S vs. Sm	30	–	Varvel (1994)
Manhatten, Kansas, USA	Si	CT	11	S-Sm vs. S	30	+	Havlin *et al.* (1990), Havlin and Kissel (1997)
Manhatten, Kansas, USA	Si	ZT	11	S-Sm vs. S	30	+	Havlin *et al.* (1990), Havlin and Kissel (1997)

Manhatten, Kansas, USA	Si	CT	11	Sm-S vs. Sm	30	–	Havlin *et al.* (1990), Havlin and Kissel (1997)
Manhatten, Kansas, USA	Si	ZT	11	Sm-S vs. Sm	30	–	Havlin *et al.* (1990), Havlin and Kissel (1997)
Manhatten, Kansas, USA	SiC	CT	12	S-Sm vs. S	30	–	Havlin *et al.* (1990)
Manhatten, Kansas, USA	SiC	ZT	12	S-Sm vs. S	30	–	Havlin *et al.* (1990)
Manhatten, Kansas, USA	SiC	CT	12	Sm-S vs. Sm	30	–	Havlin *et al.* (1990)
Manhatten, Kansas, USA	SiC	ZT	12	Sm-S vs. Sm	30	–	Havlin *et al.* (1990)
Manhatten, Kansas, USA	Si	CT	8	C-S vs. S	30	+	Havlin *et al.* (1990)
Manhatten, Kansas, USA	SI	CT	8	C-S vs. C	30	–	Havlin *et al.* (1990)
West Lafayette, Indiana, USA	SiCL	ZT, CT	28	C-C vs. S-C	100	+	Gál *et al.* (2007)
Hoytville, Ohio, USA	SiCL	ZT	19	C-S vs. C	30	–	Dick *et al.* (1977), Dick (1983)
Hoytville, Ohio, USA	SiCL	CT	19	C-S vs. C	30	–	Dick *et al.* (1977), Dick (1983)
Hoytville, Ohio, USA	SiCL	ZT	19	C-O-G vs. C	30	–	Dick *et al.* (1977), Dick (1983)
Hoytville, Ohio, USA	SiCL	CT	19	C-O-G vs. C	30	+	Dick *et al.* (1977), Dick (1983)
Wooster, Ohio, USA	SiL	ZT	19	C-S vs. C	30	+	Dick *et al.* (1997)
Wooster, Ohio, USA	SiL	ZT	19	C-O-G vs. C	30	–	Dick *et al.* (1997)
Celaya, Mexico	C	CT	5	W/C vs. W/Bn	30	NS	Follet *et al.* (2005)
Celaya, Mexico	C	ZT	5	W/C vs. W/Bn	30	+	Follet *et al.* (2005)
El Dorado do Sul, Brazil	SCL	CT	9	O/V-C/Cp vs. O-C	30	NS	Bayer *et al.* (2000)
El Dorado do Sul, Brazil	SCL	ZT	9	O/V-C/Cp vs. O-C	30	NS	Bayer *et al.* (2000)
El Dorado do Sul, Brazil	SCL	ZT	17	L+C vs. O-C	107.5	+	Diekow *et al.* (2005)
El Dorado do Sul, Brazil	SCL	ZT	17	P+C vs. O-C	107.5	+	Diekow *et al.* (2005)
Central Spain	NA	ZT, CT	11	B-V vs. B	30	+	Lopez-Fando and Pardo (2001)
Central Spain	NA	ZT, CT	11	B-Sf vs. B	30	+	Lopez-Fando and Pardo (2001)

Textural class: LS, Loamy sand; SiL, Silt loam; Si, Silt; L, Loam; SCL, Sandy clay loam; SiCL, Silty clay loam; CL, Clay loam; SiC, Silty Clay; C, Clay. **NA**, not available. **Tillage**: ZT, zero tillage; RT, reduced tillage; CT, conventional tillage; Fertilizer levels indicated between brackets where treatment effect differed between fertilizer levels. **Duration**: Duration of experiment (year) **Treatment**: A, alfalfa (*Medicago sativa* L.); B, barley (*Hordeum vulgare* L.); Bn, bean (*Phaseolus vulgaris* L.); C, corn or maize (*Zea mays* L.); Cp, cowpea (*Vigna unguiculata* L.); F, fallow; G, grass; H, hay; L, lablab (*Lablab purpureum* L. Sweet); Leg, legume; O, oats (*Avena* spp.); Pi, pigeon pea (*Cajanus cajan* L. Millsp); S, soybean (*Glycine max* L. Merr.); Sf, sunflower (*Helianthus annuus* L.); Sm, sorghum (*Sorghum bicolor* L.); V, vetch (*Vicia sativa* L.); W, wheat (*Triticum aestivum* L.). Crops separated by "/" are planted in the same year; Crops separated by '+' are planted in an intercropping system. **Depth**: Sampling depth (cm). Δ**SOC**: Difference in SOC stocks between treatments: +, positive; -, negative.

the plow layer in zero tillage than in conventional tillage soils, a trend also observed by VandenBygaart *et al.* (2002). Black and Tanaka (1997), Havlin *et al.* (1990), Mielke *et al.* (1986) and Blanco-Canqui and Lal (2008) found with some crops and some crop rotations decreased SOC in zero tillage compared to conventional tillage (Table 1). The mechanisms that govern the balance between increased or no sequestration after conversion to zero tillage are not clear. Although more research is needed, some factors that play a role can be distinguished.

