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Management options for reducing CO_2-concentrations in the atmosphere by increasing carbon sequestration in the soil

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Abstract

Management practices for increasing the storage of organic carbon in the soil deserve more attention in policies aimed at reducing national and global CO_2-budgets, similar to re- or afforestation and bio-fuel programmes (*cf.* Kyoto Protocol to the United Nations Framework Convention on Climate Change). Best management practices to build-up carbon stocks in the soil, basically are those that increase the input of organic matter to the soil, and/or decrease the rate of soil organic matter decomposition. According to this review, the most appropriate management practices to increase soil C reserves are site specific. Available best management practices will require evaluation and adaptation with reference to soil type and land use system, and this preferably by agro-ecological region. The feasibility of the various technical options available for increasing carbon stocks, in mainly agricultural soils, is discussed by agro-ecological zone. Our exploratory scenarios, which use necessarily coarse assumptions about the potential for increased carbon sequestration in the soil, show that from 14 ±7 Pg C may be sequestered over the next 25 years - with even higher potentials over a 50 year period - if the world's ‘degraded’ and ‘stable’ Agricultural Lands are restored and/or submitted to appropriate management. When the ‘degraded’ and ‘stable’ Agricultural Lands, Extensive Grasslands and Forest Regrowth categories are considered, this would be 20 ± 10 Pg C. On average, from 0.58 to 0.80 Pg C yr^{-1} can be sequestered in the soils according to these scenarios; this would correspond with about 9-12% of the anthropogenic CO_2-C produced annually. The scenarios assume that 'best' management and/or manipulation of a large portion of the globe's soils is possible; yet their implementation need not necessarily be feasible due to the economic, environmental and societal/cultural conditions. Mitigation of atmospheric CO_2 by increased carbon sequestration in the soil, particularly makes sense in the scope of other global challenges such as combatting land degradation, improving soil quality and productivity, and preserving biodiversity. Effective mitigation policies will likely be based on a combination of many modest and economically sound reductions, which confer added-benefits to society. In identifying these 'best practices', due attention must be paid also to any possible adverse environmental and socio-economic effects some of these practices may have.

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Acknowledgements

This study was commissioned by the Dutch National Research Programme on Global Air Pollution and Climate Change (NRP Project: 952282), in support of international negotiations on reducing atmospheric concentrations of greenhouse gases and aimed at combatting environmental degradation (e.g., Kyoto Protocol to UNFCCC), and as a Dutch contribution to the Special Report on Land Use, Land Use Change and Forestry of the Intergovernmental Panel on Climate Change (IPCC). The study draws heavily on materials received from a wide range of scientists, from the various continents of the world, who provided the author with re-prints subsequent to an initial e-mail questionnaire regarding the availability of published materials on 'possibilities for carbon sequestration in the soils of the world, in the broader context of global change'. I gratefully thank all those who have provided materials for this review, and sincerely hope that their contributions have been adequately acknowledged in the references. I would also like to thank the members of the project's Steering Committee, consisting of Dr. A.F. Bouwman (RIVM), Dr. P.J. Kuikman (AB-DLO), Ir. J.W. Nieuwenhuis (VROM) and Dr. Wilko Verweij (NRP), as well as Dr. W.G. Sombroek (ISRIC) for their useful advice and comments. Thanks are expressed also to two anonymous reviewers, selected by NRP, for their comments and suggestions. Finally, I thank Mr. W.C.W.A. Bomer for preparing the figures.

Executive Summary

Terrestrial ecosystems are thought to be a major sink for carbon at the present time, the 'missing sink' amounting to about 1-2 Pg C yr^{-1}. Variable sequestration of carbon by the terrestrial biosphere is a main cause of observed year-to-year variations in the rate of atmospheric CO_2 accumulation. The Kyoto protocol currently restricts the allowable terrestrial sources and sinks of carbon to strictly defined cases of 'afforestation, reforestation and deforestation'. Appropriate management of the terrestrial biosphere and especially of soils, however, can substantially reduce the buildup of atmospheric greenhouse gases.

Agro(eco)systems can be managed to reduce carbon emissions and increase carbon sinks in vegetation and soil. It appears, however, that this increased carbon uptake/storage can only offset fossil fuel emissions temporarily (on a time scale from decades to a century), and partially, after which new equilibria levels will be reached (provided these agro(eco)systems remain undisturbed).

Best management practices to build-up carbon stocks in the soil, basically are those that increase the input of organic matter to the soil, and/or decrease the rates of soil organic matter decomposition. According to this review, the most appropriate management practices to increase soil C reserves are site specific. Available best management practices will require evaluation and adaptation with reference to soil type and land use system, and this preferably by agro-ecological region. At the same time, their implementation should be socially, politically, and economically acceptable. Possible implications of the 'CO_2-fertilization effect' on crop growth, crop quality and organic matter decomposition, although less well understood, should be considered also.

Methods are needed that increase, and permit to monitor and verify, the rate at which soils sequester carbon and the quantities that can ultimately be sequestered. Soils can store carbon in pools with different turnover times. With respect to C sequestration it is best to immobilize the atmospheric CO_2-C in soil C-pools having long turnover times. As mechanisms based on C storage in live biomass or litter will sequester C only for a relatively short time, they are considered of lesser importance for long-term C sequestration strategies in terrestrial ecosystems. Sustainable management of forests, notably those in the temperate and tropical regions, however can significantly increase the biomass-C in standing forests.

Improved information on the nature and dynamics of organo-mineral associations will lead to an enhanced understanding of soil structural dynamics, soil C-cycling and C-sequestration in the soil, thus providing a better basis for developing improved approaches to soil management. A study of the processes and mechanisms that regulate the stability of organic carbon in selected 'old-agricultural' soils, enriched in phosphorus, would be useful.

Detection of small increments in soil carbon storage over the relatively short-time frames of relevance for monitoring and verification of article 3.4 of the Kyoto Protocol requires sensitive techniques such as carbon isotope fractioning and ^{13}C NMR. Carbon isotope fractioning will mainly be of use where the vegetation has changed from forest to grassland, that is from C_3- to C_4-crops, or vice-versa. Observing changes in soil carbon stocks and dynamics within C_3 or C_4 systems, however, will require other sensitive techniques. The possibilities for using remote sensing and other non-destructive techniques for monitoring changes in carbon fluxes to/from and carbon stocks in the soil need to be researched further.

World-encompassing databases of soils and land use often are partly outdated, and thus may not be adequate for modelling the potential for carbon sequestration under changing conditions of land use and climate. Hence the continued need for data consolidation and actions in support of a 'comprehensive carbon approach' (e.g., remote sensing, land and soil data) to allow for an effective implementation/verification of the Kyoto Protocol (e.g., articles 3.3, 3.4, 3.7 and 6).

The potential for C-sequestration in a given soil, and agro-ecological zone, will be proportional to the original reserves present under undisturbed conditions. Therefore, options for C sequestration should be chosen on the basis of knowledge of the nature and likely magnitude of C pools, whether organic or inorganic, in a given biome or major agro-ecological region and their responses to different land uses and management systems. Exploratory analyses were made to allow for, necessarily coarse, inferences about the potential for increased carbon sequestration in the soil, using available global databases of agro-ecological zones, soil degradation, land use, and soil carbon stocks, supplemented with literature-based data on possible increases in SOC sequestration with improved management. This information then has been translated into exploratory, GIS-based scenarios of what should be 'technically' possible, in terms of SOC sequestration potential, over the next 25 years. Using these broad assumptions, it is estimated that from 16.1 to 48.3 Pg C can be can be sequestered in the 'degraded' lands of the world over the next 25 years, upon restoration (scenario A). If soils of the Arid, Polar and Boreal AEZs are excluded from the analyses, in view of their overall limited possibilities for 'reclamation', this range would be 11.8 to 35.1 Pg C (scenario B). Scenario C only considers the restoration of degraded agricultural lands, giving a sequestration potential of 5.1 - 15.3 Pg for the region under consideration (scenario C). In scenario D, areas of 'degraded' Extensive Grasslands and Forest Regrowth are considered in addition to the 'degraded' Arable Lands; the range then is from 6.5 to 19.4 Pg C. Finally, if one would only consider potential increases, via improved management, for the Arable lands that occur on the 'stable' lands, the range is from 2.1 to 6.4 Pg C (scenario E). If like for scenario D, areas of Extensive grasslands and Forest Regrowth are considered in addition to the Arable lands, improved management of the 'stable' lands would give a possible increase of 3.5 to 10.4 Pg C over the next 25 years. Thus a likely increase would be in the order of 14 ±7 Pg C for scenarios C and E, and 20 ±10 Pg C for scenarios D and F. From scenarios C and E and scenarios D and F, respectively, it follows that, on average, from 0.58 Pg C yr^{-1} to 0.80 Pg C yr^{-1} can be sequestered in the soils of Arable Lands, resp. Arable Lands, Extensive Grasslands, and Regrowth Forest over the next 25 years; this would correspond with about 9-12% of the anthropogenic CO_2-C produced annually. Our estimates are within the range of historic C-losses, from cultivated soils, published for the last 100 years; the latter may be seen as an upper limit of what may be achievable in terms of SOC sequestration by improved and appropriate management. Our largely 'technical' scenarios assume that intensive management and/or manipulation of a large portion of the globe's soils will be possible; yet this need not necessarily be feasible due to the economic, environmental and societal/cultural conditions.

Mitigation of atmospheric CO_2 by increased carbon sequestration in the soil, particularly makes sense in the scope of other global challenges such as combatting land degradation, improving soil quality and productivity, and preserving biodiversity. Effective mitigation policies will likely be based on a combination of many modest and economically sound reductions, which confer added-benefits to society. In identifying these 'best practices', due attention must be paid also to any possible adverse environmental and socio-economic effects some of these practices may have.

Despite the uncertainties that remain associated with model-based projections, such techniques are essential for providing quantified information on possible trends is soil C levels following alternate management practices, possible simultaneous effects of global change, and changes in policies, especially when they also provide information about the range in uncertainty of the projections. Much work remains to be done in this field.

Within a given agro-ecological region, the organic carbon content of most agricultural soils is lower than that of natural peatlands or forest soils. The greatest potential to increase current soil C stocks therefore probably is through improved management of the agricultural land, particularly degraded crop- and grass-lands, as well as conservation/restoration of marginal lands and wetlands/peatlands.
Irrespective of scale and geographic location, a key objective should be to identify and implement the best available management practices to improve soil quality, thereby ensuring a sustainable use of the land, food productivity and security, while simultaneously reducing the CO_2 concentration in the atmosphere. In identifying these best practices, due attention must be paid also to any possible, adverse environmental and socio-economic effects.

1 Introduction

1.1 Background

The rate of change of atmospheric CO_2 concentration over the Holocene was two orders of magnitude smaller than the anthropogenic CO_2 increase since industrialization (Indermühle *et al*., 1999). The expanding use of fossil fuels and large-scale land-use changes have led to increased concentrations of radiatively-active trace gases in the atmosphere, affecting global climate (Watson *et al*., 1996). The associated, predicted change in average air temperatures is likely to be accompanied by changes in precipitation and storm patterns and alterations in drought intensity and frequency. These changes in global climate could significantly affect land degradation, agricultural production, water supplies, human health, and terrestrial and aquatic ecosystems in developed and developing countries (Alexandratos, 1995; Dixon, 1997; Watson *et al*., 1996). Thus global climate change may be the most critical and complex environmental issue facing humanity over the next century.

The *Kyoto Protocol*, signed by 174 countries in December 1997 during the Third Conference of the parties to the United Nations Framework Convention on Climate Change (UNFCCC), has set the first precise targets in terms of levels and dates for reducing overall greenhouse gas emissions to the atmosphere (see WBBGU, 1998, p. 56-67). Between 2008 and 2012, Europe is to cut greenhouse gases emissions by 8%, the United States by 7%, and Japan by 6% compared to 1990 levels. The aim of the Framework Convention on Climate Change is to stabilize greenhouse gas concentrations in the atmosphere at a level which would limit the adverse anthropogenic impact on the Earth's climate.

Many research institutes and industries have been developing technologies to mitigate atmospheric CO_2 concentrations. The available options include: separation and capture of CO_2 from the energy system; sequestration in the oceans, terrestrial ecosystems, and geologic formations; advanced biological processes; and, chemical approaches (e.g., Reichle *et al*., 1999; Watson *et al*., 1996). An important issue in this context is to which extent the technical measures can be effective in periods of economic growth, associated with higher uses of fuel by industry and the transport sector. And whether these measures, alone, can adequately reduce current atmospheric CO_2 levels as envisaged by the policy makers.

Terrestrial ecosystems, which consist of vegetation and soils, are thought to be a major sink for carbon at the present time, the 'missing sink' amounting to about 1.8 ±1.5 Pg C yr^{-1} (Houghton *et al*., 1998) or 1.0 to 2.2 Pg C yr^{-1} (Fan *et al*., 1998). Variable sequestration of carbon by the terrestrial biosphere is the main cause of observed year-to-year variations in the rate of atmospheric CO_2 accumulation (Lee *et al*., 1998). The Kyoto Protocol (article 3.3) limits the allowable terrestrial sources and sinks of carbon to strictly defined cases of 'afforestation, reforestation and deforestation' (Nabuurs *et al*., 1999). There are, however, many more ways in which appropriate management of the terrestrial biosphere, especially of soils, can substantially reduce the buildup of atmospheric greenhouse gases.

Despite earlier studies based on historical data (Jenkinson *et al*., 1991; Schlesinger, 1990), there now seems little reason to suppose that soils have a low carbon storage potential under both present-day and future climate change conditions (Amthor, 1995; Batjes, 1998; Lal *et al*., 1998b). World soils, especially those in the northern latitudes and highly productive soils under arable, pastoral and silvicultural land uses can be important C sinks. In addition to help mitigating the greenhouse effect, added positive effects of increased organic matter contents for soil nutrient cycling, soil water holding properties and thermal properties, and thereby in sustaining soil productivity, biodiversity and ultimately world food supply are important (Alexandratos, 1995; Batjes and Sombroek, 1997; Ingram and Gregory, 1996; Sombroek, 1995).

1.2 Objectives

This review was commissioned by the Dutch National Research Programme on Global Air Pollution and Climate Change (NRP) in support of international negotiations on reducing atmospheric concentrations of greenhouse gases and aimed at combatting environmental degradation (e.g., Kyoto Protocol to UNFCCC). In addition, it will be used in support of the Dutch contribution to the Special Report on Land Use, Land Use Change and Forestry of the Intergovernmental Panel on Climate Change (IPCC).

The current study aims to complement other projects within the Dutch National Research Programme on Global Air Pollution and Climate Change that consider effects of changes in C-fluxes to the vegetation and soil under increased ambient concentrations of CO_2 in the atmosphere (NRP, 1998). Its focus is largely on the potential for using agricultural soils as a sink to mitigate CO_2 emissions through improved management (cf. Article 3.4 of the Kyoto Protocol), since issues related to 'reforestation, afforestation, and deforestation' have been addressed elsewhere (Nabuurs *et al*., 1999). In view of the simultaneous effects of 'climate change' on possibilities for soil carbon sequestration over the time-scales considered, these possible interactions are also briefly reviewed. An evaluation of socio-economic policies designed to promote the build-up-of organic carbon in agricultural and other soils (Bouzaher *et al*., 1995; Izac, 1997; Rexler and Broekoff, 1995; Squires and Glenn, 1995), although being crucial to the viability of the available options, is beyond the scope of this review.

Chapter 2 starts by reviewing the positive properties of soil organic matter (2.1), followed by the main controlling factors of soil organic matter formation (2.2), and soil organic matter models (2.3). The third chapter presents data on global soil carbon stocks, both organic and inorganic, thus illustrating the important role soils play in the global carbon cycle. Chapter 4 discusses C stocks in predominantly agricultural soils with examples of changes therein, both negative and positive, as induced largely by changes in land use. Chapter 5 reviews management options for increasing soil carbon stocks, with special attention to reduced tillage methods, soil fertility management, soil water management and erosion control, and it also considers possible adverse environmental side-effects associated with these practices. The potential for increased carbon sequestration in the soil via improved management is reviewed in Chapter 6, by broad soil-ecological region. Possible effects of climate change on soil carbon sequestration are presented in Chapter 7. Issues of monitoring and verification of changes in soil organic carbon status, as required for implementation of Article 3.4 of the Kyoto Protocol within the commitment period(s), are presented in Chapter 8. Concluding remarks and future research directions are outlined in Chapter 9.

2 Soil Organic Matter

2.1 Favourable properties

Many research groups have studied the characteristics, fractions and dynamics of organic matter in soil, leading to a fairly wide range of definitions (e.g., Coleman *et al*., 1989; Frimmel and Christman, 1988; Greenland and Hayes, 1978; Hayes and Wilson, 1997; Kononova, 1966; Russel, 1980; Stevenson, 1982). Basically, it is accepted that the term soil organic matter (SOM) designates a 'highly heterogeneous pool that includes numerous carbonaceous compounds; these range from easily mineralisable sugars to complex and recalcitrant products of microbial transformations whose residence times vary from a few minutes to hundred of years' (Buyanovsky *et al*., 1994).

Organic matter has a stabilizing effect on soil structure; polysaccharides are involved in the formation of aggregates, and humic substances are involved in their stabilization (Hayes, 1997, p. 5). It further improves the moisture holding and release characteristics of soil, and protects soils against erosion. Organic matter essentially affects the soil's susceptibility towards 'on site' erosion, as opposed to 'off-site erosion', through its effects on surface aggregate stability, surface sealing and crusting, and soil porosity (Gabriels and Michiels, 1991). Water erosion sorts soil particles by size and weight, whereby the smallest and lightest particles) the most 'active' fractions from a biological point of view) are moved furthest, and as a result superficial organic materials in depositional areas are easily decomposable (Tiessen and Stewart, 1983; Voroney *et al*., 1981). Decaying organic matter slowly releases nutrients such as N, P, S, K, which are essential for plant and microbial growth. Some organic constituents in humic substances may stimulate plant growth under conditions of adequate mineral nutrition. Soil organic matter is also an important determinant of the water retention and cation exchange capacity of soils, particularly in coarse textured soils and so called 'low activity' clay soils (Asadu *et al*., 1997; Bennema, 1974). The generally dark colour of organic matter in topsoils can increase the absorption of solar energy. As a result, crop growth on bare soils rich in organic matter may start somewhat earlier in cool climates.

Upon deforestation or a shift in land use, the above favourable properties may be altered as a result of accelerated decomposition of a fraction of the organic matter present in soil, as well as reduced inputs of new organic matter into the soil. The often associated decrease in soil organic matter content commonly leads to structural soil degradation, causing increased compaction and decreased soil hydraulic properties, clay mobilization, increased erosion hazard, and/or enhanced losses of soil acting herbicides and pollutants adsorbed to soil organic matter (see Section 4.2). Many of these adverse, often human-induced, impacts, however, can be reduced or controlled by adoption of appropriate land management practices, thereby offering possibilities for enhanced sequestration of atmospheric CO_2 in the soil, via plant photosynthesis, and for maintaining or improving overall soil quality (see Section 4.3 and Chapter 5).

The favourable effects of soil organic matter on the physical, chemical and thermal properties of the soil and on biological activity, and thus in sustaining soil productivity and biodiversity (Table 1), may be seen as an important 'added-benefit' over direct carbon mitigation techniques that would only physically store CO_2 in the deeper subsoil (e.g., old gas fields, mines and aquifers).

Table 1 Effects of organic matter on soil productivity and biodiversity (After: Wild, 1993)

Physical	increases supply of water to crops increases aggregation of soil particles reduces negative impacts of compaction may improve drainage and hence growth of crops gives greater flexibility for cultivation
Chemical	releases N, P, S and K on mineralization retains nutrients, for example Ca^{2+}, Mg^{2+}, K^+, NH_4^+, against leaching loss chelates micronutrients, generally facilitating uptake by plants acts as a soil pH buffer reduces the hazard from heavy metal contamination
Biological	may support organisms which help to control root diseases

2.2 Processes and factors controlling SOM dynamics

Organic matter inputs

Primary production, especially the rate and quality of C transfer below ground, and soil microbial activity are recognized as the overall biological processes controlling organic matter dynamics in the soil. Overall, input rates and quality of vegetation-carbon are largely dependent on climate (especially temperature and precipitation), vegetation type and landscape, soil type, and management practices (Jastrow and Miller, 1997). Plant residues that fall on the soil as ‘fresh’ litter are gradually altered through physical fragmentation, fauna/micro flora interactions, mineralisation and humus formation (Eijsackers and Zehnder, 1990). Decomposition processes and turnover rates are largely influenced by climate, the type and quality of the organic matter, chemical or physicochemical associations of organic matter with soil mineral components, and the location of the organic matter within the soil (Jastrow and Miller, 1997; Verburg *et al*., 1995). Changes in SOM quality may be more important than changes in SOM quantity in influencing soil quality and fertility status (Clark *et al*., 1998). SOM composition can be affected by different fertilization regimes (Ellerbrock *et al*., 1999).

Mean annual mineralization in the temperate regions is about 2%, as opposed to 4-5% in the humid tropics, but the rate is more rapid with increased intensity of cultivation and is slower under sod crops (Buyanovski and Wagner, 1997). Alternatively, biomass production is highest in the humid tropics, provided nutrients are not limiting.

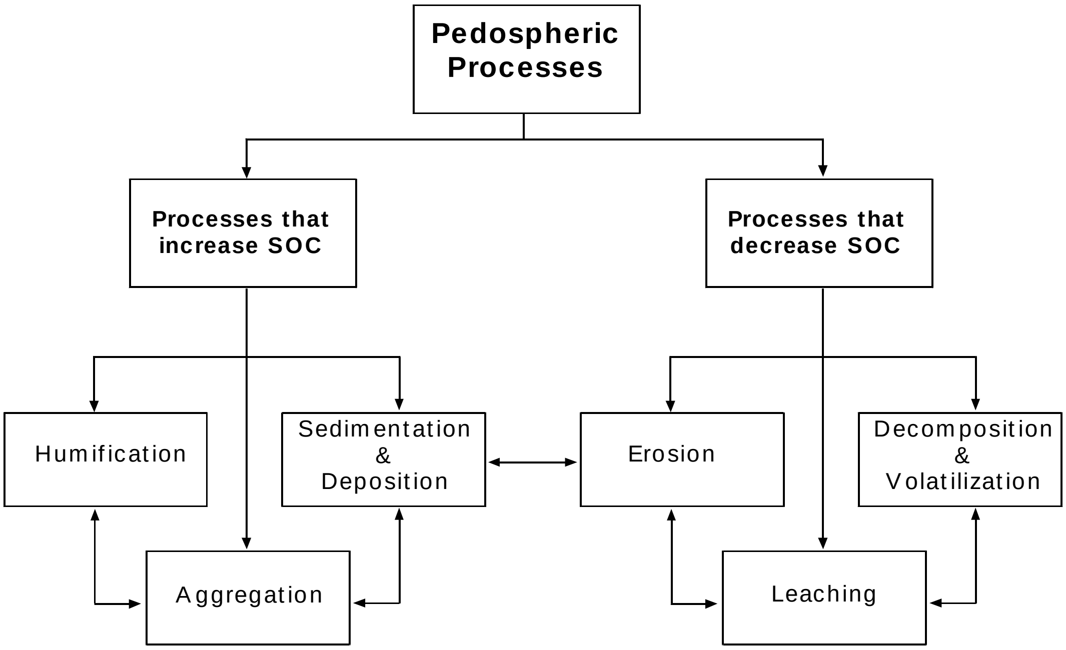

Fig. 1 Main soil processes influencing soil organic matter content (source: Lal *et al*., 1997)

SOM forming factors

Three principal processes of C sequestration in soils are humification, aggregation, and sedimentation, while processes that decrease SOC include erosion, decomposition and volatilization, and leaching (Fig.1). Soil carbon density generally increases with increasing precipitation, and there is an increase in SOC density with decreasing temperature for any particular level of precipitation (Jenny, 1941; Post *et al*., 1982). Other important environmental controls of organic matter behaviour in soil are moisture status, soil temperature, oxygen supply (drainage), soil acidity, soil nutrient supply, clay content and mineralogy (Alvarez and Lavado, 1998; Jenny, 1941; Parfitt *et al*., 1996; Russel, 1980; Tate and Theng, 1980). Main processes leading to C-emissions in upland and wetland soils are shown in Figure 2.

Changes in organic carbon content of soils have been shown to correlate with changes in the structural form and stability of soils, and the magnitude of the change in structural characteristics is often strongly dependent on soil structure (Kay, 1998). The mechanisms responsible for stabilizing SOC include physical protection, biochemical recalcitrance, and chemical stabilization (Christensen, 1996). The nature of various organo-mineral associations and their location/distribution within soil aggregates determine the extent to which SOC is physically protected and chemically stabilized (Garcia-Oliva *et al*., 1994; Gijsman and Sanze, 1998; Guggenberger *et al*., 1995; Randall *et al*., 1995), resulting in organic pools with varying input and turnover rates. Carbon decay/accretion in the surface horizon is related to biological activity, notably earthworms (Chauvel *et al*., 1999; Eijsackers and Zehnder, 1990; Linden and Clapp, 1998), varies with vegetation type (Quideau *et al*., 1998; Sanger *et al*., 1997), and organic matter and manure amendments (Atallah *et al*., 1995; Gerzabek *et al*., 1997). Manuring, for example, affects the composition and molecular size of SOM components (Ellerbrock *et al*., 1999).

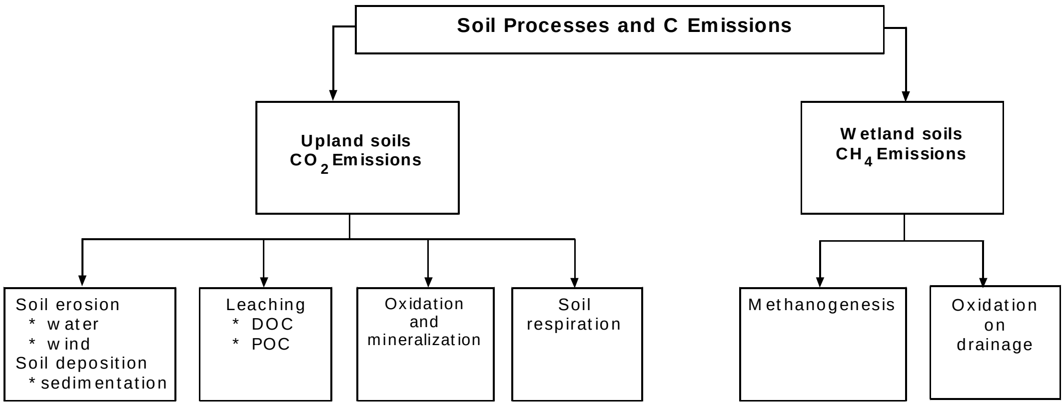

Fig. 2 Principal processes in soil affecting C-emissions from upland and wetland soils (Source: Lal, 1998)

Young organic matter is largely responsible for macro-aggregate stability (Puget *et al*., 1995). Organic carbon associated with macro-aggregates is less processed than that of micro-aggregates, as reflected by higher C:N ratios of the former (Garcia-Oliva *et al*., 1999). The active fraction of soil carbon plays an important role in determining aggregate stability and rainfall infiltration (Bell *et al*., 1999). Easily dispersed clay has lower organic contents than difficult to disperse clay (Nelson *et al*., 1999). Coarse textured soils only contain significant concentrations of physically protected organic matter within aggregates, that is mainly enriched in plant residues, as a result of which effects of land use changes on SOC are generally more apparent on coarse textured than on loamy and clayey soils (Guggenberger *et al*., 1995; Lilienfein *et al*., 1998). SOC decomposition is lower in topsoils of fine texture with a sand/clay ratio < 1, compared to topsoils of a coarse texture with a sand/clay ratio ranging from 2 to 8 (Koutika *et al*., 1999). Fine silt and coarse clay fractions contain the most refractory carbon (Feigl *et al*., 1995a). Microbial products attached to clay surfaces by a variety of physico-chemical bondings appear more stable against mineralization induced by cultivation than plant residues sequestered in aggregates (Guggenberger *et al*., 1995).

