LARGE SCALE SPATIAL MODELLING OF SOIL ORGANIC CARBON DYNAMICS

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Thesis submitted to University of Nottingham
For the degree of Doctor of Philosophy

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May 2001

ACKNOWLEDGEMENTS

I would like to thank my supervisor, Dr Pete Smith and Profs. David Jenkinson and David Powlson at IACR who were very encouraging during the early stages of this project. There are many other friends and colleagues at IACR who should be thanked, in fact too many to mention. However, I must personally thank Kevin Coleman, Nick Hesketh, Tom Addiscott, Gordon Dailey, Penny Leech, Margaret Glendining, Gill Tuck and Goetz Richter in the Soils Modelling Group. At Nottingham, I have to thank Stewart Marshall and Neil Crout for their comments on my work.

I'd also like to acknowledge funding from BBSRC, MAFF and the EU, which has supported my work at IACR.

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ABSTRACT

Under the Kyoto Protocol, participating nations are required to reduce National CO_2 emissions according to their 'reduction commitment' or 'quantified emissions limitation', over the first commitment period, 2008-2012. One way in which nations could achieve this would be by increasing soil carbon storage through different management practices. Most former estimates of regional scale C sequestration potential have made use of either linear regressions based on long-term experimental data, whilst some have used dynamic soil organic matter (SOM) models linked to spatial databases. Few studies have compared these two methods, and none have compared regressions with two different SOM models. This thesis presents a case study investigation of the potential of different land management practices to sequester carbon in soil in arable land, and preliminary estimates of other potential C savings.

Two dynamic SOM models were chosen for this study, RothC (a soil process model) and CENTURY (a general ecosystem model). RothC and CENTURY are the two most widely used and validated SOM models world-wide. Methods were developed to enhance use and comparability of the models in a predictive mode. These methods included a) estimation of the IOM pool for RothC, b) estimation of C inputs to soil, c) investigation of pool size distributions in CENTURY, and d) creation of a program to allow use of C inputs derived from CENTURY with the RothC model. This thesis has also investigated the importance of errors in C inputs to soil for predictive SOM modelling, and performed sensitivity analyses to investigate how errors in setting the size refractory SOM pools might affect predictions of SOC.

RothC and CENTURY were compared at the site scale using datasets from seven European long-term experiments, in order to a) verify their ability to predict SOC changes under changes in land use and management relevant to studies of C sequestration potential, b) evaluate model performance under European climatic conditions, and c) compare the performance of the two models.

Finally, a Geographic Information System (GIS) containing soil, land use and climate layers, was assembled for a case study region in Central Hungary. GIS interfaces were developed for the RothC and CENTURY models, thus linking them to spatial datasets at the regional level. This allowed a comparison of estimates of the C sequestration potential of different land management practices obtained using the two models and using regression-based estimates. Although estimates obtained by the different approaches were of the same order of magnitude, differences were observed. Encouragingly, some of the land management scenarios studied here showed sufficient C mitigation potential to meet Hungarian CO_2 reduction commitments.

ABBREVIATIONS

ACS	Additional carbon sequestration potential
Active SOM	Active soil organic matter pool, CENTURY model
APAR	Absorbed photosynthetically active radiation
BIO	Soil microbial biomass pool, Rothamsted carbon model
CAM	Crassulacean acid metabolism
CENTURY	CENTURY soil organic matter model
CORINE	Co-ordination of information on the environment land cover database
CoT	Conservation tillage
CT	Conventional tillage
DOC	Dissolved organic carbon
DOM	Dissolved organic matter
DPM	Decomposable plant material pool, Rothamsted carbon model
EFTEXT	Effect of soil sand content on decomposition rate of Active SOM pool, CENTURY model
ES	Emissions offset
FAOSTAT	Food and Agriculture Organisation Statistical Database
FPAR	Fraction of photosynthetically active radiation
FPS1S3	Efficiency of stabilising Active SOM into Passive SOM, CENTURY model
FPS2S3	Efficiency of stabilising Slow SOM into Passive SOM, CENTURY model
Fwloss(4)	Scaling parameter for potential evapotranspiration, CENTURY model
FYM	Farmyard manure
GCM	General Circulation Model
GCTE-SOMNET	Global Change and Terrestrial Ecosystems Soil Organic Matter Network
GHGMP	Greenhouse gas mitigation potential
GIS	Geographic information system
HI	Harvest index
HUM	Humus pool, Rothamsted carbon model
HunSOTER	Hungarian soil and terrain database
IOM	Inert organic matter pool, Rothamsted carbon model
IPCC	Intergovernmental Panel on Climate Change
ISOM	Inert soil organic matter
LFOM	Light fraction organic matter
LTE	Long term experiment
LUE	Light use efficiency
Min-T	Minimum tillage
Mul-T	Mulch tillage
NDVI	Normalised vegetation difference index
NPP	Net primary production
NT	No till
OM	Organic matter
Passive SOM	Passive soil organic matter pool, CENTURY model
PEM	Production efficiency model
PET	Potential evapotranspiration

PTTR	Potential transpiration water loss, CENTURY model
RAT	Ratio of (stored water plus precipitation) to potential evapotranspiration, CENTURY model
RMSE	Root mean square error
RothC	Rothamsted carbon model
RPM	Resistant plant material pool, Rothamsted carbon model
RSOM	Refractory soil organic matter
RT	Ridge till
Slow SOM	Slow soil organic matter pool, CENTURY model
SMD	Soil moisture deficit
SOC	Soil organic carbon
SOM	Soil organic matter
SR	Shoot to root ratio
STEMP	Soil temperature, CENTURY model
TCS	Total carbon sequestration potential
TFUNC	Decomposition rate modifier for temperature, CENTURY model
US-LTER	United States Long Term Ecological Research Network
WFUNC	Decomposition rate modifier for moisture, CENTURY model

1. INTRODUCTION

1.1 THE GLOBAL CARBON CYCLE

Figure 1.1 shows a simplified diagram of the global carbon cycle. A detailed overview of the global carbon cycle is given by Schlesinger (1995), including its biogeochemistry and an historical perspective. This chapter focuses upon the role of soils in bio-geochemical aspects of the global carbon cycle. The main pools of carbon are the terrestrial vegetation, soils, the atmosphere and, by far the largest pool, the oceans. Soils contain around twice as much carbon as is held in the atmospheric pool (≈1400-1500 Pg C: Post et al. 1982), with terrestrial vegetation containing a similar amount of carbon as the atmosphere. It is notable that large amounts of carbon are held in surface litter, particularly in forest soils, but this pool is not included in the soil organic carbon (SOC) pool (Buringh, 1984). The pools have different turnover times: the average turnover time of the soil organic matter pool is around 25 years, the vegetation around 5 years, and the mean residence time of C in the atmosphere is around 3.4 years.

The major fluxes are:

a) between land vegetation and the atmosphere – photosynthesis (fixation of CO_2) and respiration of CO_2 by plants
b) between the ocean and the atmosphere – uptake (including death of phytoplankton and dissolution) and return of CO_2 to the atmosphere
c) between soils and the atmosphere – soil respiration and decomposition
d) between the ocean and deep sediments – burial
e) finally between anthropogenic sources and the atmosphere – fossil fuel burning and destruction of vegetation

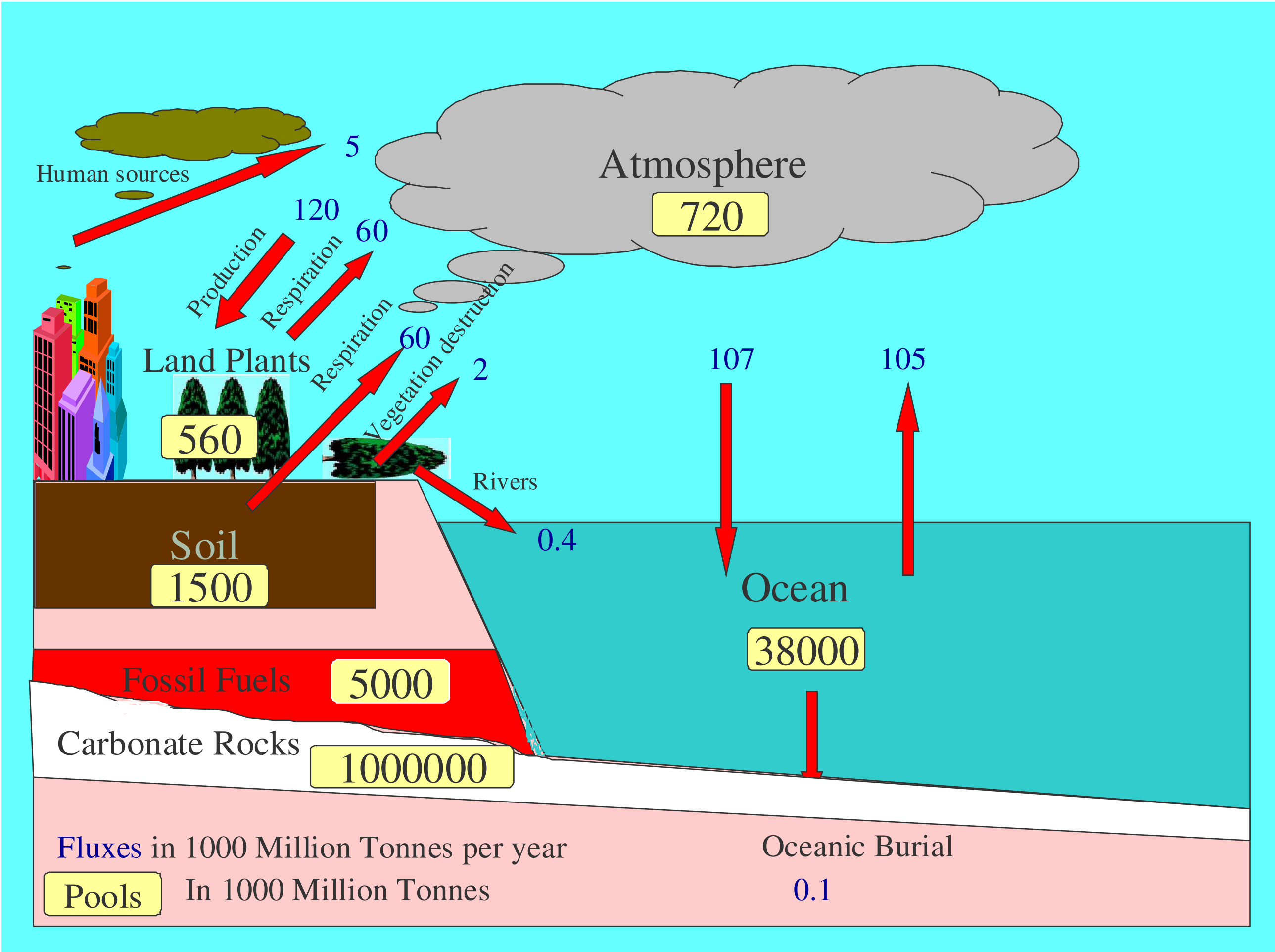
Earth's C balance is not currently in equilibrium, with a net increase of around 6 Pg C annually in the atmospheric pool. Many authors have suggested the existence of a "missing carbon sink" since the global C budget does not balance (Schimel, 1994). An additional carbon source could be a net release of carbon from land (Houghton et al. 1983, 1987; Siegenthaler and Oeschger, 1987; Leavitt and Long, 1988). Possible candidates for the missing sink have included: overestimation of CO_2 release from tropical deforestation (Lugo and Brown, 1992; Keeling and Shertz, 1992); forest regrowth on abandoned tropical (Curran et al. 1996) or temperate agricultural land (Houghton et al. 1987; Tans et al. 1990); increases in tropical (Grace et al. 1995), temperate or boreal forest stocks (Kauppi et al. 1992), or CO_2 (Strain and Cure, 1985; Allen, 1990; Bazzaz, 1990; Harrison et al. 1993a) or N fertilisation of vegetation (Nadelhoffer et al. 1999). Elevated concentration of CO_2 in the atmosphere is known to increase plant growth in short-term experiments. However, it is not yet clear whether plants respond in the same way during gradual long-term increases as they do in short-term experiments, or what the possible feedback mechanisms involved in of CO_2 fertilisation might be (Schlesinger, 1995). Alternatively, 50% of the missing sink may be accounted for by mid-latitude C accumulation in vegetation, explained by climate variability (Dai and Fung, 1993), or could be accounted for increased C storage in cropland and pastures, driven by increased C input rates to soil (Buyanovsky and Wagner, 1998).

1.2 SOIL CARBON

The world's SOC pool contains around 1400 Pg C (Buringh, 1984; Schlesinger, 1995; Batjes, 1996, 1998); the soil organic matter (SOM) has an important role in the cycling of N,P, S and other nutrients (Jenkinson, 1988), and also in the retention and behaviour of toxic chemicals. Soils also contain large amounts of inorganic carbon (SIC), largely as calcite in arid and semi-arid regions (Schlesinger, 1995). Globally, the SIC is pool slightly larger than the SOC pool, at around 1700 Pg C (Lal et al. 1995), although Batjes (1996) reports a smaller SIC pool size. Fig. 1.2 shows a diagram of the soil carbon cycle.

In the world's soils, 11.4% of SOC is under cropland, 23.1% under grassland, 30.1% under forest, and 34.8% under other uses (Buringh, 1984; IPCC, 1996). A large proportion of SOC is held in boreal forests and tundra regions (Lal et al. 1995; Schlesinger, 1995). Soil carbon density increases with increasing rainfall and decreasing temperature (Post et al. 1982). These variations are driven by greater SOM production with increasing rainfall, whilst increased dryness and waterlogging can act to limit C decomposition; decomposition is more rapid at higher temperatures. Organic soils (Histosols) contain 23% of the organic carbon of all soils, due to their high SOM content, and Brunisolic soils (Inceptisols) contain 22% of the organic carbon of all soils due to their great spatial extent, and range of SOM

Figure 1.1 The Global carbon cycle



contents (Schlesinger, 1995; Batjes, 1996). In Histosols, decomposition of organic matter is retarded under water saturated conditions, and decomposition is further slowed under cold conditions (Batjes, 1996). Andosols also store large amounts of C since their high allophane content allows great protection of organic matter (Batjes, 1996). Soils from arid regions (Yermosols and Xerosols) contain small amounts of organic C since plant growth is limited and the annual input of C is low. Figures 1.3 and 1.4 show the global distribution of SOC to 1m and 30 cm depth, respectively (Batjes, 1996). At the global scale, between 39 and 70% of total SOC is held in the top 100cm, and 58 to 81% in the top 50cm of the soil (Batjes, 1996).

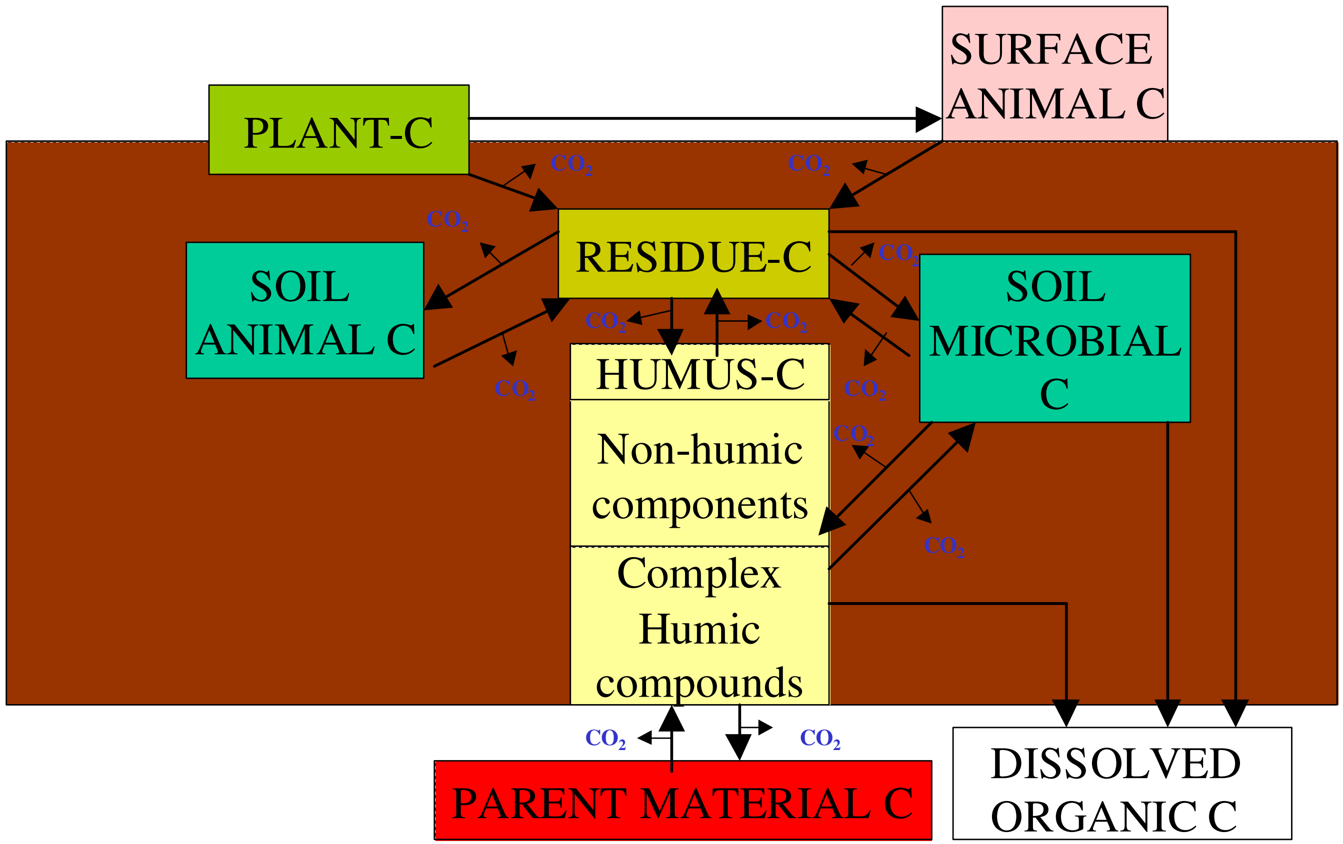
In many soils, most of the SOC is in the top 20cm of the soil (Buringh, 1984; Jenkinson, 1988), and the carbon concentration in soils tends to decrease with depth from the surface (Jenkinson et al. 1999a). However, great amounts of carbon are stored at depth in true chernozems (Buringh, 1984), podzols consist of a layer depleted in OM overlying a SOM-enriched layer, and palaeosols often contain buried OM-rich layers (Jenkinson, 1988). The spatial variation of SOC content is great (Post et al. 1982), even within soil types (Buringh, 1984). However, grouping soils according to soil conditions (productivity, temperature and moisture) can reduce this variability (Buringh, 1984).

The organic matter in soils consists of many different substances. A simple classification includes: living plant roots, dead but little-altered plant remains, partly decomposed plant remains, colloidal organic matter, living micro-organisms and macro-organisms, and inactive or inert organic matter (Buringh, 1984). Although SOM is often considered as comprising pools with discrete turnover times (e.g. Jenkinson and Rayner, 1977), in reality it is a continuum of materials of differing age, decomposition status and quality (Bosatta and Agren, 1991). A conceptual diagram of the soil C system is given in Figure 1.2.

1.2.1 Stable Isotopes of C

Two stable isotopes of C exist in nature (^{13}C: around 1.11 % of all C; and ^{12}C: 98.89% of all C). ^{14}C is another (non-stable) isotope of C discussed in Section 1.2.2. Their existence has enabled many studies of SOC turnover and dynamics, to indicate former land-use changes, and to estimate the relative contribution of different plant

Figure 1.2 The soil carbon cycle



communities to SOC. The ratio of these isotopes in natural materials varies slightly around the mean values, due to isotopic fractionation during chemical, physical and biological processes. The $^{13}C:^{12}C$ ratio is commonly expressed as a delta ($\delta^{13}C$) value in parts per mille (‰), calculated as:

$$\partial^{13}C = \frac{(r_{\text{sample}} - r_{\text{standard}})}{(r_{\text{standard}} \times 1000)}$$

where r is the mass ratio of sample and standard, and the standard is Pee Dee Belemnite (PDB) with $^{13}C:^{12}C$ =0.0112372.

In the plant soil system, the natural stable carbon isotope ratio ($^{13}C:^{12}C$) of SOC contains information regarding the presence or absence of plant species with the C3 (Calvin cycle), C4 (Catch-Slack cycle) and CAM (Crassulacean Acid Metabolism) photosynthetic pathways in past plant communities and their relative contribution to community net primary production through time (Paul et al. 1995; Boutton, 1996). C3 plants account for around 95% of all plant species, including most trees and temperate crops such as wheat and barley, and have high $^{13}C:^{12}C$ (average $\delta^{13}C$≅-27‰, range $\delta^{13}C$≅-32 to -22‰; Boutton, 1996). C4 plants are only around 5% of all vegetation, including most grasses, crops such as maize and species derived from tropical grasses and have low $^{13}C:^{12}C$ (average $\delta^{13}C$≅-13‰, range $\delta^{13}C$≅-16 to -9‰; Boutton, 1996). CAM plants (such as cacti) also have a characteristic $\delta^{13}C$ around –10 to –28 ‰ (Boutton, 1996). This plant material is incorporated in the SOM, with a relatively small degree of discrimination and the ratio of $^{13}C:^{12}C$ can be measured in SOC to determine historic vegetation changes and vegetation dynamics.

Known and dated vegetation changes can be used to study SOC dynamics in the whole soil and soil fractions (e.g. Lefroy and Blair, 1994; Balesdent, 1996), and the C inputs from different plant communities (e.g. Skjemstad et al. 1990; Puget et al. 1995; Cerri et al. 1994; Balesdent and Mariotti, 1996; Jastrow et al. 1996; Angers and Giroux, 1996; Besnard et al. 1996;). Examples of vegetation changes used in these studies include conversion of C3 tropical forests to C4 savannahs, C4 savannahs or grasslands to C3 crops, and C3 forests to C4 crops or grasslands (e.g. Condron and Newman, 1998; Golchin et al. 1995).

Figure 1.3 **Global soil C stocks to 1m depth, from ISRIC WISE database (Batjes, 1996)**

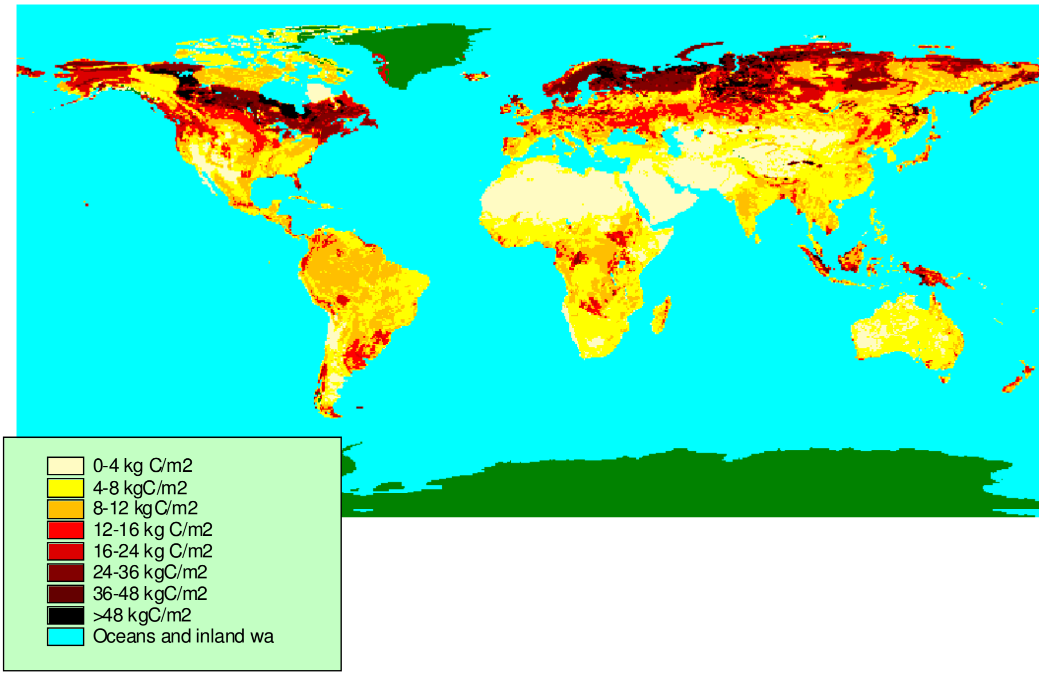
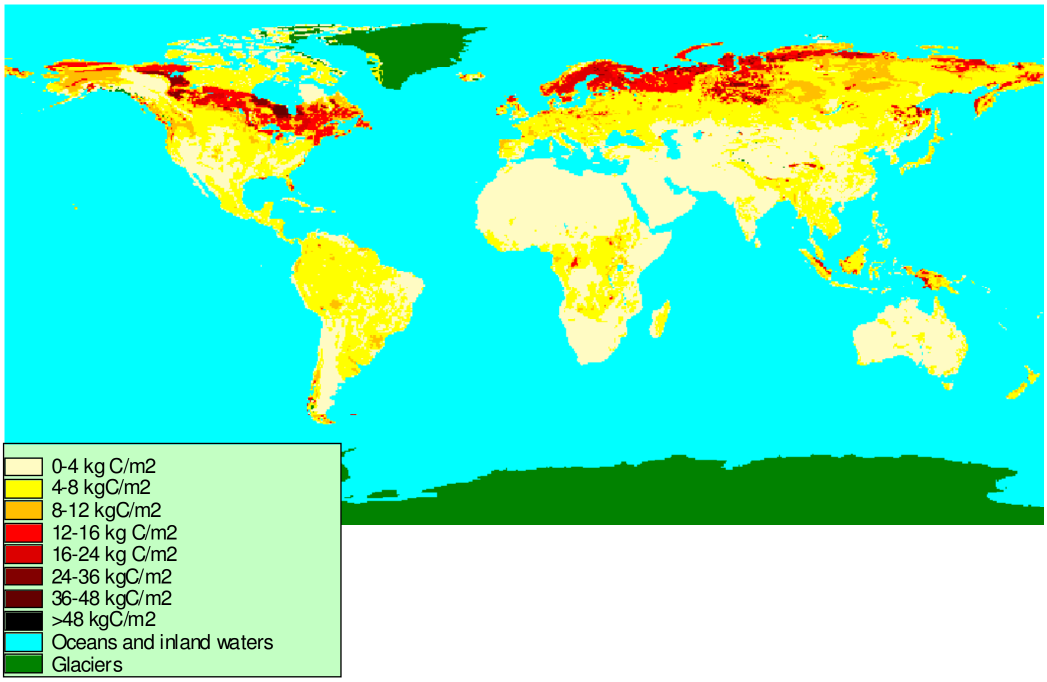


Figure 1.4 **Global soil C stocks to 30cm depth, from ISRIC WISE database (Batjes, 1996)**



Palaeo-environmental studies make use of soil or palaeosol ^{13}C:^{12}C ratios to assess past changes in vegetation and climate (e.g. Schwartz et al. 1986; Connin et al. 1997). ^{13}C:^{12}C ratios have also been used to identify sources of DOC at the catchment scale (Schiff et al. 1997), to estimate SOC turnover times (Andreux et al. 1990; Skjemstad et al. 1990; Becker-Heidmann and Scharpenseel, 1992a), in studies of global climate change (Becker Heidmann and Scharpenseel, 1992b), to estimate methane production rates in soils (Becker-Heidmann and Scharpenseel, 1992a), and for studies of soil C dynamics under different tillage regimes, (e.g. Besnard et al. 1996; Six et al. 1998).

1.2.2 Radiocarbon dating of SOM

Radiocarbon (^{14}C) dating of SOM provides opportunities to study much more than just the age of soils (Min et al. 1993; Chichagova, 1995; Liangwu, 1996) or particular soil fractions. For example, natural ^{14}C enrichment studies or artificial labelling techniques (e.g. Mercx et al. 1987) can be used in several ways. The thermonuclear bomb testing that occurred from 1960 to 1985 produced a ^{14}C signal that is useful in following medium term SOM dynamics (Jenkinson et al. 1987; Paul et al. 1995). Thermonuclear bomb testing effectively created a natural labelling experiment and caused the normally observed ^{14}C age to decrease (Jenkinson and Rayner, 1977). This increase can be used to calculate the annual input of C to soil, and requires both pre- and post-bomb soil samples. SOM at the site must be also in a steady state (Jenkinson and Rayner, 1977; O'Brien, 1984; Harrison et al. 1991). The specific ^{14}C activity of plant material entering soil must also be known, which can be derived from published data on atmospheric ^{14}C levels.

Radiocarbon dates can also be used to estimate SOC turnover times (e.g. Trumbore et al. 1990), since turnover time may be calculated as total C stock at equilibrium divided by the annual C input. The size, age and mean residence time of active and passive SOC pools can also be estimated using radiocarbon measurements on paired plots, providing that the soils had the same SOC content prior to land use change and after the change have different SOC contents due to management (Hsieh, 1992, 1994). However, ^{14}C dates should also be treated with caution, since ^{14}C dates for the same SOM samples measured by different laboratories can vary quite significantly (Martin and Johnson, 1995).

Radiocarbon dating of SOM has been widely used to show that soil age tends to increase with depth (e.g. Arslanov et al. 1970; Jenkinson, 1970,1973; Gerasimov, 1974; O'Brien and Stout, 1978; Becker-Heidmann et al. 1988; Scharpenseel et al. 1989; Harrison et al. 1991,1993; Becker-Heidmann and Scharpenseel, 1992a; Charman et al. 1992; Trumbore, 1993; Martin and Johnson, 1995; Jenkinson et al. 1999a,b). This could be explained by either a) greater amounts of old C deep with increasing soil depth (e.g. Hsieh, 2001) or b) a constant amount of old C with soil depth, with a greater amount of young C in the upper parts of the soil profile (Arslanov et al. 1970; O'Brien and Stout, 1978), due to inputs of younger C from vegetation (O'Brien, 1986). Some tropical soils, however, exhibit great age throughout the whole profile (Gerasimov, 1974). Radiocarbon dating of SOM has also shown that SOM turnover time is faster in tropical than temperate soils (Trumbore, 1993; Hsieh, 1996).