***Differences in root development and rhizodeposits*—**One formulated hypothesis on the lack of SOC increase in some cases is the deposit and decomposition of below-ground rhizodeposits (Allmaras *et al.*, 2004). Crop root-derived C may be very important for C storage in soil (Holanda *et al.*, 1998; Flessa *et al.*, 2000; Gregorich *et al.*, 2001; Tresder *et al.*, 2005; Baker *et al.*, 2007). Zero tillage practices can produce greater horizontal distribution of roots and greater root density near the surface (Ballcoelho *et al.*, 1998; Qin *et al.*, 2006). Dwyer *et al.* (1996) found that, despite the fact that total root mass was not significantly different among tillage treatments, rooting was generally shallower in zero tillage than in conventional tillage. Gregorich *et al.* (2001) observed that 10% of root residue C was retained in the plow layer versus 45% below the plow layer for both corn monoculture and corn in a legume-based rotation. On the other hand, Allmaras *et al.* (2004) demonstrated that, as buried unharvested plant materials and roots decompose, more SOC may remain in the subsurface soils of tilled plots than in zero tillage, thus compensating for SOC losses near the surface.

***Baseline SOC content*—**The effectiveness of C storage in zero tillage is reduced and can be negative when the baseline SOC content increases (Paustian *et al.*, 1997). VandenBygaart *et al.* (2003) reported an inverse relationship between SOC content and the effect of tillage on SOC in Canada, with gains due to adoption of zero tillage occurring mainly at SOC levels of less than 45 t ha^{-1}. They speculated that the lower effectiveness of zero tillage in soil with higher SOC levels was due to higher clay contents and higher soil moisture limiting growth potential and inputs of surface residues. Therefore, it can be speculated that depleted old soils have more potential to sequester carbon compared to young soils rich in carbon. However, there is limited information on the influence of tillage on C storage in the depleted soils of Africa, for instance (Table 1).

In line with these findings, VandenBygaart *et al.* (2002) concluded that soil erosion and redistribution over a prolonged period can also affect SOC storage under zero tillage. Soils that had lost SOC through soil erosion had a high potential to gain SOC when converted from conventional tillage to zero tillage, whereas in depressional landscape positions (with high SOC from a history of soil deposition) the potential to gain SOC was lower when converted to zero tillage, with some soils even losing SOC. Yang and Wander (1999) reported no overall (0–90 cm) increase in SOC storage with zero tillage compared to moldboard plowing in Illinois (Table 1) and explained that this is probably due to the fact that erosion is not as significant a factor in central as it is in southern Illinois, where the use of zero tillage practices has been shown to increase SOC conservation (Hussian, 1997).

***Soil bulk density and porosity*—**Physical properties appear to determine whether or not the use of zero tillage practices will enhance C storage by increasing physical protection of SOC. Yoo *et al.* (2006) concluded that the use of zero tillage practices only enhances physical protection of SOC where soil bulk density is relatively high (approximately 1.4 g cm^{-3}) and when the use of zero tillage management reduces the volume of small macropores (15–150 μm), thought to be important for microbial activity. This notion is supported by the findings of Strong *et al.* (2004) who observed rapid decomposition of C in the pores with neck diameters between 15 and 60 μm. There may be a threshold value for bulk density that must be exceeded before pore-dependent processes are constrained and protect SOC. By refining our understanding of the interactions between management, pore structure and SOC mineralization, we should be able to predict the influence of tillage practices on SOC sequestration (Yoo *et al.*, 2006).

***Climate*—**Ogle *et al.* (2005) found that management impacts were sensitive to climate in the following order from largest to smallest changes in SOC: tropical moist > tropical dry > temperate moist > temperate dry. For example, converting from conventional tillage to zero tillage increased SOC storage over 20 years by a factor of 1.23 ± 0.05 in tropical moist climates, which is a 23% increase in SOC, while the corresponding change in tropical dry climates was 1.17 ± 0.05, temperate moist was 1.16 ± 0.02, and temperate dry was 1.10 ± 0.03. This is confirmed by other researchers. Overall, the results indicate that the effects of tillage on soil carbon tend to be small or negative in moist, cold-temperate soils (Hermle *et al.*, 2008; Angers *et al.*, 1997; VandenBygaart *et al.*, 2002; Gregorich *et al.*, 2005). Based on this knowledge, VandenBygaart *et al.* (2003) subdivided their dataset based on whether the study was conducted in western (mean annual precipitation of less than 550 mm; mean annual temperatures of 5°C or less) or eastern (mean annual precipitation of at least 800 mm; mean annual temperatures of 4°C or greater) Canada. In western Canada, the rate of SOC storage in zero tillage soils was 32 ± 15 g m^{-2} yr^{-1}, whereas in eastern Canada the rate of storage was –7 ± 27 g m^{-2} yr^{-1}.

These results demonstrate that agricultural management impacts on SOC storage will vary depending on climatic conditions that influence the plant and soil processes driving soil organic matter dynamics (Ogle *et al.*, 2005). The biochemical kinetics of the processes involved with (1) the breakdown of SOM following cultivation, (2) the formation of aggregates in soils after a change in tillage, and (3) the increased productivity and C input with the implementation of a new cropping practice, are likely to occur at a more favorable rate under the temperature regimes of tropical regions and in more moist climatic conditions. In turn, this leads to a larger change in SOC storage (Ogle *et al.*, 2005). In cooler, more humid climates there may be a reduction in the rate of decomposition of crop residues that are buried after soil inversion by the moldboard plow (Angers *et al.*, 1997). This may limit the ability of zero tillage soils to store SOC in cool, moist climates, since residues are no longer buried after converting to zero tillage, resulting in a net loss of SOC (VandenBygaart *et al.*, 2003). Angers *et al.* (1995) observed that under eastern Canadian conditions, reduced tillage did not lead to reduced mineralization of corn crop residues or to lower SOM contents. The major factors controlling crop residue decomposition and SOM turnover under conditions in eastern Canada are soil moisture and temperature (sub-humid climate). Primary tillage in eastern Canada, which is usually done in late autumn (end of October-November), apparently does not alter the effects of these factors. Apart from cropping system and tillage differences between eastern and western Canada, the cool, moist soils of eastern Canada are often poorly drained and aeration can be limiting at depth, reducing decomposition of buried residues (Angers *et al.*, 1997). Furthermore, in eastern Canada moisture levels at the soil surface are higher for longer periods of time during the year than in the drier prairie soils, favoring greater decomposition of crop residues on the soil surface (Gregorich *et al.*, 2005). Under eastern Canadian conditions, where crop production and residue inputs are not affected by tillage, the placement of the residues would be a major factor determining the SOM balance. Soil organic matter content could be increased in surface soil under zero tillage, whereas under moldboard plow it could be increased in deeper horizons. The net effect of tillage on SOM storage in whole soil profiles would, however, be more or less equal (Angers *et al.*, 1997).