Differences in specific surface area of clay minerals — ranging from 10-15 $m^2 g^{-1}$ for kaolinite, 50-100 $m^2 g^{-1}$ for illite, and ≈ 800 $m^2 g^{-1}$ for vermiculite (Scheffer and Schachtschabel, 1984) — affect the capacity of clays to adsorb humic substances (Tate and Theng, 1980; Van Breemen and Feijtel, 1990). The resistance of clay-organic matter complexes against microbial attack decreases as follows: allophane >> 2:1 clays > 1:1 clays. Consequently, volcanic Andosols often have high organic carbon contents relative to other mineral soils under similar environmental conditions (see Batjes, 1996b; Ingram and Thingstad, 1995; Sombroek *et al*., 1993). Mean residence times for organic carbon of 2000 to 5000 yr have been reported for soils rich in allophane (Wada and Aomine, 1975).

Torn *et al*. (1997) propose the effect of mineralogy on soil carbon storage is of the same magnitude as that attributed to climate or vegetation. In selected tropical and temperate upland soils, soil mineralogy was the main determinant of the rate of carbon turnover, and organic chemical structure a secondary consideration (Meredith, 1997). Nevertheless, the organo-mineral interactions will be determined both by the nature of the mineral and of the organic component, and the preservation of certain species in certain soils and condition will be determined by these interactions (Meredith, 1997). The net rate of accumulation of organic matter depends not on the protective capacity of a soil perse, but rather on the extent to which this capacity is already occupied by organic matter (Hassink, 1996; Hassink and Whitmore, 1997).

As has been shown above, many factors influence C mineralization. Texture and soil mineralogy may be seen as ‘fixed’ factors that are not likely to change in the short-term, as opposed to factors such as temperature, moisture and substrate quality that are more directly affected by climate change (see Chapter 7). Nutrient availability will be affected through changes in mineralization upon SOC decomposition. As discussed by Verburg *et al*. (1998), the ‘fixed’ soil factors will determine the importance of the effect of the more ‘variable’ factors on C mineralization; if the rate limiting factors happen to be fixed, climate change will hardly affect C mineralization. Possible impacts of climate change on soil processes affecting SOM dynamics have been reviewed elsewhere (Bouwman, 1990; Brinkman and Sombroek, 1996; Lal, 1998; Mosier, 1998).

SOM pools

With respect to soil carbon sequestration, it is most desirable to fix atmospheric C in those pools having long turnover times. To model the cycling of C in the soil, soil organic matter must be subdivided into several compartments considered more or less ‘homogenous’ in terms of residence times. Eswaran *et al*. (1995) defined four pools based on carbon dynamics. An ‘active or labile pool’ of readily oxidized compounds, the formation of which is largely dictated by plant residue inputs (and hence management) and climate; a ‘slowly oxidized pool’ associated with soil macro-aggregates, the dynamics and pool size of which are affected by soil physical properties such as mineralogy and aggregation, as well as agronomic practices; a ‘very slowly oxidized pool’ associated with micro-aggregates, where the main controlling factor is water stability of the aggregates and agronomic practices have only little effect; and, a 'passive or recalcitrant pool' where clay mineralogy is the main controlling factor, and there probably are no effects of agronomic practices. Indicative residence times of the various C-pools, now termed 'labile', 'moderate', 'slow' and 'passive', are given in Table 2. The terms humic acids (HAs), fulvic acids (FAs), and humins are still widely used to chemically characterize fractions of *humic substances*, which may be considered to be the contents of soil organic matter (SOM) that are solubilized in base (see Hayes, 1997).

Table 2 Estimated ranges in the amounts and turnover times of various types of organic matter stored in agricultural soils (After: Jastrow and Miller, 1997, p. 22)

Type of organic matter	Proportion of total OM (%)	Turnover time (yr)	C-pool
Microbial biomass	2-5	0.1-0.4	labile
Litter	-	1-3	rapid
Particulate OM	18-40	5-20	moderate
Light fraction	10-30	1-15	moderate
Inter-microaggregate [a)]	20-35	5-50	moderate to slow
Intra-microaggregate [b)]			
Physically sequestered	20-40	50-1000	passive
Chemically sequestered	20-40	1000-3000	passive

a) Within macro-aggregates but external to micro-aggregates, including particulate, light fraction, and microbial C.
b) Within micro-aggregates, including sequestered light fraction and microbially derived C.

Figure 3 serves to illustrate the complexity of a particle size fractionation of soil organic matter, which includes both physical (size) and chemical steps. Not all the components of a given physically- or chemically-defined fraction of soil organic matter are as readily decomposable as the descriptive terminology would imply. Inert charcoal, for example, is a serious contaminant in the 'light' (< 1.6 g cm^{-3}), that is 'labile' fraction of SOM (Skjemstad *et al*., 1990). Adequate methods to experimentally establish the partitioning of soil organic matter over the different pools conceptualized in various models are still lacking (e.g., Buyanovsky *et al*., 1994; Hassink, 1995; Jenkinson, 1990; Jensen *et al*., 1997, Skjemstad, 1998 #586; Metherell *et al*., 1995; Powlson *et al*., 1996). Novel fractionation schemes that truly reflect the biological diversity of organic matter in soil thus are needed (Hassink, 1995; Johnson, 1993; Skjemstad *et al*., 1998; Van Veen, 1986).

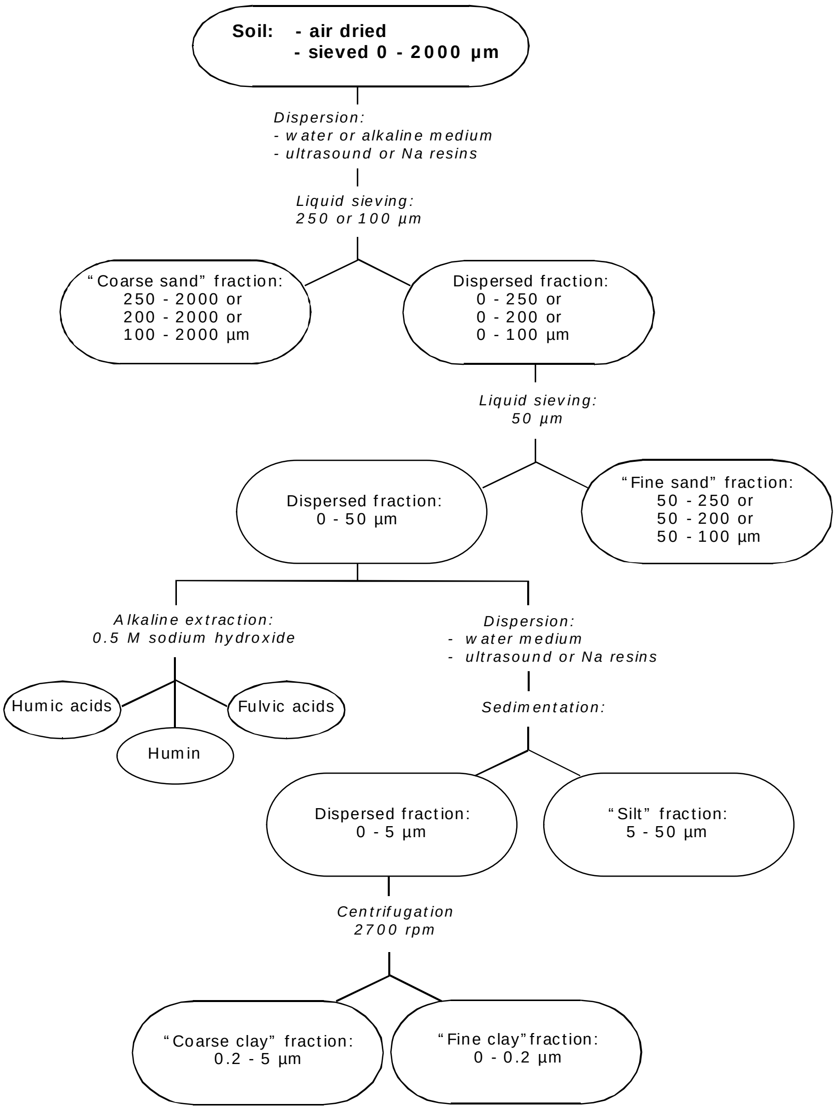

Fig. 3 Example of a combined physical and chemical fractionation procedure for soil organic matter (After: Andreux and Choné, 1993)

2.3 SOM models

Model complexity

A careful evaluation of the analytical methods and interpretation of the results is critical for process-oriented SOM models (Guggenberger *et al.*, 1995). In the absence of substantial justification for segregating SOM into different compartments based on size and chemical stability, modelling should be done with the simplest model possible (Bernoux *et al*., 1998a). A better understanding of organo-mineral associations may provide a key to better defining or quantifying the conceptual pools used by SOM simulation models (Jastrow and Miller, 1997). Whereas assumptions leading to the separation of soil C into two or more compartments increased the standard error in Bernoux *et al.*'s (1998a) study, results of Römkens and van der Plicht (1997) showed that the application of average carbon turnover rates in simulation models cannot realistically represent SOM dynamics.

Rather than making soil C models more complicated, Liski *et al.* (1998a) suggest improvement and reliability of studies on disturbance effects on soil carbon storage are best served by taking more comprehensive measurements of the actual effects of disturbance regimes on soil C. Similarly, Prentice and Lloyd (1998) pointed at the paramount importance of more comprehensive measurements against which regional C-models can be evaluated and refined. The key issue of how to obtain good parameter values for general application of a model, and its submodels, when available (measured) data are incomplete has also been raised elsewhere (Andrén and Katterer, 1997; Cramer and Fischer, 1997; Jensen *et al*., 1997). Possible reasons for differences in SOM model performance have been discussed elsewhere (Smith *et al*., 1997d; Smith *et al*., 1997e), illustrating the usefulness of long-term agroecosystem experiments and SOM networks (Powlson *et al*., 1998; Powlson *et al*., 1996; Rasmussen *et al*., 1998a).

Point-based versus spatially-oriented models

The C models described above are essentially point-based. Geographical predictions of changes in soil C stocks can be improved by relating SOC contents to available spatial climate, landform, soil and land use parameters that may influence SOC distribution, and by combining these layers into spatial models (Arrouays *et al*., 1998; Falloon *et al*., 1998; Lisksi and Westman, 1997; Paustian *et al*., 1997), especially when linked also to environmental monitoring systems (GTOS, 1995; Várallyay, 1997). Levine and Kimes (1998) give preliminary evidence that the use of neural networks with soil characterization data is promising to better understand the relationships between organic carbon and other soil properties under different scenarios and scales (see also Chapter 8). Non-parametric approaches such as the 'mollifier technique' also seem promising in this respect (Keyzer and Sonneveld, 1997).

Necessity of model evaluation

Validation of predictions from local and regional scale, ecosystem models is difficult (Oreskes *et al*., 1994; Wessman, 1992). Significant uncertainties remain in both the reliability of geographical and attribute data sets used to compute soil C stocks and to drive models (Greenland, 1995; Leemans and van den Born, 1994; Nachtergaele, 1996). Similarly, there are uncertainties in the algorithms or procedures used to scale-up ecosystem processes and estimate carbon and nutrient fluxes (Bouwman, 1999; Cramer and Fischer, 1997; Hurtt *et al*., 1998; Paustian *et al*., 1998; Potter *et al*., 1998a). These include uncertainties that are associated with the structure, and assumptions, of the applied models (Goodchild, 1994; Liski *et al.*, 1998b; Smith *et al.*, 1997e).

For use at larger scales, and for predicting future changes in SOC, calibration data such as those used by Smith (1997e) will not exist, yet a major obstacle to the confident predictive use of models in global environmental change research is the need for such model calibration (Smith *et al*., 1997d). Other uncertainties associated with the estimation and detection of global carbon fluxes, include interacting effects on plants and soils of rising atmospheric CO_2 concentrations, drought, salinity and nutrient status, as well as ozone and UV-B stresses (Mellilo, 1996; Sage, 1996; Schimmel, 1995), and inputs of nutrients from the atmosphere with dust (Chadwick *et al*., 1999). Despite the uncertainties associated with model-based projections, they are essential for providing quantified information on possible trends is soil C levels following alternate management practices and policies, especially when they also provide information about the range in uncertainty of the projections.

3 Global Soil Carbon Stocks

3.1 Soil organic carbon

On average, the upper 1 m of soil contains about 2.5 times more organic carbon than the terrestrial vegetation ($\approx$ 600 Pg C) and about twice as much carbon than is present in the atmosphere ($\approx$ 750 Pg C). Contemporary soil organic carbon mass in the first 1 m of soil is estimated at 1200-1600 Pg C (see Batjes and Sombroek, 1997), while this is 2376-2456 Pg C for the upper 2 m of soil (Table 3). The contributions of roots, charcoal, and soil organic carbon stored at greater depths should be added to these totals (Batjes, 1996b; Buringh, 1984; Liski and Westman, 1995; Sombroek *et al*., 1993), but adequate data sets for making such global assessments are still lacking.

Table 3 World soil carbon and nitrogen pools (in Pg of C and N, respectively) (Source: Batjes, 1996b)

Region	Depth range (cm)		
	0 - 30	0 -100	0 - 200
Tropical regions			
- Soil carbon			
Organic C	201 - 233	384 - 403	616 - 640
Inorganic C	72 - 79	203 - 218	-
Total	273 - 292	587 - 621	-
- Total Nitrogen	20 - 22	42 - 44	-
Other regions			
- Soil carbon			
Organic C	483 - 511	1078 - 1145	1760 - 1816
Inorganic C	150 - 166	492 - 530	-
Total	633 - 677	1570 - 1675	-
- Total Nitrogen	43 - 45	91 - 96	-
World			
- Soil carbon			
Organic C	684 - 724	1462 - 1548	2376 - 2456
Inorganic C	222 - 245	695 - 748	
Total	906 - 969	2157 - 2296	
- Total Nitrogen	63 - 67	133 - 140	

* The Tropics have been defined as the region bounded by latitude 23.5º N and 23.5º S.

Litter is not included in most calculations of soil carbon pools as it essentially lies on the surface, yet this amount can be considerable in northern/boreal and humid tropical ecosystems (Delaney *et al*., 1997;

Johnson, 1993; Vogt *et al*., 1995). The leaf litter on well drained xanthic Ferralsols occurring on forested Bereira clays in the Brazilian Amazon, for example, contained 5 to 6 kg C m^{-2} (Andreux and Choné, 1993). This amount is of a magnitude similar to the mean SOC content of 5.1 kg C m^{-2}, to a depth of 0.3 m, found for Amazonian Ferralsols (Batjes and Dijkshoorn, 1999). Forest floor mass follows a temperature gradient with accumulations in the boreal zone > temperate zone > subtropical > tropical climatic zone, with forest floor mass varying significantly with foliage life-span (Vogt *et al*., 1995), and hence rainfall regime. In general, a significantly greater proportion of total ecosystem organic matter is located in the soil in deciduous than in evergreen dominated forests, the effect of deciduous species being greater in the higher latitude forests and decreasing in the subtropical and tropical forests (Vogt *et al*., 1995). Similarly, surface residues under no-till farming should also be considered when making total C inventories and assessing C sequestration potential of croplands (Peterson *et al*., 1998). Charcoal can be an important, recalcitrant constituent in ecosystems subject to frequent fires (Sanford *et al*., 1985; Skjemstad *et al*., 1990).

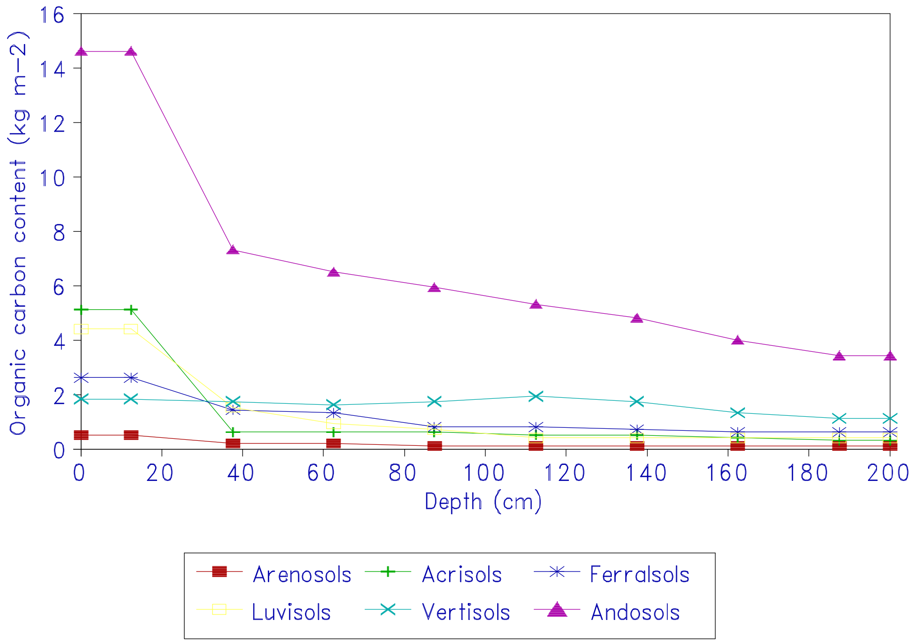

Fig. 4 Carbon density as a function of depth in selected tropical and subtropical soils (Source: Batjes and Sombroek, 1997)

Soil organic matter stored in the topsoil contributes most actively to nutrient cycling in the soil-water-plant system and to gaseous exchanges with the atmosphere, but the subsoil can also be important (Davidson *et al*., 1993; Nepstad *et al*., 1991; Trumbmore *et al*., 1995). The amount of fast-cycling carbon between 1 and 8 m depth (2-3 kg C m^{-2} out of 17-18 kg C m^{-2}) is significant compared to the amount present in the upper metre of soil (3-4 kg C m^{-2} out of 10-11 kg C m^{-2}) of deep-rooting forest of eastern Amazonia (Trumbmore *et al*., 1995). In boreal forest, soil carbon held between a depth of 1 m and the groundwater table (2.4-4.6 m; 1.3-2.4 kg C m^{-2}) represented from 18-28% of the total stock of C in the soil (Liski and Westman, 1995).

There is a great variation in the amount and vertical distribution of organic matter in boreal, temperate, and subtropical soils (see also Batjes, 1996b; Blume *et al.*, 1996 ; Kimble *et al.*, 1990; Sanchez *et al.*, 1982; Sombroek *et al.*, 1993; Tarnocai, 1998) (Fig. 4). Simple models to describe the vertical distribution of organic carbon is soil profiles have been proposed by Bennema (1974) and modified by Bernoux *et al.* (1998b).

Tables 4a and b list spatially weighted SOC pools and mean SOC densities, to a depth of 0.3 m resp. 1.0 m, computed by major Agro-Ecological Zone using preliminary data of FAO-IIASA (for details see Appendix 3). The data illustrate the large differences in organic carbon that can be stored in soils of different agro-ecological zones (see also Chapter 6). In the first 1 m of soil, these values range from about 4 kg C m^{-2} in soils of Arid regions to 21-24 kg C m^{-2} in Polar and Boreal regions. SOC density data by individual soil unit, of the Soil Map of the World Legend (FAO-Unesco, 1974), may be found in Batjes (1996b). SOC density data by land use/cover categories of IMAGE2.1 (Alcamo *et al.*, 1998) are listed in Appendix 2, and were presented by Holdridge Life-zones by Batjes and Sombroek (1997).

Table 4a Total stocks and densities of soil organic carbon (SOC) by major Agro-Ecological Zone (Pg C resp. kg C m^{-2} for upper 0.3 m)

Agro-Ecological Zone	Spatially weighted SOC pools (Pg C to 0.3 m depth)	Mean SOC density (kg m^{-2} to 0.3 m depth)
Tropics, warm humid	91.8 - 95.2	5.2 - 5.4
Tropics, warm seasonally dry	62.7 - 66.7	3.6 - 3.8
Tropics, cool	29.1 - 31.0	4.4 - 4.7
Arid	48.7 - 54.7	2.0 - 2.2
Subtropics, with summer rains	33.5 - 35.7	4.5 - 4.7
Subtropics, with winter rains	18.3 - 20.1	3.6 - 3.9
Temperate, oceanic	19.6 - 21.7	5.8 - 6.4
Temperate, continental	121.2 - 126.5	5.6 - 5.9
Boreal	202.7 - 210.3	9.8 - 10.2
Polar & Alpine (excl. land ice)	57.0 - 63.0	7.0 - 7.8

Table 4b Total stocks and densities of soil organic carbon (SOC) by major Agro-Ecological Zone (Pg C resp. kg C m^{-2} for upper 1.0 m)

Agro-Ecological Zone	Spatially weighted SOC pools (Pg C to 1.0 m depth)	Mean SOC density (kg m^{-2} to 1.0 m depth)
Tropics, warm humid	176.8 - 182.5	10.0 - 10.4
Tropics, warm seasonally dry	121.6 - 127.8	7.0 - 7.3
Tropics, cool	55.9 - 59.2	8.4 - 8.9
Arid	91.4 - 100.	3.7 - 4.1
Subtropics, with summer rains	64.4 - 68.2	8.6 - 9.1
Subtropics, with winter rains	37.1 - 40.8	7.2 - 8.0
Temperate, oceanic	39.7 - 43.7	11.7 - 12.9
Temperate, continental	233.3 - 243.0	10.8 - 11.3
Boreal	477.6 - 495.9	23.1 - 24.0
Polar & Alpine (excl. land ice)	166.9 - 188.5	20.6 - 23.2

Note: Agro-Ecological Zones of FAO-IIASA as defined in Appendix 3. Soil carbon data from WISE (Batjes, 1996b). Global totals may differ slightly from those shown in Table 3 due to rounding.

3.2 Soil carbonate carbon

While most carbon in the soil is associated with organic matter, carbonate carbon can be significant in some soils. Calcification and formation of secondary carbonates is an important process in arid and semi-arid regions, where large accumulations of carbonate carbon may occur in the soil (Nettleton, 1991; Scharpenseel *et al*., 1995), with global estimates for the upper 1 m of soil ranging from about 720 to 1738 Pg C (Batjes, 1996a; Eswaran *et al*., 1995; Sombroek *et al*., 1993). These amounts will be substantially higher if the deeper subsoil, below 1 m depth, is considered also.

Annual sequestration of carbon in soil carbonates and caliches has been estimated at 10 Tg C yr^{-1} by Scharpenseel (1995) and at 11 Tg C yr^{-1} by Schlesinger (1997) (annual accumulation rate of 0.10 to 0.60 g C m^{-2} yr^{-1}, over 18.2 x 10^{12} m^2 of deserts globally).

The dynamics of the inorganic carbon pool in relation to land use, farming/cropping systems, and management are not fully known. Carbonate carbon in the soil is relatively stable unless it is mobilised by irrigation (Schlesinger, 1986) or by acidification associated with increased nitrogen and sulphur inputs. Changes in secondary carbonate reserves may have great effects on both phosphorus (P) availability and the biogeochemical cycle of P, notably in dryland systems where most inorganic P is present as soluble Ca-phosphates in the soil (West *et al*., 1994). The rate of $CaCO_3$ accumulation is sensitive to CO_2 concentration and water content in the soil profile, both of which are a function of plant activity in desert ecosystems (Schlesinger, 1997). With rising atmospheric CO_2 concentration, the solubility of $CaCO_3$ will increase considerably; however, the associated rise in air temperature by greenhouse forcing should reduce the solubility to a similar extent (Scharpenseel *et al*., 1995).

3.3 Uncertainties

Indicators of soil quality in relation to C-sequestration include the content of soil organic and inorganic carbon, aeration porosity, aggregation and mean diameter of aggregates, available water capacity, cation exchange capacity, electrical conductivity, soil bulk density, and soil biodiversity (Lal *et al*., 1998b, p. 25). The relative importance of these indicators will vary among soils, so that site-specific information is needed for quantitative assessments. Accurate calculations of the world soil C reserves and SOC sequestration potentials are complicated by a number of factors (see Batjes, 1996b; Buringh, 1984; Eswaran *et al*., 1995; Sombroek *et al*., 1993). These include: (a) the limited reliability of areas occupied by different kind of soils; (b) the limited availability of reliable, complete and uniform profile-attribute data for these soils; (c) the high spatial variability in carbon and nitrogen content, stoniness and bulk density of similarly classified soils; (d) the limited comparability and precision of analytical methods used in different laboratories, (e) the effects of climate, relief, parent material, vegetation, land degradation and land-use, and, (f) limited knowledge about processes and properties in relation to C dynamics. At the global level, issues (a) and (b) are currently being addressed by updating the information held on the Soil Map of the World (FAO, 1995; FAO-Unesco, 1974-1981) in the 1:5,000,000 scale Soil and Terrain Digital Database project (Nachtergaele, 1996; Oldeman and van Engelen, 1993; Sombroek, 1990; van Engelen and Wen, 1995). A harmonized data set of globally distributed profiles has been compiled in the framework of the World Inventory of Soil Emissions (WISE) project (Batjes, 1997; Batjes and Bridges, 1994), providing primary soil attributes for further work by the Global Soil Data Task of IGBP-DIS (Scholes *et al*., 1995) and the update of the edaphic component in FAO's AEZ study (Batjes *et al*., 1997). As shown by the WISE project, issue (d) remains of critical importance for the development of consistent global soil databases (see Pleijsier, 1989; Rousseva, 1997; Van Reeuwijk, 1998; Wösten *et al*., 1998). This points at the need for internationally recognized reference soil laboratories, but funding for this type of work is often limited. Many groups are currently working on the process side (e.g., Hassink and Whitmore, 1997; Lal *et al*., 1997; Paustian *et al*., 1997; Powlson *et al*., 1996; Smith *et al*., 1997d; Verburg *et al*., 1998).

4 Management-induced Changes in Soil Organic Carbon Stocks

4.1 General

Due to the relatively large size and long residence time of soil organic carbon, soils are a potentially important, natural sink for carbon released to the atmosphere by fossil fuel combustion. The soil forming factors, notably climate as well the local biological activity in which man is often a predominating factor, control the amount of soil organic matter that corresponds with equilibrium conditions in a certain natural ecosystem or agro-ecosystem. Many (agro)ecosystems in the world are not in a steady state, but they accumulate dry matter during a number of years after which they are disturbed by fires and other drastic events, as a result of which their SOC levels often show ‘tooth-like’ cycles. After each disturbance, a period of constant management is required in order to reach a new steady state. In this (newly) undisturbed soil, the organic matter content will stabilize at an equilibrium level characteristic of the ‘permanent’ soil characteristics, and land use or vegetation cover and prevailing management practices. This new equilibrium may be lower, similar or higher than the original one (Figure 5). As human disturbance, induced by inappropriate land use and soil mismanagement, has caused widespread soil degradation (Oldeman *et al*., 1991), the SOC-contents in many agricultural soils are now below their potential levels (see 4.2). This again is indicative for vast opportunities for increasing present soil carbon stocks via adapted management (see Section 4.3 and Chapter 5).