The total radiocarbon age of soil is not always indicative of soil age – many arable topsoils date older than their native counterparts, due to depletion in younger C (e.g. Gerasimov, 1974; Harrison et al. 1993b; Paul et al. 1995). Cultivation leads to a homogenisation of SOM radiocarbon age in the topsoil (Becker-Heidmann and Scharpenseel, 1989; Paul et al. 1995), and it is mostly the young or labile fractions of SOM that respond to changes in land use and management (Hsieh, 1996). Grassland soils tend to be younger than forest soils (Gerasimov, 1974). Differences in turnover times between soil types have been shown using ^{14}C dating (Scharpenseel et al. 1989): in particular, soils containing allophane have much longer turnover times than other soils (see Chapter 3; Saggar et al. 1994).

Modelling SOM turnover often divides SOM into discrete fractions, and radiocarbon methods have been used in attempts to identify discrete soil C fractions with different turnover times. Acid hydrolysis residues (e.g. Martel and Paul, 1974a,b; Martel and Lasalle, 1977; Paul et al. 1995; Leavitt et al. 1996), or humins and humic acids (e.g. Becker-Heidmann et al. 1988) are often older than total soil. However, differences in the age of total SOM (Campbell et al. 1967), humic acid and residue fractions are not always apparent and may differ between soils (Martin and Johnson, 1995; Goh et al. 1977; Balesdent, 1996; Hsieh, 2001). A combination of acid hydrolysis and density fractionation or other fractionations with radiocarbon dating seems more promising, although success is partly dependent on soil texture and soil type (Trumbore et al. 1989; Trumbore and Zheng, 1996; Shang and Tiessen, 1996).

^{14}C dating and soil fractionations have also been used to show that physical protection plays an important role in SOM dynamics (Paul et al. 1995), to identify the mean age of different soil fractions (Balesdent, 1987), to demonstrate that the oldest C is associated with coarse clay fractions (Anderson and Paul, 1984; Becker Heidmann and Scharpenseel 1992b), to show that the soil microbial biomass most commonly uses fresh plant residues as a substrate, and to show temporal variations in substrate quality (Paul et al. 1995).

Other diverse applications of soil ^{14}C dating have been to identify sources of DOC at the catchment scale (Schiff et al. 1997); to show bioturbation effects on SOM turnover (O'Brien, 1984; Scharpenseel et al. 1989; Huang et al. 1996); to estimate methane production rates in soils (Becker-Heidmann and Scharpenseel 1992a), to estimate historic changes in SOC renewal, accumulation and loss (e.g. Chichagova, 1996; Cherkinsky and Goryachkin, 1992; Martel and Paul, 1974), to estimate changes in SOC turnover following land use change (Harrison and Harkness, 1993); and to identify soil forming and transport processes (e.g. Min et al. 1993; Trumbore et al. 1989; Paul et al. 1995; Becker-Heidmann et al. 1988).

1.2.3 Turnover and chemical nature of SOM

Soil C turnover may be defined as the flux of organic C through a given volume of soil, and calculated as the amount of C in a soil system when equilibrium has been reached, divided by the annual input of C into the system (Jenkinson and Rayner, 1977). Information on soil C turnover can be assimilated from a variety of sources: 1) long-term changes in biomass C and SOC from long-term experiments (10-100 y timescale), 2) incubations of labelled ^{14}C plant materials in soils (1-10 y periods), 3) radiocarbon dating (1000 y period), and 4) from the effect of thermonuclear radiocarbon on radiocarbon age (the bomb effect) – giving information on the annual input of organic C (Jenkinson and Rayner, 1977).

SOM is composed of a spectrum of compounds varying in chemical composition and turnover time, and soils contain traces of free sugars, amino acids, organic acids and lipids, as well as polymers, carbohydrate, and amino sugars (Jenkinson, 1988). The most active and youngest components of SOM are the plant residues and debris entering the soil, composed of largely unaltered (and usually cellulose dominated) plant material. Vegetative material has turnover times of the order of 0.5-3.0 y (Buyanovsky et al. 1994). Most of this material is utilised as a substrate for the microbial community, the other relatively fast turnover component of the SOM system, although some may be highly resistant and may never decompose (Hyvonen et al. 1998). The microbial biomass commonly has a turnover time of 1-10 y (Buyanovsky et al. 1994). Upon decomposition, the plant residues give rise to more microbial biomass and metabolites, with an associated release of CO_2. These products are then substrates for a much slower phase of decomposition, which forms humified and more stable products with greater turnover times. For example, SOC in whole soil has a mean turnover time around 7 y, whilst SOC associated with the silt and clay fractions turns over between 400 y and 1000 y (Buyanovsky et al. 1994).

An analysis of published data from a range of climates, land-uses and management types has shown that the chemical nature of the whole SOM is remarkably similar across all soils, despite differences in land use and management, mineral composition, climate, and organic C content (Mahieu et al. 1999). In general the order of abundance is always the same, with O-alkyl compounds being dominant (20-70%), followed by alkyls (10-50%), aromatics (10-45%), and carbonyls (5-20%). However, some differences between soil orders have been noted (Baldock et al. 1992). An increase in O-alkyl compounds in organic soils may be attributed to an accumulation of plant-derived material (Mahieu et al. 1999). Alkyl and methoxyl groups may be more resistant to decomposition than other components of SOM (Kinchesh et al. 1995; Guggenberger et al. 1995), whilst carbohydrates may be a relatively active fraction. Indeed, SOM associated with soil silt and clay fractions is often enriched in alkyl group compounds (Guggenberger et al. 1995), and more decomposed products (Baldock et al. 1992). Finally, chemical and physical fractionations of SOM indicate that variations in the rate of soil C turnover may be less due to chemical composition than physical protection and location (Balesdent, 1996).

1.2.4 The SOM system

The quantity (and quality) of SOM present in the soil is determined by a balance between inputs of organic matter and outputs, and both sides of the equation are controlled by a number of interacting factors. In particular, initial soil C stocks and soil management are critical in defining how much C can be accumulated in soil (Katterer and Andren, 1999). Normal functioning of the SOM system is crucial for nutrient supply, soil conservation, water supply and pollutant retention. Factors not considered explicitly in this section are hydrology (Buringh, 1984), soil structure, and soil contaminants.

1.2.4.1 Inputs to SOM

Inputs of C to SOM are discussed in greater detail in Chapter 4, and are one of the primary controls on soil C levels, since soil C levels and C inputs are commonly directly related (Buyanovsky and Wagner, 1998). C inputs to soil include a variety of plant-derived materials (e.g. crop residues, roots, litter, exudates and so on; Paustian et al. 1997a), and also any organic amendments (e.g. cereal straw, farmyard manure, and sewage sludge). The Net Primary Production (NPP), and hence quantity of C inputs to soil, as well as litter quality greatly influence SOC levels (Parton et al. 1987; Anderson, 1991; Agren et al. 1996; Liang et al. 1998), and plant NPP is the major factor affecting C inputs to soil (Agren et al. 1996; Liang et al. 1998). Studies using soil ^{14}C dating techniques suggest that around 1-7% of NPP in arable systems is exuded into soil via roots (Paul et al. 1995).

1.2.4.2 Outputs from SOM

The main outputs from SOM are losses from the soil through mineralisation, erosion and leaching (Lal et al. 1995; Paustian et al. 1997a). Under reducing conditions there is some methane evolution by methanogens, SOM mineralisation and carbon rhizodeposition, but most agricultural soils are net methane consumers (Cao et al. 1996; Paustian et al. 1997a). Recent work suggests that arable soils do oxidise methane, although more weakly than natural ecosystems (Goulding et al. 1998). Some authors consider erosion as a transport process rather than loss or gain of SOC (Paustian et al. 1997b), although there is little information to determine whether erosion increases or decreases SOC at the regional level.

Decomposition is controlled by abiotic factors (soil temperature, water, aeration and pH), the physicochemical nature of organic matter (chemical composition, structure and particle size), its physical exposure to decomposers, the availability of mineral nutrients needed for microbial growth and metabolism, and the nature and composition of the decomposer community itself (Jenkinson, 1988; Paustian et al. 1997c; Agren et al. 1996).

Dissolved Organic Carbon (DOC) may also be an important source of C loss, especially in acid and organic soils, although little is known about its role in agricultural soils in the long term (Paustian et al. 1997a). Possible sources of DOC are soluble by-products of microbial activity, root exudates and chemical or biological breakdown of soil (or suspended particulate) organic matter (Trumbore et al. 1992). DOC export varies among soil types and is related to SOM content (a source of potential DOC) and the presence of mineral horizons that can absorb DOC originating from organic horizons (Tipping et al. 1999). Direct decay of plant residues may also be a significant DOC source. Natural events such as fire may also have an important role to play in SOC loss in forest systems (Agren et al. 1996), as well as grasslands and heathlands.

1.2.4.3 SOM and differences in soil properties

Soil texture and mineralogy greatly influence the retention and turnover of SOC (Parton et al. 1987; Batjes, 1998). Gains in soil C are often greatest in soils with high clay content (Burke et al. 1989; Liang et al 1998; Koutika et al. 1999) and stable SOM is commonly linked with soil fine fractions (Tsutsuki et al. 1988; Leavitt et al. 1996; Quiroga et al. 1996). The largest amounts of soil C are often found in silt and clay fractions (Stemmer et al. 1998), due to protection from microbial attack, with a lower C:N ratio reflecting the mineralisation and humification status of SOM. Fine textured soils also tend to lose less C on cultivation (Tiessen and Stewart, 1983; Burke et al. 1989).

Soil pH can also have important effects on SOM accumulation - Motavalli et al. (1995) suggest that soil pH could be a significant factor limiting soil C concentrations in the tropics. SOM decomposition is often retarded under acidic conditions (Jenkinson, 1988; Jenkinson et al. 1992). Extreme soil acidity may cause the microbial biomass to be suppressed in both the short and long term (Coleman, 1983; Motavalli et al. 1995; Jenkinson et al. 1999b). More residual non-microbial biomass C is retained in slightly acid soils compared to neutral or alkaline soils (Amato and Ladd, 1992). There tends to be less C deep in most soil profiles, and turnover rates are usually slower at depth (see previous sections and Chapter 3; Trumbore et al. 1989; Skjemstad et al. 1990; Harrison et al. 1991). However, soil incubation data suggests that controls over decomposition are similar in subsoil and surface soil horizons (Parton et al. 1987).

1.2.4.4 SOM and climate

In general, SOM turnover is faster at higher temperatures (Harrison and Harkness, 1993; Hsieh, 1996) and lower altitudes, since decomposition is increased. SOC levels usually increase with increasing precipitation (Burke et al. 1989), since NPP and hence inputs of C to soil increase with rainfall. Precipitation also affects N inputs to soil and decomposition (Parton et al. 1987) – too little moisture retards decomposition, whilst under anaerobic conditions, decomposition rates are around 30% of aerobic rates (DeBusk and Reddy, 1998; Jenkinson, 1988).

Although C inputs in tropical forests can exceed five times that in temperate forests (Parton et al. 1989), temperature enhanced decomposition may also be much greater (Grisi et al. 1998; Parton et al 1989a), leading to little difference in overall soil C stocks. SOC abundance tends to decrease from cool, wet to dry, hot climates (Buringh, 1984; Burke et al. 1989; Tate, 1992; Trumbore, 1993), and the turnover time of soil C is shorter in tropical zones than at high latitudes (Bird et al. 1996; Jenkinson et al. 1999a). Climatic effects on tropical SOM are complicated by the lower stability of SOM in many tropical soils, due to different mechanisms of organo-mineral stabilisation (Shang and Tiessen, 1998).

1.2.4.5 Spatial variability in SOM abundance

Spatial variability of SOC concentration is high at the field scale (Van den Pol-Van Dasselaar et al. 1998) and at regional and global scales. Both soil survey and geostatistical methods may be of use in characterising spatial patterns in SOC concentration and a combination of the two would be ideal (Yost et al. 1993). At the field scale, spatial variations in SOC concentration are often most strongly controlled by differences in NPP, which may in turn be controlled by other factors. At the regional scale, soil texture is the often the most important determinant of soil C stocks, although mean annual temperature, mean annual rainfall, plant production, litter quality, N availability and management are also important (Parton et al. 1987; Arrouays et al. 1995). Globally, NPP is the main determinant of the spatial variation in SOC stocks, driven by differences in rainfall (Post et al. 1982).

1.2.4.6 SOM and land use

The effects of land use on SOC are discussed in greater detail in section 1.3. In general, SOC concentration tends to increase from continuous arable cropping to rotational grass-arable cropping, to grasslands to woodlands (e.g. Drury et al. 1998; Larionova et al. 1998), but the spatial distribution of SOC concentration, microflora and rhizosphere properties may affect these differences (Condron and Newman, 1998). C mineralisation in forests and grasslands is greater than in agricultural soils (Ajwa et al. 1998; Larionova et al. 1998), and grasslands often contain much more stable SOC, more evenly distributed in the soil profile, than forests (Anderson, 1991)

1.3 MANAGEMENT IMPACTS ON SOM

As discussed in the previous section, a number of interacting factors control the size of the soil C stock via their influence on the input of C to soil and the decomposition rate of C entering the soil. Soil C eventually reaches an equilibrium between carbon inputs and decomposition (Buringh, 1984). An increase in C inputs over decomposition (such applying as organic amendments or afforestation of agricultural land), or a decrease in decomposition (e.g. waterlogging, reduced tillage or reduced pH) allows SOC to increase and approach a new equilibrium (Buringh, 1984; Batjes, 1998). A decrease in C inputs or an increase in decomposition (e.g. increased temperature or increased tillage) will lead to a net loss of soil C stocks (Buringh, 1984). This change in SOC is often most rapid over the first few years of land-use or management change, and more gradual over the next 10-50 y (Buringh, 1984). Other factors can cause losses of soil C: erosion, salinisation, alkalisation and soil degradation (Buringh, 1984), whilst some soils (e.g. histosols) may naturally accumulate SOM (Buringh, 1984).

Different soil management practices offer a number of ways to control soil C stocks (Batjes and Sombroek, 1997). Some practices affect more than one facet of the soil C cycle (e.g. different controls on decomposition or inputs; Figure 1.5, Paustian et al. 1997a). The exact nature and direction of these effects depends on site-specific conditions and the management details (Paustian et al. 1997a); some effects are short-term, whilst others are long-term. Important agricultural activities that lead to changes in global SOC pools include deforestation and afforestation, biomass burning, and cultivation (Lal et al. 1995). It is mostly the labile fractions (including microbial biomass) of SOM that respond to changes in land use and management (Besnard et al. 1996; Hsieh, 1996; Quiroga et al. 1996; Bragato and Primavera, 1998), especially in the short or medium term. The actual potential of different land use and management options to increase or decrease SOC are limited by socio-economic issues (Lal et al. 1995), for example land owners will only adopt practices showing economic gains in the short term (Batjes, 1998). Figure 1.5 shows the major management controls on soil C (after Paustian et al. 1997a).

1.3.1 Land use and management

Differences in SOC stocks under different land uses have been discussed in section 1.2.4.6 – this section deals with changes in SOC stocks under changes in the management and use of land

1.3.1.1 Changes in land use - forests

Cultivation of native soils can result in losses of SOC of 10-80% of the native soil C stock (Lal et al. 1995; Paustian et al. 1997a), with the greatest loss occurring shortly after clearance, and slowing thereafter. However, changes in the bulk density of a horizon depth associated with cultivation can bias interpretations of C loss when comparing cultivated and virgin soils (Paustian et al. 1997a). Changes in soil C stocks may be caused by reduced C inputs (and changes in their amount, timing and distribution), increased erosion, physical soil disturbance and increased decomposition due to higher temperatures and differences in soil moisture (Houghton et al. 1983; Paustian et al. 1997a). Forest clearing and conversion of forests to pastures often result in losses of SOC (Cerri et al. 1994) and large losses total system C (Kotto-Same, 1997), but SOC may increase eventually. Conversion of forests to arable agriculture most often results in SOC declines of 25-50%

(Houghton et al. 1983; Buringh, 1984; Cerri et al. 1994; Schlesinger, 1995), whilst conversion of forest to grassland may result in a mean loss of about 30% of SOC (Buringh, 1984).

1.3.1.2 Conversion of land use – grasslands

In changing from long-term pasture to arable or vegetable crops, SOC may decline sharply with a mean loss of around 30% (Jenkinson, 1988; Schlesinger, 1995) to reach a new equilibrium after 60-80 years (Haynes and Tregurtha, 1999). SOC may also decline following afforestation of grassland (Harkness and Harrison, 1989), due to the rapid turnover of forest C and initial disturbance during tree planting. There also may be a shift to more recalcitrant forms of SOC in soils under forestry compared with grassland, perhaps due to changes in C inputs or mineralisation of more labile SOM pools (Condron and Newman, 1998).

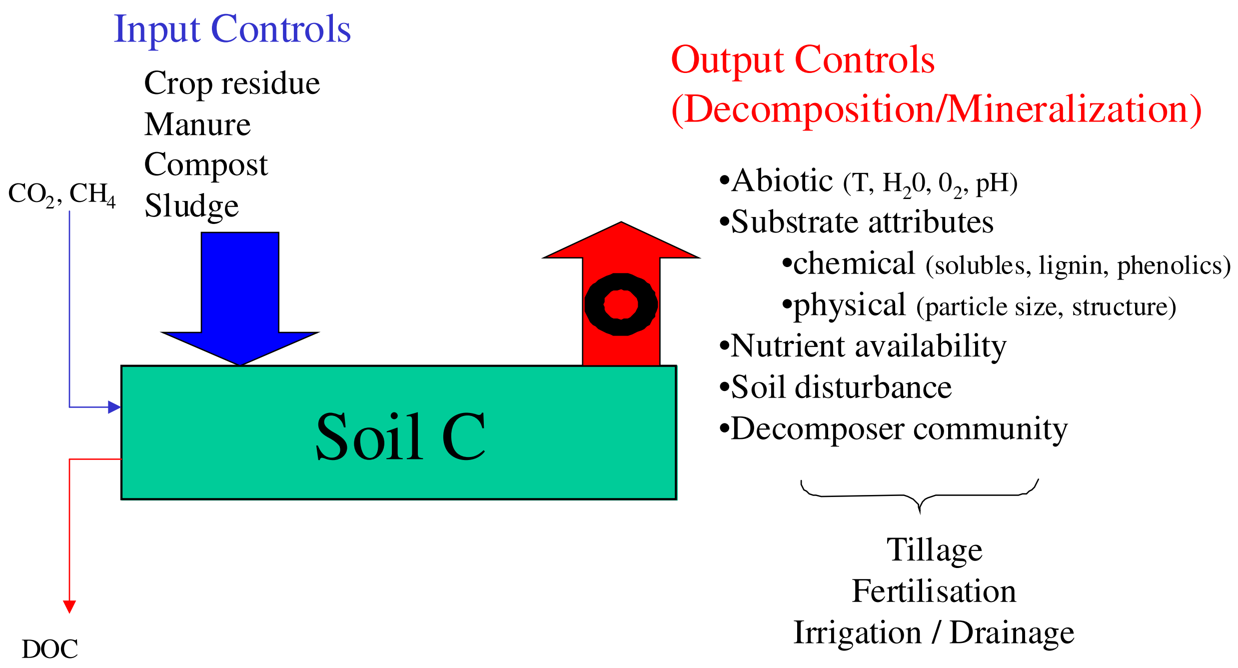
1.3.1.3 Conversion of land use – arable lands

Conversion of arable lands to perennial grasses (e.g conservation reserve program, CRP) or set-aside lands often results in accumulation of SOC (Jenkinson, 1988; Paustian et al. 1997b; Robels and Burke, 1998; Masciandro et al. 1998), due to increased C inputs, reduced litter quality and reduced decomposition. Afforestation of arable land usually results in increased SOC content (Lal et al. 1995; Paustian et al. 1997b), especially in topsoil layers, although in fine textured soils with rapid C turnover, small differences may be observed, dependent on tree type (Richter et al. 1995).

1.3.1.4 Conversion of other land use types

Afforestation of heather moor-lands may result in more rapid C turnover and increased heather SOM replacement with forest SOM up to 40 y after afforestation (Harrison and Harkness, 1993). Drainage of peatlands or lowering or peatland water tables may cause large C losses, since peat soils contain large amounts of C.

Figure 1.5 Management controls on soil C (after Paustian et al. 1997a)



1.3.1.5 Management of arable lands – cultivation and tillage

As noted in previous sections, the cultivation of native grasslands or forests commonly leads to losses of up to 50% soil C stocks (Buyanovsky and Wagner, 1998), caused by a reduction in C inputs and increased decomposition. These changes occur most rapidly in the first years after a land use or management change (Schlesinger, 1995; Mann, 1986) and there is also an initial soil C effect – soils with very little C tend gain some C, whilst high C soils may lose SOC (Mann, 1986). In addition to this, different soil tillage practices during agricultural land management have the potential to alter soil C stocks and their distribution.

Kern and Johnson (1993) suggest that conservation tillage (CoT) could have a significant role to play in offsetting US national fossil fuel emissions. CoT practices were originally developed to reduce water and wind erosion, conserve soil moisture and reduce fuel costs, compared to conventional till (CT). CoT can be either minimum (Min-T) or no-till (NT). CoT maintains >30% of residue cover. The mulch of crop residues maintained at the soil surface reduces raindrop impact, slows evaporation, increases water storage and slows organic matter decomposition (Kern and Johnson, 1993). NT is no-till from after harvest to planting time, with crops planted into narrow seed beds and weed control using herbicides. Ridge till (RT) uses no tillage from after harvest to planting; crops are planted into tilled ridges with residues left between the ridges. Mulch till (Mul-T) disturbs the entire soil whilst maintaining over 30% residue cover on the soil surface (Kern and Johnson, 1993).

1.3.1.6 Tillage and total SOC stocks

Tillage may affect total soil C stocks in a number of ways, with NT often leading to increases in plough-layer soil C of 5-20% relative to CT practices (Kern and Johnson, 1993; Lal et al. 1995; Cole et al. 1997; Paustian et al. 1997a,b,c; Six et al. 1998). However, whilst a review of 14 no-till experiments in Europe (Smith et al. 1998a) indicated a generally positive effect on SOC stocks, three experiments showed minimal or negative effects of NT on soil C levels. It is often difficult to equally compare CT and NT soils since ploughed soils are almost always less dense than no-till soils. Therefore CT and NT soils should be sampled to the same 'effective depth' (i.e. the same weight of dry soil), or measured soil C should be corrected for changes in bulk density. Changes in the plough depth can also affect results (Smith et al. 1998a).

There are several factors controlling changes in soil C under tilled or cultivated soils. Firstly, ploughing changes soil conditions that affect decomposition (temperature, moisture and oxygen availability; Six et al. 1998; Harrison et al. 1993b; Schlesinger, 1995; Paustian et al. 1997a), so under NT less C may be lost via oxidation from mixing the soil, and lower soil temperatures may slow decomposition (Kern and Johnson, 1993). Tillage-enhanced decomposition is often greatest shortly after the tillage event (Tracy et al. 1990). Secondly, the microbial community can be affected by ploughing and litter placement. Fungal decomposition may be greater than bacterial decomposition, resulting in more recalcitrant decomposition products (Kern and Johnson, 1993; Six et al. 1998). Thirdly, tillage intensity affects soil structure, mixing and aggregation (Six et al. 1998), by exposing new soil to wet-dry and freeze-thaw cycles at the surface, increasing soil disruption and surface area (Harrison et al. 1993b; Paustian et al. 1997b). Tillage also affects residue placement (Paustian et al. 1997a). C inputs under CT may also be lower than under NT (Harrison et al. 1993b; Schlesinger, 1995), and NT may reduce losses of soil C from erosion. Consequently, the mean residence time (MRT) of C in the soil may be up to 1.7x less under CT than NT (Six et al. 1998).

However, different tillage practices vary in their potential to increase soil C stocks. Whilst NT may conserve current SOC stocks, it may not consistently increase SOC stocks above CT (Kern and Johnson, 1993); Alemu et al. (1997) found no significant difference between SOC in the top 10cm of CT and Mul-T plots. Other management practices may increase or limit how much SOC is lost or gained by different tillage practices. For example, soil C increases under NT have been observed under low-N application treatments, but at high N rates soil C under CT can exceed NT-soil C (Patwardhan et al. 1995).

1.3.1.7 Tillage and SOC distribution

Many studies have shown that whilst there may be a small positive effect of NT on total soil C stocks in the profile, the effect is most pronounced in the top layers of the soil, perhaps due to distribution of C inputs (Paustian et al. 1997a). The topsoil layers of an NT profile are often cooler and wetter than one under CT, slowing decomposition, and a surface litter layer may build up under NT (Paustian et al. 1997b). NT often increases SOC in the 0-5cm or 0-10cm layer by up to 35% (Chan et al. 1992; Angers et al. 1993; McCarty et al. 1998; Campbell et al. 1998a,b; Doran et al. 1998; Wander et al. 1998; Six et al. 1998). However, SOC below 15cm depth under NT may be similar or even reduced compared to CT (Kern and Johnson, 1993; McCarty et al. 1998). As a result, no significant increase may occur over the whole 0-20cm soil layer, especially in the long-term (Chan et al. 1992; Doran et al. 1998; Wander et al. 1998; Campbell et al. 1999). The changes that occur in the transition from a typical profile under CT to a profile typical under NT often occur in the first few years of establishment (McCarty et al. 1998). Some burying of soil C below plough depth may also occur in CT soils (Kandeler et al. 1999). The mean residence time of topsoil SOC under NT may be up to three times greater than that in the subsoil, whilst residence times of topsoil and subsoil SOC are similar under CT (Six et al. 1998).

1.3.1.8 Tillage and Soil C pools

As discussed previously, changes in land use and management most commonly have the greatest impact on the more labile pools of soil C. This effect is often observed under cultivation and tillage. C losses or gains following changes in cultivation or tillage practice occur mostly from the active soil C pools (Harrison et al. 1993b; Angers et al. 1993; Lefroy and Blair, 1994; Six et al. 1998; McCarty et al. 1998; Kandeler et al. 1999). This may be because tillage destroys the macro-aggregates that protect more active forms of SOC (Wander et al. 1998). Conversely, the data of Whitbread (1994) suggest a significant loss of SOC from both the labile and non-labile pools of SOC.

1.3.1.9 Tillage and other soil factors

NT may reduce soil pH in the top layer due to increased Al brought to soil surface by inverse action (Chan et al. 1992), which could act to further lower decomposition rates. Texture is important in determining soil C changes under CT and NT - finer textured soils retain more C, and have slower turnover times (Liang et al. 1998), but the effects of NT may be inconsistent in fine textured and poorly drained soils

(Wander et al. 1998). This is largely due to problems with excess water during winter in fine textured soils under NT.

1.3.1.10 Tillage and management considerations

Although there is generally greater water capture and lower evapotranspiration under NT (Paustian et al. 1997a), there are management problems with NT under some conditions. Crop yields may be reduced under CoT since more intensive management is required, there may also be problems with weeds, insects and disease, and lower nutrient availability. Most problems occur in heavy textured or poorly drained soils in cold humid regions where the soil is extremely cold in spring (Kern and Johnson, 1993; Paustian et al. 1997a). Fossil fuel costs may also differ between systems, and grass and weed infestations may increase herbicide use (Campbell et al. 1998a). Kern and Johnson (1993) estimated that fossil fuel costs related to production of herbicides and field manipulation were 53 kg C ha^{-1} yr^{-1} under CT, 45 kg C ha^{-1} yr^{-1} under MT, and 29 kg C ha^{-1} yr^{-1} under NT. These costs represent the fossil fuel consumption, in terms of CO_2-C emitted, required in herbicide production, application and field manipulation.

1.3.1.11 Management of arable lands – other practices

In the long-term, arable rotations may alter the distribution and total amount of soil C in the profile, compared to continuous cropping (Lal et al. 1995; Monreal and Janzen, 1993; Whitehouse and Littler, 1984). Increased SOC, or reduced SOC loss is often observed under ley-arable (Wu et al. 1999; Paustian et al. 1997a,b) and legume-based rotations (Odell et al. 1984; Whitehouse and Littler, 1984; Alemu et al. 1997). If maize is the main annual crop, only rotations with grass or forage crops show consistently higher soil C levels compared to continuous maize. However, in wheat systems, only rotations including green manures or legumes, or long (>2 y) periods of hay show increases in soil C (Paustian et al. 1997a). Residue C amount and quality are important in determining soil C changes under rotations. The greatest C residues are associated with feed grains such as maize and sorghum, whilst soybeans produce around 50% of this amount (Paustian et al. 1997a).

Bare fallowing tends to reduce soil C by up to 40% relative to ordinary arable cropping (Cole et al. 1997; Paustian et al. 1997a), and may increase decomposition due to increased moisture, soil temperature and soil disturbance (from mechanical weed control) and erosion. Bare fallowing also increases erosion, and total C inputs are lower in continuous rotations compared to fallow rotations (Paustian et al. 1997a), although these changes in C inputs do not always result in soil C changes (Campbell et al. 1998b). Planted fallows may have the potential to reduce SOC losses (Lal et al. 1995), relative to bare fallow management.

Organic farming may have some potential to increase SOC relative to conventional farming (Gunapala et al. 1998). Residue burning after harvest has not been shown to have a significant effect on soil C in the short term (Ball-Coelho et al. 1993) although it may be important in the long term (Lal et al. 1995).

1.3.1.12 Management of grasslands

Pasture management has some potential to increase SOC stocks (Lal et al. 1995). Cutting and removing grass reduces C inputs and may reduce SOC stocks relative to grazed pastures (Jenkinson et al. 1987), and increased grazing intensity can reduce SOC stocks (Parton et al. 1987; Lal et al. 1995); fertilisation of grasslands can also increase SOC. However, Hassink (1994) suggests that grassland management practices have small effects on SOC.

1.3.1.13 Management of forests and other land uses

Natural changes in tree species may result in changes in SOC stocks, due to changes in C inputs and decomposition (Almendinger, 1990; Saetre et al. 1999), and fertilisation of forests can lead to increased SOC (Smith, 1999). Irrigation may also increase SOC stocks, for example in desert shrub-lands (Houghton et al. 1983).

1.3.2 Amendments and fertilisation

Fertilisation of arable lands is one practice which has the potential to increase SOC stocks (Schlesinger, 1995; Cole et al. 1997). Here the impacts of inorganic and organic fertilisation on SOC are considered.