Landscape position and erosion/deposition history—Most of the available studies on C sequestration in different management systems have been conducted on small research plots. In general, these are situated on small, level portions of agricultural fields to minimize confounding effects. However, this does not allow the study of the interaction of other factors on changes in SOC (VandenBygaart, 2006). VandenBygaart *et al.* (2002) showed that landscape position and erosion/deposition history play a significant role in the ability of soils to sequester SOC under zero tillage. They monitored the change in SOC stock at different landscape positions in farmers' fields after converting moldboard plowed soils to zero tillage. Landscape positions that had a low SOC stock due to past erosion (convex positions) generally showed gains in SOC, while positions with large SOC stocks due to deposition (concave and toeslope positions) showed losses after 15 years of zero tillage.

Suboptimal conditions at farm level—Management practices in research plots are delicately controlled. Such ideal conditions, while important to research, often contrast with the zero tillage practices in growers' fields (Blanco-Canqui and Lal, 2008). Agricultural production and farmers' decisions suffer from multiple constraints and natural resource management is tackled at the farming system level (Giller *et al.*, 2006; Smaling and Dixon, 2006) leading in many cases to sub-optimal plot management, in particular when production resources are scarce. Impacts of conservation agriculture under sub-optimal conditions are generally unknown and could delay the SOC build-up period for 2 to 5 years (Franzluebbers and Arshad, 1996). Blanco-Canqui and Lal (2008) assessed the impacts of long-term zero tillage–based cropping systems on SOC sequestration (0–60 cm soil depth) across 11 major land resource areas (containing Ultisols, Alfisols and Inceptisols) in the eastern United States. Soil was sampled in paired zero tillage and plow tillage-based cropping systems on farm. They found that zero tillage farming increased SOC concentrations in the upper layers of some soils, but did not store SOC more than tilled soils for the whole soil profile. More research is needed to determine the effect of conservation agriculture on SOC sequestration at the farm level in different conditions.

2. *The Influence of Crop Rotation on Soil Organic Carbon Stocks*

Increased moisture conservation related to conservation agriculture practices (Govaerts *et al.*, 2007; Sommer *et al.*, 2007) can result in the possibility of growing an extra cover crop right after the harvest of the main crop. Cover crops enhance soil protection, soil fertility, groundwater quality, pest management, SOC concentration, soil structure and water stable aggregates (Wilson *et al.*, 1982; Lal *et al.*, 1991; Ingles *et al.*, 1994; Hermawan and Bomke, 1997; Sainju *et al.*, 2000; Dabney *et al.*, 2001). Cover crops promote SOC sequestration by increasing the input of plant residues and providing a vegetal cover during critical periods (Franzluebbers *et al.*, 1994; Bowman *et al.*, 1999), but the increase in SOC concentration can be negated when the cover crop is incorporated into the soil (Bayer *et al.*, 2000; Table 2). Nyakatawa *et al.* (2001) reported an increase in SOC concentration in soil surface layers after using a zero tillage system with winter rye cover. Replacement of fallow with legume 'green manures' such as lentil (*Lens culinaris* M.) and red clover (*Trifolium pratense* L.) appears to be an effective C storage practice in Canada, with rates of C storage of 15 ± 11 g C m^{-2} yr^{-1} (VandenBygaart *et al.*, 2003). After 19 years of comparing zero and conventional tillage with various winter cover crop treatments in southern Brazil, the winter fallow treatment resulted in the lowest SOC stocks to the 40 cm soil depth compared to all other winter cover crop treatments, independent of soil management (Calegari *et al.*, 2008; Table 2). The inclusion of a N_2-fixing green-manure crop is, however, only a feasible option in regions without a prolonged dry season (Jantalia *et al.*, 2007).

Conservation agriculture can increase the possibility for crop intensification due to a faster turn around time between harvest and planting. Crops can be planted earlier and in a more appropriate planting time. Erenstein and Laxmi (2008) report that zero tillage wheat is particularly appropriate for rice–wheat systems in the Indo-Gangetic Plains because it alleviates system constraints by allowing earlier wheat planting, which results in increased yields. Moreover, new crops can be introduced since the actual growing period can be increased or yet another crop can be planted right after harvest of the main crop. Under irrigated conditions, permanent bed planting creates the option of increased intensification by the intercropping of legume crops with the main crop (Jat *et al.*, 2006).