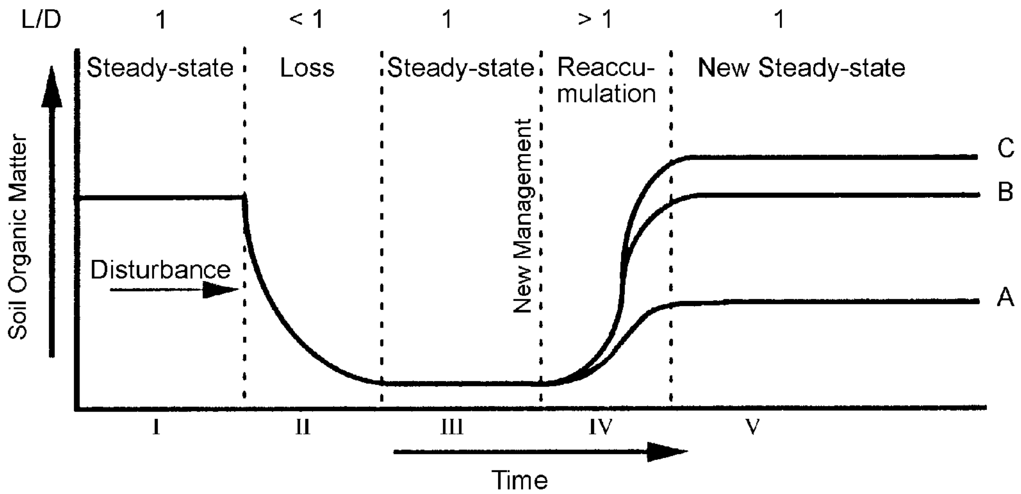

Fig. 5 at lower than original level; B- stabilization at original level; C - stabilization at above-original level; (After: Johnson, 1995b))

Generally it takes at least 15 to 30-100 years before a new organic carbon equilibrium is reached in soils (Brown and Lugo, 1990; Buringh, 1984; Buyanovski *et al*., 1997; Detwiler, 1986; Dick *et al*., 1998; Houot *et al*., 1991; Hsieh and Weber, 1984), but longer time-periods have been reported also. After 144 yr of large annual applications of animal manure (35 t ha^{-1} yr^{-1}), SOC content is still increasing at Broadbalk Wheat Experiment (Jenkinson, 1990). About 2000 yr are needed for total SOC recovery subsequent to forest fires in boreal forests (Liski *et al*., 1998a). All other factors being equal, the recovery interval is longest in northern latitudes and at higher altitudes. In addition, the rate of recovery of soil C is more rapid in wet and moist life zones, whereas soil N seemingly recovers faster in dry life zones (Brown and Lugo, 1990).

The Rothamsted data showed that in spite of crop yields of 6 to 7 Mg ha^{-1}, soil carbon in the top 23 cm has reached a steady sate only a little above the equilibrium level established for the unmanured plot (Jenkinson, 1990; Jenkinson *et al*., 1991). This observation seems to support other studies (Hassink, 1996; Hassink and Whitmore, 1997) from which it appears the net rate of accumulation of organic matter depends not so much on the ‘protective' capacity of a soil *per se*, but rather on the extent to which this capacity is already occupied by organic matter.

In the longer term, photosynthetic acclimation and growth response to increased CO_2 concentration are dependent on genotypic and environmental factors, affecting the plant’s ability to develop new sinks for C and to acquire adequate N and other resources to support an enhanced growth potential (Wolfe *et al*., 1998). Several authors (Cao and Woodward, 1998; Schimmel, 1998) have argued that the terrestrial sink for C can become saturated because photosynthesis follows a saturation function with respect to CO_2, and because plants and microbial respiration must catch up as the rate of increase in photosynthesis slows; this then would eventually reduce possibilities for incremental carbon storage to zero. Thus there is likely to be an upper limit to the biomass a forest stand, and other stands, can hold (Phillips *et al*., 1998).

4.2 Examples of historic SOC losses

There is an abundant literature quantifying C-losses associated with inappropriate management of the land, including negative impacts of erosion, deforestation, land use conversion, fire, and drainage of peatlands (Batjes, 1992; Bouwman, 1990; Dalal and Mayer, 1986; Mann, 1986; Sanchez *et al*., 1983). Selected examples are given below as an illustration.

Deforestation and changes in land use during the period 1860-1984 have been estimated to cause losses of 150 ± 50 Pg C (Bolin, 1986b), the annual losses being in the range 1-2 Pg C yr^{-1} (Bolin, 1986a; Bouwman and Sombroek, 1990) to 2-5 Pg C yr^{-1} (Houghton *et al*., 1985). Scharpenseel (1993) calculated that 30 to 60 Pg C have been lost from cultivated soils during the last 100 years. Soils of the arable lands in the former Soviet Union lost from 24 to 50 % of their carbon since the onset of agriculture (Rozanov *et al.*, (as quoted by Kolchugina *et al.*, 1995)). The magnitude of the estimated, historic C-losses vary, amongst other, due to uncertainties in acreage deforested, type of forest removed (biomass), land uses to which these forests are converted, amounts of soil C removed by water and wind erosion, and widely differing definitions of biomes, as well sampling strategies adopted (Bouwman, 1990; Greenland, 1995; Leemans and van den Born, 1994; Liski, 1995).

The quantity of organic matter in cultivated soils usually is less than in adjacent forest soils, unless improved management has been used (Gregorich *et al.*, 1995). The fraction of the soil C lost is positively correlated to the amount initially present in undisturbed soil (Davidson and Ackerman, 1993). Differences occur at the soil series level which may reflect the effects of climate and of soil pH on microbial activity (Cihacek and Ulmer, 1998; Gregorich *et al.*, 1995; Li, 1995).

North American and European types of agriculture, formerly comprising low-yielding monocultures with limited return of crop residues to soil and the use of mineral fertilizers, are generally associated with declining soil organic carbon levels (Allison, 1973; Buyanovski and Wagner, 1998b; Davidson and Ackerman, 1993; Flach *et al.*, 1997). Under such cultivation practices, soil carbon contents may decline because of a variety of processes, including the removal of topsoil by mechanical clearing, disruption of aggregates and increased oxidation of organic matter, or increased accessibility of organic matter to decomposing organisms. These processes not only reduce the total content of organic matter in the soil but also alter the fractions present, thus often reducing nutrient availability and negatively affecting soil physical properties as well (Anderson *et al.*, 1981; Wielemaker and Lanse, 1991).

The effects of land use on soil carbon content generally become negligible between a depth of 10 to 60 cm, but depths up to 100 cm have been reported also (Brown and Lugo, 1990), the wide range being attributed to differences in life zones, soil type and land use (Detwiler, 1986). Sampling by fixed depths appears to underestimate soil C losses (Davidson and Ackerman, 1993). The average loss of soil carbon (over a 1 m depth interval) after conversion of forest to cropland is about 40 to 50 per cent, of forest to grassland 20-30 per cent, and of forest to mixed cropland and grassland 35 per cent (Buringh, 1984; Detwiler, 1986; Houghton *et al.*, 1983; Schlesinger, 1986). Grassland and forest soils tend to lose from 20 to 50% of the original SOC content in the zone of cultivation, within the first 40 to 50 years of cultivation (Lal *et al.*, 1998b, p. 19). Such decreases in C storage can be attributed to reduced C inputs, as plant productivity decreases or more residues are removed, and enhanced rates of litter decay (Ellert and Gregorich, 1996; Trumbmore *et al.*, 1995). A retrospective modelling assessment of historic changes in soil carbon and impacts of agricultural development in Central USA, from 1900 to 1990, was made by Patwardhan (1998).

The rate of soil carbon loss is highest within the first 5 to 20 years following disturbance (Davidson and Ackerman, 1993; Detwiler, 1986). Schlegel and Havlin (1997), reported a decrease of 40 % to 70 % in soil organic matter content since fields were first cultivated in the early 1900s to 1960s for the Central

Great Plains, USA, associated with tillage and related soil erosion and oxidation. Tiessen *et al.* (1982) reported that organic matter levels in untilled grasslands continued to decline after 90 years, because of erosion. While deposition of eroded soil does not necessarily lead to the direct sequestration of carbon, it is likely to increase the overall sequestration of SOC leading to an accumulation of organic material which has a greater potential to be converted to the stable SOC form (Bajracharya *et al.*, 1998).

According to Mann (1986), reforestation appears to have little (negative) effects on soil carbon unless forests are converted to agriculture. Johnson (1992) reviewed the effects of harvesting, site preparation, and fertiliser treatment on soil carbon storage in forest soils, pointing at the uncertainties associated with differences in vertical and horizontal sampling intensity, and temporal trends. Most studies showed very little change in soil carbon with forest harvesting, regardless of sampling intensity or time since harvest (Johnson, 1992).

SOC in forest-to-pasture chronosequences first tends to decrease following forest clearing, after which it tends to augment again with increasing pasture age (Koutika *et al.*, 1997). Upon forest burning and pasture creation, a portion of the previously stable soil C pool is rendered less stable (Bernoux *et al.*, 1998c). Slash-and-burn affects SOC dynamics during burning, at which time fire intensity is the key factor, and during the growing season when the combination of seasonal storm characteristics, fine root dynamics, and changes in the microbial community affect soil C stabilization through soil macro-aggregate destruction (Garcia-Oliva *et al.*, 1999). Slashing-and-burning did not destroy macro-aggregates (> 250 µm), but the amount of SOC associated with macro-aggregates decreased by 32% due to combustion during burning (Garcia-Oliva *et al.*, 1999). Fire can also disrupt soil aggregate stability by changing the chemical nature of soil organic matter (Garcia-Oliva *et al.*, 1999). Burning of vegetation often has a short-term positive effect on soil biological processes by releasing nutrients and causing an increase in surface pH, through the formation of oxides of bases, and an increase in moisture content of the surface soil because of cessation of transpiration, so that microbial activity increases, and an increase in moisture content of the surface soil because of cessation of transpiration, so that microbial activity increases. Alternatively, the formation of recalcitrant coal-like substances upon burning may inhibit SOM decomposition (Humphrey *et al.*, 1987; Skjemstad *et al.*, 1990), whereas changes in microclimate may accelerate it.

Short fallows, in shifting-cultivation, will result in a fast decline and low equilibrium of soil C levels, reducing the potential yields and limiting farmers choice for other land use options which may become available with better markets. Generally, the carbon content of soils used by shifting cultivators returns to the level found under primary forest about 35 years after abandonment (Detwiler, 1986). With increasing population density, however, the length of the ‘restoration cycle’ will rapidly drop below this recommended level, as a result of which these systems will gradually degrade.

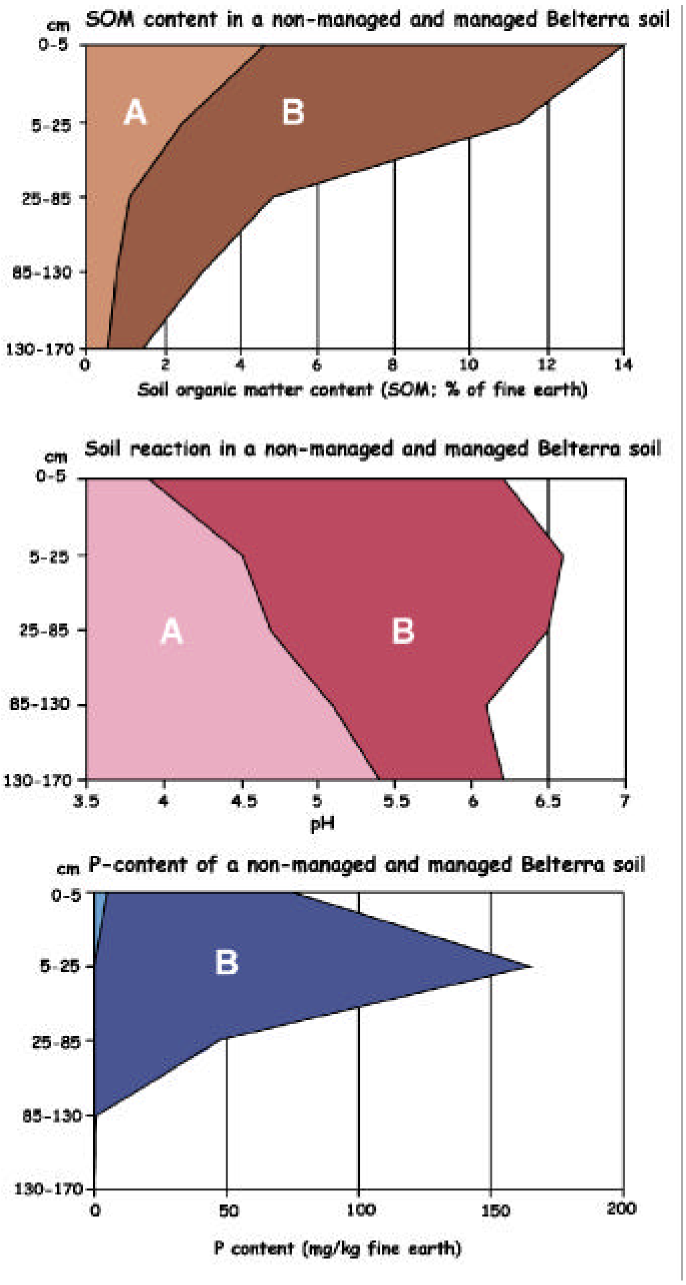

Fig. 6 Positive effects of prolonged application of 'farmyard' manure on properties of a non-managed (a) versus a managed (B) Belterra soil from Brazil (Based on data from: W.G. Sombroek, J.H. Kauffman, and O.C. Spaargaren)

4.3 Examples of increased SOC contents

Like for carbon losses, many historic examples of enhanced carbon sequestration are available from the literature. Soils of the Rothamsted Classical experiments, for example, which have received 35 t ha^{-1} farm yard manure (FYM) annually since their start in the 1840s and 1850s, now have about 3.5 % C compared to 1 % C initially (Johnston, 1991). Between, 1942-1967 annual applications of 37.5 t ha^{-1} of FYM to the sand loam at Woburn increased organic carbon from 0.87 to 1.64 % C; doubling the FYM input gave an increase to 2.26 % C. However, the soil humus content began to decline rapidly when the FYM treatment was no longer applied. In the first 5 years, soils which had received the single and double FYM lost 0.17 and 0.41 % C, respectively (Johnston, 1991), showing that some of these increases are not of a long-lasting nature since some of the organic matter was still present in a labile form. This is contrary to what is the case for the 'Terra Preta-do-Indio' found throughout the Amazon Basin (Sombroek, 1966), ancient cultivated sites in the Colca valley of Peru (Sanford *et al*., 1985), and 'Plaggen' soils in North-Western Europe (Bridges and de Bakker, 1998), in which most of the soil carbon is present in a stable form. Figure 6, may serve to illustrate the beneficial effects on soil fertility associated with increased SOM contents of selected Terra Preta-do-Indio soils.

4.3.1 Forest soils

To sequester 1 tonne of C from the atmosphere it is necessary to produce about 2.2 tonnes of wood (Chatuverdi, 1994). Compared to the cool temperate and boreal zone, carbon sequestration by trees is much faster in the tropical belt due to favourable climatic conditions. The growth rate and hence carbon sequestration potential of forests diminishes as trees approach maturity. Forests accumulate atmospheric carbon in young and middle age, but the rates decline to zero as stands mature. The greatest declines with senescence occur in boreal and in cold temperate forests, and in very infertile tropical forests in Amazonia. Current hypotheses for declining above ground net primary production (ANPP) in ageing forest stands, as reviewed by Gower *et al.* (1996), include: (a) an altered balance between photosynthetic and respiring tissues, (b) decreasing soil nutrient availability, and, (c) increasing stomatal limitations leading to reduced photosynthetic rates. Sapwood respiration costs are important only if sapwood biomass continues to accumulate as stands mature. Changes in belowground C and N cycles during stand development are so poorly understood that is difficult to speculate how this important flux may change during stand development (see Gower *et al*., 1996). The long residence time of particulate organic C from high latitude forest soils, however, provides empirical evidence that if fluxes of C from vegetation to the soil increase, as a result of global change, these soils have the capacity to act as a C sink on decadal time scales (Bird *et al*., 1996).

A full inventory of forest demographic changes is needed to estimate the terrestrial carbon sinks caused by forest regrowth (Shvidenko *et al*., 1995). Estimates of potential land available for re- and a-forestation and management of forests have been reviewed elsewhere (Brown, 1996; Ciesla, 1995; Dabas and Bhatia, 1996). The uncertainties in mitigation potentials presented in these studies are likely to be high, particularly with respect to the land area available for forestation projects, the rate at which deforestation could be slowed, and the differences in definition used (Brown, 1996; Ciesla, 1995; DiNicola *et al.*, 1998; Nabuurs *et al.*, 1999).

4.3.2 Forest to pasture conversion

Forests contain from 20 to 100 times more biomass C per unit area than agricultural land so that conversion of forest to cropland or pasture generally reduces the amount of organic carbon on land (Houghton, 1990). Effects of forest harvesting on SOC stocks are associated with five main processes: leaching loss of organic matter from the solum to streams; respiration; decay of root biomass; physical disturbance and mixing induced by logging machinery; and, accelerated decomposition and translocation of organic matter from the O horizon to the upper mineral soil (Johnson *et al.*, 1991).

Choné (1991) studied the effects of burning and deforestation on soil organic matter in the topsoil of two managed Oxisols in Manaus. Under natural forest, the organic carbon content reached maximal values of 28 t ha^{-1} in the 0-3 cm layer, 62 t ha^{-1} in the 3-20 cm layer, and thereafter decreased rapidly with depth. Burning removed about 4 t ha^{-1} of organic carbon, chiefly from the 0-3 cm soil layer, but this loss was recovered after one year of pasture (Choné *et al.*, 1991). Decreases of about 8 t ha^{-1} after 1 year and 28 t ha^{-1} after 2 years were observed in the 3-20 cm layers, probably because the humification of grass root residues did not balance the decomposition of pre-existing soil organic matter from natural forest. After 8 years of pasture, however, the initial organic carbon content of surface soil had been almost entirely restored, including the 3-20 cm layer. The introduction of graminaceous vegetation in succession to forest may stimulate microorganisms during the initial years as a result of which they also decompose residues and even humified material from the original forest (Choné *et al.*, 1991).

Clear-felling of tropical rainforest soils and subsequent cropping with pastures caused an initial decrease in microbial biomass, followed by an increase in 1-2 yr old pastures and a subsequent decrease in 8-yr old pastures (Bonde, 1991). Andreux and Choné (1993) detected the presence of C originating from pasture, subsequent to forest clearing, up to a depth of 0.4 m, with a predominant input in the upper 0.2 m. Higher net mineralization and nitrification rates in Amazonian forest suggest a higher potential for NO_3^--N losses, either through leaching or gaseous emissions, from intact forest compared with established pastures (Neill *et al.*, 1995). Clearly, the type of pasture management adopted will be critical for the observed changes in soil C stocks (Fukisaka *et al.*, 1998).

In some Brazilian Oxisols, a slight positive effect of the pasture was observed 17 years after clear-felling (Koutika *et al.*, 1997). Most of the poorly humified residues deriving from pastures ended up in newly formed aggregates in the organic-rich layers (0-0.4 m). However, it appeared that the increased incorporation of these residues (with increasing pasture age) did not improve soil aggregation, nor improve total porosity and micro-aggregate fabric porosity in these Oxisols (Koutika *et al*., 1997). Generally, however, a slight increase in SOM in the topsoil of Low Activity Clay (LAC) soils, and the associated increase of the cation exchange capacity, contribute to maintaining soil physical structure and soil nutrient retention (Asadu *et al*., 1997). Most of the negative charges responsible for the CEC of LAC soils, arise from the dissociation of carboxyl groups in organic matter molecules (Russel, 1980).

In other studies, forest conversion to pasture caused appreciable increases in soil pH and exchangeable cation content, at least until nine years after pasture installation in Rondonia. Soil carbon contents to 30 cm depth in 20-year old pastures were 17 to 20% higher than in the original forest sites. Soil carbon derived from pasture increased with time and it represented 50% of total soil carbon in the top 30 cm after 20 years of pasture (de Moraes *et al*., 1996). Similarly, Feigl *et al.* (1995a) reported that the conversion of forest to pasture can increase carbon and nitrogen stocks, and also the overall quality of surface soils. After about 80 years, about 93% of the C in the least-humified fraction of the top 10 cm of soil was of pasture origin, while in the most-humified fraction this was 82%.

In a study encompassing mature and secondary forests and agricultural sites in three subtropical life zones of the US Virgin Islands and Puerto Rico, conversion of forests to pasture resulted in less soil C and N loss than conversion to crops (Brown and Lugo, 1990). The entry of new organic C derived from pasture (upon clearing of natural forest) proceeded mainly from the soil surface, suggesting that the contribution of aerial parts of pasture was higher than that of root deposits during the initial period (Choné *et al*., 1991). After 8 yr of pasture, however, there was a great contribution of carbon from both the aerial parts and root deposits of the pasture, and soil carbon content was then again close to that observed under natural forest.

4.3.3 Grasslands

Overgrazed degraded pastures, which dominate the landscape in cleared areas of Pará (Brasil), lose C as plant productivity decreases (Trumbmore *et al*., 1995). Managed pastures, however, have the potential to increase carbon storage in surface soils. Maintaining grass productivity and root inputs to the soil is the key to maintaining or even increasing soil C stocks (Trumbmore *et al*., 1995), for example through application of phosphorus and use of productive grasses, possibly in combination with leguminous crops. In some studies, grasses were more effective in carbon sequestration in soil than leguminous cover crops (Lal *et al*., 1998a).

The average soil organic matter content under grasslands (under Dutch conditions) was shown to be 2 to 5 times higher than that of cropland, and to increase with clay and silt content (Korteleven, 1963).

Improvement in soil structure and micro-aggregation is an important factor for C sequestration by grasses (Lal *et al*., 1998a). Deep-rooted tropical grasses can sequester significant amounts of organic carbon deep in the soil, and may immobilize 2 Pg C yr^{-1} globally in biomass production (Fisher *et al*., 1994). Similarly, in the US eastern gamagrass *(Trypsicum notatum* Flugge) roots with aerenchyma and tolerance to acidity are able to grow to depth of at least 2 m in acidic clay-pan soils, which restrict roots of most crops to the upper 0.6 m. This deep rooting increases C deep in these soils where it decomposes slowly, also because it is often below the watertable (Kemper *et al*., 1998). Grass-based cropping systems, on sloping lands, have erosion control as a major side benefit over forages (Lal *et al*., 1998a).

Elevated atmospheric CO_2 concentrations cause a greater increase in carbon cycling than in carbon storage in grassland soils (Hungate *et al*., 1997). The increase in carbon allocation to labile pools below ground suggests that the net carbon balance obtained in short-term CO_2-enrichment experiments tend to overestimate the potential for grasslands to sequester carbon in soils in the long-term (Hungate *et al*., 1997). Elevated atmospheric CO_2 results in changes in soil C dynamics in agro-ecosystems that are crop species dependent (Torbert *et al*., 1997b).

Soil C storage in native grasslands in the Northern Great Plains of the US on average was 1.3 kg C m^{-2} higher than for the associated cultivated soils; this value reflects the generally higher bulk densities and lower organic C values for cultivated soils (Cihacek and Ulmer, 1998).

In Norway, a rotation with 4 years ley and 2 years spring grain resulted in 12 Mg C ha^{-1}, to a depth of 20 cm, higher soil C contents in 37 years as compared to the all-grain rotation, corresponding to a sequestration rate of 325 kg C ha^{-1} yr^{-1}. An additional sequestration of 100 kg C ha^{-1} yr^{-1} was obtained by each of the following agricultural practices: ploughing in straw; liberal application of N fertilizers, and utilization of FYM in crop production (Singh *et al*., 1998).

4.3.4 Arable land

Volume 47 of *Soil & Tillage Research* was devoted to tillage and crop management impacts on soil carbon storage, focussing largely on major agricultural regions within North America, providing important input for this section. Two fundamental management-related axioms of SOM dynamics are that SOM increases with increasing C inputs and decreases with physical disturbance of the soil, although the quantitative relationships are not well characterized and vary depending upon the situation (Paustian *et al*., 1998).

Long-term agroecosystem experiments (LTAE) in relation to assessing agricultural sustainability and global change were reviewed by Rasmussen and Oartob (1998a). Most of these studies were started to answer emerging questions about the nutrient requirements of crops in the early 1800s. In the past decades, several projects and networks were initiated to evaluate the use of LTAE data for measuring agricultural sustainability and environmental change (Paul, 1997; Paustian *et al*., 1998; Powlson *et al*., 1998; Powlson *et al*., 1996; Rasmussen *et al*., 1998a). Most of these long-term experiments are located in the temperate zone, with only few sites in tropical climates and developing countries.

Janzen *et al.* (1998) studied results from over 20 individual long-term studies in Alberta and Saskatchewan, concluding that declining C levels has abated and that gains in the order of 3 Mg C ha^{-1} can be achieved by improved practices including decreasing bare fallow, reducing tillage intensity and increasing use of perennial vegetation. Similar to Larionova *et al.* (1998) and Ryzhova (1998) in Russia, Janzen *et al*. (1998) related the potential for soil C storage across different soil and climate zones to initial C levels and potential Net Primary Production. In general, soil organic matter levels of boreal forests are about 2-3 times more sensitive to input and decomposition of plant debris than to decomposition of humus (Ryzhova, 1998). In subboreal grassland ecosystems, the sensitivity of humus decomposition increases and becomes close to the sensitivity shown by organic matter levels to variations of productivity and decomposition of plant debris.

Peterson *et al.* (1998) considered long-term studies in the great Plains region of Colorado, Nebraska, North Dakota, and Texas. In this region, the traditional cropping system is wheat-summer fallow. By implementing improved management practices, Peterson *et al.* (1998) emphasize the synergism between the use of no-till and the adoption of more intensive crop rotations with reduced or eliminated summer fallow. No-till not only reduces soil disturbance and thus decomposition but also enables more water conservation, which allo for more intensive crop rotations hence greater C inputs.

Huggins *et al.* (1998b) studied eight long-term sites on soil originally under tallgrass prairie (USA), involving different crop rotations and different fertilizer and manure application rates. The region has a strong decomposition environment but with a large capacity for C sequestration due to large historic losses of soil C. Soil C levels were closely related to C input rates, as crop residues and manure, and displayed a linear relationship between input and soil C at all but one of the sites under consideration.

In Australia, management practices designed to maximise both C inputs and maintain a high proportion of active C are seen as essential steps towards developing more sustainable cropping systems (Bell *et al*., 1999). However, where rainfall intensities are very high, for example along the wet tropical coast, or management options are limited by lack of irrigation, maintenance of soil surface cover by crop canopies (perennial crops or pastures) or crop residues during periods of high erosion risk will be essential (Bell *et al*., 1999).