1.3.2.1 Inorganic fertilisation

Since N fertiliser application generally increases crop growth, it may also result in increased residue decomposability (via higher residue N content) and production, thus increasing decomposition and carbon inputs at the same time (Paustian et al. 1997a). Combined with increased crop transpiration, there is often an increase of 25% in soil C to high levels of N fertiliser (Paustian et al. 1997a; Liang et al. 1996), although these effects are much smaller under non-N limiting conditions. Application of ammonium-based fertiliser without liming can lead to acidification and decreased decomposition rates (Paustian et al. 1997a), which would further increase soil C levels. N fertiliser can also increase microbial yield efficiencies (i.e. the amount of C fixed by microbiota versus the amount respired) and thus increase SOM, and increase the efficiency of SOM stabilisation (Paustian et al. 1997a).

1.3.2.2 Organic fertilization and amendments

Organic fertilisation and amendments may increase SOC in three ways. Firstly, fertilisation of the crop can increase residue C inputs and their quality. Secondly, the actual addition of organic amendments represents a simple addition of C to the soil. Thirdly, residues can affect soil physical properties such as bulk density, water infiltration, pore size distribution and aggregate stability, that also affect the soil C balance (Paustian et al. 1997a).

Different amendments have different potential to increase SOC, largely dependent on their quality (lignin and C:N ratio; Paustian et al. 1997a). However, the true potential of organic amendments to sequester C in soils depends on resource availability and requires full carbon accounting in evaluation as a practical option (Schlesinger, 2000; Smith and Powlson, 2000).

Farmyard manure (FYM) is pre-digested and relatively enriched in more stable compounds, and thus has the ability to increase soil C levels in the long-term. Therefore much of the C added via manure is held in the fine fractions of the soil that tend to have long turnover times for SOC (Paustian et al. 1997c). Green manures may have variable effects on SOC (Paustian et al. 1997a). Wood residue applications can result in an increase of 16-24% of SOM, mostly in the stable SOM fractions (N'dayegamiye and Angers, 1993). However, there is little difference in the ability of different straw types to increase SOC, both in the short and long term (Curtin et al. 1998). Residue placement has important effects on soil C changes: Curtin et al. (1998) showed that most SOC mineralisation occurred in plots with no straw incorporation, followed by plots with surface straw, and the least C mineralisation in plots with straw incorporated.

1.4 CLIMATE CHANGE IMPACTS ON SOM

The effects of changes in climate on SOM are highly complex and interactive (Batjes, 1998; King et al. 1997), and small changes in parameters controlling SOM storage could result in large C losses (Parshotam et al. 1995). Climatic changes may include changes in temperature, rainfall and CO_2 concentrations and therefore SOM responses to climatic change will be controlled by all or any one of these parameters.

The exact nature of SOC change under climate change depends on ecosystem properties (Wang and Polglase, 1995), and how climatic change affects the crop productivity and C input rates, and decomposition rates. However, the impact of management may be a stronger control on SOC than climatic change, so management could be used to mitigate climate change effects (Paustian et al. 1997c). As well as 'in situ' effects, climatic change could also affect both the crops grown and their management in particular regions, i.e. their zonal distribution. It is still uncertain in which direction SOC pools will change under current climate change predictions, at the global or local level.

1.4.1 Climatic warming

Increased temperatures could be expected to have a number of effects on SOC stocks, and several authors have suggested that increased temperature alone would decrease SOC storage over the 10-100 y timescale (Jenkinson et al. 1991; Tate, 1992; Smith and Shugart, 1993; Parshotam et al. 1995; McGuire et al. 1995; Rustad and Fernandez, 1998). However, total soil C and vegetation C storage may increase at equilibrium (Smith and Shugart, 1993). Higher temperatures should increase SOM decomposition (Paustian et al. 1997c), leading to less soil C storage, although this depends on whether, or how, decomposition is temperature-limited at any particular site. Warming and drying could also accelerate DOC production, although DOC export depends on the soil absorptive capacity (Tipping et al. 1999*).* Assuming water and nutrients are not limiting, plant production and hence C inputs may be greater at higher temperatures (King et al. 1997), so there may be some balancing effect to a net loss of SOC (Gifford et al. 1996). However, soil N and P could be major controls on SOC storage changes under climatic warming (McCane et al. 1995), and increased temperature could increase water loss from soil.

1.4.2 Changes in rainfall patterns

In a similar way to changes in temperature, changes in rainfall could affect both the input of C to soil and the decomposition of SOM. Increased rainfall may increase plant production, thus increasing C inputs, and SOC storage. In soils where moisture limits decomposition, rainfall would increase decomposition, thus decreasing SOC storage, but excessive moisture could limit decomposition and increase SOC storage. Increased rainfall could also increase losses from SOC via DOC export (Tipping et al. 1999). Hence whether increased rainfall actually increases or decreases SOC storage depends on whether moisture limits SOM decomposition or crop production (Paustian et al. 1997c). The effects of changes in rainfall patterns need to be considered in combination with changes in temperature and atmospheric CO_2 (e.g. King et al. 1997; Peng and Apps, 1998).

1.4.3 Elevated CO_2

Elevated CO_2 could have important effects on ecosystem C storage, but all effects of atmospheric CO_2 on the soil are indirect and affect the soil only through the vegetation. For example, elevated CO_2 could cause a net increase in SOC storage globally (Gifford et al. 1996), or regionally (Hirschel et al. 1997; Peng and Apps, 1998), or compensate for the soil C losses due to climatic warming (McGuire et al. 1995, 1997). Elevated CO_2 could 1) increase plant production (the 'CO_2 fertilisation effect'), 2) cause the C:N ratio of plant litter to increase, lowering decomposition and increasing SOM storage, and 3) enhance root turnover. Other plant-related effects include increased water use efficiency and photosynthesis, and decreased evapotranspiration (Peng and Apps, 1998). Effects on the soil microbial biomass appear to be small (Van Ginkel and Gorissen, 1998). In annual crops, increased CO_2 generally increases crop productivity (Van Ginkel and Gorissen, 1998), but in natural ecosystems, the long-term impact of increased CO_2 on ecosystem sustainability is not known (Mosier, 1998). In addition, most raised CO_2 experiments involve a direct change to doubled CO_2, not a ramp, or a more realistic gradual change, so it is hard to determine how plants might adapt in the long term (Drake et al. 1997).

1.4.4 Changes in rainfall, temperature and CO_2 - combined scenarios

The climate over land with double the current atmospheric CO_2 concentration is predicted to be an average of 5.4^{0}C warmer and 17.5 mm d^{-1} wetter than the present climate (Prentice and Fung, 1990). Changes expected include a large increase in SOC storage in tropical forests, a reduction in desert SOC stocks, expansion of cool temperate forests, and a loss of the area of boreal forest and tundra. This would lead to a net 1% increase in global SOC stocks (Prentice and Fung, 1990). At the global scale and regional scale, climatic change alone could increase NPP and decomposition, resulting in net SOC losses (King et al. 1997; Paustian et al. 1997c). Under elevated CO_2 and climate change scenarios, NPP could be further enhanced, increasing soil storage C (King et al. 1997; Peng and Apps, 1998). The exact nature of SOC changes are unclear (Gifford et al. 1996) and vary across regions dependent on local soil and climatic factors, the ecosystem type (Peng and Apps, 1998), and how climatic change affects the productivity of different crops, their C input rates, and SOM decomposition rates (Paustian et al. 1997c). The brief review of model estimates presented here describes only the overall global changes, and not locally specific effects.

1.5 MODELLING SOM

1.5.1 SOM models

There are several sources of metadata and information on SOM models. CAMASE (Plentinger and Penning de Vries, 1996; online at http://www.bib.wau.nl/camase/) contains 98 agroecosystems models having soil components, and the Global Change and Terrestrial Ecosystems Soil Organic Matter (GCTE-SOMNET) database (Smith et al. 1996a,b: online at http://saffron.res.bbsrc.ac.uk/cgi-bin/somnet) contains metadata on over 30 current operational SOM models. Several authors have previously reviewed SOM models extensively. Molina and Smith (1998) reviewed 24 SOMNET registered models for carbon and nitrogen processes in soils; McGill (1996) classified and reviewed 10 SOM models, with respect to environmental conditions, scale, regulation by soil properties, and compartmentalisation; Jenkinson (1990) classified SOM models. Smith et al. (1999) reviewed SOM modelling in tropical ecosystems, covering model use, input requirements and outputs; Smith et al.

(1998b) reviewed SOM models with respect to the description of soil biota; Donigan et al. (1994) extensively reviewed 14 soil carbon models.

There is a huge diversity in the understanding and interpretation of SOM processes in current SOM models (Molina and Smith 1998). SOM models can be broadly classified as 1) single homogenous compartment; 2) two compartment; 3) non-compartmental decay; or 4) multi-compartmental (Jenkinson, 1990). Models in categories 1) and 2) are mostly static (i.e. where the environmental variables remain constant), whereas the models in categories 3) and 4) are mostly dynamic (i.e. where environmental variables vary with time); the dynamic models can be further split into organism oriented and process oriented models (Paustian, 1994).

The majority of SOM models are discrete, deterministic and multi-compartmental, describing SOM as a finite number of subsystems (pools or compartments), each of which is homogeneous and well mixed. The compartments interact by exchanging materials, and by exchanges with the environment (Parshotam, 1996). The transfer to and from compartments is modelled using first order equations (i.e. where the rate is dependent upon the amount of substrate present), giving a linear system of equations.

The turnover rates for each compartment are often derived from empirical relationships (Molina and Smith, 1998) and almost all models include a pool that has a relatively rapid turnover time (labile SOM pool), and a pool that turns over slowly (RSOM pool).

These models are usually a mixture of mechanistic (based on the kinetics involved) and empirical; the empirical relationships used may not have any theoretical or physical interpretation, and may simply be devices to fit the model to available data (Christensen, 1996; Parshotam 1996). Empirical fitting or direct measurements are most commonly used to fit and compartmentalise models (Table 1.1; Falloon and Smith, 2000); SOM turnover is often assumed to be affected by moisture, temperature, clay content, and pH or N availability (Table 1.2; Falloon and Smith, 1998, 2000).

Continuous SOM models (e.g. Janssen, 1984; Bosatta and Agren, 1985) and continuous forms of discrete SOM models (e.g. Parshotam, 1996: Martin, 1998) or analytical solutions, have great potential in estimating future and past SOC stocks, investigating the effects of splitting and lumping pools, and validating models (Martin, 2001). However, there are relatively few continuous form models compared to discrete form models (Myers, 1994). The Analytical Biogeochemical Cycling (ABC) scheme has been used to derive a continuous form of CENTURY. ABC has been used for 1564080 runs, 10 vegetation types, 7 soil types and 21 climate regimes, to show that soil C dynamics should not amplify the direct forcing of climatic perturbations (Martin, 2001).

1.5.2 Long-term datasets

There are many long-term experiments (LTE's) with measurements of soil C that could be of use in testing and validating SOM models. The review in this section concentrates on experiments registered with the GCTE-SOMNET, which has strict conditions for dataset collection and data quality. The majority of SOMNET datasets are also over 20 y in length. One other major site network is the US Long Term Ecological Research Network (US-LTER); Steiner and Herdt (1993) give a review of other, non-European long-term experiments. In reviewing available data sets, it is important to recognise that different analytical methods of measuring soil C give quite different results (Kalembasa and Jenkinson, 1973) and some standardisation may be necessary to make datasets comparable.

Figures 1.6-1.13 show environmental and management aspects of the long-term experiments in SOMNET. The vast majority of SOMNET LTE's are located in Europe, with a relative lack of experiments in Asia, North America, Africa and Australasia. Experiments in North America are included in the US-LTER, but the lack of experiments in the other continents may be related to a lack of knowledge of experiments, or to a genuine lack of experiments in these regions. As a consequence, most experiments are also in the cool-temperate zone, with a lack of experiments in cold temperate boreal, warm temperate subtropical and tropical zones.

Table 1.1 SOM models – model fitting, compartmentalisation and rate constants

Model	Model fitting and compartmentalisation	Soil Organic Matter (SOM) rate constant, y^{-1}	Reference
Continuous models			
Q-Soil	EF	SOM	Bosatta and Ågren (1991)
Single compartment models			
Model of Humus Balance	EF	SOM	Schevtsova and Mikhailov (1992)
NAM SOM	EF	SOM	Ryzhova (1993)
O'Brien's Model	None	SOM	O'Brien (1986)
Multi-compartment models - RSOM compartment uncoupled			
APSIM	EF, CN	BIOM, HUM, INERT-C	McCown et al. (1996)
Candy	EF	Active organic matter, 0.14; Stable organic matter, 0.05; Inert organic matter	Franko et al. (1995)
DAISY	EF, CN, DM	SMB1, 0.36; SMB2, 3.6; SOM1, 0.001; SOM2, 0.05; SOM0, 0	Jensen et al. (1994)
RothC	EF, C14, Bio	Microbial biomass, 0.66; Humified OM, 0.02; IOM, 0.00002	Coleman and Jenkinson (1996)
Multi-compartment models - RSOM compartment coupled			
ANIMO	EF,DM	Root exudates, 36; Dissolved organic matter, 2.9; Stable humus, 0.2	Rijtema and Kroes (1991)
CENTURY	DM, Clay, S	Surface microbes, 6.0; Soil microbes, 7.3; Slow soil organic matter, 0.2; Passive soil organic matter, 0.0045	Parton et al. (1987)
Chenfang Lin Model	EF, CN	Labile decomposable; Resistant decomposable; Decomposable SOM; Active biomass; Inactive biomass; Humic	Lin et al. (1987)
DNDC	EF	Labile microbial mass, 120; Resistant microbial mass, 15; Labile humads, 58; Resistant humads, 2.2; Passive humus	Li et al. (1994a,b)
DSSAT	CN, DM, Bio	Fresh OM I, 0.2; Fresh OM II, 0.05; Fresh OM III, 0.0095; Humus, 8.3×10^{-5}	Hoogenboom et al. (1994)
D3R	DM,EF	Fast; Slow	Douglas and Rickman (1992)
Ecosys	DM, Bio, Hp	Soluble SOM; Adsorbed SOM; Microbial SOM; Microbial residues; Active SOOM; Particulate SOM; Acetate; Methane	Grant (1995)
EPIC	DM, Tc	Active SOM; Stable SOM	Williams and Renard (1985)
ForClim-D	EF	Fast humus, Slow Humus	Perruchoud (1996)
FERT	EF, DM	Microbial biomass; Humus	Kan and Kan (1991)
GENDEC	EF	Live microbiota; Dead microbiota, labile, 73; Dead microbiota, recalcitrant, 0.36	Moorhead and Reynolds (1991)
Hurley Pasture / ITE Forest	EF, CN, Litter	Biomass; SOM dead	Thornley and Cannell (1994)
ICBM	CN, EF, DM	Young SOM, 0.6; Old SOM, 0.13	Andrén and Kätterer (1997)
KLIMAT-SOIL-YIELD	EF, CN	Rapidly decomposable SOM; Slowly decomposable SOM; Living SOM; Old SOM1; Fresh SOM; Old SOM2	Sirotenko (1991)
McCaskill and Blair	EF, CN, Ti	Unprotected and protected biomass	McCaskill and Blair (1990)
MOTOR	EF, DM	User defined	Whitmore (1995)
NCSOIL	EF, CN, No, DM, Bio	Pool I, 120; Pool I, 15; Pool II, 2.2; Pool II, 58; Pool III, <0.04	Molina et al. (1983)
NICCE	EF, Bio, N15	Microbial biomass, Metabolic, Structural, Humic, Stable, Resistant	Van Dam and Van Breeemen (1995)
O'Leary Model	EF, N%	Fresh SOM, 5.1; Microbial biomass, 1.4; Stable humus, 0.014	O'Leary (1994)
SOCRATES	DM	Microbial biomass (unprotected); Microbial biomass (protected); Humus (stable pool)	Grace and Ladd (1995)
SOMM	EF, CN, Ash	Litter; Humic substances	Chertov and Komarov (1996)
SUNDIAL	EF, Yield	Microbial biomass, 0.66; Humus, 0.21	Smith et al. (1995)
Verberne	EF, Clay	Non-protected biomass ; Protected biomass ; Non-protected SOM; Physically protected SOM, Stabilised SOM	Verberne et al. (1990)
VOYONS	EF, Clay, LN	Active SOM, 7.3; Structural OD, 4-5; Slow SOM, 0.2; Passive SOM, 0.0067	André et al. (1992)
Wave	CN, Bio	Soil litter; Soil manure; Soil humus , 0.025-0.36	Vanclooster et al. (1996́)

EF = Empirical fitting, DM = Direct measurement, Clay = Clay content, S = sand content, No = Nitrogen mineralisation potential, LN = Lignin/N ratio, Ash = Ash content, CN = C/N ratio, Ex = Extraction, Hp = Hydrolysis potential, Tc = Time since cultivation, Litter = Litter, Bio = fumigation/incubation/biomass, Ti = Time since incorporation, C14 = Radiocarbon dating, Yield = Yield, N15 = ^{15}N enrichment

Table 1.2 SOM models - factors affecting turnover and refractory components

Model	Factors assumed to affect SOM turnover	Refractory compartment	Parameters defining refractory compartment	How parameters defined
Continuous models				
Q-Soil	Bio, N, Res	Single pool	R_c	EF
Single Compartment models				
Model of humus balance	Clay, pH, Crop, N, Nut, Lime	SOM	Clay, pH, Crop, N, Nut, Lime	F
NAM SOM	Water, Temp, Clay, Crop	SOM	Water, Temp, Clay, Crop	Lr
O'Brien's Model	None	SOM	C14	C14
Multi-compartment models - RSOM compartment uncoupled				
APSIM	Water, Temp, pH, N, Depth	INERT-C		F, Diff
CANDY	Water, Temp, Clay, N	Inert organic matter		F
Daisy	Water, Temp, Clay, N	SOM0 (Inert)	TotC, CN	Diff
RothC-26.3	Water, Temp, Clay, Crop	Inert organic matter	C14, TotC	F or Reg
Multi-compartment models - RSOM compartment coupled				
ANIMO	Water, Temp, pH, N, O_2	Stable humus	R_c, N	F
Chenfang Lin	Water, Temp, Bio	Humic material	R_c	F, Lr
CENTURY	Water, Temp, Clay, pH, N	Passive SOM	Clay, R_C	F
DNDC	Water, Temp, Clay, Till	Passive humus	Clay R_c	F
DSSAT	Water, Temp, N, Clay	Humus (SOM)	TotC	F
D3R	Crop, N, Temp, Res, Till	Slow	R_c	DM
Ecosys	Water, Temp, Clay, Crop, N, O_2, Nut	Passive SOM	HydP, Clay	F
EPIC	Water, Temp, CEC, Clay, pH, Crop, Nut	Stable SOM	Time	F
FERT	Water, Temp, pH, Crop, N	Humus C	R_c,, CN	F, DM
ForClim-D	Temp, Water	Slow Humus	Temp, water, C14	Temp, water, C14
GENDEC	Water, Temp, N	Dead microbiota, recalcitrant	Water, temp, CN, Bio	DM
ITE Pasture / Hurley Forest	Water, Temp, N	SOM (dead)	CN, Bio, R_c	F
ICBM	User defined - external factors in one parameter	Old SOM	User defined	User defined
KLIMAT-SOIL-YIELD	Water, Temp, pH, CN, Crop	Old SOM2	Water, Temp, Crop, CN	F
CNSP	Water, Temp, pH, N, Nut, Crop, Res	Protected	Res, Crop, Time	F
MOTOR	Water, Temp, Clay, pH, N	User defined	R_C	F or DM
NCSOIL	Water, Temp, Clay, pH, N,	Pool III (Stable pool)	R_c, CN, TotC	Diff
NICCE	Water, Temp, Clay, Depth, N	Resistant	R_c	
O'Leary	Water, Temp, Clay, N, Till, Res	Stable humus	TotC, R_c	Lr
SOCRATES	Water, Temp, CEC, N, Crop	Humus pool	R_c	DM
SOMM	Water, Temp, N	Humic substances of mineral topsoil	CN, R_c	F
SUNDIAL	Water, Temp, Clay, Res	Humus	Water, Temp, Clay, Res	Tex
Verberne	Water, Temp, N, Clay	Stabilised SOM	Clay	F, Diff
VOYONS	Water, Temp, Clay, Depth	Passive SOM	Lignin, R_c	
Wave	Water, Temp, N, Depth	Soil humus	R_c,, Water, Temp, Bio, CN	F, Lr

Water = water availability, Temp = temperature, pH = pH, N = Nitrogen, 0_2 = Oxygen availability, Clay = clay content, Bio = biomass, Till = Tillage factors, Crop = crop cover/growth period, Res = residue quality/quantity, CEC = Cation Exchange Capacity, Nut = other nutrients, Lime = Lime, R_c = decomposition rate constant, CN = C/N ratio, Tex = texture, TotC = total C, HydP = hydrolysis potentials, Time = time factors, lignin = lignin content, C14 = C^{14} age, Depth = soil depth, Diff = by difference/subtraction, F = fitting, DM = direct measurement, Lr = Literature values, R=regression