West and Post (2002) calculated from a global database of 67 long-term experiments that enhancing rotation complexity (i.e., changing from monoculture to continuous rotation cropping, changing crop–fallow to continuous monoculture or rotation cropping, or increasing the number of crops in a rotation system), did not result in sequestering as much SOC (15 ± 11 g C m^{-2} yr^{-1}) on average as did a change to zero tillage, but crop rotation is still more effective in retaining C and N in soil than monoculture (Gregorich *et al.*, 2001; Yang and Kay, 2001a; Table 2). The increased input of C as a result of the greater productivity due to crop intensification will result in increased C sequestration. In the same database, mean C sequestration rates, with a change to zero tillage, for rotation cropping systems were significantly greater than for continuous monocultures (West and Post, 2002). Govaerts *et al.* (2005) reported that long-term maize monoculture decreased crop yield with an attendant reduction in biomass returned to the soil. Buyanovsky and Wagner (1998) reported from a long-term experiment in Missouri that the monoculture of wheat (*Triticum asetivum* L.) with N fertilization accumulated 50 g C m^{-2} y^{-1} compared to 150 g C m^{-2} y^{-1} with the corn *(Zea mays* L.)–wheat clover (*Trifolium* spp.) rotation with manuring and N fertilization. VandenBygaart *et al.* (2003) reported in their review of Canadian studies that, regardless of tillage treatment, more frequent fallowing resulted in a lower potential to gain SOC in Canada. When fallow was removed and wheat grown continuously, SOC was stored at a rate of 15 ± 6 g C m^{-2} yr^{-1}. When hay was included in rotation with fallow and wheat, there was a potential to gain substantial amounts of SOC at a rate of 22 ± 19 g C m^{-2} yr^{-1} (VandenBygaart *et al.*, 2003). This reflects the greater inputs of above- and below-ground residues associated with hay (Campbell *et al.*, 1998). Legume-based cropping systems (lablab + maize intercropping and pigeon pea + maize intercropping) increased C and N stocks in a southern Brasil Acrisol due to the higher residue input in a long-term (17 year) zero-tillage cereal and legume-based cropping system, with an average C sequestration rate of legume-based cropping systems (with N) of 1.42 Mg C ha^{-1} yr^{-1} in the 0–107.5 cm layer (Diekow *et al.*, 2005; Table 2). Introducing legumes in rotation enhances the N pool by symbiotically fixed N (Jarecki and Lal, 2003). On the other hand, Campbell and Zentner (1993) reported that flax (*Linum usitatissimum* L.) contributed smaller amounts of residue with higher lignin contents to the soil than wheat, and flax straw tended to be more easily blown off fields after harvest than wheat straw. The decrease in residue C input may be the cause of lower C sequestration rates or possible SOC loss, as indicated by correlations found between SOC and soil residue inputs (Clapp *et al.*, 2000; Duiker and Lal, 1999; Rasmussen *et al.*, 1980). West and Post (2002) reported that changing from continuous corn to a corn–soybean rotation did not result in increased C sequestration. Continuous corn generally produces more residue and C input than a corn–soybean rotation. Replacing wheat with flax resulted in lower SOC at a rate of –15 ± 2 g C m^{-2} yr^{-1}.

The effect of crop rotation on C sequestration can be due to increased biomass C input, because of the intensified production, or due to the changed quality of the residue input. Many of the wheat experiments consisted of decreasing the fallow period (e.g., changing from a wheat–fallow rotation to a wheat–wheat–fallow rotation) or rotating wheat with one or more different crops (e.g., wheat–sunflower [*Helianthus annuus* L.] or wheat–legume rotations) (Table 2). West and Post (2002) reported that these practices appeared to be more successful in sequestering C than moving from a wheat–fallow rotation to continuous wheat. Therefore, while moving from wheat–fallow to continuous wheat may increase C residue inputs, it does not appear to increase SOC as effectively as a continuous cropping system that either rotates wheat with other crops or reduces the fallow period. VandenBygaart *et al.* (2003) reported that including legumes such as alfalfa (*Medicago sativa* L.) or red clover in rotation with corn can result in large gains in SOC content relative to corn grown in monoculture (44 ± 28 g C m^{-2} yr^{-1}). Gregorich *et al.* (2001) found that SOC below the plow layer was greater in legume-based rotations than under corn in monoculture. They observed that the legume-based rotations contained much greater amounts of aromatic C content (a highly biologically resistant form of carbon) below the plow layer than continuous corn. Crop residue mass may not be the only factor in SOC retention by agricultural soil. The mechanism of capturing C in stable and long-term forms might also be different for different crop species (Gál *et al.*, 2007).

3. *The Influence of Residue Retention on Soil Organic Carbon Stocks*

Crop residues are precursors of the SOM pool. The decomposition of plant material to simple C compounds and assimilation and repeating cycling of C through the microbial biomass with formation of new cells are the primary stages in the humus formation process (Collins *et al.*, 1997). Returning more crop residues is associated with an increase in SOC concentration (Dolan *et al.*, 2006; Wilhelm *et al.*, 2004; Paustian *et al.*, 1997; Rasmussen and Parton, 1994). Blanco-Canqui and Lal (2007) assessed long-term (10 year) impacts of three levels (0, 8, and 16 Mg ha^{-1} on a dry matter basis) of wheat straw applied annually on SOC stocks (0–50 cm depth) under zero tillage on a Crosby silt loam (fine, mixed, active, mesic Aeric Epiaqualf) in central Ohio. Overall, SOC in the 0–50 cm layer was 82.5 Mg ha^{-1} for unmulched soil, 94.1 Mg ha^{-1} for 8 Mg ha^{-1} mulch, and 104.9 Mg ha^{-1} for 16 Mg ha^{-1} mulch. Dersch and Böhm (2001) report enhanced SOC stock in one site in Austria by about 5.55 t ha^{-1} with residue retention in comparison with the removal of all crop residues. The potential to increase SOC stock was lower at another site where soil texture and suitable precipitation distribution provided favorable conditions for mineralization.

The rate of decomposition depends not only on the amount of crop residues retained, but also on soil characteristics and the composition of residues. The composition of residues left on the field—the soluble fraction, lignin, hemic (cellulose) and polyphenol content—will determine its decomposition (Sakala *et al.*, 2000; Vanlauwe *et al.*, 1994; Palm and Rowland, 1997; Palm and Sanchez, 1991, Trinsoutrot *et al.*, 2000; Stevenson and Cole, 1999; Handayanto *et al.*, 1994). The soluble fraction is decomposable (Sakala *et al.*, 2000) and can stimulate the decomposition of the (hemi)cellulose (Vanlauwe *et al.*, 1994). Lignin is resistant to rapid microbial decomposition and can promote the formation of a complex phenyl-propanol structure, which often encrusts the cellulose-hemicellulose matrix and slows decomposition of these components (Sanger *et al.*, 1996). Soybean residues decompose faster than corn and wheat residues (Wagner and Broder, 1993).