Dick *et al.* (1998) studied impacts of agricultural management practices on C sequestration in forest-derived soils of the eastern Corn Belt, USA. Annual organic C input and tillage intensity were the most important factors affecting C sequestration. The impact of rotation on C sequestration was primarily related to the way it altered total inputs of C. The removal of above-ground plant biomass and use of cover crops were of lesser importance. The most rapid changes in SOM content occurred during the first five years after a management practice was imposed, with slower changes occurring thereafter. Although certain management practices, such as no-till increased the soil's ability to sequester atmospheric CO_2, Potter *et al.* (1998b) and Dick *et al.* (1998) concluded the impact of this sequestration will be significant only when these practices are used extensively on a large percentage of cropland and when the C-building practices are maintained.

Crop yields are an important determinant of C input into the soils and ultimately of soil organic C levels. Reilly and Fuglie (1998) analysed yield trends for 11 major crops in the US for the period 1939-1994. The observed yield growth increases, of up to 3% per year under an exponential model, could lead to substantial increases in SOC in USA cropland if crop residues are retained. They also pointed at the great uncertainties associated with the extrapolation of yield data as well as those associated with possible changes in socio-economic driving forces. Similarly for England and Wales, Skinner *et al.* (1998) observed the significant reduction in soils having low organic matter contents may be the reflection of the general rise in crop yields which accompanied the survey period (1969 - 1985), with larger crops leaving larger crop residues to enter the soil each year. Similarly, in experiments (1985-1990) with N fertilization in intensively managed grasslands in the Netherlands, soil C and N contents were considerably higher under grazing conditions, which return more organic materials to the soil, than under mowing conditions (Hassink, 1996). With reference to 236 grazing experiments worldwide, Milchunas and Lauenroth (1993) however concluded that soil organic matter displayed both positive and negative changes in response to grazing.

Van Meirvenne *et al.* (1996) studied changes in SOM in Belgium soils during the last 40 years, revisiting 983 locations that were still used as arable land. Taking into account the increased ploughing depth, a significant average increase of the SOC content of 0.2% was found. Expressed on a mass basis, SOC in the topsoil rose by 9.3 t ha^{-1} on average, representing an increase of 25% over 40 years. These values are comparable with those observed upon conversion of arable land into grassland for 2 to 3 decades (Van Meirvenne *et al.*, 1996).

Hendrix *et al.* (1998) considered long- and short-term data sets for three research sites in the southern Appalachian Piedmont of Georgia having similar climate but differing in soils and management histories (i.e., long period of intensive cultivation and erosion). Intensive cultivation resulted in no observable change in total C contents at the end of 3 yr, but at the end of 16 yr there were 40% and 18% declines in C in conventional tillage (CT) and no-tillage (NT) at one of the sites, whereas no significant changes in either NT or CT were observed for the other site, possibly due to a higher content at the later site. Knowledge of the size and nature of organic pools at equilibrium is needed to estimate the ultimate fertility and C sequestering capacity of southern Piedmont soils (Hendrix *et al.*, 1998). Based on the review by Buyanovski *et al.* (1998a) at least 32 Tg C was sequestered annually during the last 40-50 yr in the USA.

In eastern Canada, the cultivated and forage sites had denser soil profiles that the forest sites studied (Carter *et al*., 1998). Based on an equivalent soil mass, to accommodate for differences in bulk density, the paired forest and cultivated sites showed that cultivation decreased the mass of organic C (33%) and total N (10%) in the Podzols, but led to higher organic C (25%) and total N (37%) in the Cambisols and Gleysols under consideration. For the Podzols, use of forages increased soil stored organic C and N by 55% and 35%, respectively. Soil microbial biomass was greater in the forested than in the cultivated soil, but the ratio of soil organic C as microbial biomass C (1.3% to 1.6%) was similar. The highest (2.1%) ratio was found for the forest soils, while it was 1.3% in the corresponding cultivated soil, suggesting that organic C was continuing to increase under the forest soils. Carter *et al.* (1998) concluded that, overall, soils in eastern Canada have a relatively high potential to store organic matter.

Powlson *et al*. (1998) describe the GCTE global Soil Organic Matter Network (SOMNET) before focussing on the European network of long-term sites. Their calculations, based on only two long-term experiments, suggest that amendment of arable soils with 10 Mg ha^{-1} of organic manure could lead to an increase in current total European soil C stock to 230 cm depth of about 4.8% over 90 yr, a scenario with limited potential for sequestering C. Similarly, afforestation through natural woodland regeneration of 30% of current arable land (surplus to requirement by 2010) could lead to an increase in current total European soil C stock of 12.4 % over 100 yr. This is equivalent to 43 Tg C yr^{-1} or about 4% of anthropogenic CO_2-C emissions for Europe. When Powlson *et al.* (1998) included temporary C storage in standing woody biomass in their scenario, the amount of C sequestered was quadrupled and could account for 15% of Europe annual CO_2-C emissions, or about 3% of annual global anthropogenic CO_2-C emissions. In order to refine these exploratory, regression-based studies, which indicate some potential for soil C sequestration in Europe, it is necessary to perform a more detailed analysis taking account of actual distribution of land-use with respect to soil type, as well as obtaining better estimates of changes in soil C, for example by using more than 2 long-term experiments (see also Smith *et al*., 1997b; Smith *et al*., 1997c; Smith *et al*., 1998).

Application of fallow and mineral fertilizer without N led to a significant decrease of soil organic matter in a long-term experiment (1956-1993) on a clay loam at Ultuna, Sweden, while green manure maintained SOM content, and animal manure and pea increased SOM content significantly. The stable portion of the added organic materials after 37 years of continuous input was 12.8% for green manure, 27.3% for animal manure and 56.7% for peat (Gerzabek *et al*., 1997). High fertilizer rates to compensate for nutrient removed by crops/biomass sequester more carbon in soil than low fertilizer rates (Lal *et al*., 1998a). As an effect of increasing SOM content under winter wheat applied with FYM resp. mineral fertilizers over long periods, the amount of N needed for maximum economic yields can be diminished significantly (Csathó and Árendás, 1997).

Rotations showed less negative effects on SOC over 100 years, at Sanborn Field, compared with non-fertilized continuous cropping (Buyanovski *et al*., 1997). In some cases, rotations had the same influence as the annual application of manure or mineral fertilizers. Soil effects associated with rotations were a reduced soil erosion and a reduced depletion in soil nutrients, especially phosphorus at Sanborn Field. Use of organic residues with low C-to-N ratios, as occurring in legume-based cropping systems, combined with greater temporal diversity in cropping sequences significantly increases the retention of soil carbon and nitrogen (Drinkwater *et al*., 1998).

Long-term crop rotations alter soil biological (CO_2 production, denitrification capacity, and microbial biomass), chemical (soil organic C and soluble organic C), and physical (soil structure) properties. Long-term fertilization also affected all of these properties except for soil organic C and wet aggregate stability in the study of Drury *et al.* (1998).

Rasmussen *et al.* (1998b) cite two weaknesses associated with using long-term studies to determine C sequestration potential, that is uncertainty in differentiating C losses from oxidation versus erosion and limited knowledge of C changes occurring at depth (< 0.3 m). The formation of an international network to set protocols for Long-Term Agroecosystem Management (LTAE) and data interpretation are considered to be invaluable for accurate projection of future global change (Rasmussen *et al*., 1998a). In order to be most useful, long-term experiments (see Powlson *et al*., 1998; Powlson *et al*., 1996) should be combined with adequate spatial databases and environmental monitoring systems (GTOS, 1995; Várallyay, 1997), to explain changes in the main controlling factors of soil organic matter formation and decomposition.

4.3.5 Peri-urban and urban land

Peri-urban soils

Few data appear to exist on carbon sequestration in peri-urban lands that are being supplied with 'night-soil' (cf. Shanghai area), or those in Oases as occurring nearby Damascus, and other old agricultural soils along the silk-road in Tashkent (W.G. Sombroek, *pers. comm.*). Studies of these soils would be very useful to help deciphering the processes and mechanisms that lead to sustained sequestration of organic carbon in the soil, notably complex organo-mineral interactions as modified by the presence of human-supplied phosphorus-compounds. In this respect, a study of the organic carbon held in a stabile form in old indian settlements in South America (Sandor and Eash, 1995; Sombroek, 1966), selected Anthrosols of China, and old 'plaggen' soils in western Europe (Bridges and de Bakker, 1998; Sombroek, 1995) is recommended, as it can provide essential insights into the crucial issue of sustainable sequestration of carbon in the soil through improved management.

Urban soils

Increasingly large areas of the world are being affected by urbanization (Bouma *et al*., 1998; Groffman *et al*., 1995). Urban areas are distinct from more natural environments in several factors that influence biogeochemical processes. Important environmental changes of biogeochemical importance that may affect carbon storage and trace gas fluxes in urban areas, include increases in temperature, enhanced deposition of N, metals and organic compounds, increased concentrations of atmospheric pollutants, and the introduction of non-native species and increased physical disturbance (Groffman *et al*., 1995). Possibilities for carbon sequestration in urban forestry are briefly mentioned by DiNicola *et al.* (1998). 'Intensively managed' systems, such as urban lawns with trees, should have a high potential for increased carbon sequestration associated with high uses of fertilizers and irrigation. Ancillary benefits from urban forestation might induce local cooling effects and water retention that would reduce emissions from fossil fuel use (Reichle *et al*., 1999). For the longer-term, the preliminary data of Groffman (1995) suggest that urban sites should sequester and store more C than rural forest sites (see also data for 'Major cities' in App. 2).

Restored and other industrial soils

Malik *et al.* (1999) observed organic matter accumulated at the surface (0-7.5 cm) of restored soils, the contents being greater than for undisturbed open-cast coal sites after 21 years. However, this increase in total organic matter did not result in proportionate increases in carbohydrates and microbial biomass, nor did aggregate stability increase markedly.

Application of coal combustion products, such as fly- and bed-ash, at rates exceeding liming requirement may cause considerable degradation of organic N in soils, while bed ash, when applied at rates > 20 g kg^{-1} soil, can cause substantial mobilization of soil organic C (Stuczynski *et al*., 1998).

Based on global data on the generation of municipal solid waste, and measured carbon storage factors, Barlaz (1998) estimated global carbon sequestration from burial of municipal solid-waste to be at least 118 million ton per year.

5 Management Options for Increasing Soil Organic Carbon Stocks

5.1 General

Topsoil management, soil water conservation and management, soil fertility regulation, and erosion control are all important aspects of carbon sequestration in soil (e.g. Carter and Hall, 1995; Lal and Kimble, 1997). Basically, crop management systems should be aimed at increasing humification and increasing the passive fraction of organic carbon in the soil, as well as reducing C emissions. Clearly, selection of the most appropriate management practices for increasing SOC content will be soil and eco-region specific. Societal and cultural values are key factors determining the farmers acceptance of any new proposed measures dictated by policy. The most appropriate options for soil management should be selected on the basis of their effects on agronomic productivity, environmental quality, and profitability (Lal *et al*., 1998b).

With reference to Fig. 5 and Kern and Johnson (1993), one could suggest three management options for reducing greenhouse gas emissions and for managing soil organic matter, being those aimed at: (a) maintaining existing levels of soil organic matter; (b) restoring depleted SOM levels, for instance in degraded soils; and, (c) enlarging SOM pools above their historic carrying capacity, keeping in mind that these increases may be of finite magnitude and duration (Hassink, 1996; Hassink and Whitmore, 1997). Soils most depleted in soil organic matter, that is degraded soils, may have the highest potential for increased C sequestration under appropriate management (see Chapter 6).

Many researchers have considered land management options for enhanced C sequestration in the soil (Table 5; Appendix 1). These generally include a clever combination of:

1) Tillage methods and residue management (e.g., conservation tillage; cover crops; mulch farming)
2) Soil fertility and nutrient management (e.g., macronutrients; micronutrients; strengthening nutrient cycling mechanisms to minimize losses)
3) Water management (e.g., supplemental irrigation; surface and subsoil drainage; soil-water management; water harvesting)
4) Erosion control (runoff management with terraces etc.; vegetative barriers; soil surface amendment and mulch farming).
5) Crop selection and rotation.

Table 5 Land use and soil management strategies to sequester organic carbon in the soil (After: Lal and Kimble, 1994; Lal *et al*., 1998b)

System	Cultural practices
1. Land use	Afforestation; permanent crops; improved pastures with low stocking rate; multiple cropping systems; land restorative measures (e.g. use of chemical fertilizers, planted fallows, erosion control); conversion of marginal agricultural land to grassland, forest or wetland; restrict agricultural use of organic soils; restore wetlands; intensification of prime agricultural land (e.g. erosion control, supplemental irrigation; soil fertility management; improved crop assortment; reduction of fallow)
2. Farming systems	Ecologically compatible farming systems with high diversity (e.g. mixed farming, agro-forestry, silvo-pastoral systems, agri-sivlopastoral systems)
3. Tillage	Conservation tillage, mulch farming, reduced intensity of plough-based systems
4. Fertility maintenance	Judicious use of fertilizers and organic amendments; improving fertilizer use efficiency; nutrient cycling through cover crops and planted fallows; enhanced biological N-fixation
5. Pest management	Integrated pest management (IPM), selective use of chemicals

5.2 Tillage methods and residue management

5.2.1 Conservation tillage

Conventional tillage includes plough-based methods, such as successive operations of ploughing or turning over of soil with a mouldboard plough, mixing with a disc plough, and pulverization with a rotovator (Lal *et al*., 1998b). Thus intensively tilled soil have a greater capacity for C mineralization and for reductions in soil organic C levels compared to less intensively tilled systems (Torbert *et al*., 1997a). Conservation tillage is a generic term implying all tillage methods that reduce runoff and loss of soil by erosion relative to plough-based (conventional) tillage (Lal and Kimble, 1997). Principal mechanisms of SOC sequestration with conservation tillage are increase in micro-aggregation and deep placement of SOC in the subsoil. Other useful agricultural practices associated with conservation tillage are those that increase biomass production, including soil fertility enhancement, improved crops and species, cover crops and fallowing, improved pastures and deep-rooted crops (Lal and Kimble, 1997; Lal *et al*., 1998b). Various conservation tillage systems, as summarized by Lal (1997), are depicted in Figure 7.

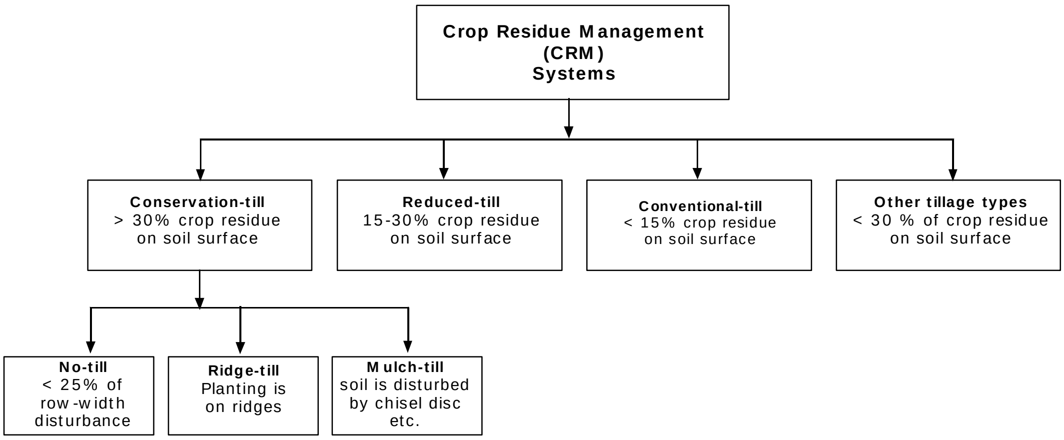

Fig. 7 Crop residue management systems and types of tillage methods (Lal, 1997)

Increased levels of crop residues on the surface of no-tillage soils will protect the soil better against the erosive forces of wind and water, and surficial runoff. The biological environment near the soil's surface (0-30 cm) with no-till is often cooler and wetter than that with conventional tillage management practices, especially mouldboard ploughing (Doran *et al.*, 1998). As result, surface soils under zero-tillage generally have a greater mass of organic C than soil under conventional tillage (Nyborg *et al.*, 1995). Overall, reduced soil disturbance and increased inputs of crop residues at the soil surface, although having little effect on the diversity of species, are conducive to increasing earthworm population in conservation tillage practices (Mele and Carter, 1999), which will have a favourable impact on soil structure and aeration. Less tillage will influence the maintenance of C in undecomposed residue and increase sequestered C in the soil (Cihacek and Ulmer, 1995). As biological activity and organic C reserves are concentrated near the soil surface with no-tillage, there is greater potential for immobilization of plant-available N in organic forms (Lyon *et al.*, 1998).

No-till fallow management increases soil water storage and reduces soil erosion potential, which are important management considerations for dryland crop production (Lyon *et al.*, 1998). Data for Vertisols with three different levels of tillage intensity, varying from no tillage to complete residue burial, suggest that in the short-term tillage systems may control soil organic C at the soil surface, while changes in plant rooting may control soil organic C storage at deeper soils depths (Torbert *et al.*, 1997b). In general, no-tillage increases particulate organic matter in surface soil (0-5 cm) at the expense of particulate organic matter stored at depth (5-17.5 cm); however, the extent of this effect varies among soils with different textures and initial C contents (Wander *et al.*, 1998).

Estimates of areas under cultivation tillage in the world are difficult to make because of lack of a standard definition, and changes in definition over time (Lal, 1997). Spread of conservation tillage in Australia, Canada, Europe, and New Zealand may be at a slower rate that in the USA, and this form of tillage may be adopted on about 50% of planted cropland in these regions by 2020. Implementation of no-till management on the 181 Mha of agricultural land, considered climatically suitable in the Former Soviet Union, would result in the sequestration of 3.4 Pg C over a 10-year period, which would represent a 10% increase in the organic carbon content in the agricultural soils of FSU (Kolchugina *et al*., 1995). In developing countries of the subtropics and tropics, however, spread of conservation tillage may be slow because crop residues are used for other purposes. The actual rate of adoption may depend on several factors, which include practices and policies on burning crop residues, availability of herbicides and appropriate machinery, market forces and other sociol-economic factors.

From 1.5 to 4.9 Pg C may be sequestered in the soils of the world if conservation tillage is adopted (Lal, 1997). While most soil physical, chemical and biological properties are improved by conservation tillage systems, the rate of change for crop yields, and SOC reserves, will differ due to climatic, soil and crop differences among sites considered. Winter-wheat fallow, for example, is probably not a sustainable production system for the Central Great Plains (USA), regardless of the fallow management system used (Lyon *et al*., 1998). In other studies, however, no-till management resulted in the greatest conservation of SOM (Lyon *et al*., 1998). A review of agricultural management impacts on soil C sequestration in eastern Canada showed that, in general management systems involving external inputs of C in the form of crop residues and farmyard manure were efficient in improving the total C content of the soil profile (Angers *et al*., 1995). However, it also showed that soil profiles under no-till generally contained less C than the conventional treatments (Angers *et al*., 1995). Inconsistent benefits of no-till practices to soil organic matter in fine-textured, poorly drained soils from Illinois have been reported also (Wander *et al*., 1998) . This is contrary to the common finding that reducing tillage intensity and in particular no-till, will lead to an accumulation of C relative to conventional mouldboard ploughing. Clearly, other management practices and environmental aspects again are important also in this context.

Annual tillage in one experiment changed the dominant plant species from grasses to annual herbs as a result of which carbon storage in the surface 15 cm of soil was reduced by 24 % (679 g C m^{-2}), 76 % of which was due to a reduction in root biomass (Richter *et al*., 1990). Relatively small changes were found in mineral soil organic C from annual tillage, i.e., about 1 mg C g^{-1} soil (Richter *et al*., 1990).

In the Argentina Pampa, no-tillage has been proposed to replace the use of the mouldboard plough to reduce soil C losses. Alvarez *et al.* (1998) propose that, after an accumulation phase, Pampa soil under no-tillage loses higher amounts of CO_2-C than under ploughing, so that the use of no-tillage would not significantly affect soil organic matter pools of the Pampa region in situations with low erosion losses. In this study, no-tillage led to about 0.7 to 1.5 Mg C ha^{-1} yr^{-1} more losses than did plough-based tillage. Taboada *et al.* (1998) observed that zero tillage is often perceived as causing shallow compaction or hardening in the soils typical of the Rolling Pampas of Argentina; they concluded such soils should be either periodically loosened by ploughing or maintained moistened through irrigation, in order to prevent the development of mechanical constraints to crops. Similarly, in parts of Australia it appears that reduced rainfall infiltration due to subsurface factors like compaction may be a more important factor than improving soil organic C for minimising erosion in cropping systems characterised by low rainfall intensity (Bell *et al.*, 1999).

In central Germany, with its 'higher rainfall', forms of plough-less cultivation, such as shallow tillage, are associated with N-deficiency symptoms in crops which may be tillage induced (Ahl *et al.*, 1998). Also long-term shallow-tillage led to a change in microbial decomposer activity towards fungi, upon termination of conventional tillage practices (Ahl *et al.*, 1998). In the drier Argentinean Pampas, Buschiazzo *et al.* (1998) observed that crops with a large N-requirement yielded less with conservation- than with conventional-tillage.

Regardless of tillage practice with wheat-fallow management at two sites near Sidney (NE), long-term (22-27 yr) losses of soil organic C from surface soil (0-30 cm) ranged from 12 to 32% (320-530 kg C ha^{-2} yr^{-1}), respectively, for no-till and mouldboard ploughing (Doran *et al.*, 1998). For these soils, Doran *et al.* (1998) observed that the decline in soil organic matter, and associated degradation in soil quality, will likely only be slowed down by increasing C inputs to soil by use of a more intensive cropping system which increases the time of cropping and reduces the time in fallow.

Other studies have shown greater pesticide leaching to groundwater under well established no-till than under conventional-till. Increased leaching in no-till is thought to be caused by preferential transport through macropores (Isensee and Sadeghi, 1996). For the native sod site studied by Mielke and Wilhelm (1998), infiltration rates were generally largest for sod and decreased as the tillage method with greater amount of soil disturbance increased. Fallow tillage systems affect soil and fertilizer N-cycling and transformations, and ultimately the N-availability to crops (Power and Peterson, 1998).

5.2.2 Residue management

Residue management, quantity and quality of biomass applied to the soil have a significant impact on soil quality and resilience, agronomic productivity, and net greenhouse gas emissions from the soil to the atmosphere (e.g., Rasmussen *et al.*, 1998a). The below-ground or root biomass, and the total biomass produced by weeds are of great importance to increasing carbon sequestration in the soil. The achievable increase in SOC content through crop residue application depends on the quantity and quality of the residue, soil properties, climatic conditions, and management. Data of Huggins *et al.* (1995) suggest fundamental differences in the dynamics of C derived from corn versus soybean residue that, in turn, may influence commonly observed differences in soil aggregate size and stability, crusting, susceptibility to erosion, and other physical characteristics.

Lal *et al.* (1997) present estimates of crop residue production for the main crops grown in the world, amounting to 3.4 Pg C per year globally. Assuming the mean C content of residues is 45%, total carbon assimilated in crop residue globally is about 1.5 Pg C yr^{-1}. Assuming 15% of the carbon assimilated in the residue can be converted to humus fractions this may lead to carbon sequestration at the rate of 0.2 Pg C yr^{-1}. In total, this would correspond with about 5.0 Pg extra C in the soil by the year 2002, corresponding with a 0.001% yr^{-1} mean increase in SOC to a depth of 1 m (Lal, 1997).

Removing a major portion of crop residue tends to reduce soil C in some climatic environments but not in others. Below-ground C input from roots and root exudates appear to be sufficient to maintain soil C in cool humid climates (Juma, 1995), but not in warm tropical or temperate semi-arid environments. Below-ground material makes up a substantial part of net annual production in most crops, approaching nearly 40% in modern wheat varieties. While returning crop residue to soil rather than removing it for other uses is (increasingly) common in most developed nations, this is often not so in developing countries where these materials are often used for animal feed, fuel or bedding (see Rasmussen *et al.*, 1998a).

Maintaining high amounts of wheat residues at the soil surface helps improve the macro-porosity of near-surface zones of soils under conservation tillage (Dao, 1998). Depending on the type of soil and climatic zone under consideration different tillage practices may be recommended in view of their differing effects on crop yields (Opoku *et al.*, 1997). For example, the acceptance of no-till systems for soybean production following winter wheat on fine-textured soils in Ontario has been hampered by soybean yield reduction due to unfavourable seedbed conditions (Vyn *et al.*, 1998). Bauer (1981) found that average organic C and total N contents were significantly higher under wheat stubble mulch than conventional tillage management. Data of Liang *et al.* (1998) are consistent with the hypothesis that the quantity of crop residue inputs and soil texture greatly influence the retention and turnover of crop residue C.

Total SOC content in the surface 20 cm was increased by 5.6 t C ha^{-1} in the continuous wheat no-till treatment and 2.8 t C ha^{-1} in the continuous grain sorghum treatment compared with the stubble-mulch treatment of Potter *et al.* (1997). Thus in this study, no-till management with continuous crops sequestered more carbon in comparison to stubble-mulch management of the southern Great Plains (Potter *et al.*, 1997). In Hungary, larger yields - implying higher returns of organic matter to the soil - were obtained for crop rotations as opposed to maize monoculture, especially in rotations which included leguminous crops (data for 1961-1996; 7 crop rotations and 5 fertilizer treatments) (Berzseny and Györffy, 1997). The yield-increasing effect of the crop rotation was inversely proportional to the ratio of wheat in the sequence (Berzseny and Györffy, 1997). Under wheat mono-crop, with mineral fertilizer added, SOC accumulated at a rate of 50 g C m^{-2} yr^{-1}, while a 3 year rotation of corn/wheat/clover sequestered 150 g C m^{-2} yr^{-1} (Buyanovski and Wagner, 1998a).

Residue burning

Fires affect the amount of C stored in the soil by burning C already in the soil and by changing the amount (of residue) available for input to the soil. In addition, fire can make a significant contribution to the distribution of nutrients in a soil by rapidly mineralising the above ground biomass into ash (Materechera *et al*., 1998). No general tendencies can be identified with respect to the extent and effects of fire-induced changes on the content and composition of organic matter in the soil (Almendos *et al*., 1990; Choné *et al*., 1991; Greene *et al*., 1990; Hernani *et al*., 1987a). Possible effects of heating on some physical and chemical parameters, in relation to soil aggregation and erodibility, are discussed by Giovanninni *et al*. (1998). During burning, the maximum soil temperature can be in the order of 150-220 ^oC at 1 to 5 cm depth (Ghuman and Lal, 1989). Reducing fire frequency and intensity generally has a positive effect on SOC stocks on the long-term, both in savannas (Jones *et al*., 1990; Materechera *et al*., 1998) and under forest (Liski *et al*., 1998a). Nitrogen mineralization appears to show a much greater sensitivity to burning frequency than total N (Jones *et al*., 1990). Concern about the sustainability of fire-managed savannas has been raised by Materechera *et al*. (1998), in view of the high losses of carbon, nitrogen and sulphur from the biomass during burning.