Figure 1.6 Location of SOMNET long-term experiments

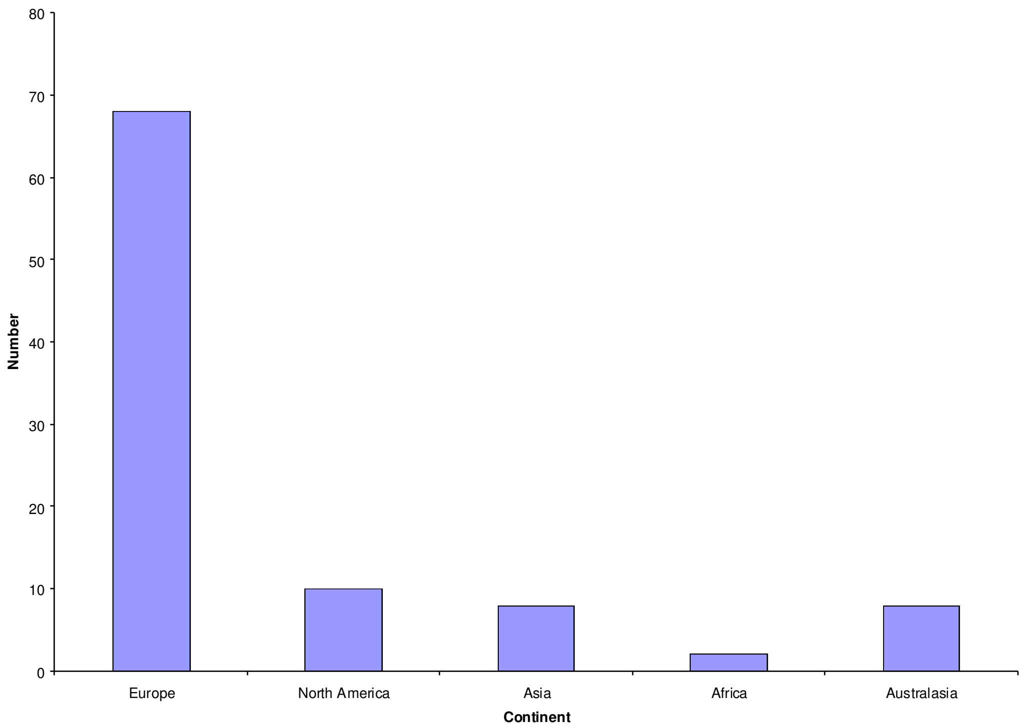


Figure 1.7 Climatic zone of SOMNET long-term experiments

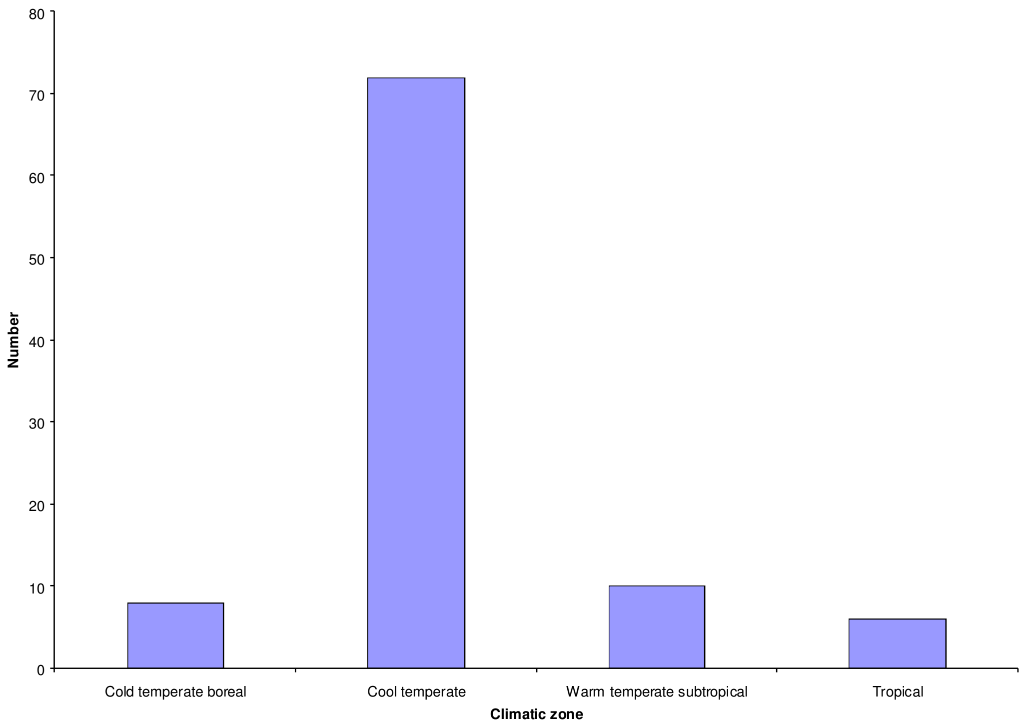


Figure 1.8 Duration of SOMNET long-term experiments

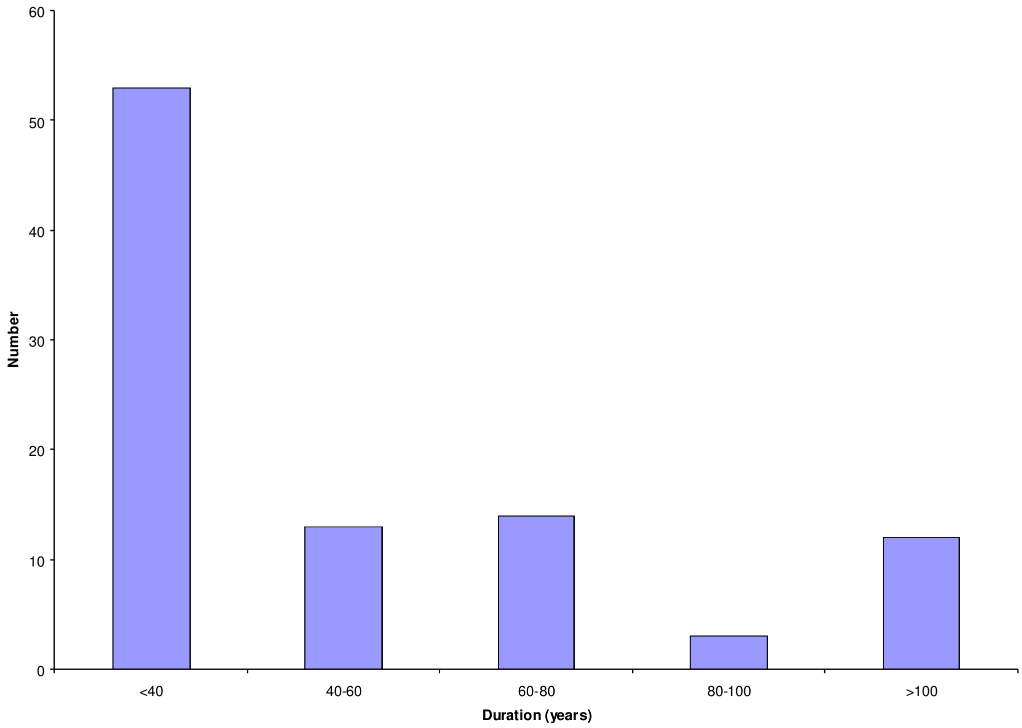


Figure 1.9 Main land use of SOMNET long-term experiments

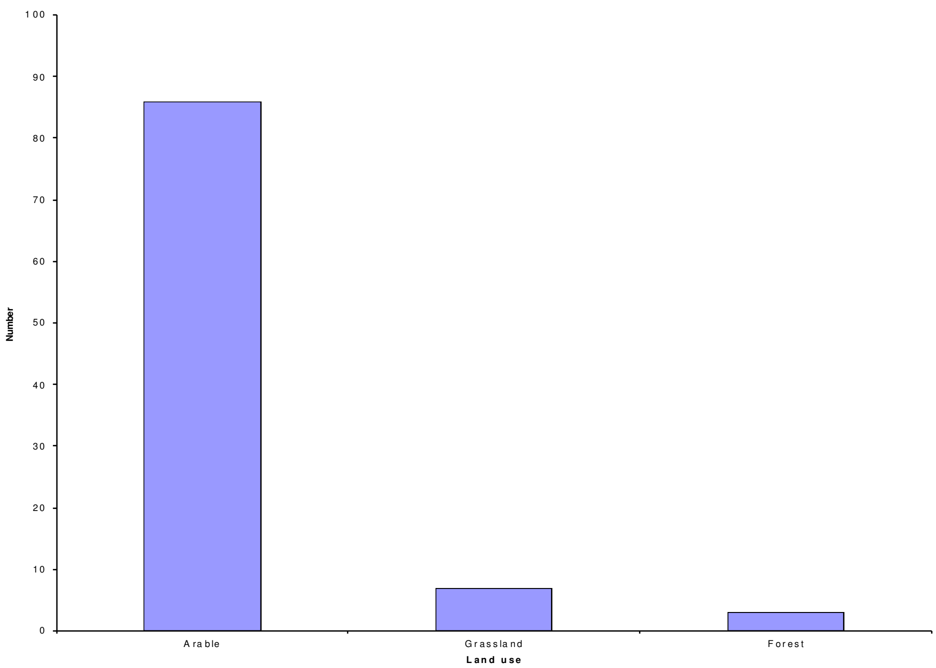


Figure 1.10 Treatments of SOMNET long-term experiments

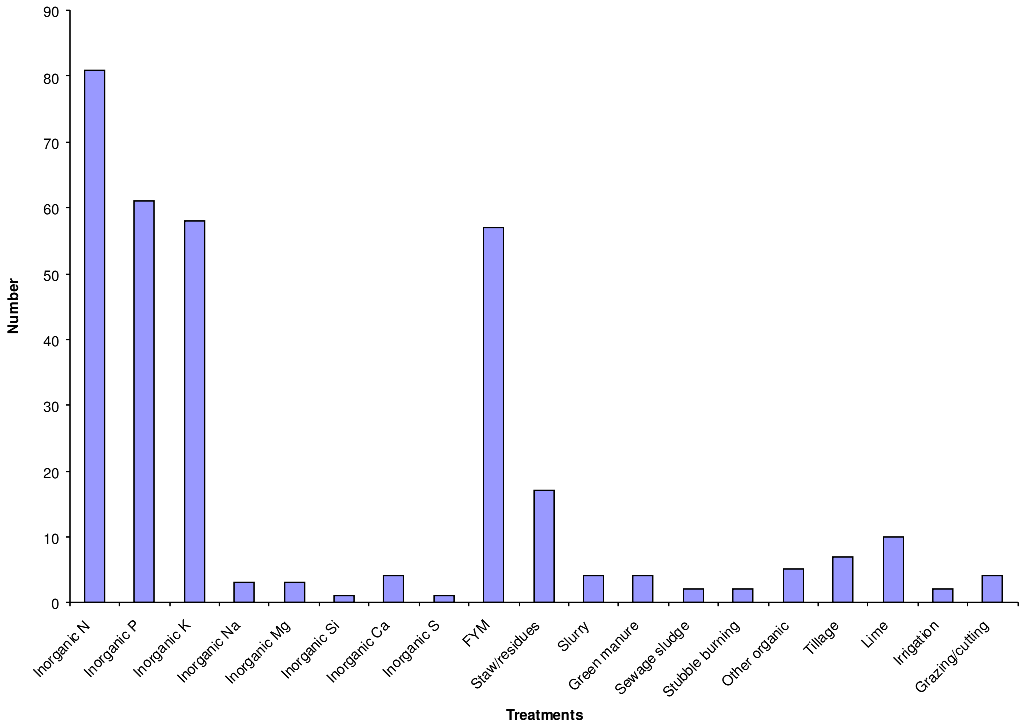


Figure 1.11 Soil clay % of SOMNET long-term experiments

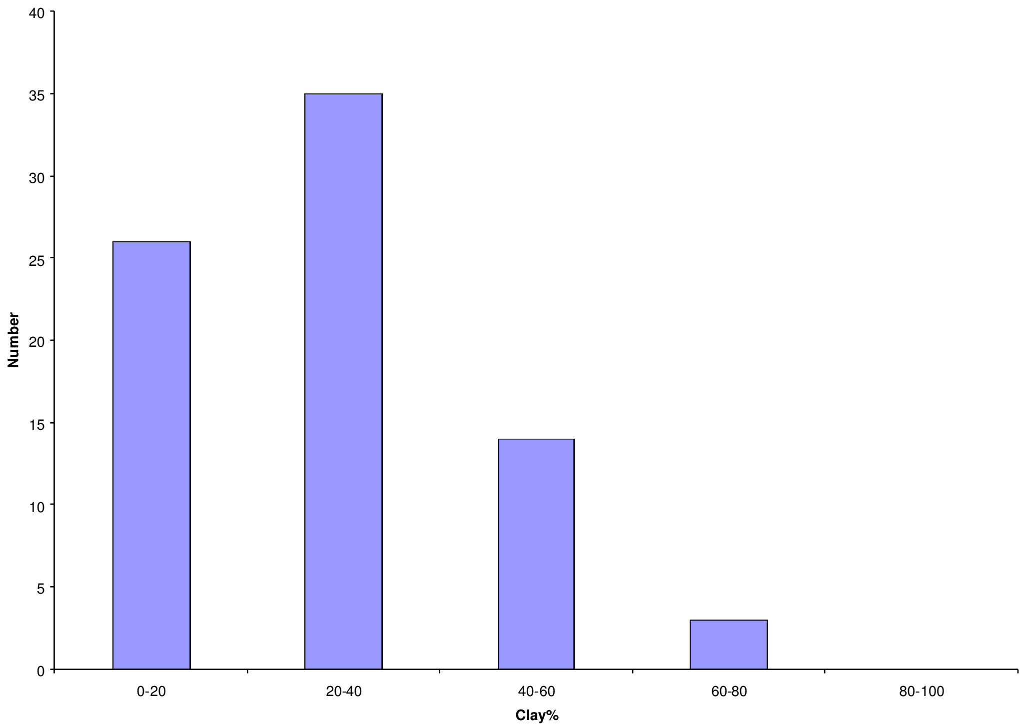


Figure 1.12 Soil C% of SOMNET long-term experiments

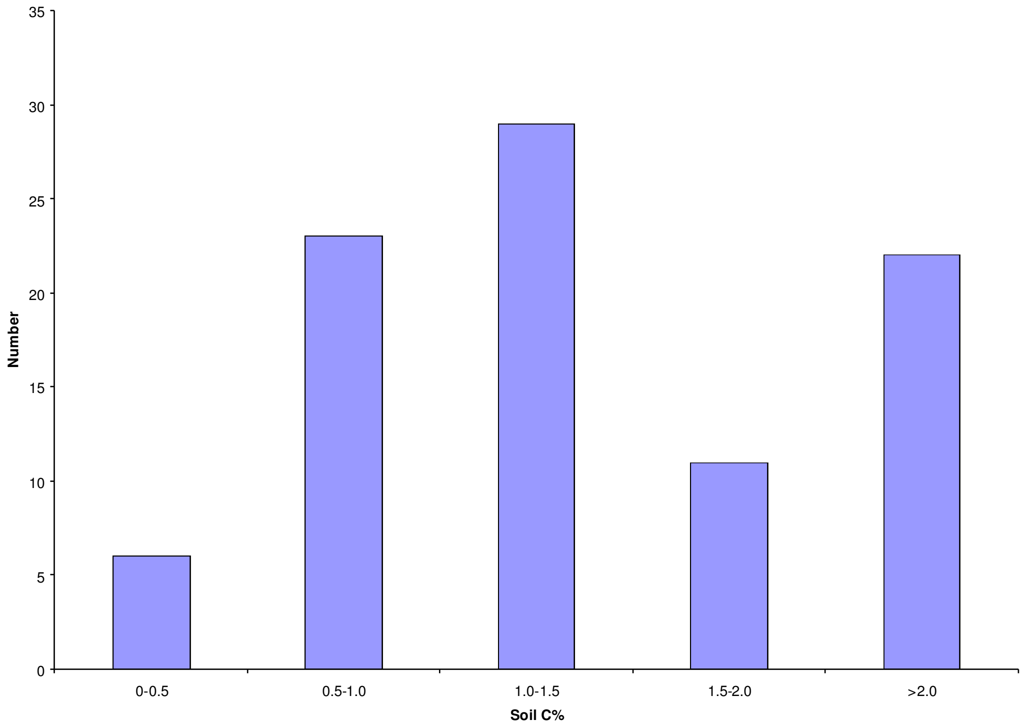
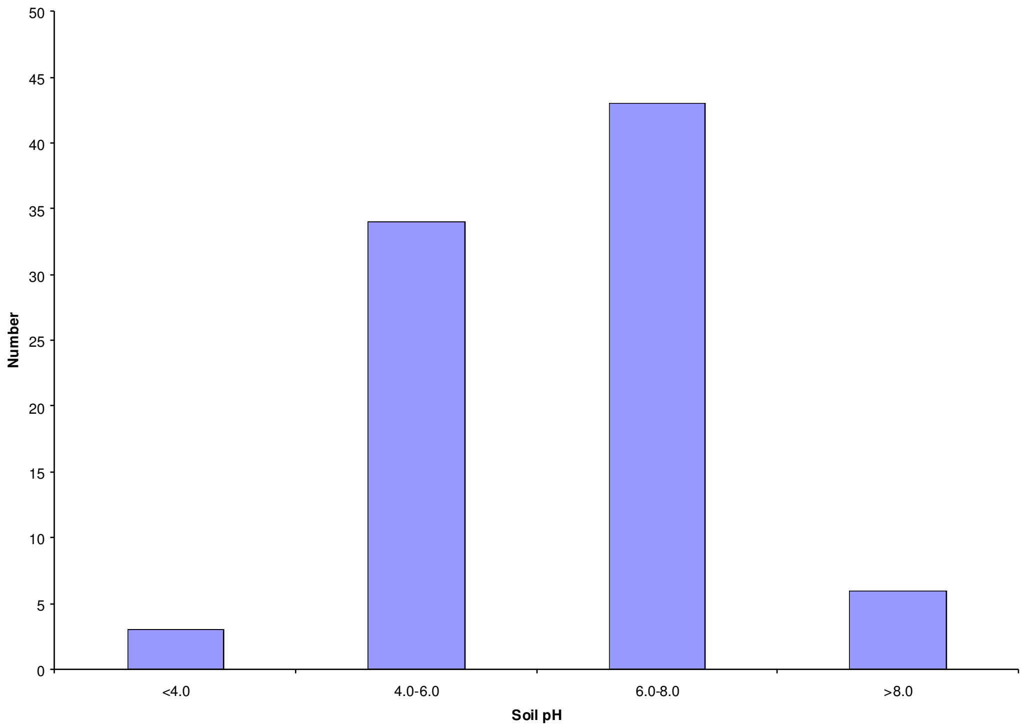


Figure 1.13 Soil pH of SOMNET long-term experiments



There is also a lack of experiments over 100 y in age – most experiments tend to be less than 40 y old. Most experiments are primarily under arable land use, and forestry and grassland are under-represented. There are many experiments with inorganic fertilization treatments (N, P and K) and FYM application, but relatively few experiments covering other nutrients, different organic amendments, irrigation or tillage. The experiments in SOMNET cover a wide range of soil textures and soil C%, mostly under 40% clay, and soil C contents mostly in the range 0.5-1.5% C. Finally, most experiments are on soils in the neutral range, with few experiments on extremely acid or alkaline soils. The range of soil types covered mainly reflects the range of normal arable soil properties.

1.5.3 Model application

Models for the turnover of SOM are currently being used to a) better understand SOM turnover processes (e.g. Jenkinson and Coleman, 1994; Smith et al. 1998c; Falloon and Smith 1998, 2000; Falloon et al. 1998a; Romanya et al. 2000); b) predict agronomic yields and make fertiliser recommendations (e.g. EPIC - Williams and Renard, 1985; DSSAT - Hoogenboom et al. 1994; SUNDIAL - Smith et al. 1995; for a review, see Falloon et al. 1999a); and c) aid policy decisions regarding carbon sequestration options, or investigate the effects of climate change and land use on SOM (e.g. Jenkinson et al. 1991; Lee et al. 1993; Donigan et al. 1994; Parshotam et al. 1995; King et al. 1997; Falloon et al. 1998b, 1999b, 2000, 2001a,b). It is therefore essential that the science underpinning SOM turnover models is robust, and that models are well validated for the full range of conditions under which they are to be used.

CENTURY and RothC are amongst the most widely tested SOM models, with generally good agreement between observed data and simulated results (Parshotam and Hewitt, 1995; Smith et al. 1997a). CENTURY and RothC are also being increasingly used for studies of climate and land use change (e.g. Jenkinson et al. 1991; Donigan et al. 1994; Parshotam et al. 1995; King et al. 1997; Falloon et al. 1998b, 1999b, 2000, 2001a,b). Section 2.5.1 of Chapter 2 gives an overview of CENTURY and RothC applications. This section therefore aims to give a general, but not exhaustive, overview of some areas of application of other SOM models.

1.5.3.1 Site-scale model applications

SOM model applications at the site scale have included calibration of NCSOIL to long-term incubations and labelled residue experiments in the UK, Nigeria, Australia, Canada, the USA (Nicolardot et al. 1994) and elsewhere (Hadas et al. 1998). ICBM has been used to analyse the effect on management on SOC in European field experiments (Katterer and Andren, 1999). Tietema and Van Dam (1996) used the NICCE model to estimate the role of N deposition in soil C and N cycling at the site scale. Continuous form models have also been developed and applied (Parshotam, 1996; Hyvonen et al. 1999).

1.5.3.2 Regional scale model applications

Burke et al. (1990) used soil and climate databases to simulate SOM levels in Colorado grasslands. Simple steady state models have been used to investigate regional changes in Oregon soil C storage under changes in climate, using different regional datasets (Homann et al. 1998). Gifford et al. (1996) used the model CQUESTN at the global scale to investigate global atmospheric change effects on terrestrial carbon sequestration. Post et al. (1996) give an overview of SOM model application at the global scale.

1.5.4 Application of models to explore land use change and climate change scenarios

Various SOM and ecosystem models have been applied to determine the effects of different scenarios of climate and land use change on SOM. For example, simple regression models have been used to estimate C fluxes resulting from moving the British cattle herd after the BSE crisis (Smith et al. 1996c). Regression models have also been used to determine SOC controlling factors in French forest soils (Arrouays et al. 1995), and to follow SOC changes in Canadian forests (Rapalee et al. 1998).

At the continental scale, Klein Goldewijk and Vloedbeld (1995) applied the IMAGE2 model to investigate CO_2 exchange between the atmosphere and biosphere in Latin America. Li et al. (1994b, 1996) used the DNDC model to estimate nitrous oxide emissions from agriculture in the United States.

At the global level, the TEM model has been used to investigate soil C and NPP responses to climate change (Melillo et al. 1993; McGuire et al. 1995) and doubled CO_2 (McGuire et al. 1997), showing that carbon storage could be further enhanced by changes in plant N content anticipated under elevated CO_2. Ecosystem models have also been used to estimate global carbon fluxes from land use change historically (e.g. Emanuel et al. 1984). A review of SOM model application to climate change scenarios is given by Post et al. (1996).

One major weakness of many modelling studies of climate and land use change is a lack of model comparisons, especially at the regional, national or global scale.

1.5.5 Model weaknesses

Weaknesses of RothC and CENTURY are discussed in Chapter 2: this section focuses on general SOM model limitations and weaknesses of other SOM models. There may be many weaknesses and limitations of SOM models, since most were parameterised under particular management or climatic regions. Ideally, SOM models should account for all major SOM controlling factors, such as parent material, time, climate, biota and management. These factors may have complex interactions, and separate analysis of controls could limit predictions of their effects on SOM (Burke et al. 1989).

As noted in section 1.5.3, many models have been evaluated under different climatic and management conditions, but rarely compared using common datasets. There is a need to assess C turnover times and pool sizes at different soil depths, under different climates and land uses, in a more comprehensive way than has been previously done. Model evaluation exercises are extremely useful for identifying model weaknesses and areas for improvement; global networks of long-term experiments should prove invaluable for this purpose (e.g. Smith et al. 1996a,b). A soil radiocarbon database with climatic, management, and other soil data could also be invaluable for evaluating SOM models (e.g. Becker-Heidmann, 1996).

Few models simulate aggregation processes, which may be important in the stabilisation of plant residues, microbial biomass and humic substances (Parton et al. 1989; Paul et al. 1995). Differences in drainage are often not accounted for in SOM models. However, SOM accumulation in grassland and forest soils may be partly attributable to reduced drainage (Jenkinson, 1988). SOM models also need better integration with landscape processes, and better description of plant root development (Paul et al. 1995), layering, dissolved organic carbon and deep soil processes (Parton et al. 1989, 1994; Jenkinson et al. 1999b). Few models explicitly account for mesofauna and macroorganisms such as earthworms (Parton et al. 1989; Woomer, 1993; Lavelle et al. 1997), which are an important part of the SOM system (Buringh, 1984; Smith et al. 1998b).

Some authors have questioned the temperature-decomposition relationships conventionally used in SOM models. Many SOM models use a Q_{10} (Ahhrenius) relationship to relate decomposition to temperature (Agren et al. 1996), as shown in equation (1)

$$(1)\ k_2 = k_1 Q_{10}^{(T_2 - T_1)/10}$$

In this equation, k_2 and k_1 are rate constants at two observed temperatures T_2 and T_1. Agren et al. (1996) states that this relationship lacks any theoretical justification. It may also be hard to apply to a system such as a population of soil organisms, where the total activity is determined by a whole range of different organisms with quite different responses to temperature. A further difficulty is that Q_{10} values also change with temperature and between systems (Agren et al. 1996; Lomander et al. 1998). This process may be better modelled using a quadratic function (Lomander et al. 1998).

1.6 AIMS AND OBJECTIVES OF THE PROJECT

Globally, CO_2 concentrations in the atmosphere have been increasing since pre-industrial times. This has been linked to changes in global climate, in particular climatic warming, or the 'greenhouse' effect. One possible way in which atmospheric CO_2 concentrations could be reduced or stabilised is via sequestration of carbon in the soil. Indeed, the Intergovernmental Panel for Climate Change (IPCC) identifies three main carbon mitigation options for agriculture: (a) reduction of agriculturally related emissions, (b) the use of biofuels to replace fossil fuels, and (c) the sequestration of carbon in soils (IPCC, 1996). C sequestration is defined as 'any net persistent increase in organic C storage' (Paustian et al. 1997b). However, a more stringent definition of carbon sequestration in relation to management practices would be 'any net persistent increase in organic C storage, directly due to to a deliberate change in land management, with the aim of changing C storage'. There are a number of possible options for SOC sequestration. These could include:

- soil erosion management
- land conversion and restoration (CRP, conservation buffers, wetland restoration, restricted use of organic soils)
- restoration of degraded soils (eroded lands, minelands and toxic soils, salt-affected soils, creation of biofuels to offset fossil fuels)
- intensification of agricultural production on prime agricultural land
- conservation tillage and residue management
- irrigation water management
- improved cropping systems,
- better use of organic manures and by-products (Lal et al. 1999).

Although the C content of agricultural soils is generally low, when compared to peat soils, or some natural ecosystems, the potential to increase current C stocks is greatest for cropland (Smith et al. 1997b). The results presented in this thesis are concerned with SOC sequestration in arable lands. Regional estimates of the carbon sequestration potential of agricultural practices are crucial if policy makers are to plan future land-uses with the aim of reducing national CO_2 emissions.

The three methods that have been previously used to estimate changes in regional soil organic carbon (SOC) stocks are outlined below. Firstly, Smith et al. (1997b, 1998a,c, 1999, 2000a,b,c,d,e, 2001a,b), and Gupta and Rao, 1994) used regressions of changes in SOC stocks under changes in land use or management based on long-term experimental data to calculate annual percent increases in SOC, and extrapolate future changes in SOC stocks under different scenarios. Regression-based estimates are limited in their ability to predict long-term soil C dynamics in a changing environment (Peng et al. 1998).

Secondly, Kern and Johnson (1991) and Kotto-Same et al. (1997) used regressions based on SOC changes in long-term experiments, but applied their relationships to spatial soil databases, allowing them to take the local soil characteristics into account. The main limitation of approaches based upon linear regressions is that they assume a constant rate of SOC change throughout the time period of the scenario under study. Long-term experiments show that there is a period following a change in land use or management when the annual change in SOC is more rapid, followed by a period of much more gradual change (see, for examples, the data sets modelled by Coleman et al. 1997).

Thirdly, studies have used dynamic models linked to spatial databases (or Geographic Information Systems, GIS). Examples are given by Williams and Renard (1985), Prentice and Fung (1990), Jenkinson et al. (1991), Lee et al. (1993), Donigan et al. (1994), Parshotam et al. (1995), Post et al. (1996), King et al. (1997), Paustian et al. (1997d), Falloon et al. (1998b, 1999b, 2001a,b), and Smith et al. (1999). This approach allows the local soil characteristics, meteorological conditions and land use to be taken into account, allows dynamic estimates, is amenable to a range of scenarios, and allows geographic areas of particular C sequestration potential to be identified.

Following regression-based estimates of soil C sequestration in Europe, more detailed calculations taking account of variations in soil type and climate are needed (Smith et al. 1997b; Batjes, 1998). This requires a spatially explicit approach. Furthermore, while SOM models have been compared and evaluated at the site scale (Smith et al. 1997b), there have been few previous SOM model comparisons at the regional scale (Falloon et al. 1999b, 2001a,b), or comparisons with regression-based approaches. RothC and CENTURY are two of the most widely used SOM models, and have been well evaluated under many conditions; both of these models have also been used in some regional applications. However, advances in modelling techniques to allow these SOM models to be used in a truly predictive mode are still required (Smith et al. 1998b).

The overall aim of the project is to develop and test tools for estimating changes in soil organic carbon stocks over large spatial areas, under changes in land use and management. A case study area in Central Hungary was chosen for large-scale, predictive SOM modelling.

This will be achieved using the following objectives:

1. Enable RothC to be used in a predictive mode by developing methods to estimate Inert Organic Matter (IOM) and plant inputs of C to soil.
2. Investigate the role of refractory SOM pools in SOM models.
3. Link RothC SOM model to CENTURY to allow model comparison at site and regional scale.
4. Evaluate the RothC and CENTURY models using a suite of datasets (from SOMNET if possible), for the scenarios to be investigated, under Hungarian/European conditions.
5. Link RothC and CENTURY to GIS databases for upscaling.
6. Use GIS linked models to evaluate changes in SOC stocks for the Hungarian test area, for a set of land use/management change scenarios.
7. Compare SOM model-based estimates of regional C sequestration with regression-based approaches.

2. REVIEW OF THE ROTHAMSTED CARBON MODEL AND CENTURY

2.1 INTRODUCTION

The Rothamsted Carbon Model (RothC; Coleman and Jenkinson, 1996) and CENTURY (Parton et al. 1988) are the most widely used and evaluated SOM models world-wide (Parshotam and Hewitt, 1995), and are now being used at scales from site to globe in carbon cycling studies. Both models were originally developed in the 1970's and 1980's, and many other SOM models are similar in structure to RothC and CENTURY. RothC was originally calibrated using data from long-term arable experiments at Rothamsted, UK, and extended to other systems, whilst CENTURY was developed for the grassland systems of the Central Great Plains and then extended to arable ecosystems and forests. The aim of this chapter is to describe the structure of both SOM models in detail, to describe model developments, and review model application and evaluation.

2.2 MODEL STRUCTURE AND THEORY

There are a number of differences in the ways in which RothC and CENTURY simulate SOM turnover processes. Before describing these differences in detail, a major difference is in the *type* of model. RothC is purely concerned with soil processes, and as such is not linked to a plant production model. The CENTURY SOM model, however, is part of a larger ecosystem model that simulates crop, grass and tree growth and the effects of different management practices on both plant production and SOM. Since this thesis is mostly concerned with SOM processes, the focus here is on differences in how the SOM models operate, so CENTURY's above ground production modules are not discussed further. Coleman and Jenkinson (1996) and Metherell et al. (1993) give full descriptions of RothC and CENTURY, respectively. Whilst both models have also been adapted to simulate N and S dynamics, only CENTURY simulates P dynamics. Model structure is discussed in detail in section 2.2.4 (Figs. 2.1 and 2.2) and CENTURY SOM models. The actual SOM model pool names are in *Italics* in these figures and RothC/CENTURY equivalents are shown in normal type for comparison.

Figure 2.1 The structure of the RothC SOM model

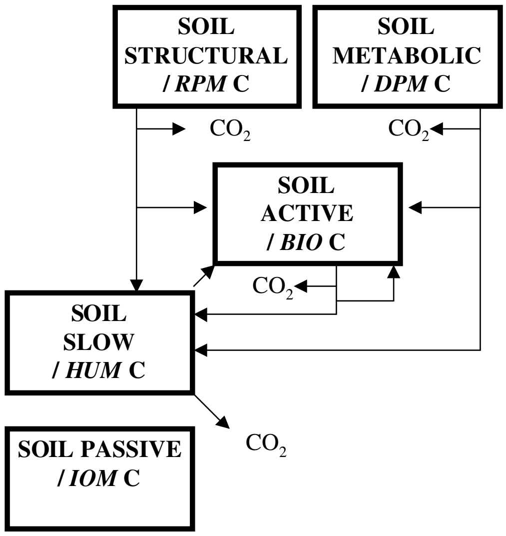
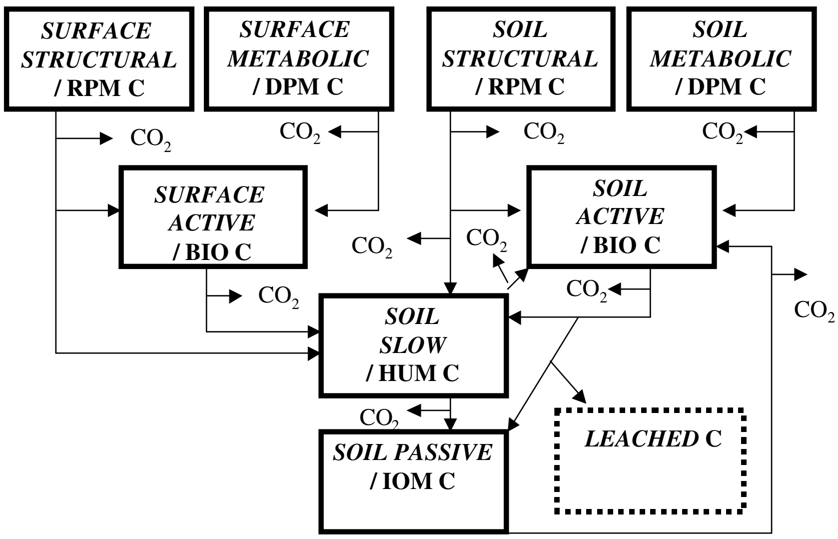


Figure 2.2 The structure of the CENTURY SOM sub-model



2.2.1 Carbon inputs to the SOM models

2.2.1.1 RothC

In RothC, the user defines C inputs to the soil in terms of both quantity and quality. The amount of C entering the soil is defined in "land management" files, and residue/litter quality is defined using the ratio of Decomposable Plant Material (DPM) to Resistant Plant Material (RPM). There are three suggested values for DPM/RPM, based on simulation of SOM dynamics in Rothamsted long-term experiments, although user-defined values may also be used. The default DPM/RPM values are: 1.44 for arable crops and managed grasslands; 0.67 for unmanaged grasslands and scrub; and 0.25 for woodlands and forests. In other words, there is a greater proportion of decomposable C input in arable crops, whereas tree litter is assumed to be much more resistant in nature. As well as plant C inputs, the user may also define any organic amendments to the soil, such as farmyard manure (FYM). The model assumes that FYM is more decomposed than normal plant C inputs, and contains DPM 49%, RPM 49% and HUM 2% (Coleman and Jenkinson, 1996).

2.2.1.2 CENTURY

In CENTURY, the relevant (crop, grass or forest) plant growth model determines plant C inputs. The user defines parameters that set the maximum potential plant production, which is then modified by temperature and water and nutrient availability, thus indirectly setting C returns to the soil. CENTURY uses lignin: N ratios to define litter quality, which is determined by the plant production model. The lignin: N ratio also determines how incoming litter is split between Structural (more resistant) and Metabolic (more labile) compartments. Organic amendments such as FYM are also partitioned between the Structural and Metabolic compartments. With material containing a greater proportion of lignin, more of the C input is partitioned into the Structural pool. The Structural compartment contains all of the lignin matter. The model has both above and below ground input compartments, which represent C inputs from litter and branch fall and root death and senescence, respectively (Metherell et al. 1993).

2.2.2 Water budget models

2.2.2.1 RothC

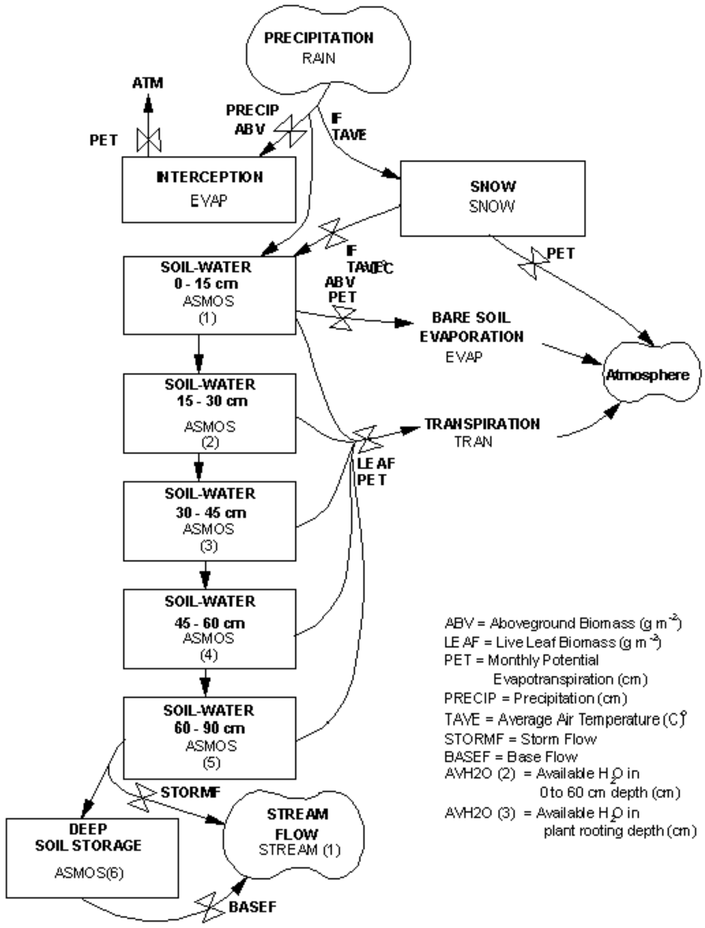
RothC uses a simple water balance model that calculates the soil moisture deficit (SMD). The maximum SMD for the 0-23 cm layer of a particular soil is calculated using a function related to soil clay content. To allow for the reduction in evaporation, this maximum SMD is divided by 1.8 for bare soils. Next, the accumulated SMD is calculated from the first month when 0.75 (open-pan evaporation) exceeds rainfall until it reaches the maximum SMD. Once the maximum SMD is reached, the soil remains at this maximum SMD until the rainfall starts to exceed 0.75 (evaporation) and the soil wets up again. The factor 0.75 is conventional for converting open pan evaporation to evapotranspiration from a growing crop (Coleman and Jenkinson, 1996).

2.2.2.2 CENTURY

The CENTURY model includes a simplified water budget model which calculates monthly evaporation and transpiration water loss, water content of a variable number of 15cm soil layers, snow water content, and saturated flow of water between soil layers (Fig. 2.3). The potential evapotranspiration rate (PET) is calculated as a function of the average monthly maximum and minimum air temperature using equations based on those of Linacre (1977). Bare soil water loss is a function of standing dead and litter biomass (lower for high biomass levels), rainfall and PET. Interception water loss is a function of aboveground biomass (increases with biomass), rainfall and PET. Potential transpiration water loss (PTTR) is a function of the live leaf biomass and PET. Interception and bare soil water losses are calculated as fractions of the monthly precipitation and are subtracted from the total monthly precipitation. Remaining water is then added to the soil.