4. *Conservation Agriculture: The Combined Effect of Minimum Tillage, Residue Retention, and Crop Rotation on Soil Organic Carbon Stocks*

Conservation agriculture is not a one-component technology but the cumulative effect of all three components it is comprised of. Based on the historical movement from conservation tillage towards zero tillage, researchers in the United States and Canada take it for granted that zero tillage or reduced tillage is always combined with sufficient retention of crop residues. However, in more arid regions, competition for residue is extremely high and farmers are struggling to keep sufficient residue on the soil. Actually, reducing tillage without applying sufficient residue cover can lead to tremendous soil degradation that results in yield declines, in rainfed semi-arid areas (Govaerts *et al.*, 2005, 2006, 2006b, 2007; Sommer *et al.*, 2007; Lichter *et al.*, 2008) as well as in arid irrigated conditions (Limon-Ortega *et al.*, 2006).

The crop intensification component will result in an added effect on C storage in zero-tillage systems. West and Post (2002) report that although relative increases in soil organic matter were small, increases due to adoption of zero tillage were greater and occurred much faster in continuously-cropped than in fallow-based rotations. Halvorson *et al.* (2002) found that zero tillage had little impact on SOC storage in dry climates if the cropping system had a year of bare summer-fallow, presumably due to enhanced decomposition during fallow that negated any benefit of reduced soil disturbance. Sisti *et al.* (2004) found that under a continuous sequence of wheat (winter) and soybean (summer) the stock of soil organic C to 100 cm depth under zero tillage was not significantly different from that under conventional tillage. However, in the rotations with vetch planted as a winter green-manure crop, soil C stocks were approximately 17 Mg ha^{-1} higher under zero tillage than under conventional tillage. It appears that the contribution of N_2 fixation by the leguminous green manure (vetch) in the cropping system was the principal factor responsible for the observed C accumulation in the soil under zero tillage, and that most accumulated C was derived from crop roots. To obtain an accumulation of SOM there must be not only a C input from crop residues but a net external input of N, e.g., including an N-fixing green-manure in the crop rotation (Sisti *et al.*, 2004). Conventional tillage can diminish the effect of an N-fixing green-manure either because the N-input can be reduced by soil mineral N release or the N can be lost by leaching (NO_3^-) or in gaseous forms (via NH_3 volatilization or denitrification) due to SOM mineralization stimulated by the disc plowing that immediately preceded this crop (Alves *et al.*, 2002). Hence, intensification of cropping practices, by the elimination of fallow and moving toward continuous cropping, is the first step toward increased C sequestration. Reducing tillage intensity, by the adoption of zero tillage enhances the cropping intensity effect.

III. CONSERVATION AGRICULTURE IN RELATION TO OTHER TRACE GASSES

The potential to offset greenhouse gas emissions from energy and industrial sources is largely based on studies documenting the CO_2 mitigation potential with conservation agriculture (Kern and Johnson, 1993; West and Marland, 2002). Consequently, some planned emissions trading between industry and producers is based on the uptake of CO_2 from the atmosphere and subsequent soil storage of C following adoption of conservation agriculture. This type of emission trading may fail to reduce the potentially-deleterious effects of greenhouse gas emissions on the climate if it does not consider the net result of fluxes for all three major biogenic greenhouse gases (i.e., CO_2, N_2O and CH_4) on radiative forcing, which is essential for understanding agriculture's impact on the net global warming potential (Kessavalou *et al.*, 1998; Robertson *et al.*, 2000; Oenema *et al.*, 2001; Smith *et al.*, 2001). Nevertheless, only a few studies have considered the fluxes of all three major greenhouse gases that are impacted by tillage management.

Increases in soil organic matter, can increase N cycling in soil and that leads to larger emissions of N_2O (Butterbach-Bahl *et al.*, 2004) as nitrification is stimulated (Gök and Ottow, 1988). As a result, more NO_3^- will be formed in soil, but when anaerobic micro-sites are formed when O_2 diffusion is inhibited, reduction of N_2O to N_2 will be reduced increasing the N_2O-to-N_2 ratio (Firestone *et al.*, 1980). However, zero tillage with residue retention improves soil structure compared to conventional tillage (Govaerts *et al.*, 2007) so that fewer anaerobic micro-sites are formed reducing denitrification and the emission of N_2O. It can be speculated that a better aeration of soil as a result of increased soil organic matter content and the resultant increased aggregate stability will inhibit denitrification and stimulate oxidation of CH_4, but it remains to be seen how emissions of NO and N_2O are really affected. Rochette (2008) concluded that N_2O emissions only increased in poorly-drained finely-textured agricultural soils under zero tillage located in regions with a humid climate, but not in well-drained aerated soils. The global warming potential of 1 kg emitted N_2O-N ha^{-1} is equivalent to 125 kg C ha^{-1} so that gain of C sequestered in soil might be offset by the increased emission of N_2O. Converting natural soils to agriculture soils reduce their capacity to serve as sink for CH_4 (Boeckx *et al.*, 1998). Mosier *et al.* (1991) found that the capacity of a pasture soil to oxidize CH_4 decreased by 41% when 22 kg N ha^{-1} of NH_4NO_3 was added. Steudler *et al.* (1989) found that N fertilization also resulted in an inhibition of CH_4 oxidation in a forest soil. However, N fertilizer applications to a regularly fertilized wheat field had no effect on CH_4 oxidation (Mosier *et al.*, 1991). A better aerated soil with no tillage and residue retention would also favor CH_4 reduction and inhibit CH_4 production. However, soil as a sink for CH_4 is far less important than as a source for N_2O.