5.3 Soil fertility and nutrient management

5.3.1 NPK Fertilization

Improved N, P, and S status and overall soil fertility can result in concomitant improvements in soil organic matter. To sequester 10,000 kg of carbon in humus, about 833 kg of nitrogen, 200 kg of phosphorus, and 143 kg of sulphur are needed (Himes, 1998). As such, adequate management of soil fertility plays an important role in carbon sequestration. Changes in N-storage are dependent on management of N-fertility, and cultivation-induced narrowing of C/N ratios indicate preferential maintenance of N relative to C storage, while increases in P storage are due to fertilization (Ellert and Gregorich, 1996). Given that P and S contents are intrinsic soil properties (unless increased by fertilizer application), they may set a limit to the level of soil organic matter which can be attained, especially if the levels of these elements are particularly low (Tate and Salcedo, 1988). In most agricultural systems in the tropics the concentration of P in the soil is insufficient for crop growth and must be provided as an external input. Many researchers have shown the importance of N, P, K and S supply, either through fertilizer addition, application of phosphate rock, manuring or legume cover crops, in increasing C inputs and maintaining soil C levels (Fairhurst *et al*., 1999; He *et al*., 1997; Mitchell and Entry, 1998). The implementation of improved farming systems will be aided by improved soil analytical services, increased access to quality soil and crop data, and more work to quantify the extent of depletion of soil P, and other nutrients, in Asia, Africa and Latin America through integrated nutrient management (Smaling *et al.*, 1996; Stoorvogel *et al.*, 1993).

Sustainable farming systems that contribute directly to agricultural production and to the preservation of natural resources must be developed, along with socially, politically, and economically acceptable methods of implementation. It is likely that these systems will be characterized by methods of: (a) incorporating P into the farming systems, mediated by perennial, annual and fodder crop plants (b) avoiding losses and minimising P removal, (c) recycling P within the farming systems and between the production and consumption sectors, and, (d) funding increased use of P and other fertilizers (Fairhurst *et al.*, 1999). The importance of P-enrichment for formation of stable SOC in ancient cultivated soils has been document by Sombroek (1966) and Sandor and Eash (1995). In the case of the Colca valley, Peru, P-enrichment in the soil was a consequence of the intensive use of guano which was an important P-fertilizer during Spanish Rule (Rosell and Galantini, 1998). Present governments and private sectors have been requested to co-operate in the exploitation of indigenous P fertilizer deposits, the provision and distribution of cost effective imported P fertilizers in addition to appropriate advisory and extension services in order to overcome the widespread occurrence of P deficiency in tropical soils (Fairhurst *et al.*, 1999).

5.3.2 Organic inputs and manuring

Regular application of livestock manure can induce substantial changes in SOC over the course of a few years (Sommerfeldt *et al.*, 1988). Jenkinson (1990) reported that after 144 yr of large annual applications of animal manure (35 t ha^{-1} yr^{-1}), SOC content is still increasing at Broadbalk Wheat Experiment. Planting winter-crop or using manure at 1000 kg C ha^{-1} can sequester an equivalent of 2000-5000 kg CO_2 ha^{-1} yr^{-1}, according to model studies by Chansheng Li (1995). Bronson *et al.* (1998) reviewed management options, available to farmers, to improve synchrony of nutrient mineralization of organic inputs and crop demand, with respect to quality of organic inputs, placement and transfer of organic inputs, and timing of organic and inorganic applications. Woomer *et al.* (1998), reviewed management options for African smallholder agriculture with respect to carbon sequestration and organic resource management. The results from the long-term on-station experiment considered by Woomer *et al.* (1998), and the reality of the smallholder resource availability, indicate that the improvement in soil organic matter content tends to offset losses from SOC accumulated prior to land management rather than lead to a net increase in soil carbon. In another study, the rate of decline in soil organic carbon once the application of peat, applied as organic amendment, ceased was faster than the rate of decline observed where a range of different organic amendments, with small C:N ratios, had been tested (Johnston *et al.*, 1997).

5.3.3 Improved pastures

Changes in SOM quality may be more important than SOM quantity in influencing soil quality and fertility status (e.g. Clark *et al*., 1998; Drinkwater *et al*., 1998). Trujillo (1998) confirmed the potential of improved pastures to increase total SOC. The observed differences in total SOC contents between improved pastures and native Colombian savannas may be the result of a greater growth rate and net primary productivity of the improved pastures, probably stimulated by the applied fertilizers (50 kg P ha^{-1}) at the establishment and coupled with lower quality and decomposition dynamics of the below ground residues (Trujillo *et al*., 1998). The C:N ratios for introduced pastures in the Colombian savannas are in the range 72-194 for above-ground and 158-224 for below-ground litter, while the C:N ratio after 9 years is about 33 (Fisher *et al*., 1995). This implies the newly formed SOC is highly resistant to microbial decomposition in these savannas, pointing at the need for additional N-fertilizer management or introduction of legumes (Fisher *et al*., 1994). The usefulness of legume-based cropping systems for maintaining soil fertility in many agro-ecological zones are well known from other studies (e.g., Drinkwater *et al*., 1998; Grace *et al*., 1998 ; Tilman, 1998; Trumbmore *et al*., 1995). According to Drinkwater *et al*. (1998), application of legume-based cropping systems in the major maize/soybean growing region in the USA would increase soil carbon sequestration by 0.01-0.03 Pg C yr^{-1}, corresponding with about 1-2 % of the annual carbon released into the atmosphere from fossil fuel combustion in the USA.

5.4 Water management

5.4.1 Drainage

A favourable structure, with at least 10% air-filled porosity at field capacity, improves soil conditions including soil moisture regime, root growth and nutrient cycling. Soil drainage is an important management practice for agricultural lands across the mid-western United States, western Europe and many other regions of the world. Drainage may strongly interact with tillage methods to influence soil physical properties. The content of organic matter in a soil is influenced by hydrological processes, notably the water regime and its management through surface and subsurface drainage. Subsurface drainage generally leads to increased crop yields, and thus possibly higher returns of residues to the soil. Yet, as a result of increased aeration in comparison to poorly drained conditions, SOC contents of (artificially) drained soils are often lower than those of similar undrained soils (Fausey and Lal, 1992).

5.4.2 Irrigation

Irrigation is important in increasing crop productivity, and hence potential returns of crop residues to the soil carbon pool, notably in arid and semi-arid regions where the inherent SOC levels are low. Possible negative side effects of irrigation include the enhanced risk of salinisation and erosion (Szabolcs, 1989). Improved plant growth under irrigation as well as use of mineral fertilizers, such as urea, may enhance acidifying effects, thus leading to the dissolution of some carbonate-C. Some organic and inorganic carbon may be leached out of the system into aquatic systems. Increased ploughing of irrigated soils may also enhance oxidation of the organic matter present. Thus it is possible that the processes leading to SOC-loss under irrigation may mask the beneficial effects of an increased water-supply for C-sequestration on irrigated fields (Lal *et al.*, 1998b), notably where ambient air temperatures are high. Nonetheless, Lal *et al.* (1998b) estimated the total C sequestration potential of the 21 Mha of irrigated land in the US at 4.2 Tg C yr^{-1}, this total including both SOC-C and the C sequestered as secondary carbonates.

5.5 Erosion control

Soil C losses can occur both as a result of mineralisation as well as through erosion, often making it difficult to interpret soil carbon responses to management practices in long-term experiments (Paustian *et al.*, 1998). Gregorich *et al.* (1998) discuss the effects of erosion and deposition on C balances for cultivated soils, including field sites where attempts have been made to differentiate C losses due to oxidation versus erosion by water. The effects of erosion on C sequestration are complex and conflicting. For example, deposition and burial of eroded C can reduce decomposition and thereby contribute to soil C increases (off-site), while productivity losses due to erosion can reduce the amount of C input to the soil which reduces C stocks on-site (Gregorich *et al.*, 1998). Management practices will further complicate the picture. Along with reduced sediment loads, total P, total N, and total fertilizer N loss with sediment, for example, was reduced with the no-till compared to chisel-till in the study of Torbert *et al.* (1996).

In Western Nigeria, C losses from bare-fallow Alfisol plots with slopes of 1, 5, and 10 % varied from 54 to 3080 kg ha^{-1} (Lal as cited by Gabriels and Michiels (1991)). Hernani (1987b) studied C losses from Yellow Latosol plots with a 14 % slope. The annual losses of organic matter varied with management practices, amounting to 497, 32, 7.2 kg ha^{-1} for 'total clearing by bulldozer', 'total burning', 'all forest material left on site (mounding)' as compared to only 0.9 kg ha^{-1} for the intact forest plots. Rainfall simulators showed that continuous fallow plots have significantly higher C concentrations in runoff and higher sediment concentrations (Lowrance and Williams, 1988). The highest C content measured in runoff was 5.83 % from a four-row peanut plot. Total loads of C and sediment and total runoff were about twice as high as for their bare bedded plots. About 1 % of the total soil C, from the 0-29 cm zone, could be moved annually from the continuous fallow plots. The total amount of C transported depended on surface runoff, and the C content of the eroded sediments was related to the crop residue cover (Lowrance and Williams, 1988).

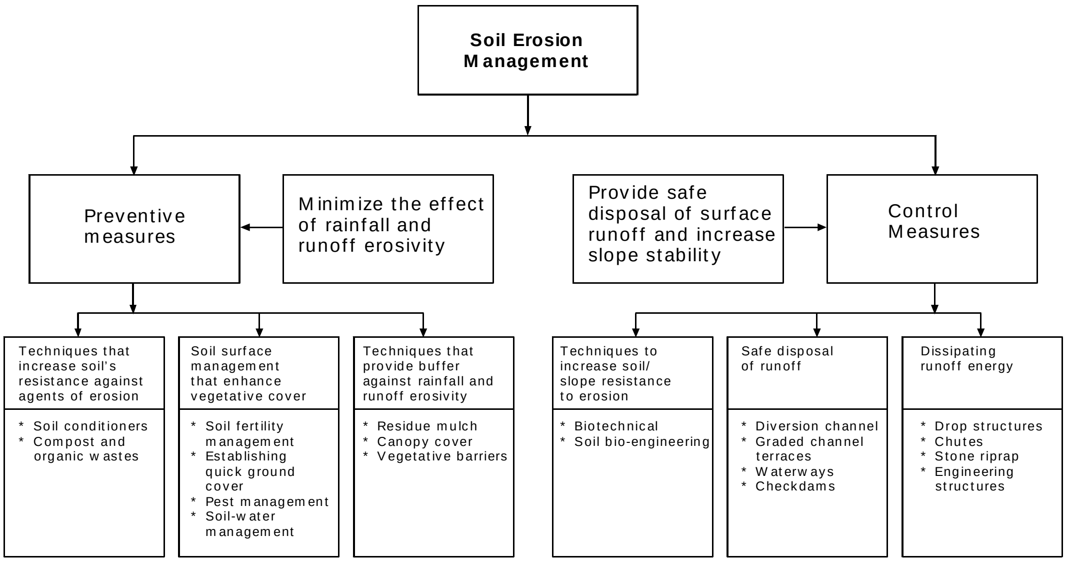

Fig. 8 Strategies of soil erosion management though preventive and control measures (After: Lal, 1990)

Erosion management is important to sustainable use of natural resources and involves preventive and control measures (Fig. 8). The first involve adoption of judicious land use and appropriate land management systems that maintain the ecological balance between soil, vegetation, and water, and minimize the impact of rainfall and runoff erosivity. There are three main types of erosion-preventive techniques (Lal, 1990): (a) those that increase the soil's resistance against agents of erosion; (b) soil surface management techniques that help establish quick ground cover; and (c) techniques that provide a buffer against rainfall and runoff-erosivity. Common examples of conservation approaches that reduce erosion include the use of soil conditioners, compost and organic amendments, residue mulch and vegetative materials that provide canopy cover and bind soil through root action. Others are soil surface management systems that help establish a quick ground cover such as, soil water and nutrient management, early sowing. Additional information on the subject of erosion management, in relation to maintaining or increasing soil organic carbon content, may be found elsewhere (Gabriels and Michiels, 1991; Gray and Sotir, 1996; Lal, 1990; Lal *et al*., 1998b). The Overview of Conservation Approaches and Technologies, (WOCAT, 1998), which contributes to the sustainable development of natural resources worldwide by presenting lessons learned from successful examples of soil and water management, should be mentioned also in this context.

There are about 1964 x 10^6 ha of degraded soils in the world (Oldeman, 1994). At regional or global levels the greatest impact of soil degradation on C storage may be through reducing soil productivity, and hence C inputs to the soil as plant residues. The use of management practices that prevent or reduce soil erosion may be the best strategy to maintain, or possibly increase, the world's soil C storage. Elevated clay content may enhance sequestration in eroded surfaces (Hendrix *et al*., 1998). The carbon content of these degraded soils could be improved by afforestation or planting of cover crops, leading to an overall improvement of soil quality. Lal (1997) estimated that if SOC content of the currently degraded soils could be improved by 0.01% in the top 1-m, this would lead to a carbon sequestration at the rate of 3.0 Pg C yr^{-1}, if a mean bulk density of 1.4 Mg m^{-3} is assumed, which is of the same order of magnitude as the present annual increase in atmospheric C concentrations.

Conservation Reserve Program (CRP)
Taking fragile lands out of agricultural production will help reduce soil degradation, help to restore soil functions, and thus lead to enhanced sequestration of carbon the soil. In the USA, the Conservation Reserve Program (CRP) has been implemented to convert highly erodible land from active crop production to permanent vegetative cover for a 10-year period. Practices used include the creation of conservation buffers (vegetative filter strips in conjunction with recommended practices), restoring wetlands and restricting the use of organic soils, the restoration of degraded soils including eroded lands and mine-lands and toxic soils, as well as reclamation of salt affected soils, and soil fertility restoration. Appropriate practices, with geographical focus on the USA, have been discussed at length by various authors (Follet, 1998; Huggins *et al*., 1998a; Lal *et al*., 1998b). The adoption of soil restorative measure on US cropland has a potential to sequester from 17 to 39 Tg C yr^{-1} (Lal *et al*., 1998b). The 'set aside' policy now pursued in the European Community may also lead to increasing organic matter levels in those soils no longer cultivated, provided they are allowed to develop a vegetative cover (see Smith *et al*., 1997b). Restricting agricultural use in areas previously farmed intensively by converting them into permanent or rotation fallow, can result in measurable changes in deep percolation (ground-water recharge) of N-compounds and water quality in less than one year (Meissner *et al*., 1999).

5.6 Forest management

Schiffman (1989) found that natural reforestation of old agricultural fields increased carbon storage from about 55,000 to 185,000 kg ha^{-1} over a 50-70 yr period. Carbon in the phytomass accounted for 76 % of the increase, the forest floor for 13 per cent, and surface soil for 10 per cent.

Management practices for enhanced carbon sequestration in forests include: (a) a halt to deforestation, (b) an expansion of the land area of forests, also to reduce erosion; (c) an increase in the stock of C in existing forests; (d) more efficient harvest and greater use of wood in long-lasting products; (e) the substitution of wood fuels for fossil fuels; (f) removal of marginal lands from agricultural production followed by re- or a-forestation; (g) retention of forest litter and debris after silvicultural or logging activities; and, (h) addition of amendments to improve soil fertility (Dabas and Bhatia, 1996; DiNicola *et al*., 1998; Dixon and Krankina, 1995; Houghton, 1996; Krankina *et al*., 1996; Moura Costa, 1996). The role of C in nutrient cycling processes, with special attention to temperate regions, and research needs for carbon management in forest soils have been reviewed elsewhere (Johnson, 1993; Johnson, 1995a; Nabuurs *et al*., 1999).

Several studies show there is great potential for economically-sound increased carbon fixation through improved forest management. Slowing land use change, expanding current forest area and improving productivity of existing stands could potentially conserve or sequester about 2.9 Pg C yr^{-1} in Brazil, 6.5 Pg C yr^{-1} in Russia and 1.3 Pg C yr^{-1} in the USA (Dixon and Krankina, 1995). Shvidenko *et al.* (1995), reported a potential for an increase in carbon fixation in Russian forest ecosystems of 24.4 Pg C over 100 yr, the net sink ranging from 16.5 Pg C in the 'low' and 42.5 Pg C in the 'high' estimate. Possible implications for soil C stocks in Brazilian, Russian and US forest soils were not considered explicitly in the above studies (Dixon and Krankina, 1995; Shvidenko *et al*., 1995), but this topic has been addressed elsewhere (Nabuurs *et al*., 1999). Unless the rate of global warming is gradual enough to avoid widespread mortality of forests, the additional releases of C caused by the warming itself - through increased respiration, decay and fires - may dilute/neutralize the aims of forest management (Houghton, 1996).

The potential storage of C in forest trees appears largest in temperate and tropical forests, and lowest in boreal regions mainly as a result of the shorter growing-period and photosynthetic season. Large potentials for increased C-sequestration in the soil of well-managed forests under exist both in the cold temperate and tropical regions (Vogt *et al*., 1995).

5.7 Biomass energy and fuelwood

Net carbon sequestration in the tropics is much less than actual CO_2-assimilation as it is partly neutralized due to predominant use of wood for meeting energy needs (Dabas and Bhatia, 1996). Use of wood for making durable products, such as paper, pulp, and veneer, will not return the absorbed CO_2 back to the atmosphere immediately (Dabas and Bhatia, 1996). Lal *et al.* (1998b) give a good overview of possibilities of using bio-fuels for off-setting fossil fuel use. Ethyl alcohol and sugar cane bagasses as fuel substitutions for gasoline, and natural gas, fuel or coal, can have an important role in reducing greenhouse gas emissions (Rosa and Ribeiro, 1998). Cultivation of large quantities of modern biomass is feasible, but its effectiveness to reduce emissions of greenhouse gases has to be evaluated in combination with many other environmental land use and socio-economic factors (Leemans *et al*., 1996). Large-scale, liquid bio-fuel production is not considered a viable option as direct use of biomass, for instance in fuel production, may be a better alternative (Giampetro *et al*., 1997). Net C accumulation for

10 Mha of cropland converted to dedicated bio-fuel cropland is about 50-90 Tg C yr^{-1} (Lal *et al*., 1998b, p. 54). According to Cole *et al.* (1996), agricultural bio-fuels, including energy crops, agroforestry, and crop residues, have the potential to substitute for 0.40 to 1.50 Tg C of fossil fuel per year. Since bio-energy crops essentially are an emission reduction, through substitution of fossil fuels, this aspect is not regarded further in this paper on potentials for C sequestration in the soil. Opportunities and constraints for carbon dioxide offsets in the Asia-Pacific region by usage of wood waste of sawmills have been discussed elsewhere (DiNicola *et al*., 1998).

5.8 Potential environmental side-effects

Potential (adverse) environment side-effects, such as an increased risk of nitrate leaching or trace gas fluxes from the soil as well as the economic and policy implications of changes in land management, so far, although recognized have largely been ignored in model studies of C sequestration (Smith *et al.*, 1997c). For example, increasing nitrogen fertilization in intensive agriculture is likely to cause an upward trend in total N_2O and NO emissions (Erickson and Keller, 1997; Hénault *et al*., 1998), despite available options for mitigation (Smith *et al*., 1997a). Alternatively, liming of acidic forest soils may cause a reduction in emissions of the greenhouse gases N_2O and CH_4, due to changes in chemical, biological, and physical conditions of the soil (Borken and Brumme, 1997). Increasing the area under rotation fallow, for example subsequent to the EU's set-aside policy, may substantially increase the nitrogen load of streams within a catchment area (Meissner *et al*., 1999). Part of this nitrogen may later be re-emitted to the atmosphere (Bouwman, 1990).

Various authors have suggested that peat drained for agricultural and other purposes should be returned to their 'natural' wetland state as a mechanism for increasing SOC stocks, maintaining biodiversity and restoring natural habitats. Although appropriate in theory, this approach may lead to a range of adverse environmental impacts such as enhanced CH_4 emissions (Willison *et al*., 1998), increased heavy metal solubility (Römkens and Salomons, 1998), and time-delayed releases of pollutants as 'Chemical Time Bombs' (Stigliani *et al*., 1991). Identification of areas considered vulnerable to chemical pollution would be useful in this respect (Batjes, 1999; Batjes and Bridges, 1993).

Sewage sludge appears to be a suitable soil conditioner, for instance by increasing aggregate stability to raindrop impact (Sort and Alcañiz, 1999). Long-term use on grassland, however, may lead to a build up of metals in the topsoil giving rise to potential risk for grazing animals through ingestion (Aitken and Cummins, 1997). Much of the variation in concentrations and export of dissolved organic carbon (DOC), which contributes to the acidity of water, the mobility and toxicity of metals and organic pollutants, and the availability of nutrients in soils and aquatic systems, may be explained by differences in soils within catchments (Moore, 1998).

Irrigated rice soils are a special issue in the current context. When supplied with organic matter, much of the C added evolves as methane basically making them unsuitable for carbon sequestration. Bronson *et al.* (1998) reported carbon levels in the lowland rice soils of tropical and subtropical Asia generally remain constant, despite straw removal in the low-producing rainfed regions and burning of straw in more productive, irrigated rice areas. Mitigation options for reducing CH_4 fluxes from flooded rice fields have been reviewed by Neue (1997).

Some cultivated prairie and forest soils can act as CH_4 sinks (Wang *et al.*, 1999). Tillage and land use effects on methane oxidation rates in soils have been discussed by Hütsch *et al.* (1998) and Boeckx *et al.* (1998). Forest-to-pasture conversion can promote the anaerobic mineralization of organic matter by changing the physical properties of soil, thus leading to enhanced anaerobic production of methane (Chauvel *et al.*, 1999).

6 Potential for Soil Carbon Sequestration by Agro-Ecological Region

6.1 General

Chronosequence studies, as shown in Table 6, give indicative values for long-term rates of organic carbon accumulation is soils since the Holocene. These values illustrate differences in soil type, their suitability for different uses and the factors of soil formation must be considered when identifying management options for enhanced C-sequestration. This chapter starts with a review of the literature on SOC sequestration potential by broad agro-ecological zone. Subsequently, exploratory scenario studies are made, using available global data sets and necessarily coarse assumptions, to infer likely ranges in SOC sequestration potential with improved/appropriate, management over the next 25 years (see Section 6.8).

Table 6 Long-term organic carbon sequestration rates in soils since the Holocene (After: Schlesinger, 1990)

Ecosystem	Accumulation phase (10^3 yr)	Accumulation rate (g C m^{-2} yr^{-1})
Polar desert	8 - 9	0.2
Tundra	9	1.1 - 2.4
Boreal forests	3 - 5	5.7 - 11.7
Temperate forests	2	0.7 - 2.5
Tropical forests	4 - 8	2.3 - 2.5
Temperate grasslands	9	2.2

6.2 Polar and boreal regions

There still is some organic matter accumulation (0.2-0.4 %) in many soils of Antarctica, particularly in old penguin rookeries characterized by high contents of gravel and apatite (up to 5% SOC) (Blume *et al.*, 1996). The average amount of organic carbon held to a depth of 1 m in various soils of arctic (28.6 kg C m^{-2}), subarctic (50.4 kg C m^{-2}), and boreal (28.6 kg C m^{-2}) eco-climatic provinces of Canada, have been discussed by Tarnocai (1998).

A high percentage of hemi-cellulose and the relatively low extractability by alkali solutions reflect the environment characterized by cold climates and high water content (in summer), which retard SOM transformations (Ping *et al.*, 1998). Arctic tundra has been a net sink for carbon dioxide during historic and recent geological times, and large amounts of organic carbon are stored in soils of northern ecosystems (see also Table 6).

Tundra soils lose or accumulate carbon because of changes in temperature and precipitation, but are also affected by soil processes and vegetation changes due to mechanisms independent of climate, such as drainage, disturbance, nutrient availability, or plant succession (Eisner, 1998). In the boreal and cold temperate climatic regions, foliage life-span and site nutrient status are important criteria in separating sites as to their carbon sequestering potential in the forest floor and soil (Vogt *et al.*, 1995). These organic materials have a high potential for bioactivity with increasing soil temperatures and increased nutrients inputs (Ping *et al*., 1998). Consequently, with global warming Cryosols or 'frozen soil' ecosystems are likely to exert a positive feedback, implying CO_2 release, on atmospheric concentrations of carbon dioxide (Oechel *et al*., 1993). Different feed-backs may occur depending on whether tundra soils are acidic or non-acidic, and these responses need to be studied further (Bockheim *et al*., 1998; Walker *et al*., 1998). Inherently, soils in these regions should best be left in their natural state.

6.3 Cold and cool temperate regions

Smith (1997c) examined the potential for C sequestration in the wider Europe, that is all Europe excluding most of the former Soviet Union but including Byelorussia and the Ukraine, over 100 years. Using the linear regression equations of Smith *et al.* (1997b), they explored the effect of amending *all* arable soils in Europe with organic manure at 10 t ha^{-1} yr^{-1}, leading to an annual increase of 0.326% yr^{-1}. In a second scenario, arable soils are amended with sewage sludge at a realistic application rate to increase SOC content. Application at a rate of 1 t ha^{-1} yr^{-1} would increase total SOC content by 4.5% over 100 years, on the basis of the necessarily broad assumptions used. In a third scenario, cereal straw (5.07 t ha^{-1} yr -1) was ploughed back into the soils in which it was grown, which would increase SOC content by 7.5% over a 100 yr period. The fourth scenario, assumes all surplus arable land is allowed to revert to natural woodland over a century, assuming an afforestation rate of 30% of current arable land. This would lead to an estimated 10% increase in SOC, or 35.8 Tg C yr^{-1}, for the region. If the potential for the wood produced to substitute for fossil fuel is also considered, greater gains can be achieved (Smith *et al*., 1997c). The combined scenario in which 50% of wood used for durable bio-products and 50% for energy production as biofuels yields a C- mitigation potential of 73.8 to 83.9 Tg C yr^{-1}. Finally, the last scenario of Smith *et al.* (1997c) considers agricultural extensification through ley-arable farming, so that at any one time 30% of the land is in a ley phase. This would lead SOC to increase by 1.02% yr^{-1}. Conversion of current arable land to a ley-arable system would increase total SOC stocks in Europe by 21% over 100 years, the greatest effect on SOC content of any of the scenarios examined by Smith *et al.* (1997c).

Table 7 C mitigation potential of five scenarios examined for Europe (Source: Smith *et al*., 1997c, p. 34)

Scenario	Yearly increase in SOC ($Tg\ C\ yr^{-1}$)	% of anthropogenic CO_2-C produced annually in Europe	% of anthropogenic CO_2-C produced annually globally
10 $t\ ha^{-1}\ yr^{-1}$ animal manure to all arable soils all arable soils	23.41	2.08	0.38
1 $t\ ha^{-1}\ yr^{-1}$ sewage sludge to arable soils (11.2% land)	15.72	1.39	0.25
5.1 $t\ ha^{-1}\ yr^{-1}$ straw incorporated into all cereal land	26.08	2.31	0.42
Afforestation of 30% surplus arable land by natural regeneration	119.64	10.62	1.93
Conversion of all current arable land to 33% ley-arable farming with extensification of pig and poultry farming	72.92	6.47	1.18

Summarizing, Table 7 shows that the afforestation and extensification scenarios show the greatest potential for C-mitigation in Europe. They can account for 6.5-10.6% of anthropogenic CO_2-C produced in Europe, corresponding to 1.2-1.9% of that produced globally. The underlying assumptions and uncertainties are clearly outlined by Smith *et al.* (1997c); the calculated potentials for C-sequestration may overestimate the level of C-sequestration attainable in practice. For example, no account was taken yet of the edaphic and climatic suitability in a given region for the management types under consideration, nor whether the natural resources such as animal manure are locally available. Possible effects of changes in EU agricultural policy for set-aside practices are also important in this respect (Bouma *et al*., 1998). To account for such complex interactions, more sophisticated model approaches than linear regressions, although based on the best available long-term data, are needed. This will involve the use of spatial databases of soil, climate, ground cover coupled with suitable dynamic simulation models.