Water is distributed to the different layers by adding the water to the top layer (0-15 cm) and then draining excess water (water above field capacity) to the next layer. Transpiration water loss occurs after the water is added to the soil. Water loss occurs first as interception, then by bare soil evaporation and transpiration. The maximum monthly evapotranspiration water loss rate is equal to PET. Water leached below the last soil layer is not available for evapotranspiration and is a measure of inter-flow, runoff or leaching losses from the soil profile.

Figure 2.3 The CENTURY water sub-model (after Metherell et al. 1993)



Water going below the profile can be lost as fast stream flow or leached into the subsoil where it can accumulate or move into the stream flow. CENTURY's water model also simulates leaching of the labile mineral N (NO_3 and NH_4), P, and S pools.

2.2.3 Soil temperature models

2.2.3.1 RothC

In RothC, air temperature is used rather than soil temperature. At Rothamsted, UK, monthly air temperature satisfactorily represents the monthly mean soil temperature in the surface 20 cm, showing a difference of only +1°C of the annual minimum and -1°C of the annual maximum. Furthermore, air temperature data is more easily obtainable for most sites. Since the model is designed to simulate SOM turnover in the topsoil, it does not have a separate soil temperature sub-model.

2.2.3.2 CENTURY

In CENTURY, average monthly soil temperature near the soil surface is calculated using equations developed by Parton (1984). This method calculates maximum soil temperature as a function of the maximum air temperature and the canopy biomass (lower for high biomass), and minimum soil temperature is calculated using a function of the minimum air temperature and canopy biomass (higher for larger biomass). The actual soil temperature used for decomposition (and plant growth rate) functions is the average of the monthly minimum and maximum soil temperatures.

2.2.4 Model SOM pools, exchanges and turnover rates

2.2.4.1 RothC

In RothC (Fig. 2.1), SOC is split into four active compartments that decompose by a first-order process and have their own characteristic rate constants. The model also assumes a small amount of SOC is resistant to decomposition (inert organic matter, IOM). The four active compartments (and their maximum decomposition rates) are Decomposable Plant Material (DPM; $k=10.0y^{-1}$), Resistant Plant Material (RPM; $k=0.3y^{-1}$), Microbial Biomass (BIO; $k=0.66y^{-1}$) and Humified Organic Matter (HUM; $k=0.02y^{-1}$). These k values were set by calibrating the model to data from long-term field experiments at Rothamsted (Jenkinson et al. 1987; Jenkinson et al. 1992) and are not normally altered when using the model.

Both DPM and RPM decompose to form CO_2 (lost from the system), BIO and HUM. The proportion that goes to CO_2 and to BIO + HUM is determined by the clay content of the soil. The BIO + HUM is then split into 46% BIO and 54% HUM. BIO and HUM both decompose to form more CO_2, BIO and HUM. If an active compartment contains Y t C ha^{-1}, this declines to Y e^{-abckt} t C ha^{-1} at the end of the month. In this equation, a is the rate modifying factor for temperature, b is the rate modifying factor for moisture, c is the plant retainment rate modifying factor, k is the decomposition rate constant for that compartment and t is 1 / 12, since k is based on a yearly decomposition rate. Therefore Y (1 - e^{-abckt}) is the amount of the material in any compartment that decomposes in a particular month (Coleman and Jenkinson, 1996).

The rate modifying factor for soil moisture, (b), used each month is calculated as follows. If the accumulated SMD is less than 0.444 of the maximum SMD, then b is set to 1.0, i.e. no effect of moisture. Otherwise, b is calculated as in the following equation (and shown in Fig. 2.4):

$$b = 0.2 + (1.0 - 0.2) * \frac{(\max.SMD - acc.SMD)}{(\max.SMD - 0.444\max.SMD)}$$

The model uses average monthly air temperature (tm) to calculate the decomposition rate modifier for temperature (a) according to the following equation (Fig. 2.5):

$$a = \frac{47.9}{1 + e^{\left(\frac{106}{tm + 18.3}\right)}}$$

The plant retainment factor (c) slows decomposition if growing plants are present. If soil is vegetated, c=0.6 and if soil is bare c=1.0. Finally, RothC allows for differences in soil texture by changing the proportion of CO_2 evolved and (BIO+HUM) formed during decomposition, rather than by using a rate modifying factor, such as that used for temperature. The ratio CO_2 / (BIO + HUM), x, is calculated from the clay content of the soil using the equation below: -

$$x = 1.67(1.85 + 1.60\exp(-0.0786\ \%clay))$$

Then x / (x + 1) is evolved as CO_2 and 1 / (x + 1) is formed as BIO + HUM. The scaling factor 1.67 is used to set the CO_2 / (BIO+HUM) ratio in Rothamsted soils (23.4% clay) to 3.51, based on experimental data: the same scaling factor is used for all soils. Fig. 2.6 shows how the clay content of the soil affects the soil texture factor (Coleman and Jenkinson, 1996).

2.2.4.2 CENTURY

The CENTURY SOM sub-model (Fig. 2.2) includes three soil organic matter pools and four litter pools (two surface and two sub-surface), and a surface microbial pool, that decompose by first order kinetics and have characteristic decomposition rate constants. The soil organic matter pools (and their maximum decomposition rates) are active (k=7.3y^{-1}), slow (k=0.2y^{-1}) and passive (k=0.0066y^{-1}). The litter pools are surface structural (k=3.9y^{-1}), surface metabolic (k=14.8y^{-1}), soil structural (k=4.9y^{-1}) and soil metabolic (k=18.5y^{-1}), and the surface microbial pool is surface active (k=6y^{-1}; Metherell et al. 1993). In a similar way to RothC, the decomposition of both plant residues and SOM are driven by the microbial pools and decomposition results in CO_2 loss via respiration.

The loss of CO_2 on decomposition of the active pool increases with increasing soil sand content. The effect of sand content on the decomposition rate of active SOM (EFTEXT; Fig. 2.7) is calculated according to Metherell et al. (1993):

$$EFTEXT = 0.25 + 0.75 \quad (\%SAND)$$

Decomposition products flow into a surface active pool or one of three SOM pools. The maximum potential decomposition rate is reduced to an actual decomposition rate by functions of soil moisture and soil temperature.

Figure 2.4 The rate modifying factor for moisture in RothC

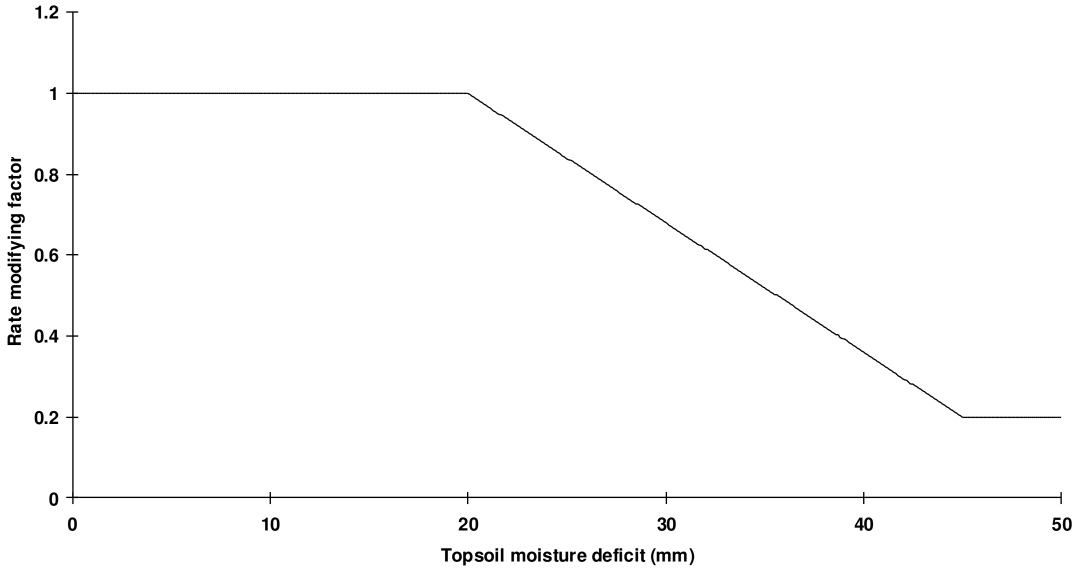


Figure 2.5 The rate modifying factor for temperature in RothC

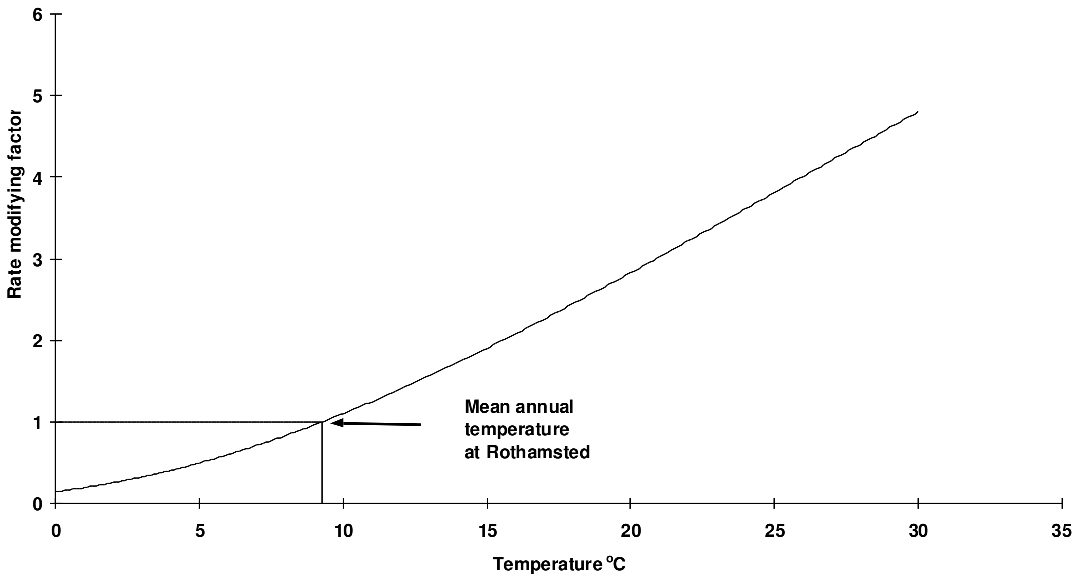


Figure 2.6 How clay % affects the ratio CO_2 / (BIO + HUM) in RothC

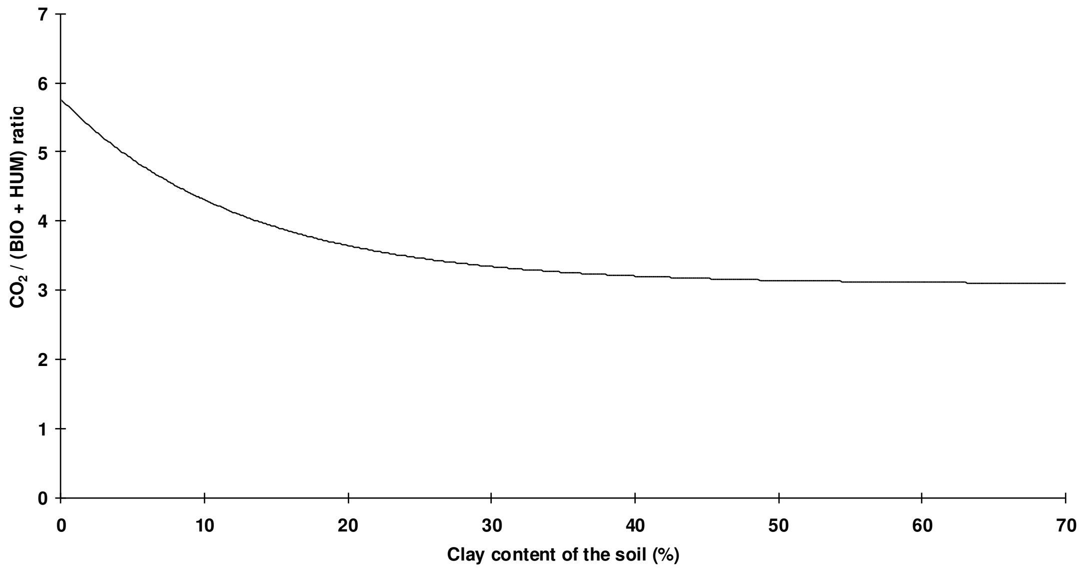


Figure 2.7 The active SOM decomposition rate constant modifier for texture in CENTURY

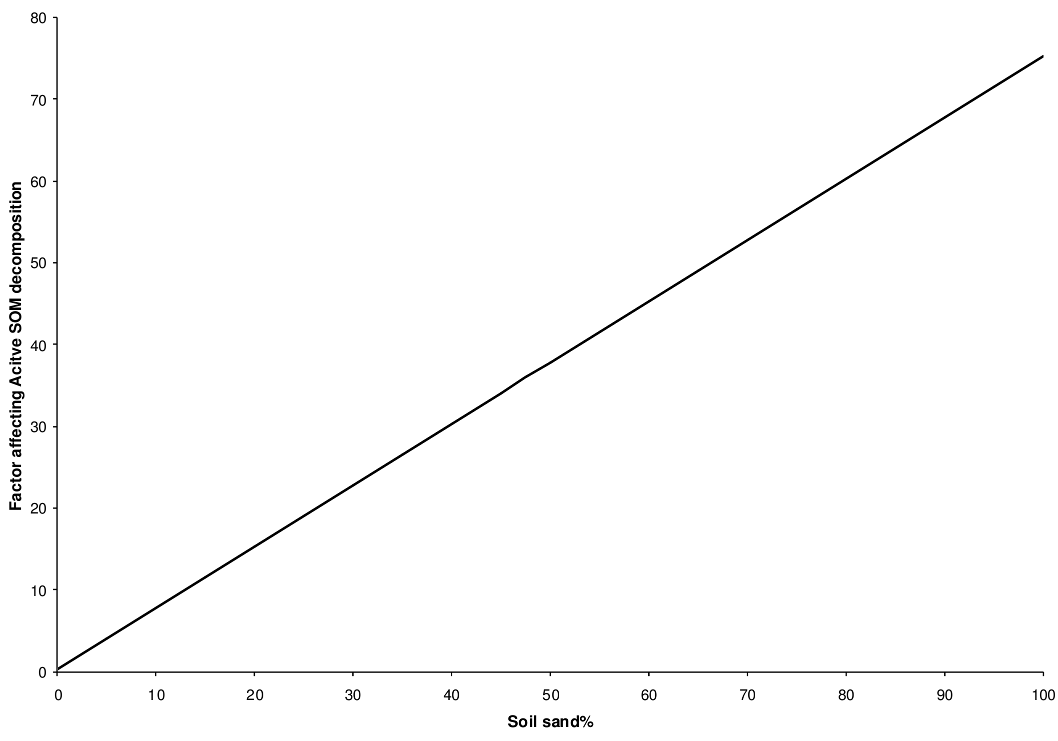


Figure 2.8 The decomposition rate constant modifier for temperature in CENTURY

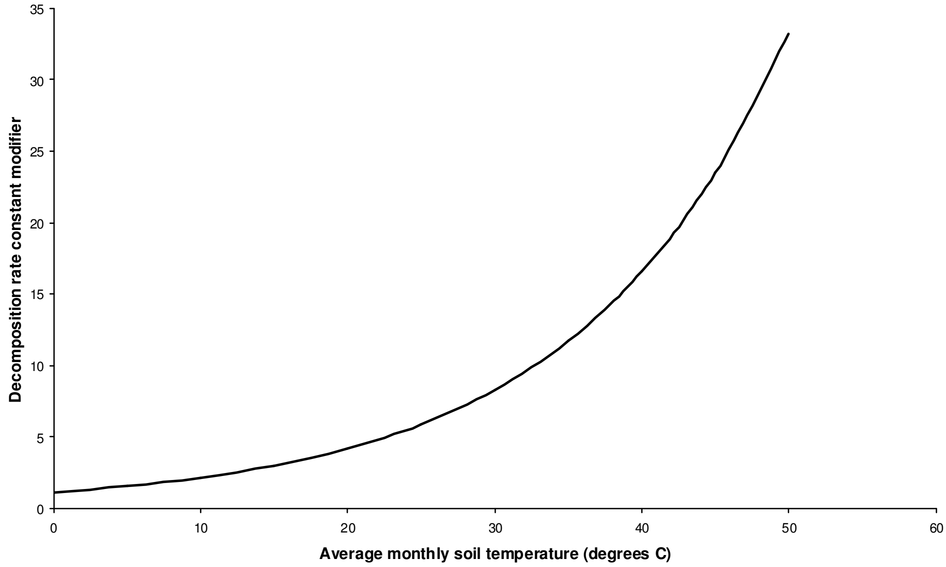


Figure 2.9 The decomposition rate constant modifier for moisture in CENTURY

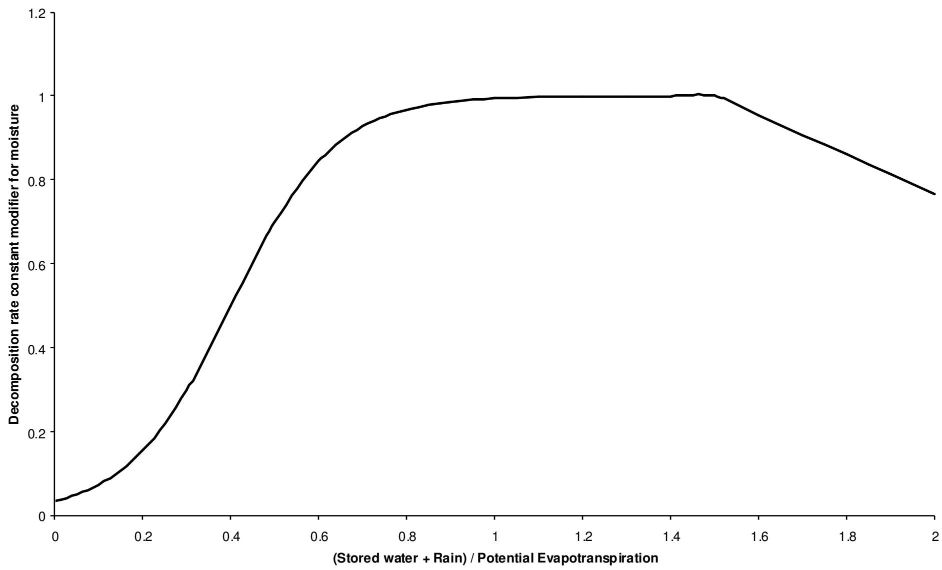
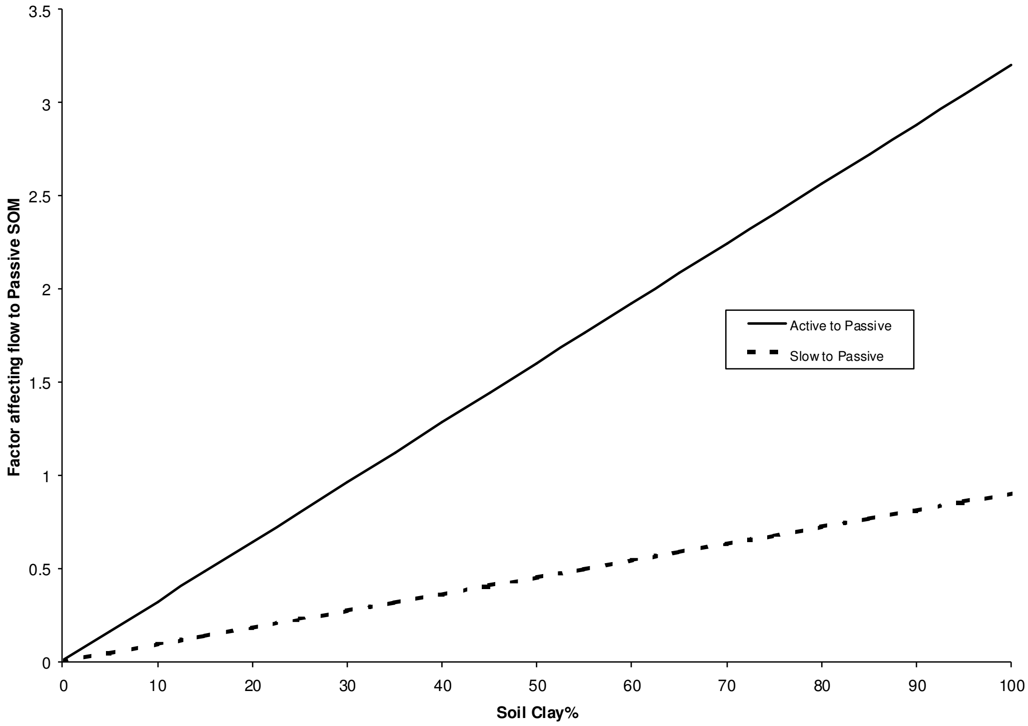


Figure 2.10 The texture factors affecting the flow of Active and Slow SOM to Passive SOM in CENTURY



Decomposition may also be increased due to cultivation. The impact of temperature on the actual decomposition rate is driven by average monthly soil temperature. The decomposition rate modifier for temperature (TFUNC; Fig. 2.8) is calculated from the average monthly soil temperature (STEMP) as follows (Metherell et al. 1993):

$$TFUNC = 0.125 + e^{(0.07\,STEMP)}$$

The ratio of (stored topsoil water + current month's precipitation) to potential evapotranspiration determines the impact of moisture on decomposition. The decomposition rate modifier for moisture (WFUNC; Fig. 2.9) is calculated from the ratio of (stored water + precipitation) to potential evapotranspiration (RAT), from the equations below (Metherell et al. 1993):

$$WFUNC = \frac{1.0}{1.0 + 30 * e(-8.5 * RAT)}$$

if RAT > 1.5

$$WFUNC = 1.0 - 0.7 * \frac{(RAT - 1.5)}{1.5}$$

Lignin material is assumed to go directly to the slow C pool during the decomposition of structural plant material. Soil clay and silt content also affect the efficiency of stabilising active SOM into slow SOM and passive SOM. This has the effect of creating more stable SOC in finer textured soils. The efficiency of stabilising active SOM into passive SOM (FPS1S3; Fig. 2.10) is calculated (Metherell et al. 1993) from:

$$FPS1S3 = 0.003 + 0.032(CLAY\%)$$

The efficiency of stabilising slow SOM into passive SOM (FPS2S3; Fig. 2.10) is calculated (Metherell et al. 1993) from:

$$FPS2S3 = 0.003 + 0.009(CLAY\%)$$

CENTURY also simulates leaching of organic matter. Finally, the model uses a soil drainage factor that allows a soil to have different degrees of wetness. Anaerobic conditions have the effect of reducing the decomposition rate (Metherell et al. 1993).

2.2.4 Other considerations

CENTURY is able to simulate the fate of ^{14}C and ^{13}C tracers in the soil-plant system, whilst RothC can model the ^{14}C age of the topsoil. CENTURY simulates N, P and S dynamics, and recent developments to a daily version of RothC have enabled N and S dynamics to be modelled. CENTURY is also capable of modelling DOC loss from soils, whilst RothC currently cannot. Both models are applicable to arable, grassland and forest systems. However, neither model has been developed to accurately simulate the development of pronounced litter layers, so applicability to forested systems may be limited to the mineral soil only. Both RothC and CENTURY can simulate the effects of FYM application on SOM. The CENTURY model is also able to simulate the impact of several management practices and other factors on SOM, such as tillage, harvesting, different organic amendments, irrigation, grazing, erosion, fire and tree removal.

2.3 DEVELOPMENTS AND IMPROVEMENTS MADE TO THE MODELS

Both RothC and CENTURY have undergone considerable changes and development since their inception. This section aims to document some of the more significant changes to the models.

2.3.1 RothC

The original version of RothC ran on an annual timestep, and was a steady-state SOM model (Jenkinson and Rayner, 1977), specifically for Rothamsted soils. This version of the model did not have an IOM pool, but had two pools of 'protected' organic matter, physically protected OM (POM) and chemically protected OM (COM). The later version of RothC described by Jenkinson et al. (1987) was modified for application to soils and climates other than Rothamsted. This model had four SOM pools, BIOZ, BIOA, HUM and IOM, and ran on a monthly timestep. The current version of RothC (Coleman and Jenkinson, 1996) has only three SOM pools (BIO, HUM and IOM) and runs on a monthly timestep. One of the major changes made to this version of the model is in the effect of soil clay % on SOM decomposition. The current model uses a different equation to relate the CO_2 / (BIO+HUM) ratio to %clay than that given by Jenkinson et al. (1987) or Jenkinson (1990).

2.3.2 CENTURY

The current CENTURY model (Metherell et al. 1993) has undergone many improvements since the original model of Parton et al. (1987). The current model now allows the user to specify many management options not previously available.

2.4 DATA REQUIREMENTS AND OUTPUT VARIABLES

2.4.1 RothC

RothC uses two input files, a land management file (with C inputs, FYM and plant cover), a weather file (with rainfall, temperature, evaporation, clay content and sample depth) and input from the command line, and can be run in DOS batch mode. There is also a windows 3.1/windows 95 user interface. The only data required to run the model are:

1) Monthly total rainfall (mm)
2) Monthly total open pan evaporation (mm).
3) Average monthly air temperature (^oC).
4) Clay content of the soil %.
5) An estimate of the decomposability of the incoming plant material - the DPM/RPM ratio.
6) Soil cover (whether the soil is bare or vegetated in a particular month)
7) Monthly input of plant residues (t C ha^{-1}).
8) Monthly input of farmyard manures (FYM) (t C ha^{-1}), if any.
9) Depth of soil sample (cm)

RothC produces output data on a monthly timestep including total C, C in DPM, RPM, BIO and HUM, CO_2 evolved, IOM and radiocarbon age. Predictive use of the model requires estimation of a) the IOM pool for the model, and b) C inputs to soil.

2.4.2 CENTURY

The user may specify many input variables and parameters in CENTURY. Major input variables for the model include:

(1) Monthly average maximum and minimum air temperature (^oC).,
(2) Monthly total precipitation (cm),
(3) Lignin content of plant material (%),
(4) Plant N, P, and S content (%),
(5) Soil texture (%sand, %silt, and % clay),

(6) Atmospheric and soil N inputs (g m^{-2}), and

(7) Initial soil C, N, P, and S levels (g m^{-2}).

The CENTURY model takes input from monthly weather data files, files containing fixed parameters primarily relating to organic matter decomposition (not normally adjusted between runs), and files containing site-specific parameters such as precipitation, soil texture, and the initial conditions for soil organic matter. The user can also specify crop, cultivation, fertilisation, fire, grazing, harvest, irrigation, organic matter addition, tree, and tree removal options. Radiocarbon data may also be input. Runs of the model are executed using 'schedule' files that specify the order and timing of events such as harvesting and fertilisation. The model runs using a monthly or daily (DAYCENT; Parton et al. 1998) time step. The model has many output variables, of interest here are labelled, unlabelled and total C in SOM pools and total C, C inputs and CO_2 loss. Predictive use of CENTURY requires setting of the distribution of SOC between model pools, especially the 'passive SOM' pool, which changes very little throughout the course of normal simulations.

2.5 MODEL EVALUATION AND LIMITATIONS

2.5.1 Model evaluation

2.5.1.1 RothC

RothC has been used to better understand SOM turnover processes (e.g. Jenkinson and Coleman, 1994; Smith et al. 1997c, 1999; Falloon et al. 1998a), aid policy decisions regarding carbon sequestration options, and investigate the effects of climate change and land use on SOM (e.g. Jenkinson et al. 1991; Parshotam et al. 1995; King et al. 1997; Falloon et al. 1998b, 1999b, 2000, 2001a,b). At the site scale, RothC has been extensively evaluated using twelve arable, grassland and forest datasets world-wide (Coleman et al. 1997; Smith et al. 1997a), but has been most widely evaluated for UK arable, grassland and forest sites (Jenkinson et al. 1992). Jenkinson et al. (1999a,b) also evaluated impacts of errors in input parameters on RothC predictions. RothC has been used in conjunction with radiocarbon data to estimate C inputs and NPP in the UK (Jenkinson et al. 1992; Jenkinson and Coleman, 1994) and in Kenyan savannahs and Zambian woodlands (Jenkinson et al.

1992, 1999b). Other RothC applications include determination of SOC turnover times under New Zealand forests and grasslands (Tate et al. 1995), estimating SOC recovery times in degraded New Zealand soils (Parshotam and Hewitt, 1994), simulation of SOC changes in ley-arable rotations of Thailand (Wu et al. 1998), estimation of new SOM production under maize in France (Balesdent, 1996), simulation of ^{14}C-labelled decomposition studies (Ayanaba and Jenkinson, 1990) and of SOC accumulation during afforestation in Spain (Romanya et al. 2000).

At the regional level, RothC has been used in New Zealand to estimate climate change effects on SOM (Tate et al. 1996) and to determine whether natural grasslands and forests are sinks or sources of C (Parshotam et al. 1995). King et al. (1997) used RothC with a simple NPP model at the global scale to determine the response of C storage to changes in climate and atmospheric CO_2. At the national scale, Parshotam et al. (1995) used RothC in New Zealand to determine the possible effects of climate change and CO_2 on SOM. Falloon et al. (1998a, 2000, 2001a,b) have used RothC for studies of regional scale SOM dynamics in Hungary. Jenkinson et al. (1991) used RothC to determine climate-warming effects on global SOC stocks.