Chatskikh *et al.* (2008) found that the average daily soil CO_2 respiration was significantly higher for conventional tillage than for zero-tillage, whereas the N_2O emissions did not show consistent differences. Mosier *et al.* (2006) found that nitrous oxide fluxes from unfertilized conventionally-tilled soils were small, yet were significantly greater than from zero-tillage soils. Zero-tillage soils were net sinks of global warming potential when adequate fertilizer was added to maintain crop production. The conventionally tilled soils were smaller net sinks than zero tillage soils (Mosier *et al.*, 2006). Liu *et al.* (2005) found that during the growing season tillage did not significantly affect the emission of N_2O, but zero tillage resulted in a much lower emission of NO compared to conventional tillage; and during the fallow period, much more N_2O (2.0–3.1 times) and NO (13.1–16.8 times) were emitted from conventionally tilled soils than from zero tillage. Patiño-Zúñiga *et al.* (2008) reported from laboratory incubation experiments that conservation agriculture, in its version of permanent raised bed planting with crop residue retention, decreased emissions of N_2O and CO_2 compared to soil under conventionally tilled raised beds.

IV. FARMERS MANAGING SOIL CARBON

A. The Economic Potential of Conservation Agriculture for Carbon Sequestration

The technical potential of conservation agriculture for C sequestration has been established in the previous sections. The discussion now turns to the economic potential of conservation agriculture for C sequestration considering the profitability and the cost of C sequestration, and the prospects for widespread adoption. Generally, the off-site public benefits of conservation agriculture exceed the on-farm private benefits (Knowler and Bradshaw, 2007). Micro-economic analyses in the United States indicate that the economic potential for C sequestration under CA ranges from 22% to 78% of the technical potential (Bangsund and Leistritz, 2008). It should be noted that the profitability of conservation agriculture varies widely, depending on the characteristics of farming systems, local markets and institutions, and relevant agri-environmental policies. Knowler (2003) reports a positive Net Present Value from on-farm benefits in approximately 90% of 29 studies of conservation agriculture in various farming systems in Latin America and sub-Saharan Africa. In southern Brazil, conservation agriculture profitability dominates conventional tillage (Sorrenson, 1997). Erenstein and Laxmi (2008) report the reduction of field operational costs by 15–16% by the adoption of zero-tillage drills in rice–wheat farming systems in northwest India and the Pakistani Punjab. Long-term trials in Mexico suggest continuously higher and more stable yields for both wheat and maize compared to the farmers conventional practice of tillage and residue removal, even with optimal levels of inputs in both cases (Govaerts *et al.*, 2005). In Zambia, Haggblade and Tembo (2003) report that, on average, hand-hoe conservation agriculture in smallholder maize-based farming systems in Zambia attained higher maize and cotton yields than conventional hand-hoe or animal draft systems respectively: 1.5 tons more maize per ha and 460 kg more cotton per ha. In the case of maize, it is estimated that 1.1 tons of this increase can be attributed to the improved practices. Even though the cost of production and labor use increased under conservation agriculture (at least initially) in this area, the gross margins and returns to labor are larger than under conventional agriculture (Haggblade and Tembo, 2003). Stonehouse (1997) estimates higher returns for conservation agriculture in large-scale commercial wheat farming systems in Canada. In conclusion, conservation agriculture practices seem to have a higher relative profitability compared with conventional tillage in diverse farming systems, climates, and regions.

There are relatively few studies of the cost of C sequestration per se, and they are mostly confined to commercial mechanized farming systems. McCarl and Schneider (2001) report substantial scope for C sequestration on cropland for less than US$100 per ton C. Murray *et al.* (2004) concludes that improved crop management (and afforestation) is competitive with nonagricultural C sequestration at low C prices of US$5 per ton CO_2 but not at higher C prices. On a national U.S. scale, however, Lewandrowski *et al.* (2004) find that improved crop management is competitive over a range of C prices US$10–125 per ton. Antle *et al.* (2003) and Capalbo *et al.* (2004) show that C sequestration through conversion from summer-fallow to continuous, reduced-tillage cropping in eastern Montana commercial cereal systems occurred with average marginal costs of less than US$50 per ton CO_2, which they claim are comparable with marginal costs of C sequestration in Iowa. C sequestration through improved crop system management is competitive with nonagricultural C sequestration.

Carbon markets offer the potential of additional income for farmers including, under certain conditions, smallholders in developing countries. The Clean Development Mechanism of the Kyoto Protocol provided both the framework and the stimulus for carbon trading which currently occurs in the Chicago Climate Exchange and the European Exchange. Although the price is low, this offers another potential source of income for farmers; and may provide added incentive for the adoption of C sequestration technology. The effective development of C markets requires functional techniques to assess baselines and changes in the soil C pool at a field and landscape level over short time periods. Trading of C credits may be facilitated by effective community organization to minimize transaction costs.

B. Farmers Managing Soil Carbon: Beyond Direct Incentives

The management of the majority of soil C lies in the hands of the farmers, pastoralists, and forest managers whose decisions are determined by multiple goals. Whilst future income-earning capacity is often given a high priority by households, the relative importance of current and future consumption depends on the farm household type and their current situation (Corbett, 1988; Frankenberger and Coyle, 1993). The major potential for conservation agriculture as a climate mitigation strategy is based on its related agronomic and economic productivity gains. As described above, the additional private benefits from the partial or full adoption of conservation agriculture are generally substantial even in the absence of incremental profits arising from market or subsidy payments from C sequestration. Soil and water conservation technologies have been barely implemented by farmers (in many cases only during periods when direct incentives were provided) because such practices do not always result in soil erosion reduction or do not increase yields (Hellin and Schader, 2003). Conservation agriculture has that win-win combination of being a soil and water conservation technology that can also increase productivity in most cases. Higher yields in (Govaerts *et al.*, 2005) are the result of an increase in soil quality, especially in the topsoil (Govaerts *et al.*, 2006). Increased aggregation and soil organic matter at the soil surface lead to increased water and nutrient use efficiency (Franzluebbers, 2002) as well as reduced soil erosion (Dickey *et al.*, 1985; Barton *et al.*, 2004; Scopel *et al.*, 2005; Zhang *et al.*, 2007). The increased production profitability can be the major driving factor for farmers to implement conservation agriculture and as such the soil carbon sequestration strategy, and thus, go beyond ineffective and expensive direct incentives.