Key strategies for increasing SOC sequestration in temperate regions include increasing cropping frequency and reducing bare fallow, increasing the use of perennial forages (including N-fixing species) in crop rotations, retaining crop residues and reducing or eliminating tillage (so-called no-till practices). In North America and Europe, conversion of marginal arable land to permanent perennial vegetation, to protect fragile soils and landscapes and/or reduce agricultural surpluses, will provide additional opportunities for SOC sequestration (Paustian *et al*., 1997).

6.4 Arid and semi-arid regions

Dominant soils of drylands, of mainly arid and semi-arid tropical and mid-latitude regions of the world, contain about 251 Pg C of SOC and 509 Pg C of carbonate carbon, amounting to 760 Pg C in total in the upper 1 m of soil (Batjes, 1995). These totals do not account for SOC held in minor associated soils. Based on soils of life-zones corresponding with 'dryland' areas of the world, the estimates are 300-369 Pg C SOC and 473-546 Pg C carbonate carbon to a depth of 1 m (Batjes, 1995).

With comparatively small stores of biomass and SOM and limited NPP, plants occurring on arid lands make only a small contribution to the global carbon cycle through photosynthesis. Even the large pool of carbon, found in dispersed and solidified soil carbonates, turns over relatively slowly (Schapendonk *et al*., 1998; Schesinger, 1985). The possibility of enhancing carbon sequestration through management of drylands has been the topic of an international workshop organized by UNEP (Squires, 1998; Squires *et al*., 1995), while aspects of sustainability of agroecosystems in semi-arid regions have been discussed by Stewart and Robinson (1997). The potential for sequestering carbon in the soils of arid regions decreases as annual precipitation decreases, and generally as mean temperatures increase (Grace *et al*., 1998; Rasmussen and Oartob, 1994; Stewart, 1995).

Even in semi-arid regions, significant amounts of carbon can be retained in soil organic matter if crop residues are carefully managed, and tillage is used as sparingly as possible (Stewart, 1995). Based on a literature review of studies in semi-arid regions, Rasmussen and Oartob (1994) reported that soil organic C levels typically increase at a rate of 10 to 25% of the amount of C added, and that greater C retention rates are expected with increasing precipitation and decreasing cropping intensity. Pierri (1995) summarized data from semi-arid, Francophone Africa and reported that on very sandy soils, annual ploughing with fertilizers led to an annual loss of organic matter of 5% or more. Without exception, only the methods with manure application prevented a decline in soil organic matter. Although the effects of ploughing were less clear, the studies reported by Pierri (1995) indicated that ploughing increased the decline rate. Pierri (1995) proposed that there is a critical level for soil organic matter that is dependent on the soil organic matter content (%) and the sum of clay plus silt. Once organic matter levels fall below this critical level, maintenance of soil structure is difficult to achieve. Pierri (1995) states also that in semiarid Africa, where there are so many technical and economic constraints to crop performance, it is fruitless to aim for soil organic matter levels above the critical value.

Organic matter maintenance in semiarid regions is clearly one of the greatest constraints in the development of sustainable agroecosystem, particularly in many developing countries where the crop residues are so important as a source of animal feed, fuel and for cooking (Stewart and Robinson, 1997). Whenever feasible, Stewart and Robinson (1997) recommend to let animals graze the crop residues so the manure will be distributed over the area. When it is necessary to utilize crop residues as animal feed away from the land, every attempt should be made to return manure to the land whenever feasible. Otherwise, the soil organic matter level will continue to decline to the point that long-term sustainability of the soil resource will be threatened (see also: Stoorvogel *et al*., 1993). Steen (1998) recommends extensive grazing of low-productive rangelands should be abandoned in favour of more stationary feeding systems. Fertiliser costs for farming systems with an inherently low biological production potential, are likely to exceed the benefits in terms of increased crop and livestock output (Powell *et al*., 1994). Reducing nutrient losses via runoff, leaching, volatilization etc. can enhance the profitability of using external nutrient sources.

Groot *et al.* (1998) reviewed dynamics of organic matter in soils of the Soudano-Sahelian zone. West (1994) reviewed the possible effects of climatic change on the edaphic features of arid and semiarid lands of western North America. They expect that both soil C and N levels will decrease, especially under increased temperatures, whereas the C/N ratio will decline to its lowest range of possible values. Both free and symbiotic N fixation should decline unless there is a shift to greater absolute precipitation during summers. Only slight changes in soil P, N, S and trace elements are expected by West *et al.* (1994) under any of the five scenarios of climatic change considered. Soil organic matter levels in arid regions could potentially increase if changes in mean annual temperature and mean annual precipitation are minimal, and either elevated CO_2 results in higher efficiency of water use by plants or the N deposition from the atmospheric pollution increases N availability (West *et al*., 1994). Production of litter will change relatively little, but its chemical quality will decrease and nutrient cycling will be accelerated when the vegetation shifts from perennials to annuals. Main processes of land degradation in dryland areas, and options for combatting land degradation and thus carbon sequestration have been reviewed elsewhere (see Squires *et al*., 1995). Lioubimtseva (1998) also discussed possible impacts of climatic change on carbon storage variations in African and Asian deserts.

Drylands have the potential to be a sink of 1.0 Pg C yr^{-1} over the next 50 years (Squires, 1998), provided they can be restored to their ecological potential. Dryland restoration thus could have a major impact on global climates since the non-forested world's drylands (excluding hyper-arid regions) is 5.2×10^9 ha, or about 43% of the earth's surface. Model simulations by Grace *et al.* (1995), however, suggest that even in high input, conservation tillage farming systems, it will be an onerous task to sequester significant amounts of organic C in these regions. Stewart and Robinson (1997) observe also that while maintenance of soil organic matter is of critical importance in semiarid areas, this poses a tremendous challenge because insufficient rainfall seriously limits carbon inputs into the system, and the warm conditions accelerate the decomposition of native soil organic matter during the limited period when soil water conditions favour growth. Under these conditions, extensive tillage generally increases the rate of decomposition (Stewart and Robinson, 1997).

Rasmussen *et al.* (1998b) conclude that the main factors influencing C levels in semi-arid farmland of the region are the frequency of summer fallow and the amount of C input by crop residue. Where summer fallow is practised, soils continue to lose C except where large amounts of manure are added. Rasmussen *et al.* (1998b) report that C levels in the semi-arid US Pacific Northwest can be increased by continuous cropping, returning all crop residues and minimizing erosion.

6.5 Seasonally dry and humid tropics

Tropical and temperate natural grasslands play a significant but often poorly recognized role in the global carbon cycle. Grassland ecosystems store most of their carbon in soils, where turnover times are relatively long (100-10,000 yr), and so changes, though they may occur slowly, will be of significant duration (Parton *et al.*, 1995). Carbon in grasslands, grassland NPP data and the role of grasslands as carbon sinks have been reviewed by Scurlock and Hall (1998). Based on this study, the authors conclude that tropical grasslands and savannas 'soil carbon sinks' may contribute more to the 'missing sink' than was previously appreciated, perhaps as much as 0.5 Pg C yr^{-1} (Scurlock and Hall, 1998). The future sink under climate change, however, is much less certain, given its dependence upon future management regimes, with extreme values of -2 and + 2 Pg C yr^{-1} (Scurlock and Hall, 1998).

Since the end of the 1960's, deforestation and agricultural extension have increased markedly in the tropics, notably in the Amazon basin where over 100,000 km^2 of pastures have been created (Malingreau and Tucker, 1988). In Brazilian Amazonia, logging crews now degrade 10,000 to 15,000 km^2 yr^{-1} of forest that are not included in deforestation mapping programmes, while surface fires burn additional large areas of standing forest the destruction of which is neither documented (Nepstad *et al.*, 1999). In many instances, this has lead to widespread soil degradation and loss in soil quality, including in organic matter content. Increases in SOM under appropriate management, however, have been reported also for the region (Andreux and Choné, 1993; Neill *et al.*, 1998; Theng, 1991).

It appears no generalisation can be made regarding the rate of recovery of SOM content following clear-cutting and pasture installation, as it depends on several factors including the parent material, local climate, type of pasture, and more generally the overall management of the pasture (Andreux and Choné, 1993). About 100% of the C in above-ground biomass is lost within 20 years of clearing (Neill *et al.*, 1998). In the experimental farm near to Manaus, after 8 years, total C content had returned to its initial value, contrary to what was observed in older degraded pastures managed by traditional farmers in the same region (Andreux and Choné, 1993). However, N contents decreased from the first year, and the pasture soil had C/N ratios of 2-3 units higher than those of the reference forest soils (Andreux and Choné, 1993). The best long-term mean accumulation rate observed under pasture for soil organic C, determined from chronosequence studies in Rondônia, is 2 kg C m^{-2} to a depth of 50 cm over about 80 years (Neill *et al*., 1998). The dynamics of organic matter in tropical soils has attracted much interest and created much discussion for many years (Greenland *et al*., 1992).

Soil organic carbon dynamics in native and cultivated ecosystems of South America have been reviewed by Rosell and Galantini (1998). In weathered tropical soils, rates of C losses caused by (improper) cultivation are many times faster than those for temperate soils, with a substantial deterioration/increase of soil quality often in less than 10 years (Shang and Tiessen, 1997). The most variable climatic attribute in the Humid Tropics is probably the seasonal distribution of rainfall, which determines the length of the dry and wet periods, and thus influences litter fall accumulation and decomposition pathways, thus SOC content. Knowledge about processes and properties in relation to C dynamics in the tropics remains scanty.

Unlike the majority of the world's soils which consist of minerals the surface charge of which is constant and negative, the surface charge of the minerals in most soils of the humid tropics is variable and pH-dependent, whereby it can be net positive, zero, or net negative (Uehara, 1995). The potential for high productivity of isoelectric soils resides in their good physical characteristics. When the characteristics are combined with high soil fertility (which they now lack, unless fertilized or manured), adequate rainfall and year-long growing season, the isoelectric soils of the humic tropics can be as productive as any in the world, creating options for increased SOC by appropriate management (Greenland, 1995; Sombroek, 1995; Uehara, 1995). In order to overcome Mn^{2+} and Al^{3+} toxicities, these soils generally have to be limed to a pH of about 6.2 (Rosell and Galantini, 1998).

In the tropics, possibilities to increase SOC sequestration are mainly through increasing C inputs to soil by improving the fertility and productivity of cropland and pastures (Paustian *et al*., 1997). In extensive systems with vegetated fallow periods, such as shifting cultivation, planted fallows and cover crops can increase C levels over the cropping cycle. Use of no-till, green manures and agroforestry are other beneficial practices, for instance when compared with continuous sugarcane cropping with burning (Bronson *et al*., 1998; Skjemstad *et al*., 1999). Overall, improving the productivity and sustainability of existing agricultural lands is crucial to help reduce the rate on new land clearing, from which large amounts of CO_2 from biomass and soil are emitted to the atmosphere (Paustian *et al*., 1997).

6.6 Wetlands

Organic carbon stocks in upland soils generally are far below those of wetland soils which, when organic in nature such as Histosols, may hold up to 60-78 kg C m^{-2} in the upper 1 m (Batjes, 1996b; Sombroek *et al*., 1993). This is a reflection of the fact that natural peatlands, and many wetlands soils, have been a net sink for atmospheric CO_2 from the Last Glacial Maximum to the Present (Adams *et al*., 1990).

The total content of carbon in organic soil varies with peat fibre and ash contents. Reserves of carbon in the top 100 cm of the world's peat soils have been estimated variously at 300 Pg of C (Sjors, 1980), 202-377 Pg of C (Adams *et al*., 1990), and 357 Pg of C (Eswaran *et al*., 1993). The figures of Sjörs (1980) and Eswaran *et al.* (1993) are comparable with the 330 Pg of C found elsewhere (Batjes, 1996b). SOC estimates for the first 0-30 cm are 120 Pg of C, while it is 679 Pg of C for the upper 200 cm, respectively (Batjes, 1996b).

Changes from anaerobic to aerobic environments, through drainage of wetlands due to natural or anthropogenic factors, can drastically influence the carbon balance of organic soils, and thus the global carbon cycle. Changes in land use, deforestation, drainage, and cultivation generally lead to a rapid loss of C from organic soils by oxidation, volatilization, and mineralization. Dried peats, which are often hydrophobic, are prone to burning and wind erosion. Possible controls of CO_2, N_2O and CH_4 emissions by different management of organic soils have been discussed by Kasimir-Klemedtsson *et al.* (1997).

Due to natural accretion, Histosols may absorb about 0.2 Pg C yr^{-1} (Buringh, 1984). The net release of CO_2 from wetlands upon drainage amounts to 0.15-0.18 Pg C yr^{-1} (Armentanao and Menges, 1986). Schlesinger (1984) calculated drainage and cultivation may release 0.03 Pg C yr^{-1} from organic soils of the world, transforming peatlands areas from a sink to a source of atmospheric CO_2. Additional releases of about 1.6 to 3.1 Pg C yr^{-1} of CO_2 and somewhat greater than 0.22 Pg C yr^{-1} of CH_4 to the atmosphere may occur as a result of the climate change scenarios modelled by Post (1990). These figures reflect the potential significance of climate change, notably if climate predictions of increased evapotranspiration and reduced soil water status are realized, on CO_2- emissions from tundra regions (Malmer, 1992; Oechel *et al*., 1998; Post, 1990).

Natural wetlands have large organic carbon stocks that need to be preserved by conservation. 'Re-flooding' of previously drained/reclaimed wetland/peat soils can sequester large amounts of atmospheric CO_2, but some adverse environmental side-effects may occur also (see section 5.8).

6.7 Mountains and highlands

In view of steep slopes, limited possibilities are foreseen with respect to enhancing C sequestration in these regions via management, unless by re- or afforestation of degraded sloping lands. In the subalpine forests of the cold temperate zone, forest stand development sequences caused soil organic matter to accumulate at < 0.4 Mg C ha^{-1} yr^{-1} over a 160-year time span (Vogt *et al*., 1995). These rates may be modified by anthropogenic nitrogen deposition in that these lead to changes in plant community structure as a result of competitive displacement (Bowman and Steltzer, 1998).

6.8 Exploratory studies of SOC sequestration potential

The potential for C-sequestration in a given soil, and agro-ecological zone, will be proportional to the original reserves present under undisturbed conditions. Therefore, options for C sequestration should be chosen on the basis of knowledge of the nature and likely magnitude of C pools, whether organic or inorganic, in a given biome or major agro-ecological region and their responses to different land uses and management systems. The analyses underlying Table 9 were made to allow for, necessarily coarse, inferences about the potential for increased carbon sequestration by major agro-ecological zone (see App. 3), with special reference to degraded soils as defined in the GLASOD study (Oldeman *et al*., 1991), and Agricultural Lands as considered in IMAGE 2.1 (Alcamo *et al*., 1998).

The GLASOD map (Oldeman *et al*., 1991) considers four degrees of soil degradation. They are expressed in terms of declining agricultural productivity, biotic functions and restoration costs, as follows:

1) light degradation: The terrain has somewhat reduced agricultural productivity, but is suitable for use in local farming systems. Restoration to full productivity is possible by modifications of the management system. The original biotic functions are still largely intact.
2) moderate degradation: The terrain has greatly reduced agricultural productivity but is still suitable for use in local farming systems. Major improvements are required to restore productivity. Original biotic functions are partially destroyed.
3) strong degradation: the terrain is non reclaimable at farm level. Major engineering works are required for terrain restoration. The original biotic functions are largely destroyed.
4) extreme degradation: The terrain is unreclaimable and beyond restoration. The original biotic functions are fully destroyed.

These four broad categories in combination with information on the relative extent of degradation, expressed as severity classes (see UNEP, 1992), can be used for global assessments of the possible gains in SOC with improved management and/or conservation. Nonetheless, making inferences about realistic possibilities for increased carbon sequestration in the soil, through improved appropriate management, is difficult because many of the factors and processes that control the flow of carbon between soils and plants are still poorly understood (see Chapter 7).

Despite the variability of possible responses to improved management, conservation practices, and climate change, the average rate of carbon sequestration can be estimated as being about 0.3 Mg C ha^{-1} yr^{-1} based on available published data, with a maximum of around 1.0 Mg C ha^{-1} yr^{-1} for reclamation of 'disturbed, eroded soils' most depleted in SOC (Bruce *et al.*, 1999). Conservatively, it is assumed that this increase in C-mass will apply to the first 1 m of soil. Assuming a mean organic carbon content of 10 kg C m^{-2} to 1 m depth for an 'average' soil, the value of 0.3 Mg C ha^{-1} would correspond with an annual increase of:

a) 0.3 Mg C ha^{-1} = 0.3×10^3 kg x 10^{-4} m^2 = 0.03 kg C m^{-2}

b) Expressed as percentage of annual increase of the organic carbon mass of the 'average' soil, this is:
(0.03 kg C m^{-2}) / (10 kg C m^{-2}) x 100 % = 0.3 % yr

c) Assuming this trend can be sustained over 25 years, this would amount to:
$SOC_t \approx SOC_c \times 1.075$ (rounded-off to 8% for Medium scenario in Table 8).

Table 8 Broad scenario's for possible increases in SOC of degraded lands by improved management over next 25 years

Scenario for SOC increase	Severity of degradation			
	Low	Medium	High	Very High
Low	4.0*	2.0	1.0	0.6
Medium	8.0	4.0	2.0	1.0
High	12.0	6.0	3.0	1.5

* Percentage (%) increase expressed as added-mass of SOC per area unit (and type) of degraded land, subsequent to appropriate management, relative to contemporary organic carbon levels (kg C m^{-2}) in the upper 1 m of soil.

A Low and a High scenario for the possible increase in SOC over the next 25 years are introduced to show a 'window' for the possible increases. The assumption is that with the best available remediation/conservation techniques, it should be feasible) at least from a technical point of view) to increase the SOC mass from 4 to 12% over the next 25 years where the severity of soil degradation is 'low' or the arable land is termed 'stable'. Arbitrarily, and with reference to the definition of the GLASOD-degradation types, the 'possible' increase is set at ½ of the potential increase where the severity is 'medium'; 1/4 of this potential where it is 'high'; and, 1/8 of it where it is 'very high' in GLASOD-terms. This then leads to three broad exploratory scenarios, termed 'Low'; 'Medium', and 'High', for the possible increase in contemporary SOC levels over the next 25 years (Table 8). Differences in SOC-sequestration potential by soil type, severity of soil degradation, and agro-ecological zone thus are implicitly considered in the current approach.

Table 9 lists the inferred increase in SOC over the next 25 years, expressed as range in possible annual increase (as: Pg C yr^{-1}) respectively as total increase (as: Pg C) for the regions under consideration.

Table 9 Exploratory scenarios for increases in organic carbon sequestration in the soil over the next 25 years (first value is in Pg C yr^{-1}, second value in *italics* is given as Pg C)

Scenario	Inferred SOC increase		
	Low	Medium	High
A) Restoration of all degraded lands, irrespective of land use/cover	0.65 *16.1*	1.29 *32.2*	1.9 *48.3*
B) Restoration of degraded lands, excl. those in Arid, Boreal, and Polar regions, irrespective of land cover/use	0.47 *11.8*	0.93 *23.4*	1.4 *35.1*
C) Restoration of degraded, Agricultural Lands only	0.21 *5.1*	0.41 *10.2*	0.61 *15.3*
D) Restoration of degraded Agricultural Lands, Extensive Grasslands, and Regrowth Forest	0.26 *6.5*	0.52 *13.0*	0.78 *19.4*
E) Improved management of 'non-degraded' Agricultural Lands*	0.09 *2.1*	0.17 *4.3*	0.26 *6.4*
F) Improved management of 'non-degraded' Agricultural Lands, Extensive Grasslands, and Regrowth Forest*	0.14 *3.5*	0.28 *6.9*	0.42 *10.4*

[1] Land cover categories from IMAGE2.1; [2] Degraded land (severity) from GLASOD; [3]Contemporary SOC stocks from WISE; [4]Major Agro-Ecological Zones after FAO-IIASA (see App. 3); [5]For general assumptions used for 'Low', 'Medium' and 'High' sequestration potential for scenarios A, B, C & D see Table 8; [*] For scenarios E & F, corresponding with the 'stable' lands of GLASOD, the increases for Low, Medium and High are set at 4, 8 and 12 %, respectively.

Based on our simple assumptions it is estimated that, from a 'technical point of view', from 16.1 to 48.3 Pg C can be can be sequestered in the degraded lands of the world over the next 25 years, upon restoration (scenario A in Table 9). If soils of the Arid, Polar and Boreal AEZs are excluded from the analyses, in view of their overall limited possibilities for 'reclamation', this range would be 11.8 to 35.1 Pg C (scenario B). Scenario C only considers the restoration of degraded agricultural lands, giving a sequestration potential of 5.1 - 15.3 Pg for the region under consideration (scenario C). In scenario D, areas of Extensive Grasslands and Forest Regrowth are considered in addition to the Arable Lands of IMAGE2.1; the range then is from 6.5 to 19.4 Pg C. Finally, if one would only consider potential increases, via improved management, for the Arable lands that occur on the 'stable' lands of GLASOD, the range is from 2.1 to 6.4 Pg C (scenario E). If like for scenario D, areas of Extensive grasslands and Forest Regrowth are considered in addition to the Arable lands, improved management of the 'stable' lands would give a possible increase of 3.5 to 10.4 Pg C over the next 25 years. Thus a likely increase would be in the order of 14 ±7 Pg C for scenarios C and E, and 20 ±10 Pg C for scenarios D and F.

From scenarios C and E and scenarios D and F, respectively, it follows that, on average, from 0.58 (= 0.41+ 0.17) Pg C yr^{-1} to 0.80 (= 0.52 + 0.28) Pg C yr^{-1} can be sequestered in the soils of Arable Lands, resp. Arable Lands, Extensive Grasslands, and Regrowth Forest over the next 25 years; this corresponds with about 9-12% of the anthropogenic CO_2-C produced annually. By comparison, Reichle *et al.* (1999) estimated from 0.85 - 0.90 Pg C yr^{-1} (to 1 m depth) can be sequestered over the next 25-50 years in soils of agricultural lands.

The estimates in Table 9 are within the range of historic C-losses estimated for the last 100 years, which may be seen as an upper limit of what may be achievable in terms of SOC sequestration by improved and appropriate management. As indicated earlier in this report, deforestation and changes in land use during the period 1860-1984 have been estimated to cause losses of 150 ± 50 Pg C by Bolin (1986b). Scharpenseel (1993) calculated that 30 to 60 Pg C have been lost from cultivated soil during the last 100 years; assuming that 50% of historic losses of SOC can be 're-sequestered' via improved management of mainly croplands, this would amount to a maximum of 15-30 Pg C. Cole, (1996) estimated from 20 to 30 Pg C may be re-sequestered over the next 50 to 100 years in the world croplands.

From a theoretical point of view, the potential exists for even higher contributions than those shown in Table 9 over the next 50 years. However, in order to realize the potentials modelled for the various scenarios, intensive management of and/or manipulation of a significant part of the globe's terrestrial surface will be needed. Thus, although often feasible from a purely technical point of view, some of these increases may well be unrealistic in terms of economic, environmental and societal/cultural implications.

6.9 Concluding remarks

There are many uncertainties in the field data on SOC sequestration potentials, global databases used, and assumptions made in the model, as well as possible impacts of climate change on soil carbon sequestration over the time-period under consideration (see Chapter 7).

Within the framework of this short exploratory study, no attempt has been made to consider different carbon sequestration potentials by type of soil degradation, for instance for water erosion versus salinisation, by broad agro-ecological zone and land use type. These aspects, however, might be considered in a follow-up study.

Table 10 gives an overview of the feasibility of the numerous strategies, as reviewed in Chapter 5, available for increasing the organic carbon content in cultivated soils. These include four broad classes (Bruce *et al*., 1999): (a) reduction in tillage intensity; (b) intensification of cropping systems; (c) adoption of yield-promoting practices, including improved nutrient amendments; and (d) re-establishment of permanent perennial vegetation.

Table 10 Feasibility of management practices that can increase organic carbon content in the soil (Partial list; after: Bruce *et al.*, 1999)

Management Practice	Feasibility	Relative Carbon Gain
Cultivated land		
Adoption of reduced- or no-till	H	M
Use of winter crops	M	L
Improved crop nutrition and yield enhancement	H	L
Elimination of summer fallow	M	M
Use of forages in rotation	M	M
Use of improved varieties	H	M
Use of organic amendments	M	M
Irrigation	L	H
Re-vegetated or Set-aside land		
Improved grazing regime	L	H
Soil/water conservation measures	H	H
Reversion to woodland	L	H
Pasture land		
Improved grazing regime	M	M
Fertilizer application	H	M
Use of improved species/varieties	M	M
Irrigation	L	M
Rangeland		
Improved grazing regime	L	L
Degraded land		
Reversion to native vegetation	M	H
Establishment of fast-growing crops	M	H
Application of fertilizers	H	M
Application of organic amendments	M	H
Drainage/leaching of saline soils	L	L

Note: Feasibility of Management practice is expressed per unit area (L=low ; M= Medium; H= High).

Recommendations in Table 11 are based on the findings of an international workshop (Carter and Hall, 1995) and list anticipated effects of additions and management options in increasing carbon sequestration in terrestrial ecosystems within a 50 yr time frame.

Table 11 Additions and management options to increase carbon sequestration in terrestrial ecosystems within a 50 yr timeframe[1] (Source: Carter and Hall, 1995)

Options	Terrestrial Ecosystems				
	Forest Ecosystems			Grassland & Savannas	Agricultural land
	Boreal	Temperate	Tropical		
A - Additions					
(1) Nutrients	2-3	2	1	3	2
(2) Water	1	1	1	1	2
(3) Pest control	3*	3*	1	1	1
B - Forest Management					
(1) Afforestation	4-5	5-4	5-4	4-5	5
(2) Reducing deforestation	4-5	1-2	5	n/a	1
(3) Enhanced Management	1-2	2-3	1-2	1	n/a
C - Agronomic Management					
(1) Enhanced management	n/a	n/a	n/a	n/a	3-4
(2) Change agroecosystem	n/a	n/a	n/a	n/a	3
D - Bio-products	4-5	5	5	1	2-3
E - Disturbance and Protection					
(1) Fire	3	3	3**	3**	1
(2) Erosion	2	2	3	2-3	3-4
F - Herbivores	n/a	n/a	n/a	2	1
G - Land Use Policy	4-5	5	5	5	5

Notes:

1 - Based on a global potential for mean carbon storage over 50 years. Options not listed in order of importance.

2- Qualitative rating from 1(small) to 5 (large potential). For additional details on rankings see Carter and Hall (1995).

* Socially unacceptable from an environmental point of view.

** Overriding influence of Land Use Policy

Although the C content of most agricultural soils is low compared to that of natural peatlands or forest soils, for example, the greatest potential to increase current soil C stocks will probably be through improved management of agricultural land, particularly degraded croplands (e.g., Lal *et al.*, 1998b; Smith *et al.*, 1997b). There is good potential for C mitigation in the tropics, where many soils are somewhat degraded, through improved management (Batjes and Sombroek, 1997; Goreau, 1990; Sauerbeck, 1993), as well as for C-sequestration in the temperate regions (Smith *et al.*, 1997c). Forests soils of the cold temperate and tropical climatic zones have the greatest potential to sequester carbon compared to the other climatic zones (Vogt *et al.*, 1995). Management and restoration of wetland and peat soils also deserve careful consideration as viable mitigation options.