2.5.1.2 CENTURY

CENTURY has been extensively evaluated using the same twelve arable, grassland and forest datasets world-wide (Kelly et al. 1997; Smith et al. 1997a) as RothC. An extensive grassland NPP and SOM modelling exercise using CENTURY covered temperate and tropical sites in Kenya, Thailand, Mexico, Ivory Coast, Colorado, Kansas, Ukraine, Russia, and Kazakhstan (Parton et al. 1993). Other grassland sites modelled using CENTURY include the US (Kelly et al. 1996), Tanzania (Parton et al. 1989), Australia and Cote D'Ivoire (Woomer, 1993). Thirteen forest sites in Colombia, Costa Rica, Brazil and Peru have been modelled using CENTURY, with simulation of incubation experiments and experimental pool size estimation (Motavalli et al. 1994). CENTURY has also been evaluated for Canadian (Peng and Apps, 1998; Peng et al. 1998), Hawaiian (Parton et al. 1989a), Australian, Peruvian, Puerto Rican, Sri Lankan, Venezuelan and Zimbabwean forests (Woomer, 1993). CENTURY has been applied to residue and fertilization treatments of an arable experiment in Thailand (Myers, 1994), to fertilization and tillage treatments in North

America (Patwardhan et al. 1995) and Canada (Liang et al. 1996), and to arable ecosystems in Peru (Parton et al. 1989a; Woomer, 1993), Australia and Venezuela and Zimbabwe (Woomer, 1993). CENTURY has been used at the field scale to simulate decomposition of labelled plant residues in tropical forest soils (Motavalli et al. 1995). Parfitt et al. (1997) used CENTURY to determine land use effects on SOC under different New Zealand soils. Paustian et al. (1997c) used CENTURY at six US arable sites to analyse the impact of management intensity, climate change and elevated CO_2 at the site level. In the UK, Falloon and Smith (2000) used CENTURY to simulate SOC dynamics in the Broadbalk arable experiment.

Various US regional scale applications of the CENTURY model have considered soil differences only (e.g. Parton et al. 1987, 1988; Schimel et al. 1989; Cole et al. 1988), although Burke et al. (1990) used soil and climate databases to simulate SOM levels in Colorado grasslands. Parton et al. (1987) determined SOM controlling factors, and Parton et al. (1989b) simulated soil C, N and S patterns in Central US grassland. CENTURY has also been used in extensive US investigations of soil carbon sequestration options (Donigan et al. 1994). Cole et al. (1988) used CENTURY to estimate historic SOC changes in the Central US Great Plains. Falloon et al. (1999b, 2001a,b) have used CENTURY to simulate regional scale SOC dynamics in Hungary.

2.5.2 Model weaknesses

2.5.2.1 General limitations

Parshotam and Hewitt (1995) noted that the monthly timestep used by models such as RothC may be too long for simulating short-term C turnover processes. At the other end of the scale, Hyvonen et al. (1998) suggest that even after 10,000y, many soils are still far from steady state, whilst models such as RothC reach equilibrium after a period of 5-10,000y. One of the major limitations in using the CENTURY model is accurately describing land use history in initialisation (Parton et al. 1987).

2.5.2.2 Environmental limitations

Environmental limitations of models are taken here to mean primarily climatic and soil limitations on model application. In terms of soils, it is generally suggested that models such as CENTURY (Parton *et al*. 1989a; Motavalli et al. 1995) and RothC (Jenkinson et al. 1992) may be limited by failing to account for pH effects on soil C turnover (Jenkinson, 1988). This may be important in soils with pronounced litter layer development, or grasslands with constant mineral fertiliser applications (Kelly et al. 1997). SOM models generally predict faster C turnover than observed in very acid soils (Motavalli et al. 1995), since decomposition is up to two-thirds slower (Kelly et al. 1997; especially in the early stages) under acid conditions, although overall model errors may be small (Jenkinson et al. 1999b). Model improvement needs information on the both the short and long-term effects of pH on SOM, in particular the effects on the microbial biomass (Jenkinson et al. 1992).

Few models are able to predict SOC changes in variable charge and allophanic soils (Oades et al. 1989, Jenkinson et al. 1991; Motavalli et al. 1994; Smith et al. 1997c; Falloon and Smith, 1998, 2000; Falloon et al. 1998a, 2000). Some authors have used 'effective' clay contents, based on %clay, %ferrihydrite and %allophane to get model agreement with measured data (Parshotam et al. 1995; Tate et al. 1996); others have suggested using surface area measurements (Saggar et al. 1994), or accounting for mineralogy (Parton et al. 1989). There may be other problems related to soil texture with SOM models, for example, CENTURY tends to overestimate soil C and N levels for fine textured soils by 10-15% (Parton et al. 1987). Most models are also unable to simulate SOC changes in highly organic and recent volcanic soils (Falloon et al. 1998a; Jenkinson et al. 1991; Smith et al. 1997c).

RothC should not used on sites where erosion is active, and cannot be applied to soils containing large amounts of recent inert organic matter, such as charcoal (Jenkinson et al. 1999b). In models such as RothC and CENTURY, the effect of temperature on decomposition is modelled by an increase in rate constant for the different SOM pools (eg. Coleman and Jenkinson, 1996; Parton et al. 1987). There is, however, evidence to suggest that there is little change in the rate constant, but a change in the utilisation of substrate pools and a shift in function and composition of microbial communities (Zogg et al. 1997). Decomposition-moisture relationships are often described by a log-linear function of soil water potential, although quadratic or other functions may be more effective (Lomander et al. 1998). CENTURY uses a curve for moisture effects on decomposition, whilst RothC uses a split line (Parton et al. 1987).

Some authors have suggested changes in SOM pool decomposition rates. Parton et al. (1994) noted that CENTURY underestimated Active SOM levels in labelled plant decomposition experiments, and the decomposition rate could need to be reduced. Errors in CENTURY simulation of boreal forest C dynamics may be limited by current representation of the dynamics of the slow SOM pool (Peng et al. 1998).

Several authors have suggested that current SOM models may be limited in their applicability to tropical systems, and few models have been tested under arid conditions. Possible reasons could include differences in soil fauna, the much faster turnover time of slow and passive SOM pools in the tropics, different temperature and moisture relationships with metabolic rates, and differences in mineralogy (Woomer, 1993; Shang and Tiessen, 1998) and solution chemistry (Parton et al. 1989) in tropical soils. Inadequate description of nutrients (N, P and K) and inability to account for Al toxicity may also limit SOM model predictions in tropical soils (Parton et al. 1989; Shang and Tiessen, 1998).

Litter quality may also present problems for SOM modelling. In an evaluation of CENTURY using litter decomposition studies, Parton et al. (1994) showed that much lower decomposition rates were required for their slow SOM pool for initially high N litter compounds. The decomposition rate of slow SOM may need to be a made a function of either a) initial N content of decomposing litter or b) the fraction of structural plant material that is lignin (Parton et al. 1994).

2.5.2.3 Crop/vegetation system limitations

While SOM models such as CENTURY may generally predict SOC and NPP well, the monthly timestep often used may limit simulation of detailed differences in plant production. Application of CENTURY to 11 temperate and tropical grasslands showed underestimation of live plant biomass in high production years (Parton et al. 1993). This may be due inability to predict shifts in plant species composition, inter-annual variations in solar radiation, the effects of rainfall on a sub-monthly timestep, or the lag effects of nutrient or photosynthate storage in plants. DAYCENT (Parton et al. 1998), the daily timestep version of CENTURY, may solve these problems. CENTURY may also have difficulties in simulating several crops in the same year, mixed cropping or intercropping systems (Woomer, 1993), which could limit applicability to tropical systems. Comparison of CENTURY simulations in US shortgrass prairie have suggested that CENTURY's passive pool may turn over faster than originally thought (Kelly et al. 1996). Application of RothC to different vegetative systems may be limited since different systems may have different litter qualities (Jenkinson et al. 1991) – only three DPM/RPM ratios are specified in RothC, although the user can define other values.

2.5.2.4 Management limitations

Some management treatments pose particular problems for SOM model simulations. For instance, few SOM models are specifically set up to simulate sewage sludge application to land, biofuel crops, or tillage (although the model of Lee et al. 1993 does account for tillage explicitly; Smith et al. 1998a). Indeed, Patwardhan et al. (1995) showed that CENTURY's description of tillage effects on SOC could be improved, particularly with respect to layering and subsoil processes. Myers' (1994) application of CENTURY to an arable experiment in Thailand showed consistent underestimation of SOM accumulation, especially in highly fertilized and organic treatments. An application of RothC to the UK Park Grass Experiment gave a poor simulation of the NPKMg + straw/FYM/fishmeal treatment, especially during early years of the experiment (Coleman et al. 1997). This could, however, have been due to sampling error, or the presence of an organic mat, which would have increased soil pH and earthworm activity. Peng et al. (1998) noted that CENTURY's description of fire disturbance effects on soil C (especially slow pool C) dynamics were inadequate for boreal forests. CENTURY may be further limited in forest simulations since it cannot accurately simulate SOM accumulation in a forest with a well-developed litter layer (Kelly et al. 1998). Highly decomposed litter that actually remains on the surface is automatically transferred to the slow SOM pool, leading to an unrealistically high build up of SOM.

2.5.2.5 Model pools and measurements

Many SOM models require the user to specify SOC distribution between the different model pools in initialisation, although this may be done indirectly, using 'equilibrium' model runs. This, coupled with the fact that most multi-compartmental models contain at least one pool without a measurable analogue, has led many attempts to either measure model pools or vice-versa. For example, in RothC, Total C, and biomass C are measurable by standard analytical techniques, IOM may be estimated from radiocarbon dating, and only HUM need be estimated by difference. It is unclear whether it will be better to 'measure the modellable' or 'model the measurable' (Elliot et al. 1996; Smith et al. 1998b), and no single satisfactory method exists to separate soil C into components with different turnover times (Trumbore, 1990). Of course, SOM model pools do not necessarily have 'real world' analogues.

However, a better understanding of SOM turnover processes does need better estimates of SOM fractions and their dynamics (Paul et al. 1995). Understanding the size and age of the different conceptual pools of SOM is also of great importance in understanding the global carbon cycle, for example when estimating how much C is gained or lost annually from the atmosphere as a result of large scale land-use change. Issues related to model pools and measured fractions with special reference to refractory SOM are discussed in section 4.1.2. Some approaches appear to show more promise than others. For example, Sparling et al. (1998) developed a relationship between hot water soluble C and microbial biomass C on a range of New Zealand soils. This relationship could be developed as a simple test for the BIO pool in RothC or the active SOM pool in CENTURY, indeed Motavalli et al. (1994) have already used hot-water soluble C to estimate active SOM. Attempts to relate physical, biological and chemical fractionations to SOM model pools (with special reference to refractory SOM) are discussed in greater detail in Section 3.1.2.3.

2.6 DISCUSSION

This chapter has reviewed the structure and use of RothC and CENTURY. As noted in the introduction, the main difference between the two models is the fact that RothC is purely a soil process model, whereas CENTURY is a more general ecosystem model. There are also differences in the ways in which SOM is separated into different model compartments, their turnover rates, and how external factors like litter quality, temperature, moisture and soil texture affect the decomposition and turnover of soil C.

Neither model is set up to simulate pronounced litter layers in forests, allophanic soils, organic soils, waterlogged soils, soil C turnover in different soil layers or at great depth, or the effects of pH on soil C turnover. Future research should be directed to expand the models' capabilities into these areas. Both models have been extensively evaluated using long-term data and have been shown to perform generally well. However, for regional applications, it is important to verify model performance using long-term site data in the environmental conditions of the region to be simulated, and also under management regimes reflecting those of interest.

In terms of estimating C sequestration at the regional scale, it is important that the models can simulate a range of management scenarios considered practical for C sequestration. RothC can simulate the effects of FYM applications, cereal straw incorporation, land use conversion (afforestation and so on), and ley-arable rotations on SOC, but not specifically the effects of tillage or sewage sludge incorporation. CENTURY can simulate the effects of FYM, cereal straw incorporation, afforestation, ley-arable rotations, and tillage on SOC, but not specifically the effects of sewage sludge incorporation.

It would therefore be desirable to develop and test both models with data from long-term experiments with sewage sludge incorporation, and to develop RothC for tillage effects and inorganic fertilisation effects.

Finally, whilst the models have been compared at the site scale, no comparison of RothC and CENTURY has been conducted at the regional scale. Neither have former site-scale comparisons of the two models used the same C input values. Use of RothC in a predictive mode requires methods to set IOM and C inputs to soil, whilst predictive use of CENTURY requires knowledge of initial SOC pool distributions (especially the 'passive SOM' pool). It would therefore also be desirable to compare the two models with the same input data and develop methods to use them in a predictive mode.

3. THE ROLE OF REFRACTORY SOIL ORGANIC MATTER POOLS IN MODELS

3.1 INTRODUCTION

This chapter describes methods for estimating the most refractory pools of soil organic matter represented in the widely used SOM turnover models, RothC and CENTURY. These two models represent two distinct types of multi-compartmental SOM model – RothC having a refractory SOM (RSOM) pool uncoupled from the model SOM system, and CENTURY describing RSOM as being coupled other SOM pools. As discussed in Chapters 1 and 2, the RSOM pool in RothC is "Inert Organic Matter" (IOM), and in CENTURY it is "Passive SOM". The RSOM content of soil varies markedly between sites (Tate et al. 1995; Falloon and Smith 1998, 2000; Falloon et al. 1998a, 2000), and knowledge of its size, behaviour and variability are essential for determining whether soils behave as sources or sinks of atmospheric CO_2 (Emanuel et al. 1984; Schlesinger, 1995; McGuire et al. 1997).

It has also been suggested that the accurate specification of RSOM pools is essential to modelling studies (Skjemstad et al. 1996) and that uncertainty in IOM estimates could be a major source of error in modelling SOC (Jenkinson et al. 1991; Parshotam et al. 1995). In this chapter the use of RSOM in models is reviewed, and a method for estimating the size of the IOM pool for RothC in the absence of radiocarbon data is presented. Estimating IOM for RothC is a significant step towards using the model in a predictive mode, and contributes to estimating C inputs to soil (Chapter 4), model evaluation (Chapter 5) and regional-scale modelling (Chapter 6). The influence of climate, soil and management parameters on the RSOM parameter in the two models is also explored, to investigate how model assumptions relate to experimental determinations of RSOM. The importance of RSOM pools in predictive SOM modelling using RothC and CENTURY is investigated using a simple sensitivity analysis at the field scale. A simple land-use change scenario for European soils is used to demonstrate the significance of RSOM to global change studies.

3.1.1 Refractory soil organic matter

3.1.1.1 Definitions

It is first important to distinguish between refractory SOM (RSOM) and inert SOM (IOM), as determined experimentally and in models. True IOM (hereafter referred to as inert soil organic matter; ISOM) is not biologically decomposable, and therefore has no decomposition rate. In models such as RothC, IOM is a pool uncoupled to the other SOM pools, effectively a constant. RSOM may be very stable SOM, with a slow but finite turnover rate, and in models this pool is coupled to the rest of the SOM system. RSOM and IOM differ in their significance when using models in predictive studies. IOM remains unaffected by changes in climate, land-use or management, and whilst its size is of importance, it cannot, by definition, act as a sink for atmospheric CO_2 (e.g. Emanuel et al. 1984). RSOM interacts with the rest of SOM and the ecosystem, and may be a very long-term CO_2 sink.

Definitions of RSOM from the literature are quite different: a) "the inert carbon portion is the minimum carbon content under field conditions if no fertiliser is given and humus-consuming crops or no plants at all (i.e. black fallow) are grown" (Korschens, 1997b). Passive SOM in CENTURY is "a fraction that is chemically recalcitrant and may also be physically protected, with the longest turnover time (200-1500 y; Parton et al. 1987) or c) IOM in RothC is a SOM compartment "totally resistant to decomposition, with an arbitrary age of 50,000 y" (Coleman and Jenkinson, 1996).

3.1.1.2 What is RSOM?

It is difficult to define a RSOM fraction physically. There is much evidence from modelling and from the radiocarbon dating of various chemically isolated fractions that soils contain small amounts of recalcitrant materials of great age (Table 3.1 and references therein; Falloon et al. 1998a, 2000). Whether we consider these to be truly inert organic matter (ISOM) or refractory organic matter (RSOM) will affect the nature and proportion of SOM it accounts for. A wide range of compounds (e.g. charcoal, non-degradable N free plant components, acid-hydrolysis residues, humin, humic acids, and interlayer organic complexes: Table 3.1 and references therein, Gifford et al. 1996) have been identified as passive components of SOM, but model passive or stable pools are not defined with respect to chemical composition.

Residues after acid hydrolysis are often older than total soil organic carbon (SOC) (Jenkinson, 1970; Martel and Paul, 1970; Jenkinson and Rayner, 1977; Trumbore et al. 1989), with radiocarbon ages of several thousand years. ^{13}C-Nuclear Magnetic Resonance (NMR) spectra of soils indicate that alkyl and methoxyl groups may be more resistant to decomposition than other components of SOM (Kinchesh et al. 1995; Guggenberger et al. 1995). Truly inert fractions of SOM are almost certainly a mixture of charcoal, ancient coal, and trapped organic materials (Frink, 1992; Skjemstad et al. 1996; Falloon and Smith 1998,2000; Falloon et al. 1998a, 2000). Modelling has also suggested the presence of highly resistant, undecomposed plant matter in soils (Bossatta and Agren, 1991), confirmed by experimental evidence (Augris et al. 1998).

3.1.1.3 How large is the RSOM pool and how slowly does it turnover?

Estimates of the age and size of the RSOM pool are variable (Table 3.1), with ages from several hundred years to several thousand years, constituting from less than 15% of SOM to over 94% of SOM (Falloon et al. 1998a, 2000 – Table updated to include new results). Humic acids, residues from 6 molar acid hydrolysis, and OM associated with coarse clay are most stable (decay rate constant, k=0.005-0.009 y^{-1}); acid hydrolysable C, OM associated with the fine clay fraction, and DOC from streams, lakes, and wetlands decays more rapidly ($k \leq 0.001$ y^{-1}). Residues from 0.5 molar acid hydrolysis, coarser clay-associated OM and other fractions have varying rate constants (Molina and Smith, 1998).

3.1.1.4 How is RSOM formed?

The processes of RSOM formation and protection are not clearly understood (Skjemstad et al. 1996). Whitbread (1994) and Körschens (1997) suggest that the distribution of SOM between the labile and stable fractions is influenced by management factors; Tsutsuki et al. (1988), Metherell et al. (1993), Leavitt et al. (1996) and Körschens (1997) stress the importance of soil texture, with stable SOM being linked to fine fractions. Extreme protection of SOM also has been attributed to its chemical structure, to reaction with other soil constituents (e.g. formation of Al-humus; Tate, 1992, Tate et al. 1995), intermittent waterlogging (Tate et al. 1995) and to physical inaccessibility (Balesdent, 1996). Skjemstad et al. (1996) suggest two mechanisms for protection. The first mechanism could be physical incorporation into micro-aggregates that are able to withstand physically disruptive forces, a process that operates in all soils but is a major factor in strongly structured soils. The chemistry of this protected OM is similar to that of SOM in general. A second mechanism could be the formation of charcoal or 'charred' OM that is highly resistant to microbial and chemical decomposition. True IOM (ISOM) could be derived from the inorganic fabric that originally made up the soil (with the exception of soils derived from recent volcanic ash falls), or be created during the early stages of pedogenesis, perhaps by irreversible sorption of organic matter on sites that were hence effectively blocked for ever (Arslanov et al. 1970; Chicagova et al. 1995; Theng et al. 1991; Falloon and Smith 1998, 2000; Falloon et al. 1998a, 2000). In the latter sense, ISOM is conceptually important as the age of ISOM ultimately determines the time elapsed since humus development, i.e. the true age of the soil (Theng et al. 1991; O'Brien and Stout, 1978).

3.1.1.5 Which factors affect the size and properties of the RSOM pool?

The size and nature of the RSOM pool in soil may be related to soil and ecosystem properties. An increase in radiocarbon age or turnover time with depth in the soil has been noted by several authors (Arslanov et al. 1970; O'Brien, 1986; Becker-Heidmann et al. 1988; Scharpenseel et al. 1989; Skjemstad et al. 1990; Becker-Heidmann and Scharpenseel, 1992b). This could correlate with either a larger proportion of inert C in the lower part of the soil profile, or slower decomposition for other reasons. Indeed O'Brien and Stout (1978) observed a very old fraction of soil C that was uniformly distributed with depth, indicating less 'young C' at greater depths. The effects of cultivation, soil texture (Goh et al. 1977; Anderson and Paul, 1984; Becker-Heidmann and Scharpenseel, 1992b), land use, land use history (Tate

Table 3.1 Estimates of the age and size of recalcitrant fractions in soil

Reference	Method	Pool	Age of pool (y BP)[*]	Size of pool (% of SOC)
Anderson and Paul (1984)	Radiocarbon dating and hydrolysis	Non hydrolysable organic carbon	2820±45	Not determined
Anderson and Paul (1984)	Radiocarbon dating and alkali pyrophosphate extract	Humic acid A	1425±95	32
Anderson and Paul (1984)	Radiocarbon dating and particle size fractionation	Coarse clay	1255±60	31
Arlsanov et al. (1970)	Radiocarbon dating and chemical fractionation	Sum of bound humic acids and humin	1140 to 6700	Not determined
Balesdent (1987)	Radiocarbon dating and hydrolytic fractionation	Non-hydrolysable material of clay size humin	approximately 280	<15
Buyanovsky and Wagner (1998)	Long-term experimental data: relationship between C inputs and SOC	Stable pool	>100	1.2
Campbell et al. (1967)	Radiocarbon dating and hydrolysis	Non hydrolysable residues of humic acid	1400	33.2
Campbell et al. (1967)	Radiocarbon dating and hydrolysis	Non hydrolysable residues of humin	1230	23.5
Harrison et al. (1993a)	Modelling of bomb-^{14}C incorporation in soils (Global review of data)	Slow pool of natural soils	3700	25
Harrison et al. (1993a)	Modelling of bomb-^{14}C incorporation in soils (Global review of data)	Slow pool of cultivated soils	3700	50
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	250	69.7
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	250	94.5
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	331	71.2
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	331	45.5
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	380	77.0
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	380	52.2
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	388	78
Hsieh (1996)	Paired-plots radiocarbon dating	Stable pool	335	73
Hsieh (1999)	Paired-plots radiocarbon dating	Resistant SOC	4405	51
Hsieh (1999)	Paired-plots radiocarbon dating	Resistant SOC	4405	70.4
Hsieh (1999)	Paired-plots radiocarbon dating	Resistant SOC	899	59
Hsieh (1999)	Paired-plots radiocarbon dating	Resistant SOC	899	41.3
Hsieh (1999)	Paired-plots radiocarbon dating	Resistant SOC	899	88.6
Jenkinson and Rayner (1977)	Radiocarbon dating and acid hydrolysis	Acid hydrolysis residues	2560	35
Jenkinson (1970)	Radiocarbon dating and acid hydrolysis	Acid hydrolysis residues	1995	43
Kobak and Kondrasheva (1988)	Estimates of NPP[†] and C pool sizes	Stable soil humus pool for Global C cycle	Approximately 1360	approximately 33
Leavitt et al. (1996)	Radiocarbon dating and 6N HCl hydrolysis	Total acid hydrolysis residue	1485 to 2282	Not determined
Martel and Lasalle (1977)	Radiocarbon dating and acid hydrolysis	Unhydrolysable C fractions	1530±110	59
Martel and Lasalle (1977)	Radiocarbon dating and humic acid fractionation	Humic acids	1220±150	17
Martel and Paul (1970)	Radiocarbon dating and acid hydrolysis	Acid hydrolysis residues	1100±100	40
O'Brien and Stout (1978)	Steady state radiocarbon model	Stable organic matter	at least 5700	16
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	4547	46.3
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	3623	41.7
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	6934	51.4
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	462	70
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	1110±648	61
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	905±550	62
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	2245±175	58
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	1700±25	56
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	2600±300	43
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	2314	23
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	2918±68	29
Paul et al. (1997)	Radiocarbon dating and acid hydrolysis	Non-hydrolysable C	2400	54
Theng et al. (1991)	Radiocarbon dating, fractionation and peroxide treatment	Inter-layer clay organic complex	6716±524	approximately 13
Trumbore et al. (1989)	Radiocarbon dating, density separation and acid hydrolysis	Heavy fraction acid hydrolysis residues	3530	21

et al. 1995), soil type, and climate on soil radiocarbon age, SOC residence time and turnover have also been noted. In particular, a larger proportion of stable SOM has been observed in grasslands, compared with forests (Harkness et al. 1991; Tate et al. 1995; Chichagova, 1996) although Condron and Newman (1998) noted a shift to more recalcitrant forms of SOC under fast growing forest, compared to grassland. Cultivated soils often have greater radiocarbon ages than native soils, as a result of depletion in 'modern' C (Theng et al. 1991).

Organic matter in temperate zones (at high latitudes) is generally more stable than in the tropics (Bird et al. 1996; Trumbore, 1993), as higher mean annual temperatures increase decomposition in the tropics. Stable organic matter is commonly associated with soil clay content (e.g. Tiessen and Stewart, 1883; Quiroga, 1996), and the coarse clay fractions of soil (Tsutsuki et al. 1988), although differences in fine fraction stability between soil types have been also been observed (Tiessen and Stewart, 1983). Differences in the age and distribution of organic matter fractions within and between soil types have also been observed (Goh et al. 1977).

3.1.1.6 RSOM and soils of variable charge

Soils of variable charge, especially Andisols and Oxisols, present a special case for the RSOM. The SOM in these soils is much more stable than in other soils of comparable cropping regime (Saggar et al. 1994; Parfitt et al. 1997). In these soils, OM is stabilised by iron oxides, ferrihydrite and allophane, a highly reactive clay mineral with a small particle size and large surface area. Modelling Andisols commonly requires substantial adjustment to SOM models (Oades et al. 1989; Jenkinson et al. 1991; Motavalli et al. 1994; Tate et al. 1996; Parfitt et al. 1997; Smith et al. 1997c), and large IOM or Passive SOM pools have been used in RothC and CENTURY to compensate for this (Parfitt et al. 1997; Smith et al. 1997a). Alternatively, Saggar et al. (1994) suggested that using surface area in place of soil clay content would improve model estimates of C turnover in allophanic soils. Andisols cover between 0.76%-0.84% of the world's land area (Leamy et al. 1980), and are common in areas of recent volcanic ash falls (e.g. New Zealand, Japan, Hawaii and Chile) but allophane may also be found in soils derived from sedimentary rocks, soils of glacial origin, and in some Spodosols (Parfitt, 1990).

3.1.2 Refractory organic matter in SOM models

3.1.2.1 What is RSOM in SOM models?

In RothC, IOM is a SOM compartment totally resistant to decomposition, with an arbitrary age of 50000 y (Coleman and Jenkinson, 1996). It is a device to allow the model to represent short-term changes in SOM brought about by changes in land management, and at the same time to account for the great radiocarbon ages measured in surface soils collected before the thermonuclear tests (i.e. prior to 1960; Jenkinson et al. 1987). In particular, IOM is needed in RothC to allow the model to match the rapidity of measured declines in SOC when land is bare fallowed (Jenkinson et al. 1999b). In most SOM models, the refractory pools are larger (and correspondingly younger) than the IOM pool in the current version of RothC. This is demonstrated by Figures 3.1 and 3.2 (Falloon and Smith, 1998, 2000), which show runs of RothC and CENTURY for the Broadbalk Winter Wheat Experiment (FYM treatment) at Rothamsted, UK (Dyke et al. 1983), using long-term weather, soil and land-management data.

The Passive SOM pool of CENTURY is very resistant to decomposition, includes physically and chemically stabilised SOM, has a turnover time of 400-2000 y, and is influenced by soil texture (Metherell et al. 1993). The transfer of SOM to the Passive SOM pool from other pools in CENTURY is influenced by soil clay content. In most SOM models, texture (especially clay content), total SOC, biomass, or C:N ratios (see section 2.5.1) define the refractory compartment.

3.1.2.2 What range of values has RSOM taken, and how is it set?

Previous model runs for RothC have used IOM values in the range of 1 %-26 % of total SOC (Falloon et al. 1998a). IOM values for RothC-26.3 have previously been set in one of two ways. The first method has been used in the absence of soil radiocarbon data for the site. In this case, the IOM content of the soil has been set arbitrarily, usually at around 10% of total soil organic carbon (eg. Jenkinson et al.

Figure 3.1 RothC simulated and measured SOC for the Broadbalk Experiment, FYM treatment

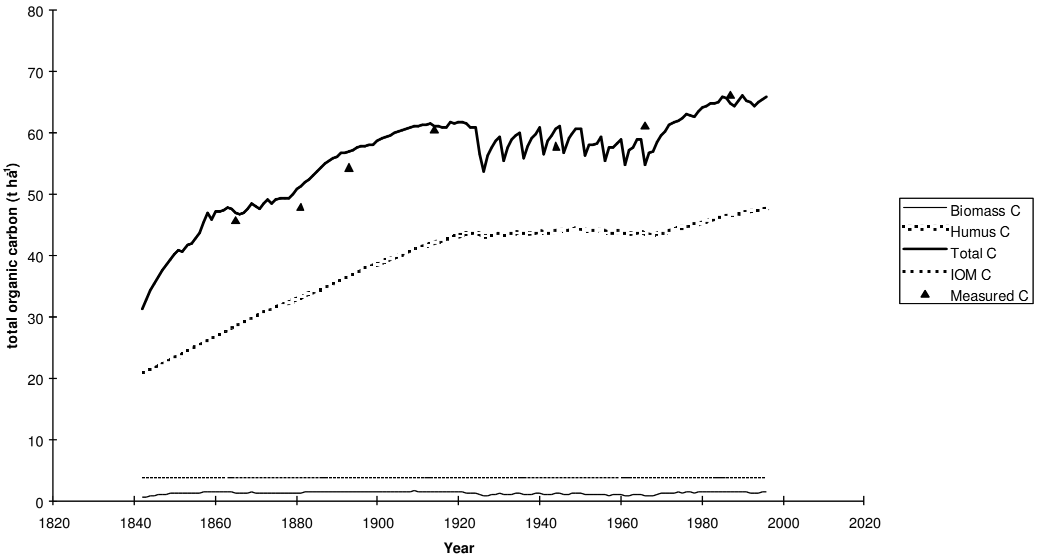
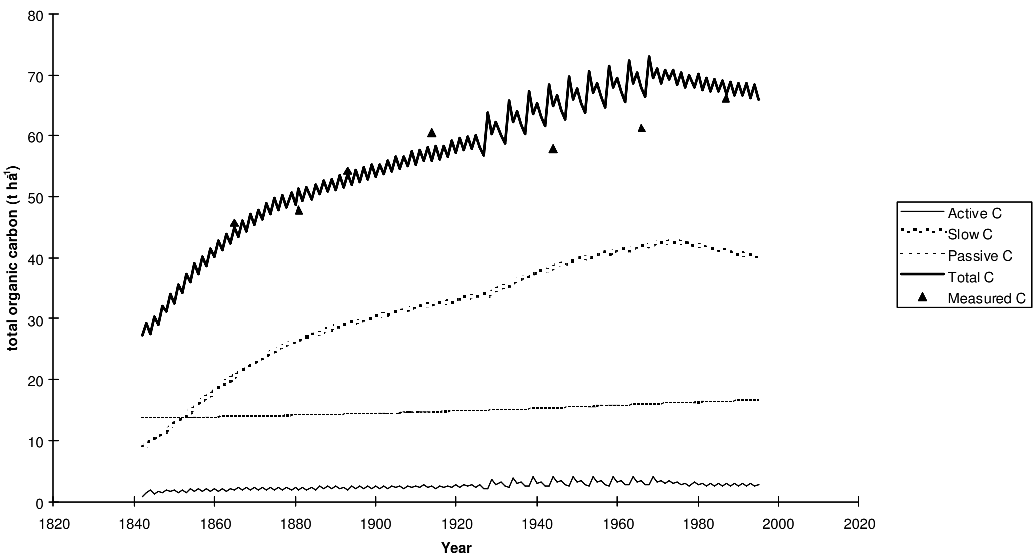


Figure 3.2 CENTURY simulated and measured SOC for the Broadbalk Experiment, FYM treatment



1991; Parshotam et al. 1995; Balesdent, 1996; Tate et al. 1996; Wu et al. 1998), or set arbitrarily to zero (e.g. Post et al. 1982; Wang and Polglase, 1995). Global-scale models for C cycling have also ignored IOM pools in their predictions (e.g. McGuire et al. 1997).