Competitive demands on resources at farm level, such as crop residues, can constitute serious bottlenecks to conservation agriculture implementation, particularly in semi-arid rainfed agriculture, as opposed to cropping systems in wetter conditions or under irrigation (Erenstein, 2002). Although exposure to market forces can be advantageous in the sense that farmers can reap the benefits of cost savings, access to fodder markets can create disincentives for the retention of residues on farm. This is often the case in densely populated regions with heavy demand on biomass and strong demand for livestock products, such as in India. Such market opportunities for straw and stover accentuate the value of crop residues in alternative uses for livestock fodder, household energy and construction purposes (alternative to leaving on the field to protect the soil surface). Moreover, the majority of smallholders are mixed crop–livestock farmers who traditionally use crop residues for maintaining their livestock. In these cases farmers are reluctant to leave residues on the surface despite the demonstrable yield advantages. More research is needed to establish minimum residue retention levels (thresholds) with positive impacts on carbon storage.

C. Constraints and Pathways for Adoption

Given that conservation agriculture appears to generate attractive private and social benefits, as well as improving the stability of yields and the productivity of inputs, it behooves us to ask why the adoption rates are not faster. Moreover, the dichotomy between large commercial farm adoption rates and smallholder adoption rates merits discussion. There are a variety of characteristics of smallholders which slow down adoption of any new productivity-enhancing technology, and which become a critical ("killer") constraint in the case of conservation agriculture. Smallholders have a multiplicity of household livelihood goals which extend beyond production and profit (Ellis, 1999). There are also a set of constraints related to the natural assets of the farm (Dixon *et al.*, 2001). Many poor smallholders cultivate poorer soil or steeper slopes than their better-off neighbors in the same community which may have heavier infestations of weed or respond more slowly to conservation agriculture practices. With small landholdings and therefore limited food production entitlements (they produce only a portion of household food consumption needs), many smallholders are risk averse and avoid introducing new practices with the perceived additional risk to household food security (Binswanger, 1980). With its added complexity and onerous management requirements, conservation agriculture is often perceived as more risky than improved varieties or fertilizer. The small size of holdings and often unclear land tenure precludes borrowing so that smallholders have little access to financial capital for new equipment or the purchase of inputs such as herbicides (Soule *et al.*, 2000). The reliance on family and often shared labor is a significant impediment to adoption of conservation agriculture. Workers generally lack the understanding of conservation agriculture even if the farmer does have a good appreciation of its principles and practices. Quite apart from field workers, the farmers themselves often lack education and thus are excluded from some of the knowledge streams which provide information on conservation agriculture. Small scale itself can also be a constraint for the efficient utilization of much conservation agriculture farm equipment, e.g., zero till drills, which forces the commercialization of machinery provision services or the formation of farmer groups for resource/equipment sharing—and each has attendant additional transaction costs (Laxmi *et al.*, 2007).

It is unlikely that complex, multicomponent technologies such as conservation agriculture can be successfully scaled out through traditional linear models of research and extension: instead they require the development of innovation systems to adapt technologies to local conditions. Experience in commercial and noncommercial farming systems show that it is essential that an innovation systems approach includes functioning networks of farmer groups, machinery developers, extension workers, local business, and researchers (Hall *et al.*, 2005). For this purpose, decentralized learning hubs within different farming systems and agro-ecological zones should be developed (Sayre and Govaerts, 2008). In those hubs, an intense contact and exchange of information is organized between the different partners in the research and extension process. Because of the multifaceted nature of conservation agriculture technology development and extension, activities should be concentrated in a few defined locations representative of certain farming systems rather than have lower intensity efforts on a wide scale. Through the research and training, regional conservation agriculture networks are established to facilitate and foment research and the extension of innovation systems and technologies. Research at the hubs also provides an example of the functionality of conservation agriculture systems, helping to break down the culture of the plow. The hubs are linked to the strategic science platforms operated by international centers and national research institutes to synthesize a global understanding of conservation agriculture, and its adaptability to different environments, cropping systems, and farmers' circumstances.

D. The Consequences of Rotating Tillage Practices for Carbon Sequestration

Because SOC responds dynamically to management, a policy to promote C sequestration presupposes the maintenance of the practices which promote the accumulation of organic matter (Paustian *et al.*, 1997). Little is known about the year-to-year soil management practices of individual farmers (VandenBygaart and Kay, 2004). Tillage systems are rotated for various reasons, including optimizing yields and managing pest and disease problems (Hill, 2001) and controlling weeds (Kettler *et al.*, 2000). Farmers have experimented with zero tillage practices on a relatively high percentage of the fields in the central and northern Corn Belt region in the United States (as much as 77% of the fields sampled in Indiana), but the longest average time that fields were maintained in continuous zero tillage is slightly less than 2.5 years in Illinois and Indiana and only 1.4 years in Minnesota.