Opportunities for carbon sequestration in the biomass and soils of terrestrial ecosystems will vary with the (agro)ecosystems and/or agro-ecological zone. These can be summarized as follows (Reichle *et al.*, 1999):

a) On forest lands, the focus should be on below-ground carbon (in stable pools), and on long-term management and utilization of standing stocks, under-story, ground cover, and litter.
b) For agricultural lands, that is mainly cropland and grasslands but also biomass/fuel croplands, the focus should be on increasing organic carbon in the stable SOC pools.
c) In case of degraded lands, restoration can offer significant benefits in terms of carbon sequestration potential, both in the soil and above ground.
d) In case of wetlands and peatlands, the focus should be on conservation and/or re-turning reclaimed-wetlands to their natural state, keeping in mind any potential adverse environmental effects (see 5.8).

7 Possible Effects of Climate Change on Soil Carbon Dynamics

7.1 General

Changes in soil carbon content, associated with higher ambient CO_2, are difficult to measure due to the vast amount of carbon already present in the soil. Prentice and Fung (1990) suggested that terrestrial vegetation and soils would act as a large sink of atmospheric CO_2 if its concentration were twice the present level. Various evidence supports a substantial terrestrial carbon sink in the north (Cao and Woodward, 1998; Fan *et al*., 1998), while data confirming a tropical sink are still limited (Keeling *et al*., 1996). Direct measurements and modelling of CO_2 fluxes, however, show that tropical ecosystems are taking up carbon (Phillips *et al*., 1998; Tian *et al*., 1998), but part of this 'tropical' carbon sink may be obscured by C-releases from deforestation, logging, increased fragmentation and edge-mortality, regional changes in climate (Fearnside, 1997; Keeling *et al*., 1996; Nepstad *et al*., 1999; Phillips *et al*., 1998). Fertilization by anthropogenic N emissions likely constitutes a significant portion of the 'missing' CO_2 sink (Hudson *et al*., 1994), but this effect now seems to have been overestimated for northern temperate forests (Nadelhoffer *et al*., 1999). Alternatively, it is possible that northern temperate forests that were stimulated by nitrogen in the past are now saturated with N (Schindler, 1999). The fact that the average growing season has extended by 10.8 days in Europe since 1960 (Menzel and Fabian, 1999), which likely contributes to increased biomass formation, may also account for part of this sink. Hulme (1999), however, argue that for some regions the impacts of human-included climate change (by 2050) will be undetectable relative to those due to natural multi-decadal climate variability. Clearly, the potential effects of changing climate and higher atmospheric CO_2 levels on soil carbon sequestration are highly interactive and complex. Modern methods are still inadequate to identify global sinks of less than 0.5 Pg C yr^{-1} with confidence (Schindler, 1999).

7.2 Interactions between NPP, nutrient and water availability, and organic matter decomposition

Both net primary productivity (NPP) and organic-matter decomposition, being (micro)biologically mediated, are likely to be enhanced by increasing temperature, provided water and nutrient supply are not limiting. Idso and Idso (1994) showed that the percentage increase in plant growth produced by raising the atmospheric CO_2 content is generally not reduced by less that optimal levels of light, water or soil nutrients, nor by high temperatures, salinity or gaseous air pollution. In fact, more often than not, the relative growth-enhancing effects of atmospheric CO_2 enrichment are greatest when resource limitations and environmental stresses are most severe (Idso and Idso, 1994). Generally, under elevated CO_2

concentrations, a greater proportion of fixed carbon is allocated below-ground, potentially increasing the capacity of below-ground sinks (Canadell *et al*., 1996). The biological control mechanisms which plants have evolved to acclimatise to shifts in source-sink (photosynthate) balance caused by elevated CO_2 concentrations are complex, and will only be fully elucidated by probing all scales along the hierarchy from molecular to ecosystem (Wolfe *et al*., 1998).

A strong relationship occurs between soil N and SOM accumulation but not with the soil C/N ratio (Vogt *et al*., 1995). Under elevated CO_2-levels, most plants are found to produce tissues that contain more carbon and less nitrogen. The assumption then is that this CO_2-induced increase in C/N ratio, and possibly increased lignin content, will lead to reduced rates of decomposition (Ball, 1997), and thereby facilitate carbon sequestration in the soil. A synthesis by Cotrufo *et al.* (1998) showed an average 14% reduction of N concentrations in plant tissue generated under elevated CO_2 regimes. However, elevated CO_2 concentration appears to have different effects on the N concentrations of different plant types (Johnson *et al*., 1997), as the reported reductions in N content have been larger in C_3 plants than in C_4 plants (Cotrufo *et al*., 1998). Under elevated CO_2, plants changed their allocation of N between above- and below-ground components: root-N concentrations were reduced by an average of 9% compared to a 14% average reduction for above-ground tissues. Although the CO_2 concentration treatments represented a significant source of variance for plant N concentration, no consistent trends were observed between them by Cotrufo *et al.* (1998). There is still a need to explore what implications the reported changes in N-concentrations will have for litter decomposition, N-mineralization, and related feedbacks on soil carbon pools (see Watson *et al*., 1996). N-poor ecosystems may respond more to elevated CO_2 concentration than N-rich ones in the long term (Cannel and Thornley, 1998; Idso and Idso, 1994). In the experiment by Johnson *et al.* (1997), elevated CO_2 concentration caused reductions in soil available N and P and increases in exchangeable Al^{3+}, but it had no consistent or significant effects upon total C, total N, exchangeable Ca^{2+}, K^+, or Mg^{2+}.

Körner (1996) illustrated there is another pathway by which CO_2-fertilization may influence the soil environment. That is, through priming-effects of increased rates of turnover of fine roots and higher exudation of low-molecular-weight organic compounds to the rhizosphere (Paterson *et al*., 1997). Increased root production under high ambient CO_2, could also enhance rhizodeposition (Gardon, 1996) which, in turn, could affect rhizosphere processes and enhance rates of carbon cycling (Fitter *et al*., 1996; Rogers *et al*., 1994; Schapendonk *et al*., 1997). There may be also a feedback mechanism in which elevated CO_2 causes an increase in substrate release in the rhizosphere by non-mycorrhizal plants, leading to mineral nutrient sequestration by the expanded microflora and a consequent nutritional limitation on plant growth (Diaz *et al*., 1993). Quantitatively, root exudation may be much less important in forests than in grasslands. In a study with *Calluna vulgaris,* Verburg *et al.* (1998) found that it is not the inputs of readily decomposable rhizodeposits but rather continuous inputs of more recalcitrant structural root material, at elevated CO_2, which may cause the soil to become a sink for C. A wide range of shoot-to-root ratios, in response to elevated CO_2 concentrations, may occur depending on species, and nutrient and water conditions, as shown by a review by Norby and Cotrufo (1998). Increased belowground allocation of C, at high ambient CO_2, could cause soils to act as a sink for atmospheric C in so far soil C levels are limited by carbon inputs rather than by the soil's C-protection capacity (Hassink, 1996).

Norby and Cotrufo (1998) observed that in order to evaluate the effect of rising atmospheric CO_2 concentrations and associated climate variables on the sequestration of carbon, decomposition must be studied in relation to plant productivity and total litter input. Shifts in plant community composition in response to high CO_2 may have dramatic effects on litter chemistry, and on carbon accumulation in soil. Litter chemistry does influence decomposition, but the changes that have been observed in experiments of increased atmospheric CO_2 concentration are simply too small to have a detectable effect on decomposition. Similarly, studies on the transition from conventional to organic or low-input practices showed that increases in SOM occur slowly and take several years to detect (Clark *et al*., 1998). Total litter input is much more important in this respect according to Norby and Cotrufo (1998). Species composition has more influence on C and N cycles than elevated CO_2 so that any effects of increasing atmospheric CO_2 on species composition may be a more important feedback from CO_2 concentration to SOM than CO_2 effect on litter quality of a given species (Gahrooee, 1998). Ecosystem development in Hawaii has been shown to be affected greatly by long-range transport of aerosols rich in nutrients, showing the dynamics of long-term soil and ecosystem development cannot be evaluated as local phenomena (Chadwick *et al*., 1999). Changes in the dynamics of soil organic matter could strongly affect nutrient mineralization and soil acidity, which are among the most important factors that determine the diversity in species-rich communities (Berendse, 1999).

Changes induced by high CO_2 in the allocation of carbon to leaves, wood or roots will alter decomposition dynamics. For example, increased root length may lead to a more intensive soil exploration by roots (Janssens *et al*., 1998). Other environmental changes that will accompany the increasing concentration of CO_2 , such as warming and increased nitrogen deposition, will certainly influence decomposition rates, but their interactions with rising CO_2 concentrations are difficult to predict (Cotrufo *et al*., 1998). According to Nadelhoffer (1999), however, it is unlikely that elevated nitrogen deposition is a major contributor to the putative CO_2 sink in forested northern temperate regions. With respect to effects of a limited water supply, observations of Schapendonk *et al.* (1997) led to the conclusion that the increase in NPP and soil carbon sequestering did not require more water because the CO_2 effects on the Leaf Area Index (LAI) and Water Use Efficiency (WUE) were compensating each other over a wide range of varying irradiances. Work by Bettarini (1998), does not support the hypothesis that long-term exposure to elevated CO_2 concentration may cause adaptive modification in stomatal number and in their distribution.

Organic matter decomposition is likely to be stimulated more than NPP by increased temperatures, which would release more CO_2 from the soil to the atmosphere upon global warming (Kirschbaum, 1995; Schimmel *et al*., 1990). Kätterer *et al.* (1998) showed the temperature dependence of organic matter decomposition at a low temperature (< 5 ^{o}C) should not be based on Q_{10}-functions. Boone (1998) determined that the Q_{10} value between soil respiration is 4.6 for autotrophic root respiration plus rhizosphere decomposition, 2.5 for respiration by soil lacking roots, and 3.5 for respiration by bulk soil. As plants in a higher CO_2 atmosphere increase their allocation of photosynthate to roots, the findings of Boone *et al.* (1998) suggest that soil respiration should be more sensitive to elevated temperatures, thus limiting carbon sequestration by soils. Model studies by Klein Goldewijk *et al.* (1994), however, suggest the effects of increased temperature and water availability on soil respiration - implying CO_2 release - are smaller than those associated with the CO_2-fertilization effect. Liski *et al.* (1998b) pointed at the fact that the decomposition of 'old' organic matter is less sensitive to increased temperature than is the decomposition of young litter, predicting the C storage of boreal forests to increase.

Jenkinson (1991) concluded that their model should be coupled to models predicting changes in global plant production, while (Gorissen, 1996) indicated that it may be just as important to implement changes in plant quality into simulation models describing C dynamics in a changing climate. The later because these effects are likely to be substantial in counteracting the stimulated decay of plant debris and SOM under increasing temperatures. According to Van Ginkel *et al.* (1996), the combination of higher root yields at elevated CO_2 combined with a decrease in root decomposition will lead to a longer residence time of C in the soil and probably to a higher C storage. Nutrient availability and the response of the soil microbial biomass (size and activity) (Niklaus, 1998), as well as mycorrhizal fungal diversity (Van der Heijden *et al*., 1998) play a major role in the processes involved. These require further attention to clarify plant/soil responses in the long-term with regard to sustained simulation of carbon inputs to the soils and the decomposability of roots (Gorissen, 1996). Issues of sustainability in relation to long-term dynamics of soil organic matter and nutrients under alternative management strategies are also important in this context (van Keulen, 1995).

Changes in mycorrhizal community structure of functioning, under climate change, could have important implications for the carbon cycle at both the ecosystem and the global level (Staddon and Fitter, 1998). In particular, the amount of carbon being transferred to the soil might be altered, leading to changes in the amount of carbon stored there. The loss of biodiversity in soils, including that of arbuscular mycorrhizal fungi (AMF) in agricultural systems, represents an understudied field of research (Van der Heijden *et al*., 1998). Effects of elevated CO_2 on rhizosphere carbon flow and soil microbial processes also need further consideration (Paterson *et al*., 1997).

It appears that little or only a small portion of the exuded labile-carbon can be stabilized in soil organic matter through interactions with clay minerals (Murata *et al*., 1995), contrary to other more recalcitrant plant constituents such as lignin and cellulose (Hungate *et al*., 1997). However, even a small change/increase in physical protection of labile organic substrates from microorganisms could be important for carbon sequestration in the soil (Norby and Cotrufo, 1998). Increased allocation of C below ground, resulting from CO_2 fertilisation, may profoundly affect microbial processes, as will increased soil temperatures and changes in soil moisture. Little is known about the possible impacts (Kamplicher *et al*., 1998). According to Davidson (1994) direct microbial responses to these changes are reasonably well known, while complex interactions, such as effects on N fixation, N mineralisation, denitrification, and ratios of trace gas emissions are equally important but are difficult to predict at present. Similarly, effects of increasing CO_2 and temperature on leaf nutritional quality and palatability and implications for herbivorous insects, plant litter decomposition, and on soil C cycle are still poorly understood (Brooks and Whitakker, 1998; Dury *et al*., 1998 ; Penuelas and Estiarte, 1998).

Pielke *et al.* (1998) reviewed the short-term (biophysical) and long-term (up to around 100 yr time scales; biogeochemical and biogeographical) influences of the land surface on weather and climate. Effects of land use practices on regional climate may overshadow larger-scale temperature changes commonly associated with observed increases in concentrations of CO_2 and other greenhouse gases (e.g., Stohlgren *et al*., 1998). The response of nitrogen cycle processes to anticipated changes in climate will depend both on the direct effects of these changes - including increased temperature, changes in rainfall amount and distribution, and increased atmospheric CO_2 - and on indirect effects such as modifications of land use cropping patterns prompted by the warmer environment (Bradbury and Powlson, 1994).

7.3 Equilibrium and transient models

Carbon cycling models for the terrestrial biosphere have been reviewed by Klein Goldewijk and Leemans (1995). Various researchers (King *et al*., 1997; Post *et al*., 1996) reported an increase in soil organic carbon content in their transient and equilibrium climate change scenarios in which NPP was varied as a function of ‘climate and atmospheric CO_2-concentration’, as opposed to a decrease for scenarios which considered solely ‘climate change’ respectively ‘climate change and concomitant NPP change’. The results of Post *et al.* (1997) suggest that the temporal evolution of the missing C sink over the period 1990-1998 could be a response of the terrestrial biosphere to changes in climate and atmospheric CO_2 concentration or perhaps climate alone. King *et al.* (1997) modelled that with both ‘climate and a doubling of atmospheric CO_2' the increases in NPP and litter inputs to the soil more than compensated for any climate stimulation of decomposition and lead to increases in global ecosystem carbon of 15-18%. King *et al.* (1997) and other authors have pointed at the need for including possible shifts in vegetation types, induced by global change, in ecosystem carbon models. Possible implications of increased litter production for plant biodiversity, and implicitly soil C dynamics, have been reviewed by Berendse (1999). Cao and Woordward (1998) incorporated submodels of soil biogeochemistry and vegetation distribution in their ‘terrestrial ecosystem model’, making it possible to investigate the interacting effects of climate, vegetation and soils on the carbon exchange between terrestrial ecosystems and the atmosphere.

Cao and Woodward (1998) considered three scenarios. First, they modelled that NPP, NEP and soil carbon stocks would increase substantially under ‘CO_2 increase but no climate change’. In the scenario ‘climate change with no CO_2 increase’, they predicted wide intern annual fluctuations in the carbon fluxes and differential responses among various ecosystems. Finally, when ‘both CO_2 concentration and climate change’ were considered, NPP, NEP and carbon stocks all increased. From 1861 to 2070 simulated global NPP increased by 36%. Global NEP varied between -6.5 and +5.3 Pg C yr^{-1} responding positively to changes in CO_2 and precipitation, but negatively to increases in temperature. NEP increased significantly from the 1950s along with the rapid increase in atmospheric CO_2 concentration, but it steadied after the year 2020 as the CO_2 fertilization effect saturates and is diminished by climate change. The decadal-mean NEP becomes positive from the 1910s and increases to 1.1 Pg C yr^{-1} in the period 1950-1990, and to 3.2 Pg C yr^{-1} in the period 2030-2070. Based on the scenarios of Cao and Woodward (1998), this would cause an increase of 309 Pg C in soils (≈130 Pg C) and vegetation (≈ 175 Pg C) from 1861 to 2070, in contrast with an increase of 490 Pg C under ‘CO_2 increase’ alone, and a loss of 87 Pg C for ‘climate change alone’ scenario.

7.4 Concluding remarks

The complexity of terrestrial ecosystem response to increased atmospheric CO_2 concentrations and changes in climate, as illustrated by the examples above, shows the need for modelling approaches. Few studies consider interactive effects of temperature and moisture limitations on plant responses and decomposition rates under ‘elevated CO_2', as extensively reviewed by Bazzaz *et al.* (1996). Transient effects of ‘CO_2 and climate change’ on vegetation composition and distribution generally are not yet considered (e.g., Cao and Woodward, 1998). Possible impacts of increased atmospheric deposition of nitrogen (Jefferies and Maron, 1997) and other substances on soil chemical, physical, and biotic processes and ultimately carbon-sequestration should be considered also in the models. This also applies for impacts caused by changing land-management practices, causing possible shifts in species diversity and functions (e.g., Betts *et al*., 1997; Bouma *et al*., 1998; Brinkman and Sombroek, 1996; Jefferies and Maron, 1997; Norby, 1997). The possibility of unanticipated, sudden releases of pollutants, previously believed to be held firmly in soils into the environment as ‘Chemical Time Bombs’, should also be considered in this context as they may negatively affect soil carbon dynamics (Batjes and Bridges, 1992; Stigliani *et al*., 1991). Clearly, the data requirements for running such complex, integrated terrestrial ecosystem models will be high (Cramer and Fischer, 1997), and the possibilities for crucial model evaluation may be limited.

8 Scope for Monitoring and Verification of Changes in Soil Carbon Content

8.1 General

The carbon balance is still ill-defined for agricultural lands so that their role in the global C balance cannot be accurately estimated (Buyanovski and Wagner, 1998a). Several sources of uncertainty may be encountered when modelling changes in the organic carbon content of soils upon a predefined change in land use. The following discussion mainly concerns the soil attribute data. Information will also be required on the type and extent of the various types of land use, and seasonal changes therein, which form an additional source of uncertainty. Inter-annual variations in environmental conditions, for example the occurrence of severe dry years or ENSO events (Nepstad *et al*., 1999; Prince *et al*., 1998) or periods with higher than normal temperatures (Kindroth *et al*., 1998), through their various effects on NPP make it cumbersome to obtain a clear picture of trends in SOC irrespective of spatial scale. With respect to the 'missing carbon' sink, Schindler (1999) observed most modern methods are still inadequate to identify global sinks of less than 0.5 Pg C yr^{-1} with confidence.

8.2 Uncertainties associated with sampling methodology

Changes in average carbon pools, particularly in the soil, cannot be derived directly (Wolf and Janssen, 1991). Jonhson (1992), for example, pointed at the uncertainties associated with chronosequence studies. They cannot completely replace long-term experimental chronological studies where randomization can control for pre-existing differences between sites and most aspects of post-clearing site history and land management can be controlled (Neill *et al*., 1998). Carbon concentration (g C g^{-1} soil or wt%) is an unreliable indicator of C loss/gain, because the total amount (mass) of C present within the soil also depends on bulk density, stoniness, and depth of soil. Each of these attributes have their own associated suites of uncertainties (Batjes, 1996b; Eswaran *et al*., 1995; Sombroek *et al*., 1993).

Sampling intensity

Random sampling errors for composite soil cores taken from plots used in most agronomic studies are in the order of (at least) ±10 to 15 % of the mean for surface soils on plots established on a relatively uniform area, where all the soil belong to one series (Greenland, 1995). Buol *et al.* (1997), quote a study by Wilding and Drees (1983), which indicates that for the 'more variable' soil property which soil organic carbon is, estimates of the mean within ±10% at the 95% confidence level are unrealistic because of the large number of samples are needed.

The error in using one single soil profile will be much larger, as is illustrated by the high coefficients of variation in SOC observed for major soil groupings of the FAO-Unesco legend (Batjes, 1996b; Sombroek *et al*., 1993) and for orders of USDA Soil Taxonomy (Eswaran *et al*., 1995; Kimble *et al*., 1990). In view of the nature of the frequency distribution of soil carbon data in Australian Great Soil Groups, Spain and Isbell (1983) considered it preferable to present summaries of these data in terms of medians and interquartiles instead of averages. Land use will mostly affect/dominate the more variable content of carbon in the surface soils, while the amount of carbon in the whole soil profile will be determined more by the more ‘stable’ or ‘intrinsic’ soil characteristics reflected in the soil classification (Greenland, 1995; Verburg *et al*., 1995).

Huntington *et al*. (1988) assessed whether an intensive sampling programme can provide estimates of soil C and N pool size with sufficiently small confidence limits that changes of 20 % of the means can be detected. To this avail 60 plots were selected out of a total of 360 from a 25 x 25 m grid, located in a northern hardwood forest ecosystem. They stratified the plots into six elevation bands to ensure adequate representations of the elevation gradient, soil mapping units, topography and tree species. The number of plots per band was weighted by the proportion of the watershed represented by the elevation band. The location of pits within plots was also randomly selected. Sampling was according to depth strata as this was considered more repeatable than per genetic horizon (0-10, 10-20, and 20 cm to bottom of B horizon). Mann (1986) found that a 30 cm sampling depth provided a less variable estimate (r^2 = 0.9) of changes in carbon than a sampling depth of 15 cm (r^2= 0.6). This indicates that a judicious selection of sampling depth will be needed to adequately reflect changes in soil C content upon changes in land use (see under sampling methodology).

Huntington *et al*. (1988) calculated the minimum detectable changes (defined as 1.96 times the standard error) in total C pool sizes for the forest floor and 0-10 cm depth stratum were 5.9 and 2.4 Mg C ha^{-1}, respectively. Based on these findings they concluded that large scale disturbances, such as harvesting of timber, may produce statistically measurable changes within a period of 5 to 20 yr. In practice, however, few studies of modifications in soil organic matter content following a change in land use have considered the kind of sampling intensity as used by Huntington at al. (1988). In view of the high costs, many studies of changes in soil C induced by modifications in land use practices do only consider single pit (e.g., in ‘paired’ studies) or average carbon data. Knowing the intensity of tillage in a given field is also critical when sampling for soil physical and soil chemical properties (Tsegaye and Hill, 1998a; Tsegaye and Hill, 1998b).

Besides the sampling density/scheme the technique of analysis can be important in identifying changes in soil carbon pools. Jones *et al*. (1990), for instance, found that while changes between burned and unburned sites were sufficiently marked after a period of 34 years to be detectable by total analysis, differences between annual, biennial and triennial burnings were more subtle requiring the measurement of mineralization rates and microbial biomass, a more technique (see also: Feigl *et al*., 1995b). In order to determine the effects of land use on soil C, the soil sampling scheme must distinguish clearly between effects on plant roots from those on mineral-soil organic matter (Richter *et al.*, 1990).

During routine soil surveys, soil samples are taken according to the horizonation which reflects the pedogenesis. These depth-layers do not necessarily coincide with the depth zone that will be affected by a change in land use. As land use practices may influence horizonation and soil bulk density in any soil, similar soil depth ranges ‘before’ and ‘after’ cultivation need not correspond with similar volumes of soil, making the comparison for changes in organic carbon content cumbersome. An additional problem is that bulk density data are cumbersome to measure, an alternative being to develop pedotransfer functions. Soil structure/porosity, soil texture/mineralogy and soil carbon content are important determinants of bulk density which ideally should be considered when developing pedotransfer functions for bulk density. Additionally, the volume of coarse fragments should be considered in calculations of carbon density, but this information is not always available.

Spatial averaging

Spatial averaging, as commonly used in computations of regional C stocks, can be improved by a systematic stratification of environmental factors that affect SOM dynamics in a Geographical Information Systems coupled to up-to-date databases and process models (see also 2.3 and 3.3). Improvements in remote sensing (Seitzinger *et al*., 1999), neural networks (Levine and Kimes, 1998), and non-parametric techniques to characterize data sets and models, such as the 'mollifier method' (Keyzer and Sonneveld, 1997), will provide much needed information about the spatial distribution of SOC sources and sinks, and possibilities for scaling (Bouwman, 1999). In principle, changes in SOC can be determined by measuring the change in stocks or the changes in fluxes for given functional types (Bouwman *et al*., 1999). The need for good ground data, for example on soil bulk density and soil hydraulic parameters by major soil type, cannot be overemphasized within this context, often requiring new ground-based surveys (Batjes and Dijkshoorn, 1999; Sombroek *et al*., 1997). Such data sets will also be critical for developing pedotransfer functions for soil hydraulic conductivity for soils of different mineralogical composition (e.g., Rousseva, 1997; Tomasella and Hodnett, 1997; Tomasella and Hodnett, 1998; Van den Berg *et al*., 1997; Wösten *et al*., 1998).

Analytical methods

The way according to which soil organic matter and soil organic carbon content are determined in the laboratory has repercussions on the estimates of soil organic C-reserves. With respect to the commonly used Walkley-Black method, Allison (1965) recommends using a conversion factor of 1.33 to account for incomplete oxidation unless more accurate methods - for example, dry combustion, an estimator of actual organic C - are available for comparison. Similarly, in their study of organic matter in surface soils from Australia, Spain and Isbell (1983) scaled all the Walkley-Black data by this arbitrary (but widely used) average factor to convert them to a level comparable with that obtained by dry combustion. A correction factor > 1 indicates the presence of organic compounds that are resistant to chemical digestion but which are readily broken down by high temperature dry combustion, while a correction factor of < 1 possibly indicates the presence of ferrous iron in the samples (Allison, 1965). Statistical analyses by Grewal *et al.* (1991), for 40 samples from 8 New Zealand soils with a wide range in texture (20-56 %) and organic matter content (2-16 %), suggest an average conversion factor of 1.25 irrespective of the position in surface or subsurface horizon. Re-computation of the individual conversion ratios for the respective horizons, however, indicates a fairly wide range between the individual conversion factors of about 1.1 - 1.6, corresponding with individual recoveries in the range 62 to 85 per cent. In the original study of Walkley and Black (1934) the individual recoveries ranged from 60 to 80 per cent. Bornemisza *et al.* (1979) recommended that 75% oxidation be used as the basis for conversion of Walkley-Black C to total C. These differences in analytical measures mean that there may be a systematic underestimate of total soil carbon, of the order of 10 to 30 %, in many of the analyses reported for surface soils, and rather greater discrepancies for subsoils (Greenland, 1995).