The second method requires soil $\delta^{14}C$ values for the site to be modelled. RothC is used to iteratively determine the soil IOM content needed to match the soil $\delta^{14}C$ value and SOC level (e.g. Coleman et al. 1994; Jenkinson and Coleman, 1994; Jenkinson et al. 1994; Parshotam and Hewitt, 1995; Tate et al. 1995; Coleman et al. 1997; Jenkinson et al. 1999a,b). Where soil $\delta^{14}C$ values are available, the model can be used to estimate the plant inputs (and net primary production) needed to maintain the measured level of SOC.

The Passive SOM pool of CENTURY is often set indirectly, using an 'equilibrium run' of 2000 y to 5000y to distribute SOM between the model pools (e.g. Kelly et al. 1997). Alternatively, the initial distribution of SOM between the model pools may be set arbitrarily, with Passive SOM as 40-55% of SOM (Metherell et al. 1993).

3.1.2.3 Can RSOM pools be measured experimentally?

There are a number of potential problems in actually 'measuring' IOM for RothC and Passive SOM for CENTURY. Firstly, in both RothC and CENTURY pools were not defined by any experimental measure but by fitting to long-term data. Secondly, the turnover time of IOM and Passive SOM excludes verification or direct measurement on the timescale of even the oldest long-term experiment. Thirdly, attempts to relate physical, biological and chemical fractionations to SOM model pools have had mixed success (Nicolardot et al. 1994; Motavalli *et al.* 1995; Balesdent, 1996; Christensen, 1996).

Problems related to fractionation of SOM are illustrated by the physical 'light fraction OM' (LFOM). LFOM is generally considered labile, but may include charcoal, due to its low density. This conflicts with the assumption that charcoal could make a significant contribution to the RSOM pools in SOM models (Skjemstad et al. 1996). Jenkinson et al. (1992) also noted that carbonized C should be removed from soil samples before determining SOC, and that it could account for a considerable proportion of IOM.

The use of particulate soil organic matter (POM) to calculate the size of the intermediate, 'Slow SOM' pool of CENTURY (e.g. Cambardella and Elliott, 1992), has also proven inapplicable to highly weathered tropical oxisols (Gijsman, 1996), perhaps due to faster turnover processes in these soils. Only the RPM compartment of RothC agreed well with separated organic matter >50μm in an evaluation of various chemical and physical fractions (Balesdent, 1996). A combination of density and chemical fractionation techniques, coupled with ^{14}C analysis, may prove more promising (Trumbore and Zheng, 1996). However, the fact remains that no single satisfactory method exists to separate soil C into components with different turnover times (Trumbore, 1990).

3.1.2.4 Why are the RSOM pools important?

The size of the RSOM pool in SOM models will, by definition, affect the size and dynamics of the more active pools, which are able to exchange CO_2 with the atmosphere (Schlesinger, 1995). It has been suggested that accurate specification of the size of any inert or passive pool is essential (Jenkinson et al. 1991; Skjemstad et al. 1996), and that uncertainty in its size could be a major source of error in calculating C inputs to soils (Jenkinson et al. 1991). However, few models specify the origin or nature of the IOM or refractory pool (McGill, 1996).

As discussed, direct measurement of IOM for RothC requires radiocarbon dating, which is costly and rarely performed routinely in long-term experiments. It is, therefore, essential for predictive modelling of SOM dynamics using RothC, to estimate IOM in the absence of radiocarbon data. This chapter also describes an investigation of methods for estimating Passive SOM for CENTURY, and an investigation of how the model assumptions underlying IOM and Passive SOM are borne out by experimental evidence. A field-scale investigation of the sensitivity of RothC and CENTURY to IOM and Passive SOM is presented. Finally, a simple land-use change scenario for European soils is used to demonstrate the significance of RSOM to global change studies.

3.2 ESTIMATING IOM FOR ROTHC IN THE ABSENCE OF RADIOCARBON DATA

3.2.1 Methods

In order to establish a statistically robust way of estimating IOM content in the absence of radiocarbon data, a data-set of soil, land management, and climatic factors was assembled for 28 steady-state treatments at long term experimental sites where radiocarbon data were available (Table 3.2; Falloon et al. 1998a). Estimates of the size of the IOM pool using radiocarbon data were from previous runs of RothC, with the exception of the Morrow Plots site, which had not previously been modelled using RothC. Climate, management and soil data were collected for the Morrow Plots site, which was run to equilibrium to give the steady-state IOM content, SOC value and input of C from vegetation. The dataset enabled a statistical examination of IOM, with the possibility to examine the effect of factors such as soil clay content, total soil carbon content, soil pH, mean annual rainfall and temperature, and management on IOM. Investigations included:

1. Simple linear regressions of individual variables against IOM, and
2. Stepwise multiple linear regressions of IOM against other variables to maximise the variance accounted for by the statistical model.

3.2.2 Results and discussion

Simple linear regressions between IOM and a range of soil and environmental parameters were first constructed, and are shown in Table 3.3. Since only total SOC and mean annual precipitation showed significant relationships with IOM, only these factors were included in stepwise multiple linear regressions. However, this resulted in a decrease in the proportion of variance accounted for by the model (65.4% for total SOC only, 64.2% for total SOC and mean annual precipitation), showing that

Table 3.2 Soil properties, land management, and climate for steady-state treatments in long-term experimental sites

No	Country	Site	Management	Sampling depth (cm)	Clay (%)	pH	Organic C (t ha^{-1})	$\delta^{14}C$	Annual input (t C ha^{-1})	IOM (t C ha^{-1})	Mean annual temperature (^{o}C)	Mean annual rainfall (mm)	FAO soil group	Reference
1	UK	Park Grass 3d un-manured	Grassland	0-23	23.0	5.70	75.6	-119.0	3.00	6.9	9.1	728.4	Luvisols	Jenkinson et al. (1992)
2	UK	Broadbalk 0.8 NPK	Arable	0-23	23.0	7.70	30.4	-166.0	1.45	3.8	9.1	693.0	Luvisols	Jenkinson et al. (1992)
3	UK	RES Drain Gauge	Arable	0-23	23.0	7.90	38.0	-175.0	1.73	6.3	9.1	717.0	Luvisols	Jenkinson and Coleman 1994
4	UK	Woburn Sp. Barley	Arable	0-23	10.3	5.60	47.8	-82.0	2.44	3.5	9.3	620.0	Arenosols	Jenkinson and Coleman 1994
5	UK	Gleadthorpe EHF	Arable	0-20	6.0	6.10	38.5	-258.0	2.95	10	8.8	1642.0	Arenosols	Jenkinson and Coleman 1994
6	UK	Rosemaund EHF	Arable	0-15	26.0	7.80	32.9	-82.0	2.47	2.2	9.1	679.0	Luvisols	Jenkinson and Coleman 1994
7	UK	Bridgets EHF	Arable	0-15	20.5	7.80	41.6	-138.0	3.60	4	9.5	815.1	Cambisols	Jenkinson and Coleman 1994
8	Syria	ICARDA wheat/wheat rotation	Arable	0-20	62.5	8.00	16.5	-53.9	0.94	3	18.0	324.0	Cambisols	D. S. Jenkinson, pers. comm.
9	USA	Calhoun exp. forest	Arable	0-15	15.0	4.30	5.8	7.0	0.38	0.5	16.0	1170	Arenosols	Coleman et al. (1997)
10	Kenya	Nairobi N. Park	Savannah	0-15	44.5	6.90	31.5	60.0	2.91	2.9	20.0	834.0	Vertisols	D. S. Jenkinson, pers. comm.
11	Kenya	Machange	Savannah	0-20	24.0	6.47	21.5	33.8	1.56	2.1	23.0	795.0	Cambisols	Unpublished
12	Kenya	Mutuobare	Savannah	0-20	18.0	6.54	25.2	101.7	2.28	1.2	25.0	855.0	Cambisols	Unpublished
13	Zambia	Misantu Res. Stn.	Dry Woodland	0-15	7.8	4.60	20.7	152.0	2.40	0.4	20.0	1245.0	Ferralsols	D. S. Jenkinson, pers. comm.
14	Ghana	Lower Tano Basin	Sec. Rainforest	0-15	45.0	4.40	47.0	69.0	6.30	3.1	26.1	1496.0	Ferralsols	Jenkinson (1973)
15	Ghana	Ho-Keta Plains	Savannah	0-23	7.0	6.40	20.0	41.0	2.00	0.7	27.0	1110.0	Calcisols	Jenkinson (1973)
16	New Zealand	Station Creek	Forest	0-36	28.0	4.60	138.0	85.0	8.00	6.3	9.8	1520.0	Podzols	Coleman et al. (1994)
17	New Zealand	Conroy	Grassland	0-23	12.0	5.85	27.0	-21.0	3.40	0.3	8.5	350.0	Leptosols	Tate et al. (1995)
18	New Zealand	Tima	Grassland	0-23	16.0	6.00	45.0	-81.0	0.90	6.1	8.5	630.0	Luvisols	Tate et al. (1995)
19	New Zealand	Tawhiti	Grassland	0-23	19.0	4.70	75.0	-39.0	2.50	10	5.5	1100.0	Cambisols	Tate et al. (1995)
20	New Zealand	Carrick	Grassland	0-23	11.0	4.60	73.0	-82.0	1.80	12	3.5	1230.0	Podzols	Tate et al. (1995)
21	New Zealand	Obelisk	Grassland	0-23	5.0	4.40	164.0	-107.0	3.80	29	2.0	1570.0	Podzols	Tate et al. (1995)
22	New Zealand	Kaikoura	Grassland	0-23	24.0	5.40	98.0	-21.0	3.30	9.9	5.5	1300.0	Cambisols	Tate et al. (1995)
23	New Zealand	Bealey	Forest	0-23	30.0	5.50	86.0	52.0	2.80	3.5	6.0	1300.0	Cambisols	Tate et al. (1995)
24	New Zealand	Maruia I	Forest	0-23	27.0	4.13	136.0	85.0	8.00	8	9.8	2000.0	Podzols	Tate et al. (1995)
25	New Zealand	Maruia II	Forest	0-23	26.0	4.13	151.0	9.0	8.20	18	9.8	2000.0	Podzols	Tate et al. (1995)
26	New Zealand	Puruki	Forest	0-23	12.0	5.50	148.0	20.0	8.90	15	10.5	1500.0	Podzols	Tate et al. (1995)
27	UK	Geescroft wilderness	Forest	0-23	21.0	5.60	21.0	-131.0	1.00	2.5	9.1	734.2	Luvisols	Coleman et al. (1997)
28	USA	Morrow Plots	Arable	0-23	25.0	6.00	71.7	-108.4	6.06	6.85	11.1	922.0	Gleysols	Hsieh (1992)

Table 3.3 Simple linear regressions between IOM and soil and environmental parameters

Environmental parameter	r^2 value	p value	Significance
Soil clay %	0.06	1.0	Not significant
Total SOC (t C ha^{-1})	0.66	<0.001	Highly significant
FAO soil code	0.05	1.0	Not significant
Mean annual temperature (^{0}C)	0.31	1.0	Not significant
Mean annual precipitation (mm)	0.32	<0.001	Highly significant
Annual C inputs (t C ha^{-1} y^{-1})	0.18	1.0	Not significant
Land use code	0.03	1.0	Not significant
Soil pH	0.16	1.0	Not significant

IOM is best correlated with SOC alone. Next, the relationship between total SOC and IOM was further investigated.

The simple linear regression of IOM on total SOC indicated a strong correlation, but the data showed variance increasing with the mean. Hence a model for IOM based upon total SOC (both in t C ha^{-1}) was constructed (Fig. 3.3), using log functions, which accounted for 63.4% of the variance in IOM (t_{26}=6.91, p<0.001). The model equation (Falloon et al. 1998a) is given below:

$$\log IOM = -1.31(S.E.0.28) + 1.139(S.E.0.165) \times \log SOC \quad \textbf{(1)}$$

which is equivalent to

$$IOM = 0.049 \times SOC^{1.139} \quad \textbf{(2)}$$

The equations for the model plus and minus the standard error, and for the upper and lower 95% confidence intervals (plus and minus 1.96 standard errors) are given by equations (3), (4),(5) and (6) below, respectively.

$$IOM\,(+1S.E.) = 0.0933 \times SOC^{1.304} \quad \textbf{(3)}$$

$$IOM\,(-1S.E.) = 0.0257 \times SOC^{0.974} \quad \textbf{(4)}$$

$$IOM\,(+95\%C.I.) = 0.1733 \times SOC^{1.4624} \quad \textbf{(5)}$$

$$IOM\,(-95\%C.I.) = 0.01384 \times SOC^{0.8156} \quad \textbf{(6)}$$

It should be noted that the confidence limits of the model are wide (equations 1, 5 and 6), reflecting the large variability of the data. The model cannot, therefore, provide a precise prediction. Separate models of log (IOM) and log (total SOC) content for each land-use show significant relationships for all land-uses except savannah. The regression equations for separate land uses are given below:

Grassland (n=7): $IOM = 4.178 \times 10^{-4} \times SOC^{2.269}$ (t_5=4.60, f=0.006) **(7)**

Arable (n=9): $IOM = 0.1151 \times SOC^{1.002}$ (t_7=4.25, f=0.004) **(8)**

Savannah (n=4): $IOM = 1.259 \times 10^{-4} \times SOC^{2.220}$ (t_2=1.44, f=0.286) **(9)**

Forest (n=8): $IOM = 0.0236 \times SOC^{1.223}$ (t_6=4.34, f=0.005) **(10)**

However, given the small sample sizes for each land-use (n < 10 in each case), and larger associated standard errors, it is suggested that the most robust way to calculate default IOM values for RothC in the absence of radiocarbon data is to use equation (2), regardless of land use.

The close relationship between total SOC and IOM is puzzling, since the presence or absence of charcoal, coal, or carbon from ancient sediments is largely independent of the present stock of organic C in a soil, which is determined by soil type, land use, and climate. It may be that a substantial part of IOM is not truly inert, but of very great turnover time. Although the relationship presented here was derived using data from a wide range of soil types, preliminary studies suggest that this relationship will not hold for soils with a high allophane content (Andosols; Smith et al. 1997c). Similarly, it is not expected to be valid for waterlogged (or highly organic) soils (Histosols); nor should it be used for sub-soils - an increase in radiocarbon age with soil depth, attributed to a larger proportion of inert C in the lower part of the soil profile, has been noted by several authors (Arslanov et al. 1970; Becker-Heidmann et al. 1988; Becker-Heidmann and Scharpenseel, 1992b).

Using the separate equations to predict IOM based on SOC for each land use, for a SOC content of 50 t C ha^{-1}, the following IOM contents are predicted: grassland, 2.99 t C ha^{-1}; forest, 2.83 t C ha^{-1}; savannah, 7.44 t C ha^{-1}; arable 5.80 t C ha^{-1}. Although there were insufficient data points to make statistical conclusions, this qualitatively suggests that there may be more inert carbon in savannahs than arable soils, and more in grasslands than forests.

This is consistent with experimental evidence that has inferred more stable SOC in grasslands than in forests (Harkness et al. 1991; Tate et al. 1995; Chichagova, 1996) and that cultivation resulted in depletion in modern C (Theng et al. 1991). There is no experimental evidence to compare with savannahs. The only variable showing a significant correlation in the initial investigations was mean annual rainfall, which showed a positive relationship. This is likely to be related to larger net primary production at higher levels of mean annual rainfall, which would result in a greater stock of SOC, and hence more IOM-C. It is also possible that intermittent periods of waterlogging could lead to increased SOC residence times (Tate et al. 1995).

3.3 INFLUENCE OF SOIL AND ENVIRONMENTAL PARAMETERS ON IOM AND PASSIVE SOM

3.3.1 Methods

As the IOM pool in RothC is truly inert, it is not affected by environmental factors, and so only CENTURY was used in this section. In order to investigate the influence of soil and environmental parameters on the Passive SOM pool in CENTURY, the long-term (3000y) dynamics of SOC were simulated, whilst varying one parameter at a time, keeping all other factors constant. The SOC pools of CENTURY were left empty at the beginning of the run.

The parameters investigated were:

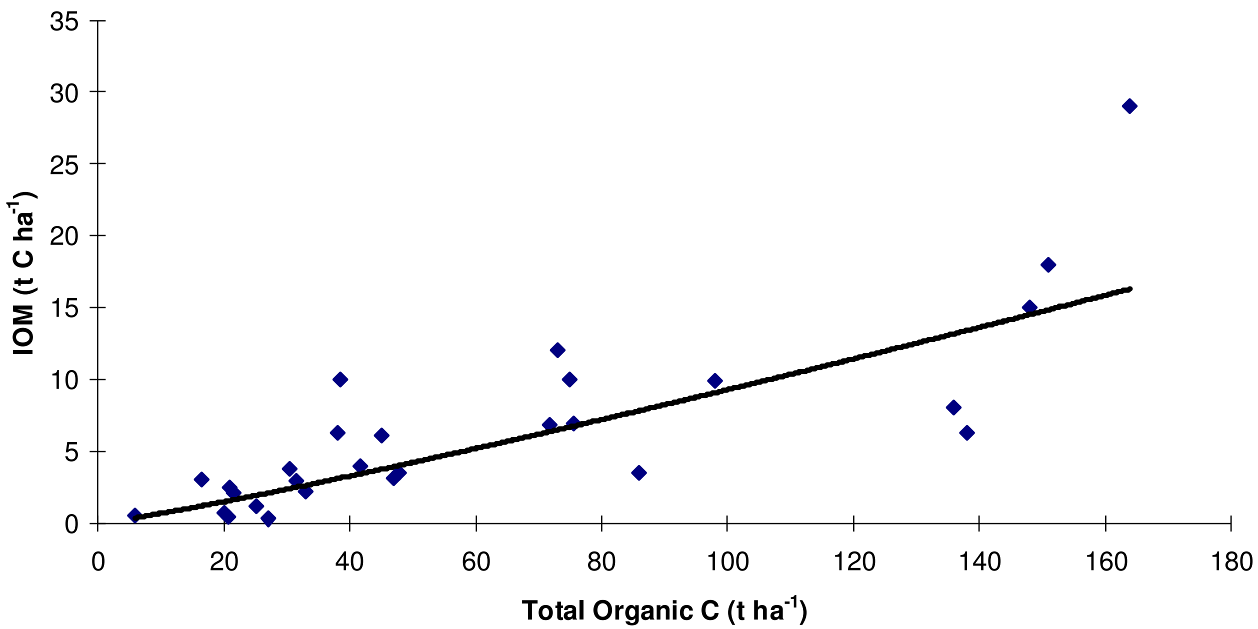
a) Soil clay % (0-100%)
b) Total annual rainfall (0-8000mm)
c) Mean annual temperature (0-30^{o}C)
d) Management (arable-till, arable-no-till, forest and grassland)

The runs used long-term averaged weather data for Rothamsted, and management files created to simulate the Rothamsted Geescroft Wilderness Experiment (Poulton, 1995a). The management runs used land management files set up for the Broadbalk

**Figure 3.3	Fitted model and IOM values estimated by RothC**

♦	IOM values estimated by RothC

—	Fitted model



Winter Wheat Experiment (arable – with hypothetical till vs. no till scenarios), the Rothamsted Geescroft Wilderness Experiment (natural woodland regeneration), and the the Rothamsted Park Grass Experiment (grassland; Poulton, 1995b).

The aim of this exercise was to investigate the relative importance of different factors in determining the long-term build up or decay of RSOM in soils as represented by CENTURY, and to compare this to experimental evidence from the literature.

3.3.2 Results and discussion

3.3.2.1 Influence of soil clay content

Figure 3.4 shows CENTURY predicted total SOC against clay content after 3000y runs using land management and weather data for the Geescroft Wilderness Experiment, against soil clay content from 0-100%. Figure 3.5 shows the proportion of Passive SOC to total SOC for the same runs. At 0% clay content, the model predicts that there is approximately 25% of SOC as Passive SOC, which increases with the clay content up to a maximum of about 65% of SOC as Passive SOC at 100% clay content. Fitting a model to the curve in Figure 3.5 gave the equation below (r^2=0.97).

$$\frac{Passive\ SOM}{Total\ SOM} = -4e^{-5(clay\%^2)} + (0.0079 \times clay\%) + 0.2446$$

In CENTURY, the following equations define the flow of SOM from the active and slow pools, respectively:

$$Flow\ from\ Slow\ SOM\ to\ Passive\ SOM\ = 0.003 - 0.009(clay\%)$$

As would be expected from the equations defining the flow to the Passive SOM pool from the other SOM pools in CENTURY, the model runs assuming different clay contents showed an increase in both the proportion of Passive SOC: total SOC and the amount of total SOC with increasing clay content. RothC also assumes that the build up of SOC is clay-related, but clay is a rate-modifying factor for SOC decomposition in general and not linked to the IOM

Figure 3.4 CENTURY modelled Total SOC vs. clay content, 3000y runs

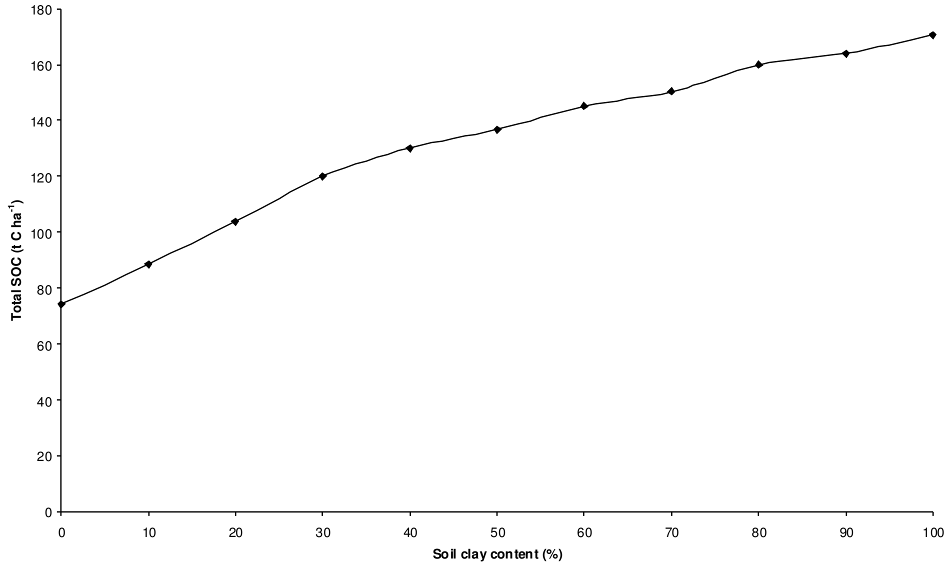
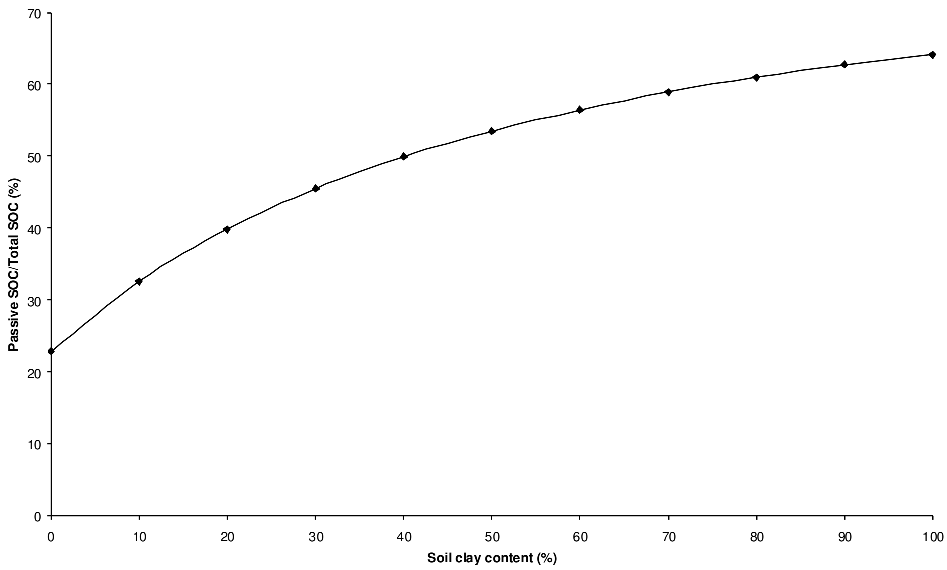


Figure 3.5 CENTURY modelled Passive SOC/Total SOC vs. clay content, 3000y runs



pool. Experimental evidence suggests more RSOM in soils with higher clay contents (Goh et al. 1977; Anderson and Paul, 1984; Becker-Heidmann and Scharpenseel, 1992b; Buyanovsky et al. 1994; Kaiser et al. 1998), as previously discussed.

3.3.2.2 Influence of total annual rainfall

Figure 3.6 shows total SOC against the total annual rainfall, after 3000y runs using land management and weather data for the Geescroft Wilderness Experiment. Figure 3.7 shows the proportion of Passive SOC to total SOC for the same runs, against total annual rainfall from 0-8000mm. At 0mm annual rainfall, about 5% of total SOC builds up as Passive SOC, even though the amounts of total SOC are very small. The amount of total SOC and the proportion of Passive SOC increase sharply with mean annual rainfall, with Passive SOC reaching about 38% of Total SOM at about 500mm of rainfall p.a. Above 500mm-rainfall p.a., there is a much more gradual increase in total SOC, and the proportion of Passive SOC to Total SOC, with Passive SOC reaching 60% of total SOC at around 4500mm of rainfall p.a. Above 4500mm rainfall p.a., the amount of total SOM and the proportion of Passive SOC remain constant, at around 450 t C ha^{-1} and 60%, respectively.

The model runs with variable total annual rainfall show a clear increase in the proportion of Passive SOC: total SOC and also the amount of total SOC with increasing rainfall. The initial analysis of single variables on IOM for the dataset for IOM prediction for RothC indicated a similar relationship. This implies that more SOC is formed as a result of elevated net primary production at higher levels of rainfall. However, as the proportion of Passive SOC: total SOC also increases with rainfall, this suggests that there is not simply more SOM, but more stable SOM. Tate et al. (1995) noted that intermittent periods of waterlogging could lead to increased SOC residence times. Waterlogging could also be a factor leading to higher levels of stable SOM at higher mean annual precipitation, since decomposition rates may be reduced under anaerobic conditions.