The effect of rotational zero tillage on C sequestration is not well documented. To our knowledge, there are no reports about field experiments where a long-term zero tillage field is brought back permanently under conventional plowing. A model analysis for three sites with fine-textured soils and different crop rotations in North America (Conant *et al.*, 2007) simulated zero tillage until equilibrium was reached and ran experimental models for 220 years thereafter. Model results demonstrated that changing the management from zero tillage to continuous conventional tillage (moldboard plowing to a depth of 20 cm) reduced soil C content by an average of nearly 27% across the three sites compared to the zero tillage equilibrium. All the model runs in the study indicated a loss of soil C following even one tillage event. Losses increased as the frequency of tillage increased and were greater following more intensive tillage (Conant *et al.*, 2007).

Immediately after a tillage event, CO_2 emissions are greatly increased compared to a zero tillage field (Reicosky *et al.*, 1997). An amount equivalent to approximately 30% of the annual crop residue carbon was transferred to the atmosphere in a period of 4 weeks after moldboard tillage (30 cm deep) in a sugar cane field on a clay Oxisol in southern Brazil that had been under zero tillage with high crop residues for 6 years, an increase of 160% compared to the zero-tillage control (La Scala *et al.*, 2006).

In the longer term, the effect of a single tillage operation on SOC content in fields that have been under zero tillage for an extended period seems to differ between soils. Koch and Stockfisch (2006) studied three adjacent fields on a Stagnic Luvisol in a temperate climate in Germany, that had been under reduced tillage for 7–9 years, consisting of 30 cm deep non-inverting tillage combined with shallow-mixing (10 cm deep) operations. Moldboard ploughing (30 cm deep) was conducted once in each field and then reduced tillage was re-established. The single ploughing operation resulted in a substantial loss of organic matter: 1.5–2.5 years after ploughing, losses from the 0–45 cm depth accounted for 6% of the initial total mass of SOC.

Kettler *et al.* (2000) assessed the influence of moldboard plowing (15 cm deep) and secondary tillage operations on SOC content of a silt loam soil that had been cropped in a zero-tillage winter wheat–fallow system for more than 20 years in the western United States. Results of this study showed that SOC in surface soil was redistributed within the top 30 cm of the soil profile by plowing, but 5 years after tillage the SOC content from 0 to 30 cm did not differ between the plowed and the continuous zero tillage soil.

VandenBygaart and Kay (2004) determined the change in SOC when a long-term (22 yr) zero tillage field in southern Ontario, Canada, was moldboard plowed once (20 cm deep). Four plots were located within three textural classes (sandy loam, sandy clay loam, and silty clay loam) within two hydrologic conditions in the field (well- and poorly-drained). When calculated on an equivalent mass basis beyond the plow depth, there was no significant change in SOC 1.5 years after plowing the sandy clay loam, silty clay loam, and sandy loam with high SOC plots. However, in the sandy loam plot with low SOC, there was a loss of about 3 Mg SOC ha^{-1} after 1.5 years, a loss that may have accounted for as much as two-thirds of the SOC gained from zero tillage. They attributed the different effect of plowing to differences in forms of SOC that may have been sequestered during the zero tillage period, since earlier research in this field had shown that about 62% of the increase in SOC under zero tillage was in the humified fraction of the sandy clay loam soils, while almost 70% of the gain in SOC under zero tillage in the coarser-textured soil had been in the occluded particulate fraction (Yang and Kay, 2001b). Another possible reason for the difference was the probably lower proportion of microaggregates in the coarser-textured soil, providing less protection of the particulate SOC fraction from microbial decomposition (VandenBygaart and Kay, 2004).

V. CONCLUSIONS AND FUTURE PERSPECTIVES

Today's global cultivated area has been strongly degraded. Even in high-yielding areas where soils are not considered to be degraded, crops require ever increasing input to maintain yields. In the frame of the recent food crisis, it is clear that agriculture should not only be high yielding, but also sustainable. Conservation agriculture is a cropping system both characterized by short-term maximization of crop production as well as by potential long-term sustainability (i.e., carbon storage) at micro-site (i.e., soil aggregation studies) and farm level (i.e., yields analysis, profitability). Concerning the potential of conservation agriculture as a strategy for C sequestration, important gaps still need to be covered. In general, it is striking that, for the developing world and the more tropical and subtropical areas, information is lacking on the influence of tillage and crop rotation on C storage. Most results were obtained at the plot level, and more holistic research at the farm level, including agro-ecosystem constraints, is needed, as well as total carbon sequestration budgets at the regional and global levels. It is of course clear that most information is coming from areas where reduced tillage and conservation agriculture have been implemented already for a substantial time. This confirms the need for the development of an international network of hubs that install working examples of conservation agriculture within the different agro-ecological areas and farming systems. Only by connecting all these different sites in a working network of excellence, the underlying mechanisms of C storage as influenced by conservation agriculture can be revealed. The increasing evidence points to the validity of conservation agriculture as a carbon storage practice and justifies further efforts in research and development.

However, even if carbon sequestration is questionable in some areas and some cropping systems, conservation agriculture remains an important technology that improves soil processes, controls soil erosion, and reduces tillage-related production costs, and these are sufficient reasons to promote the step-by-step conversion by adopting resource conserving technologies with conservation agriculture as the final goal. Although a more detailed knowledge of functional relationships is required to determine the real potential of conservation agriculture as a carbon off-set technology, it is safer to adopt agricultural practices that preserve and restore soil functionality than practices that destroy it. Global food security, global environmental preservation, as well as farmer-level increased livelihood, should be the main goals of a sustainable farming system.

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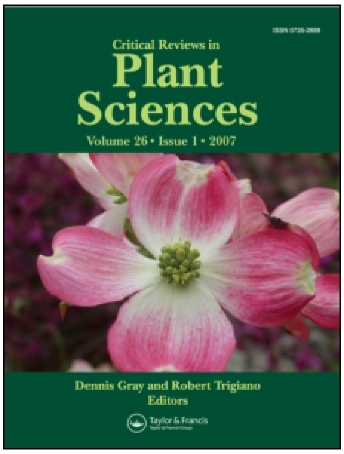

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