In the 8 soils studied by Grewal *et al.* (1991) there was no clear tendency in the recovery ratio to either decrease or increase consistently with soil depth. Alternatively, Edwards and Todd-Ross (1983) used correction factors for ‘high temperature dry combustion / Walkley and Black’ of 1.08, 0.90 and 1.38 for the 0-15, 15-30 and 30-45 cm depth zones, respectively. Basically, the higher correction factor for the 30-45 cm zone, as compared to the 0-15 and 15-40 cm zone reflects the fact the ratio of relatively inert organic compounds to readily decomposable compounds should increase with depth from the readily decomposable litter and root detritus near the surface. In the soil studied by Edwards and Ross-Todd (1983) the 30-45 cm layer was almost devoid of roots. Richter *et al.* (1990, p. 79) used the ‘uncorrected’ Walkley and Black values when studying the effect of annual tillage on soil organic carbon levels; according to these researchers the correction factors vary too widely. Tiurin (as quoted in the Transactions of the 1930 International Congress of Soil Science, p. 124) suggested a comparison of the titration value with the amount of CO_2 actually liberated by oxidation might be used to characterise the type of organic matter present.

In addition to the above, the organic matter (OM) content has frequently been estimated by loss-on-ignition and organic C content by using a ‘C/OM’ ratio. Considerable error is expected when a single C/OM ratio is used for the conversion, because the ratio varies depending upon soil type and soil depth as the quality/type of organic matter present changes with depth. The review by Nelson and Sommer (1982) showed a range of 0.40 to 0.59 for the C/OM ratio. Schlesinger (1977) criticized his own use of the commonly adopted ‘standard’ correction factor of 58 % (1/1.724) for converting from soil organic matter to carbon because there should be a qualitative change in the nature of organic matter with depth. This aspect may be illustrated by the study of Huntington *et al.* (1989), who observed the carbon concentration decreases more rapidly with depth than the organic matter concentration, resulting in a decrease in C/OM ratio from 0.56 in the Oi + Oe horizon of the forest floor to 0.44 in the Bs2 horizon of a forested Spodosol. In an earlier study, Huntington *et al.* (1988) measured a conversion factor of 0.55 ± 0.02 for 'C/OM loss on ignition' for forest floor samples, and 0.45 ± 0.09 for the mineral soil. The C/OM ratio commonly changes with the type vegetation present. Wolf and Janssen (1991), for instance, used different C concentrations for crop residues, tree litter, manure and soil organic matter (average C/OM ratios of 0.45, 0.48, 0.50 and 0.58, respectively) in their modelling study.

Although the above discussion shows the use of 'standard' conversion factors for 'incomplete oxidation' of 1.33 and 'C/OM conversion' of 58 %, irrespective of the vegetation/crop cover, type of humus and soil depth need not be justified, it will often be necessary to use these conversion factors in studies at the global and regional level as there are no other valid alternatives.

8.3 Detectability of SOC changes

Soil C only responds gradually to changes in agricultural management such as crop rotation, fertilizer input, manure application or tillage. Long-term experiments throughout the world provide useful information on the long-term effects of agricultural practices on soil conditions and function, as influenced by the ever-changing conditions of climate, social, cultural, and technological considerations with time. Although time scales of at least 15 are required for C concentrations in soil to reach equilibrium when a new cropping system is imposed on a soil, as little as 5 years can be sufficient to begin to observe trends in (bulk) organic C levels in some situations (Dick *et al.*, 1998). Lyon (1998) pointed out this type of information can not be obtained from 2 to 3-year duration experiments. The increase in SOC may be nil or slight in the first 2 to 3 years following introduction of conservation-tillage (Franzluebbers and Arshad, 1996) and large in the next 5 to 10 years, following a sigmoid response curve. The majority of the research reported in scientific journals, however, consists of such short-term type of studies.

Conclusions based on 10 to 20 years of data can be very different from those based on 50-plus years of data. According to Rasmussen *et al.* (1998a), most soil C changes require at least 20 years to be detectable by present analytical methods because of the small yearly inputs of C into a much larger soil matrix, a substantial portion of which is relatively inert. With respect to C gains in 'reclaimed' eroded soils it is possible to measure changes in soil C in a period of 5 years, using sensitive techniques such as determination of light-fraction organic matter (Izaurralde *et al.*, 1998). McCarty *et al.* (1998) found that transformations of a soil profile from that of a typical plough soil to one characteristic of no-tillage occurred rapidly within a 3-yr period; during this period stratification of organic matter in the profile progressed significantly towards that which occurs 20 years of no-tillage treatments. Much of the early changes in SOC upon a change in land use occur in the so-called light-fraction rather than in the total pools of C and N, so that measurements of changes in these 'reactive' pools are needed (Bell *et al.*, 1999; Janzen *et al.*, 1992; McCarty *et al.*, 1998). This will allow for more detailed investigation of the effect of agricultural management systems on SOM changes at the molecular level (Arai *et al*., 1996; Ellerbrock *et al*., 1999; Madari *et al*., 1997). The use of radio-isotopic tracers is discussed in greater detail below.

Radio-isotopic tracers

Soil C dynamics) over the time scales ranging from a few years to several centuries that are relevant to understanding the consequences of human-induced land use changes) can be studied using stable ^{13}C tracers (e.g., Bernoux *et al*., 1998c; Boutton *et al*., 1998; Buyanovski *et al*., 1997). Plants with C_3, C_4 and CAM photosynthesis have unique $\delta^{13}C$[1] values which are not altered significantly during decomposition and soil organic matter formation. C_3 plants, with a Calvin cycle, are more discriminative against the natural radioisotope ^{13}C than those with a Hatch-Slack cycle (C_4). This means that changes in climax vegetation from either C_4 to C_3 plants or from C_3 to C_4 plants provide a natural labelling of organic matter.

Many researchers have used radioisotopic labelling to study the dynamics of organic matter incorporation into soil, both in time and with depth. For instance, where forest is cleared to grassland, that is for a C_3 to C_4 vegetation (Balesdent *et al*., 1988; Bernoux *et al*., 1998c; Cerri *et al*., 1985; Choné *et al*., 1991; Skjemstad *et al*., 1990), or where a climax grassland vegetation is reafforested that is from C_4 to C_3 vegetation (Balesdent, 1987; Martin *et al*., 1990). Mean $\delta^{13}C$ values of C_3 plants are in the order of -23 to -28 $^{o}/_{oo}$, while this is about -12 to -16 $^{o}/_{oo}$ for C_4 plants (Balesdent *et al*., 1988; Choné *et al*., 1991; Lobo *et al*., 1990; Martin *et al*., 1990).

A difficulty in applying the $\delta^{13}C$ technique in soil comparative studies of organic matter dynamics in a particular soil (series) is to ensure that the 'disturbed' soils were similar to the reference soils at time zero, in terms of original soil carbon content between different sites under forest (Choné *et al*., 1991). Also leaves, stems and roots of the same plants show small differences in $\delta^{13}C$ values (Medina *et al*., 1986), and ^{13}C isotopic composition of organic matter in soil is not constant with depth (Bertram, 1985; Volkoff and Cerri, 1988). Two scenarios are possible for the deeper layers depending on whether the soil is under forest or savanna. In the case of forest there is a progressive enrichment in ^{13}C with depth (Bonde *et al*., 1991; Volkoff *et al*., 1978), whereas under savanna the opposite pattern is observed (Volkoff and Cerri, 1988). Changes with depth may also be due to carbon isotope fractionation during microbial degradation of soil organic matter. Microbes would preferentially 'consume' the isotopically lighter material because it is easier energetically to 'decompose' its bondings, whereby the residual organic matter becomes enriched in ^{13}C (Bertram, 1985). The decrease in $\delta^{13}C$ values from fresh under-canopy litter to decomposing litter and soil organic matter is also related to changes in the chemical composition during the decomposition process (Medina *et al*., 1986). Bol *et al.* (1999) showed soil forming processes are significant in determining the carbon isotope signatures retained in SOM.

1 $\delta^{13}C$ (in %o) = 1000 %o x{ $(^{13}C/^{12}C)_{sample}$ - $(^{13}C/^{12}C)_{PDB}$) } / $(^{13}C/^{12}C)_{PDB}$; with PDB the standard.

8.4 Monitoring systems

Global monitoring systems are needed that register changes in soil quality with time in conjunction with changes in the driving biophysical and socio-economic forces. A Global Terrestrial Observing System (GTOS, 1995), conceived by FAO, UNEP, UNESCO, WMO and ICSU/IGBP, has started, albeit still at a low pace in view of limited funding. With respect to carbon turn-over and sequestration, monitoring systems should consider SOM fractionation schemes that produce repeatable, interpretable and meaningful data on carbon pools, whereby the impact of different land-uses and climate change on the dynamics and size of soil carbon pool can be studied in space and time. Remote sensing techniques, while providing a powerful tool for assessing the dynamics of the biosphere and climate (Burrows, 1999; Seitzinger *et al*., 1999), are not yet suited to monitor changes in most soil chemical and physical properties. Although soil organic carbon status can be related to normalized difference vegetation index (NDVI) data, additional collection and manipulation of site information remains necessary for local, regional and global scale prediction (Merry and Levine, 1995). Operational and scientific problems associated with the accuracy and regional representativeness of the various spatial and attribute data in global databases are well known, yet difficult to remedy in studies based on available data (Bouwman *et al*., 1999; Cramer and Fischer, 1997; Leemans and van den Born, 1994).

There is an unfortunate tendency to neglect ground surveys of soil and terrain conditions and the supporting laboratory analyses of soils at representative sites. Compensation and recognition to so-called 'data collectors' should become a standardized practice to encourage critical data collection, harmonization and analysis efforts, critical for monitoring changes in SOC pools as needed for verification of targets (to be) proposed for the enforcement articles 3.3 and 3.4 of the Kyoto Protocol within the commitment period.

9 Conclusions

! Terrestrial ecosystems are thought to be a major sink for carbon at the present time, the 'missing sink' amounting to about 1-2 Pg C yr^{-1}. Variable sequestration of carbon by the terrestrial biosphere is a main cause of observed year-to-year variations in the rate of atmospheric CO_2 accumulation. The Kyoto protocol currently restricts the allowable terrestrial sources and sinks of carbon to strictly defined cases of 'afforestation, reforestation and deforestation'. Appropriate management of the terrestrial biosphere and especially of soils, however, can substantially reduce the buildup of atmospheric greenhouse gases.

! Best management practices with respect to soil carbon sequestration, basically are those that increase the input of organic matter to the soil, and/or decrease the rates of soil organic matter decomposition. Possible implications of the 'CO_2-fertilization effect' on crop growth and organic matter decomposition, although less well understood, should be considered also.

! According to this review, the most appropriate management practices to increase soil C reserves are site specific. Available options will require evaluation and adaptation with reference to major soil type and land use system, and this preferably by agro-ecological region. At the same time, these practices should be socially, politically, and economically acceptable.

! Methods are needed that increase the rate at which soils sequester carbon and the quantities that can ultimately be stored. Soil carbon can be stored in pools with different turnover times. With respect to C sequestration is best to immobilize the atmospheric CO_2-C in soil C-pools having long turnover times. Mechanisms based on C storage in live biomass or litter will sequester C only for a short time and are therefore considered of less importance for long-term C sequestration strategies (Verburg *et al*., 1995).

! Better information on the nature and dynamics of organo-mineral associations will lead to a better understanding of soil structural dynamics, C cycling, and C sequestration in soils, providing a better handle for developing improved approaches to soil management as process-based models. A study of the processes and mechanisms that regulate the stability of organic carbon in selected 'old-agricultural' soils, enriched in phosphorus, would be useful in this context.

! Detection of small increments in soil carbon storage over the relatively short-time frame(s) of relevance for monitoring and verification of article 3.4 of the Kyoto Protocol, in soil will require new techniques, such as ^{13}C Nuclear Magnetic Resonance (NMR) and the use of stable isotopes. Carbon isotope fractioning will mainly be of use where the vegetation has changed from forest to grassland, that is from C_3- to C_4-crops, or vice-versa. Observing changes in carbon stocks and dynamics within C_3 or C_4 systems, however, will require other sensitive techniques. The possibilities of using remote sensing and other non-destructive techniques for monitoring changes in soil carbon stocks need to be researched further.

! Our exploratory scenarios show that, on average, from 0.58 Pg C yr^{-1} to 0.80 Pg C yr^{-1} can be sequestered in the soils of the 'degraded' and 'stable' Arable Lands, resp. Arable Lands, Extensive Grasslands, and Regrowth Forest over the next 25 years; this would correspond with about 9-12% of the anthropogenic CO_2-C produced annually. Our estimates are within the range of historic C-losses, from cultivated soils, published for the last 100 years; the latter may be seen as an upper limit of what may be achievable in terms of SOC sequestration by improved and appropriate management. These largely 'technical' scenarios assume that 'best' management and/or manipulation of a large portion of the globe's soils will be possible, but their implementation need not necessarily be feasible due economic, environmental and societal/cultural conditions.

! Mitigation of atmospheric CO_2 by increased carbon sequestration in the soil, particularly makes sense in the scope of other global challenges such as combatting land degradation, improving soil quality and productivity, and preserving biodiversity. Effective mitigation policies will likely be based on a combination of many modest and economically sound reductions, which confer added-benefits to society. In identifying these 'best practices', due attention must be paid also to any possible adverse environmental and socio-economic effects some of these practices may have.

! Existing global databases of soils and land use are partly outdated, and thus may not be adequate to help quantifying the potential for carbon sequestration at the global and national levels, confirming the need for data consolidation (Cramer and Fischer, 1997; Leemans and van den Born, 1994; Nachtergaele, 1996; Scholes *et al*., 1995).

! Large uncertainties remain associated with rule-based and model-based scenarios for carbon sequestration. Such projections, however, are essential for providing quantified information on possible trends is soil C levels following alternate management practices, possible simultaneous effects of global change and changes in policies. Analyses of feedbacks between different processes and their temporal dynamics need to be considered in integrated models. Predictions of such (process) models will be most useful when they also provide information about the range in uncertainty of the projections; much work remains to be done in this field.

! In conclusion, efforts should be made to allow soil carbon sequestration under the Kyoto protocol. Irrespective of scale and geographic location, a key objective should be to identify and implement best available management practices to improve soil quality, thereby ensuring food productivity and sustainability, while simultaneously reducing CO_2 concentrations in the atmosphere. In identifying these 'best practices', due attention must be paid also to any possible adverse environmental and socio-economic effects they may have.

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Greenhouse Effect (Ed. by R. Lal, J. Kimble, E. Levine and B. A. Stewart), pp. 159-178. CRC & Lewis Publishers, Boca Raton.

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Appendices

App. 1 Selected references on effects of different cropping systems and cultivation practices for carbon sequestration in agricultural soils

Subject	Reference
! slash-and-burn; burning; shifting cultivation	Almendos *et al.*, 1990; Garcia-Oliva *et al.*, 1999; Giovanninni *et al.*, 1998; Greene *et al.*, 1990; Jones *et al.*, 1990; Shang and Tiessen, 1997
! cultivated versus nartive/uncultivated soils	Andreux *et al.*, 1990; Batjes, 1998; Bauer and Black, 1981; Bridges and de Bakker, 1998; Buyanovski *et al.*, 1987; Carter *et al.*, 1998; Davidson and Ackerman, 1993; Ellert and Gregorich, 1996; Rosell and Galantini, 1998; Sandor and Eash, 1995; Sombroek, 1966
! post-harvest residue input to cropland; plant residues as a source of SOM; composting; farmyard manures; organic farming	Atallah *et al.*, 1995; Buyanovski and Wagner, 1986; Buyanovski and Wagner, 1997; Cihacek and Ulmer, 1995; Ellerbrock *et al.*, 1999; Gerzabek *et al.*, 1997; Gijsman and Sanze, 1998; Gregorich *et al.*, 1996; Lal, 1997; Linden and Clapp, 1998; Loess and Ogaard, 1997; Rasmussen and Oartob, 1994; Sommerfeldt *et al.*, 1988; Torbert *et al.*, 1998; Vyn *et al.*, 1998; Woomer *et al.*, 1998
! cropping systems; crop rotations vs. monoculture; legume-based systems; green fallows	Berzseny and Györffy, 1997; Bronson *et al.*, 1998; Drinkwater *et al.*, 1998; Huggins *et al.*, 1995; Lal *et al.*, 1998; Schlegel and Havlin, 1997; Skjemstad *et al.*, 1999
! conventional versus conservation and no-tillage	Ahl *et al.*, 1998; Alvarez *et al.*, 1998; Angers *et al.*, 1995; Bajracharya *et al.*, 1997; Bonde, 1991; Buschiazzo *et al.*, 1998; Carter and Hall, 1995; Cihacek and Ulmer, 1995; Cihacek and Ulmer, 1998; Cole *et al.*, 1996; Dao, 1998; Davidson and Ackerman, 1993; Doran *et al.*, 1998; Franzluebbers and Arshad, 1996; Grace *et al.*, 1995; Grace *et al.*, 1998; Hendrix *et al.*, 1998; Huggins *et al.*, 1998a; Janzen *et al.*, 1998; Kern and Johnson, 1993a; Kern and Johnson, 1993b; Lal and Kimble, 1994; Lal and Kimble, 1997; Lal *et al.*, 1998; Lyon, 1998; Mann, 1985; Mann, 1986; Opoku *et al.*, 1997; Paustian *et al.*, 1998; Peterson *et al.*, 1998; Potter *et al.*, 1997; Puget *et al.*, 1995; Rasmussen *et al.*, 1998b; Richter *et al.*, 1990; Singh *et al.*, 1998; Smith *et al.*, 1998; Taboada *et al.*, 1998; Torbert *et al.*, 1997; Torbert *et al.*, 1998; Watson *et al.*, 1996

! Effects of soil fertility and nutrient management	Batjes and Sombroek, 1997; Berzseny and Györffy, 1997; Borken and Brumme, 1997; Brinkman and Sombroek, 1996; Cannel and Thornley, 1998; Chadwick *et al.*, 1999; Clark *et al.*, 1998; Csathó and Árendás, 1997; Dalal and Mayer, 1986; Doran *et al.*, 1998; Fairhurst *et al.*, 1999; Gregorich *et al.*, 1995; Gregorich *et al.*, 1996; Himes, 1998; Johnston, 1991; Kononova, 1966; Lal, 1997; Lal and Kimble, 1994; Lal *et al.*, 1998; Lilienfein *et al.*, 1998; Loess and Ogaard, 1997; Murata *et al.*, 1995; Nyborg *et al.*, 1995; Powell *et al.*, 1994; Power and Peterson, 1998; Randall *et al.*, 1995; Rasmussen *et al.*, 1998b; Sandor and Eash, 1995; Schwab *et al.*, 1990; Simonson, 1995; Singh *et al.*, 1998; Skinner and Todd, 1998; Sombroek, 1995; Stoorvogel *et al.*, 1993; Tate and Salcedo, 1988; Torbert *et al.*, 1996; Tsegaye and Hill, 1998b; Uehara, 1995; van Keulen, 1995; Van Meirvenne *et al.*, 1996
! Management effects on soil physical properties	Bernoux *et al.*, 1998; Buyanovsky *et al.*, 1994; Chauvel *et al.*, 1999; Huntington *et al.*, 1989; Linden and Clapp, 1998; Mele and Carter, 1999; Mielke and Wilhelm, 1998; Taboada *et al.*, 1998; Thomas *et al.*, 1996; Tiessen and Stewart, 1983; Tsegaye and Hill, 1998a; Wielemaker and Lanse, 1991
! Erosion control and restoration	Doran *et al.*, 1998; Gabriels and Michiels, 1991; Gray and Sotir, 1996; Gregorich *et al.*, 1998; Lal, 1990; Lal, 1997; Lal *et al.*, 1998; Lowrance and Williams, 1988; Squires *et al.*, 1995; Torbert *et al.*, 1996; WOCAT, 1998
! Water management (drainage; irrigation)	Fausey and Lal, 1992; Lyon *et al.*, 1998; Oechel *et al.*, 1998; Stewart, 1995; Sullivan *et al.*, 1997; Szabolcs, 1989
! Environmental side-effects	Batjes and Bridges, 1993; Boeckx *et al.*, 1998; Borken and Brumme, 1997; Hénault *et al.*, 1998; Hütsch, 1998; Isensee and Sadeghi, 1996; Kasimir-Klemedtsson *et al.*, 1997; Neue, 1997; Oldeman, 1994; Römkens and Salomons, 1998; Sauerbeck, 1993; Smith *et al.*, 1997a; Stigliani *et al.*, 1991; Wang *et al.*, 1999
! Long-term experiments/studies	Christensen and Johnston, 1997; Dalal and Mayer, 1986; Huggins *et al.*, 1998b; Mitchell and Entry, 1998; Németh, 1997; Paul and *et al.*, 1997; Paustian *et al.*, 1998; Pierri, 1995; Potter *et al.*, 1998; Powlson *et al.*, 1998; Powlson *et al.*, 1996; Rasmussen *et al.*, 1998a; Skjemstad *et al.*, 1999; Smith *et al.*, 1997b; Smith *et al.*, 1997c; Smith *et al.*, 1997d; Várallyay, 1997; Witter and Kanal, 1998

App. 2 Total stocks and densities of soil organic carbon (SOC) by major categories of contemporary land cover/use

Table A Total stocks and densities of soil organic carbon (SOC) by major categories of contemporary land cover/use (Pg C resp. kg C m^{-2} for upper 0.3 m)

Land cover class (IMAGE2.1)	Spatially weighted SOC pools (Pg C to 0.3 m depth)	Mean SOC density (kg m^{-2} to 0.3 m depth)
Tundra	45.5 - 50.0	7.4 - 8.2
Wooded tundra	20.2 - 21.7	8.8 - 9.4
Boreal forest	154.5 - 160.2	10.3 - 10.7
Cool conifer forest	24.1 - 24.8	11.6 - 11.9
Temp. mixed forest	16.7 - 17.4	8.8 - 9.2
Temp. deciduous forest	9.0 - 9.9	5.3 - 5.8
Warm mixed forest	10.2 - 10.9	5.5 - 5.9
Grassland/steppe	33.0 - 35.6	3.4 - 3.7
Major cities (Peri-urban and urban)	5.1 - 5.3	5.5 - 5.8
Extensive grassland	43.0 - 47.0	3.5 - 3.8
Hot desert	28.1 - 31.6	1.8 - 2.1
Scrubland	12.0 - 12.9	3.6 - 3.9
Savanna	39.2 - 41.1	4.4 - 4.6
Tropical woodland	31.6 - 32.8	5.0 - 5.2
Tropical forest	37.4 - 38.7	6.3 - 6.5
Agricultural land	170.3 - 180.0	5.0 - 5.3
Regrowth forest	4.7 - 5.0	8.3 - 8.7
Major cities (Peri-urban and urban lands)	5.1 - 5.3	5.5 - 5.8

Note:
- Land cover categories are after IMAGE2.1 (Alcamo *et al*., 1998) and soil carbon data from WISE (Batjes, 1996b).
- Global totals differ slightly from those shown in Table 3 due to rounding.
- Agricultural Land comprises both crop-lands and pasture. Regrowth forest is an ill-defined category with respect to soil organic C stocks..

Table B Total stocks and densities of soil organic carbon (SOC) by major categories of contemporary land cover/use (Pg C resp. kg C m^{-2} for first 1.0 m)

Land Cover class (IMAGE2.1)	Spatially weighted SOC pools (Pg C to 1 m depth)	Mean SOC density (kg m^{-2} to 1 m depth)
Tundra	136.1 - 152.8	22.2 - 24.9
Wooded tundra	53.5 - 58.2	23.3 - 25.3
Boreal forest	365.2 - 377.7	24.4 - 25.3
Cool conifer forest	52.9 - 54.1	25.4 - 26.0
Temp. mixed forest	33.0 - 34.5	17.5 - 18.3
Temp. deciduous forest	17.1 - 18.7	10.1 - 11.0
Warm mixed forest	19.7 - 21.3	10.6 - 11.5
Grassland/steppe	67.2 - 72.1	7.0 - 7.5
Extensive grassland	88.6 - 96.6	7.1 - 7.8
Hot desert	52.1 - 58.1	3.4 - 3.8
Scrubland	24.0 - 25.5	7.3 - 7.7
Savanna	74.9 - 78.0	8.4 - 8.7
Tropical woodland	59.1 - 61.1	9.4 - 9.7
Tropical forest	74.2 - 76.7	12.5 - 13.0
Agricultural land	327.5 - 344.5	9.7 - 10.2
Regrowth forest	10.3 - 10.8	18.1 - 19.0
Major cities (Peri-urban and urban)	9.5 - 10.0	10.3 - 10.7

Note:

- Land cover categories are after IMAGE2.1 (Alcamo *et al*., 1998) and soil carbon data from WISE (Batjes, 1996b).
- Global totals differ slightly from those shown in Table 3 due to rounding.
- Agricultural Land comprises both crop-lands and pasture. Regrowth Forest is an ill-defined category with respect to soil organic C stocks.

App. 3 Criteria for Agro-Ecological Zones

Agro-Ecological Zone[1)]	Length of Growing Period (days)	Monthly mean temperature	Daily mean temperature
Tropics, Warm Humid	> 270	all months > 18 ºC	> 20 ºC
Tropics, Warm Season. Dry	60 - 270	all months > 18 ºC	> 20 ºC
Tropics, Cool	> 60	all months > 18 ºC	5 - 20 ºC
Arid Regions	< 60	-	-
Subtropics, summer rains	> 60	one or more months between 5 and 18 ºC	-
Subtropics, winter rains	> 60	one or more months between 5 and 18 ºC	-
Temperate, oceanic	> 60	one or more months between 0 and 5 ºC	-
Temperate, continental	> 60	one or more months < 0 ºC	-
Boreal	-	less than 4 months > 10 ºC	-
Polar and Alpine	-	warmest month < 10 ºC	-

1) After preliminary work by FAO-IIASA (de Pauw *et al.,* 1996) and ISRIC (Kauffman *et al.,* 1996). Corresponds with map of Agro-Ecological Zones as used for study on 'Food Production and Environmental Impact, FAO World Food Summit 1996 (see Fig. 1.3 in: Fresco, L.O., Bridges, E.M. and M.H.C.W. Starren, 1998.Food Production and Environmental Impact (Based on Draft Report 'Food Production and Environmental Impact, FAO Food Summit 1996). Report 98/04, Wageningen Agricultural University and International Soil Reference and Information Centre, Wageningen.

App. 4 Project information

Abstract, NRP Project 952282:

The current baseline study concerns a review of published data on the capability of soils for enhanced carbon sequestration, be it due to: (1) improved and sustainable soil management practices or (2) the physiological 'CO_2-fertilization' effect, associated with increased atmospheric CO_2-levels, and the associated improved water-use efficiency and more favourable temperatures and increased anthropogenic nitrogen emissions. The emphasis in this study will be on item (1). First, a wide range of international experts, known to ISRIC, will be asked to provide information on topics such as 'where have long-term SOC field experiments been carried out' and 'what did this give in terms of increased CO_2 sequestration'. This information, supplemented with a survey of the peer-reviewed literature, will be presented as a 'state-of-the-art' review into a Technical Paper. In a third stage, preliminary GIS-based studies will be made to attempt a global quantification of possibilities for enhanced carbon sequestration in the soil, in first instance using broad assumptions and necessarily coarse scenario's.

Attendance and presentations at national and international meetings (partially in framework of NRP project):

- World Congress of Soil Science, Montpellier (20-26 August 1998)
- Annual Meeting, Soil Science Society of America, Baltimore (18-22 October 1998)
- National NRP Symposium, Garderen (29-30 October 1998)
- Terrestrial Biospheric Sinks in the Kyoto Protocol: What tools do we need to monitor a European carbon balance? (Sponsored by: DG XII, NOP, WURC). SC-DLO, Wageningen (March 2-3, 1999).