3.3.2.3 Influence of mean annual temperature

Figure 3.8 shows total SOC against the mean annual temperature after 3000 y runs,

Figure 3.6 CENTURY modelled Total SOC vs. Total Annual Rainfall, 3000y runs

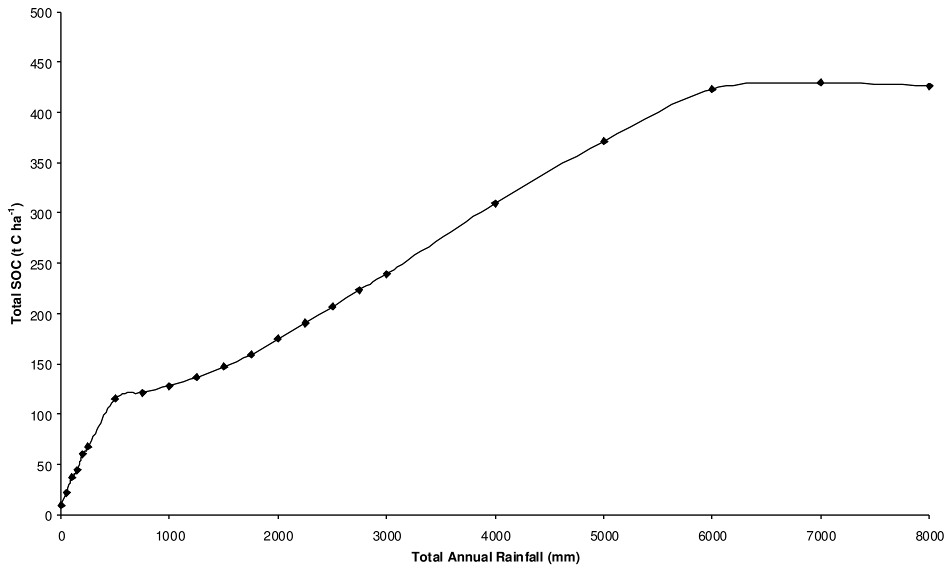


Figure 3.7 CENTURY modelled Passive SOC/Total SOC vs. Total Annual Rainfall, 3000y runs

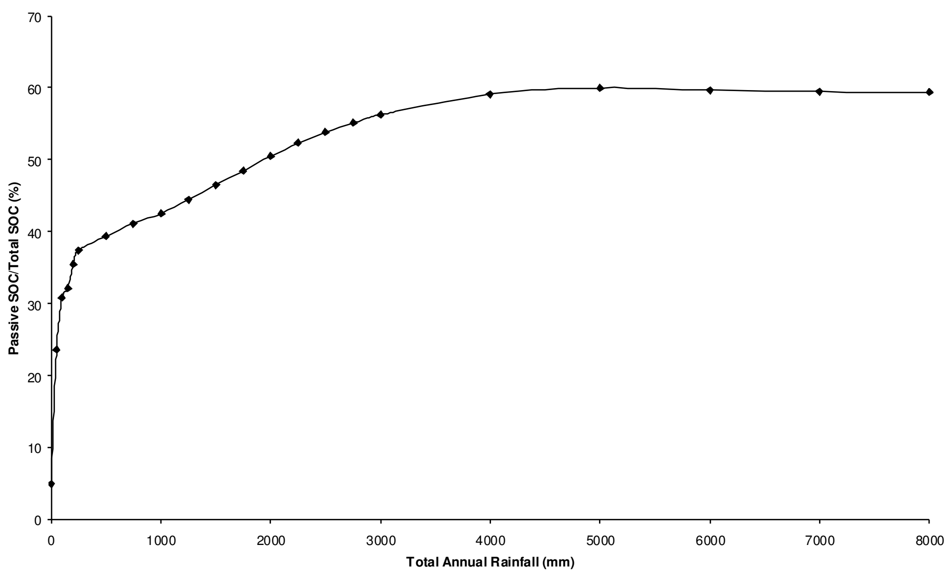


Figure 3.8 CENTURY modelled Total SOC vs. Mean Annual Temperature, 3000y runs

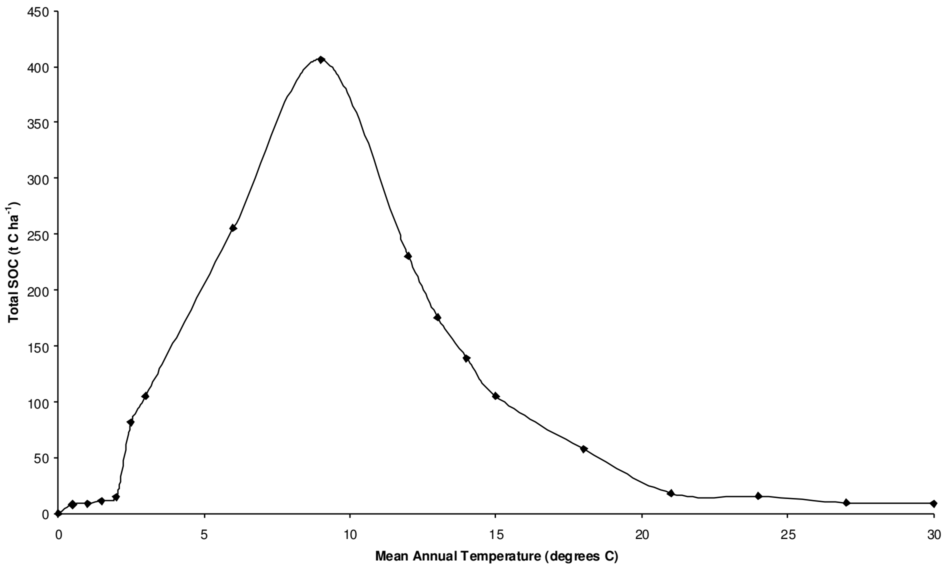
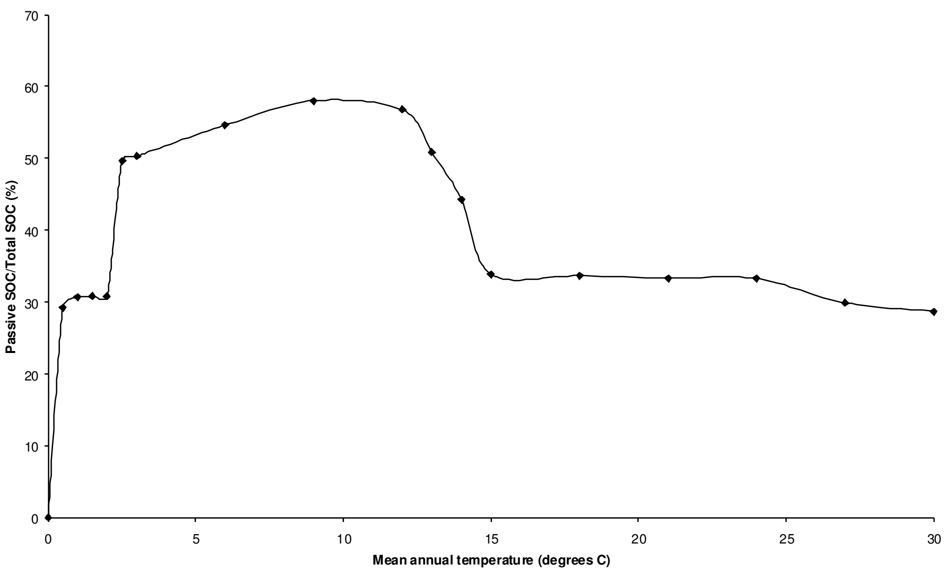


Figure 3.9 CENTURY modelled Passive SOC/Total SOC vs. Mean Annual Temperature, 3000y runs



using land management and weather data for the Geescroft Wilderness Experiment. Figure 3.9 shows the proportion of Passive SOC to total SOC for the same runs, against mean annual temperature from 0-30^{o}C. At a temperature of 0 ^{o}C, very small amounts of total SOC and Passive SOC build up, but between about 2-3 ^{o}C, the amount of total SOC and proportion of Passive SOC increase dramatically, with Passive SOC reaching 50% of total SOC. Between approximately 3 ^{o}C and 7 ^{o}C, there is a smaller increase in the proportion of Passive SOC to total SOC, and the amount of total SOC. From approximately 7 ^{o} C to 15 ^{o} C, the proportion of Passive SOC to total SOC, and the amount of total SOC decrease to a fairly constant level of around 35% of total SOC.

The model runs with variable temperature showed a roughly bell-shaped response curve of total SOC to mean annual temperature, with the peak at around 10^{o}C. In CENTURY, the temperature response curve for relative plant production is set using a Poisson Density function (Metherell et al. 1993). For C3 plants, the lower limit for production is between 0 and 5^{o}C, and the peak of the Poisson Density curve is around 15^{0}C. This would explain the pattern of total SOC accumulation shown by the model runs, which would be modified by the plant production parameters set for the Rothamsted Geescroft Wilderness Experiment. However, the greatest proportion of Passive SOC:Total SOC was also observed close to the maximum plant production levels. The reasons for this are unclear. From experimental evidence, less RSOM would be expected at higher temperatures, as the SOC in tropical zones is less stable than that in temperate zones (Bird et al. 1996; Trumbore, 1993).

3.3.2.4 Influence of management

Figure 3.10 shows CENTURY modelled total SOC for the four management scenarios, with 3000y runs. Figure 3.11 shows the proportion of Passive SOC to total SOC for the same runs. Figure 3.11 shows that over a 3000y timescale, CENTURY predicts the proportion of Passive SOC to total SOC to be: Arable-tilled > Arable no-till > Grassland > Forest. The amount of total SOC accumulating under the different management scenarios was: Grassland > Forest > Arable no-till > Arable-tilled. As previously discussed, experimental evidence suggests more stable SOC in grasslands than in forests (Harkness et al. 1991; Tate et al. 1995;

Figure 3.10 CENTURY modelled Total SOC for different management scenarios, 3000y runs

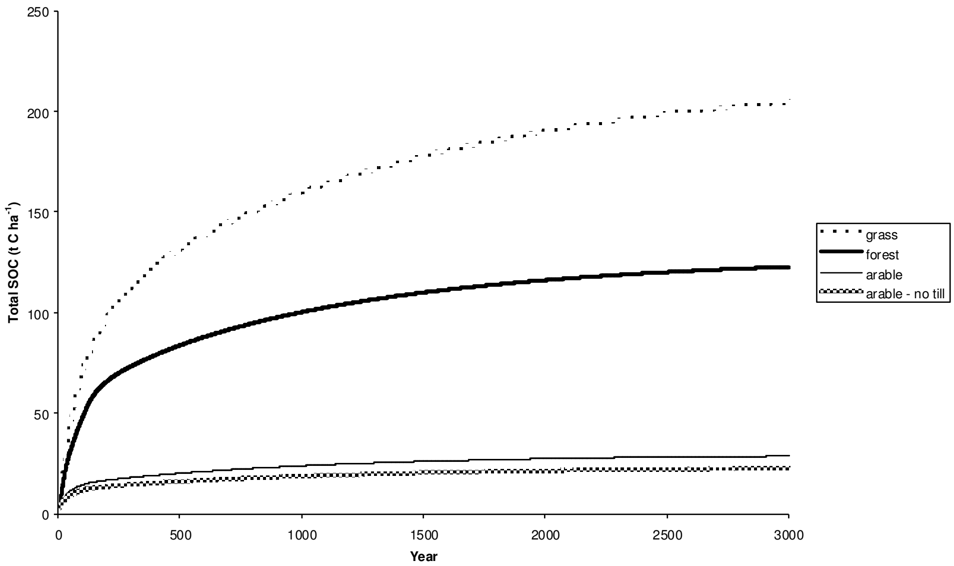
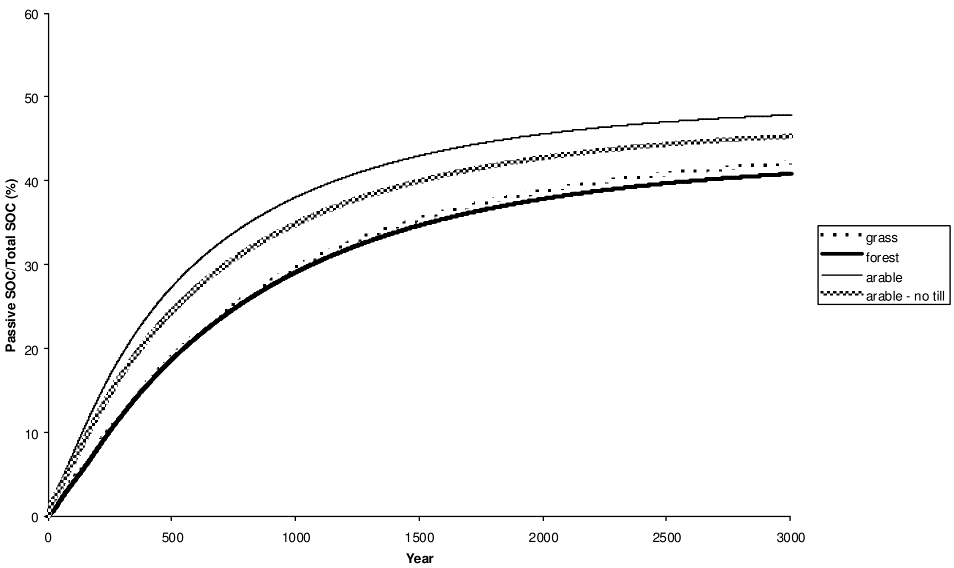


Figure 3.11 CENTURY modelled Passive SOC/Total SOC for different management scenarios, 3000y runs



Chichagova, 1996) and that cultivation results in depletion in modern C (Theng et al. 1991). Physical fractionation of conventional till and no-till soils has shown a decrease in total SOC, and an increase in the proportion of stable SOC to total SOC remaining (Six et al. 1998). The management impacts on Passsive SOM were also confirmed by the dataset used for estimating IOM for RothC in the absence of radiocarbon data (see section 3.2.2)

3.4 SENSITIVITY OF ROTHC AND CENTURY TO IOM AND PASSIVE SOM AT THE FIELD SCALE

3.4.1 Methods

The aims of the work reported in this section were to:

1. Estimate errors introduced in modelling SOC turnover, by methods of setting IOM in RothC and Passive SOM in CENTURY, and
2. Investigate the sensitivity of model predictions of SOC turnover to IOM and Passive SOM.

RothC and CENTURY were again run for the Rothamsted Geescroft Wilderness Experiment, but this time varying the proportion of IOM to total SOC and Passive SOM to total SOC, respectively. For RothC, the IOM values used were: IOM set using radiocarbon data, IOM from the regression of IOM on SOC (equation 2, section 3.2.2), IOM from the upper and lower 95% confidence limits of the regression (equations 5 and 6, section 3.2.2), and 0%, 25%, 50%, 75%, 90%, 110%, 125%, 150%, 175%, 200% and 300% of IOM set by radiocarbon data (Falloon et al. 2000). For CENTURY, the Passive SOM values used were: the value set by a CENTURY model developer for a previous comprehensive evaluation exercise (Kelly et al. 1997), and 0%, 50%, 75%, 90%, 110%, 125%, 150% of this value, and Passive SOM = 100% of SOM. The pool sizes used for both models are given in Table 3.4. The resultant simulated total SOC values were studied, keeping C inputs to the soil and all other factors constant.

3.4.2 Results and discussion

Figure 3.12 shows RothC-modelled total SOC for the Geescroft Wilderness Experiment, varying the soil IOM content as follows: a) that given by fitting the model to radiocarbon data; b) that from the regression (equation 2, section 3.2.2), and c) that from the upper and lower 95% confidence limits of the regression (equations 5 and 6, section 3.2.2). Figure 3.13 shows RothC-modelled total SOC for the same experiment, varying the soil IOM content as fixed proportions of the value set using radiocarbon data. Figure 3.14 shows CENTURY-modelled total SOC, also for the Geescroft Wilderness Experiment, varying the initial proportion of Passive SOC from the original value. Table 3.5 shows the Root Mean Square Error (RMSE; Smith et al. 1996d) values for selected model runs, and the impact of changing the relative size of the IOM and Passive SOM pools on SOC simulations.

Using the regression to predict IOM for RothC gives results close to those obtained using radiocarbon data (RSME=6.41, Table 3.5). At the lower 95% confidence level of the regression, RothC slightly under-predicts total SOC (RMSE=6.66), but using IOM from the upper 95% confidence interval results in a large over-prediction in total SOC (RMSE=19.46). The SOC values obtained, after 100y, by using IOM from the lower and upper 95% confidence levels of the regression result in differences of (-) 2.59% and (+) 18.52% from the radiocarbon predicted value, respectively. The differences in total SOC from the value set by radiocarbon also increase with time.

The model runs giving the best fit to the experimental data, indicated by the lowest RMSE values are for RothC with IOM at 75%-90% of the IOM value set using radiocarbon data, and for CENTURY at around 50% of the value used by Kelly et al. (1997). Using 0% IOM for RothC, there is a difference of –1.63 t C ha^{-1} (-2.79%) from the final measured total SOC value. Using 300% of the radiocarbon-derived value for IOM for RothC results in a difference of –4.9 t C ha^{-1} (-8.3%) from the final measured total SOC value. The differences using CENTURY are much wider. With 0% Passive SOM, the difference from the final measured total SOC value is –12.31 t C ha^{-1} (-18.61%); with 100% of SOC as Passive SOC (300% of the value used in Kelly et al. 1997 would have been greater than total SOC)), the difference is 9.86 t C ha^{-1} (14.29%).

Table 3.4 Pool sizes (t C ha^{-1}) used in initializing RothC and CENTURY for the sensitivity runs

	RothC IOM value[1]														
	0%	25%	50%	75%	90%	100%	110%	125%	150%	175%	200%	300%	Lower 95%[4]	Upper 95%[5]	Regression[6]
DPM	0.05	0.05	0.05	0.05	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.03	0.05	0.01	0.05
RPM	3.71	3.62	3.52	3.43	3.38	3.34	3.30	3.25	3.15	3.06	2.97	2.22	3.68	0.86	3.43
BIO	0.55	0.53	0.52	0.51	0.50	0.49	0.49	0.48	0.46	0.45	0.44	0.33	0.54	0.13	0.51
HUM	20.63	20.12	19.60	19.08	18.77	18.57	18.36	18.05	17.53	17.01	16.50	12.36	20.48	4.81	19.05
IOM	0.00	0.63	1.25	1.88	2.25	2.50	2.75	3.13	3.75	4.38	5.00	10.00	0.19	19.13	1.91
Total	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94	24.94
	CENTURY Passive SOM value[2]														
	0%	50%	75%	90%	100%	110%	125%	150%	100% of SOC[3]						
Active surface SOM	2.77	2.02	1.64	1.41	1.26	1.11	0.88	0.50	0.00						
Active soil SOM	2.69	1.95	1.59	1.37	1.22	1.07	0.85	0.49	0.00						
Slow SOM	22.02	16.01	13.00	11.20	10.00	8.80	7.00	3.99	0.00						
Passive SOM	0.00	7.50	11.25	13.50	15.00	16.50	18.75	22.50	27.48						
Total SOM	27.48	27.48	27.48	27.48	27.48	27.48	27.48	27.48	27.48						

[1] IOM value used for RothC model run, % of IOM value set using radiocarbon data
[2] Passive SOC value used for CENTURY model run, % of Passive SOM value set for a model evaluation and comparison exercise described in (Smith et al. 1997a)
[3]Passive SOM value equal to 100% of total SOC
[4]IOM set from the lower 95% confidence interval of the regression
[5]IOM set from the upper 95% confidence interval of the regression
[6]IOM value set using the regression

The simple sensitivity analyses performed for RothC and CENTURY, using data from the Geescroft Wilderness Experiment, showed that CENTURY-predicted total SOC is much more sensitive to changes or errors in the size of the Passive SOM pool than was RothC-predicted total SOC to changes in IOM.

This is not surprising for several reasons. Firstly, the IOM pool in RothC is much smaller than CENTURY's Passive SOM pool, and so a comparable proportionate change in both will result in a larger proportion of total SOC being affected in CENTURY. Secondly, Geescroft was one of the RothC model development sites. Thirdly, CENTURY attempts to estimate inputs of carbon to the soil independently using its plant production model, whereas for RothC, inputs are fitted iteratively to measured total SOC data.

Finally, predicting IOM for RothC in the absence of radiocarbon data within the 95% confidence limits of the regression of IOM on SOC could result in significant errors in SOC predictions: up to 18.54% difference in final predicted SOC values, or 37.02% difference in the predicted change in SOC stocks. Therefore accurate specification of IOM for RothC is important at the field scale.

3.5 SENSITIVITY OF ROTHC TO IOM FOR LARGE-SCALE SOC PREDICTIONS

3.5.1 Methods

The effect of using the regression developed for estimating IOM for RothC (section 3.2) on a simple carbon sequestration estimate was investigated (Falloon et al. 2000). To estimate the maximum error, the scenario used was that most effective in increasing the SOC stocks of European arable land (Smith et al. 1997b, 1998a,c), i.e. afforestation of 30% of surplus arable land. C inputs, weather and land management were based upon the Geescroft Wilderness Experiment (Jenkinson et al. 1991).

The change in SOC stocks over 100 y was calculated using model runs with IOM set by: a) radiocarbon data, b) the regression of IOM on SOC (section 3.2.2), and c) from the upper and lower 95% confidence limits of the regression (section 3.2.2).

Figure 3.12 RothC modelled total SOC for the Geescroft Wilderness Experiment

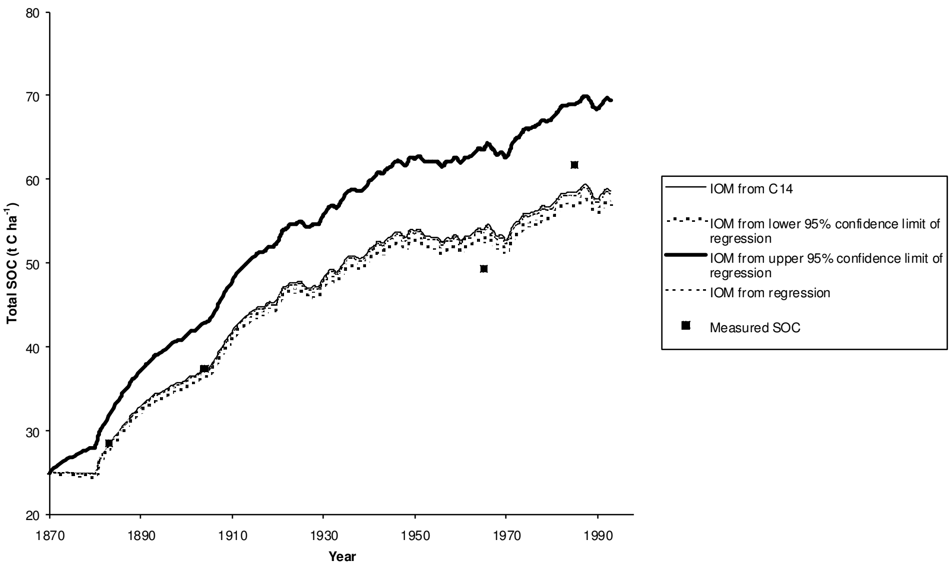


Figure 3.13 RothC modelled total SOC for the Geescroft Wilderness Experiment

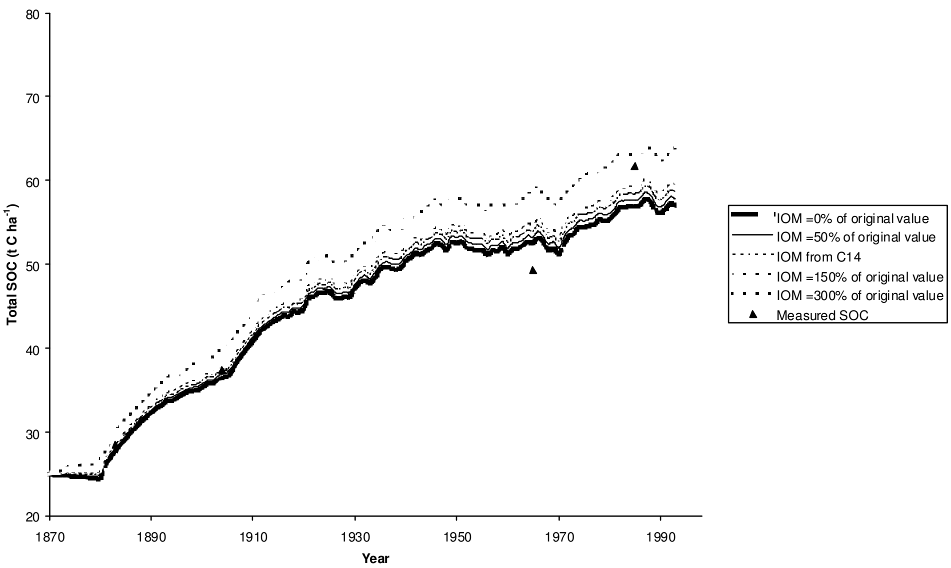
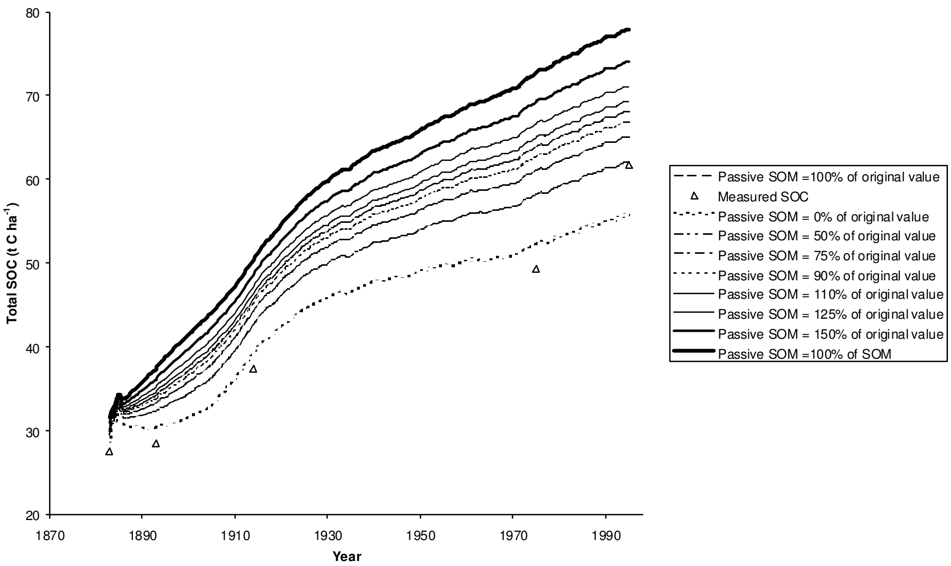


Figure 3.14 CENTURY modelled total SOM for the Geescroft Wilderness Experiment



The proportionate change in SOC stocks from these values were then applied to the current SOC stock of European arable land (Smith et al. 1997b, 1998a).

3.5.2 Results and discussion

Table 3.6 shows differences in SOC, and increases in total European SOC stocks predicted by various IOM values, over 100 years. The largest deviation from the change in SOC predicted using IOM set with radiocarbon data is that using IOM set from the upper 95% confidence interval of the regression. Given a total SOC stock for this area to be 34.64 Pg C (Smith et al. 1997b, 1998a), the difference in the predicted change in SOC stocks over 100 years is 35.59% (0.85 Pg C) for the *area affected by the land use change* (i.e. the 30% arable land afforested). However, the difference in the predicted change in SOC stocks for the *whole of Europe* is 2.29% over 100 years.

This simple investigation using RothC, the difference in the predicted change in SOC stocks may be as large as 35.59%, within the area to which a land use/management change is applied, when IOM is set at the limit of its 95% confidence interval.

Table 3.5 Root Mean Square Error to measured data and difference from final SOC values for sensitivity runs using RothC and CENTURY

RothC				CENTURY			
IOM value[1]	RMSE[2]	Difference[3] (t C ha^{-1})	% Difference[4]	Passive SOC value[5]	RMSE[6]	Difference[1] (t C ha^{-1})	% Difference[8]
Regression[12]	6.41	-0.39	-0.66				
Lower 95%[10]	6.66	-1.51	-2.58				
Upper 95%[11]	19.46	10.86	18.54				
0%	6.72	-1.63	-2.79	0%	10.30	-12.31	-18.16
25%	6.55	-1.22	-2.09				
50%	6.44	-0.82	-1.39	50%	8.19	-6.07	-8.96
75%	6.41	-0.41	-0.70	75%	11.03	-3.01	-4.44
90%	6.42	-0.16	-0.28	90%	13.26	-1.20	-1.77
100%	6.44	0.00	0.00	100%	14.86	0.00	0
110%	6.47	0.16	0.28	110%	16.51	1.19	1.76
125%	6.54	0.41	0.70	125%	19.03	2.98	4.39
150%	6.72	0.82	1.39	150%	23.19	5.87	8.67
175%	6.94	1.22	2.09				
200%	7.22	1.63	2.79				
300%	10.76	4.90	8.36	100% of SOC[9]	28.82	9.68	14.29

[1] IOM value used for RothC model run, % of IOM value set using radiocarbon data
[2]RMSE of paired modelled-measured points
[3]Difference from final SOC value of RothC run using radiocarbon data to set IOM value
[4]% Difference of [3] above
[5] Passive SOC value used for CENTURY model run, % of Passive SOM value set for NATO workshop (Kelly et al. 1997)
[6]RMSE of paired modelled-measured points
[7]Difference from final SOC value of RothC run with Passive SOM value set for NATO workshop (Kelly et al. 1997)
[8]% Difference of [7] above
[9]Passive SOM value equal to 100% of total SOC
[10]IOM set from the lower 95% confidence interval of the regression
[11]IOM set from the upper 95% confidence interval of the regression
[12]IOM value set using the regression

Table 3.6 Differences in predicted SOC increase for European arable land due to the size of the IOM pool, assuming afforestation of 30% arable land

	Initial SOC (t C ha^{-1})	Final SOC (t C ha^{-1})	SOC difference over 100y (t C ha^{-1})	Increase in total European C stocks over 100y[1] (P g)	Predicted total SOC stock after 100 y of afforestation of 30% arable land (P g)
IOM from radiocarbon data	24.94	52.73	27.79	2.4000	37.0400
IOM from regression	24.94	52.37	27.43	2.3690	37.0090
IOM from upper 95% CI[2]	24.94	62.62	38.08	3.2543	37.8943
IOM from lower 95% CI[2]	24.94	51.35	26.41	2.2809	36.9209

[1]Assuming the same area of land and scenario as in Smith et al. (1997b,1998a) and using RothC for the Geescroft Wilderness Experiment to predict SOC increases.
[2] Confidence interval of the regression of IOM on SOC (Section 3.2.2)

Hence uncertainties in IOM estimates could cause significant errors in SOC prediction. However, only a small error is introduced for the change in SOC stocks for Europe as a whole, since only 8.3% of the land area is affected by this scenario. The maximum error introduced by setting IOM using the upper 95% confidence interval of the regression is less than the error inherent in previous approaches for estimating regional C sequestration potential. Smith et al. (1998a), used a regression-based estimate in their scenario of conversion of all European arable lands to no-till farming. The difference between the mean value and the upper 95% confidence interval was 53.23% for the change in SOC stocks of the affected area, or 5.93% for the change in the total European SOC stock. Therefore, the errors introduced by uncertainties in the IOM pool using a dynamic modelling approach are less than those introduced by methods previously used for estimating carbon sequestration.

3.6 GENERAL DISCUSSION AND CONCLUSIONS

The work described in this chapter has led to the derivation of a relationship to enable IOM to be estimated for RothC in the absence of radiocarbon data, and investigated the way in which Passive SOM is set and is influenced by other parameters in CENTURY. The CENTURY model simulations and the IOM database showed model assumptions on RSOM to be consistent with experimental evidence, with the exception of the influence of increased mean annual temperature. The regression for estimating IOM for RothC in the absence of radiocarbon data will prove extremely useful for predictive modelling work, particularly when using large databases, where radiocarbon measurements are scarce or non-existent. However, the regression cannot be used with any confidence for waterlogged, highly organic, recent volcanic soils, or sub-soils. Extrapolation to these soil types would only be possible with model development for these conditions. The regression, and indeed, the RothC model, should also not be used for soils containing large amounts of recently added IOM, such as charcoal (Jenkinson et al. 1999b).

The regression could be further tested and developed with the collection of more paired SOC-radiocarbon measurements from steady-state sites. Passive SOM for CENTURY is generally set indirectly, using an 'equilibrium' run to distribute C between the pools. Although there is some relationship between measured fractions and the model pools in CENTURY, Passive SOM can only be estimated by difference (Metherell et al. 1993). The collection of model pool sizes used for long-term CENTURY model runs, with site, soil, management and weather data in a database could help investigation of simple relationships between Passive SOM and other variables, similar to the IOM-SOC relationship presented here.

Whether it is better to model SOM turnover using a small inert pool or a larger pool of very slow turnover rate will depend upon several factors. Certain changes in land management are impossible to simulate without the use of a small inert pool (Coleman et al. 1997), but inert SOM pools cannot simulate the influence of soil or environmental parameters on RSOM. Using either radiocarbon data or the regression, it is a simple and transparent process to set IOM for RothC. However, the regression does have wide confidence limits. In practical terms, it is possible that there are both very small amounts of SOM that is truly inert (ISOM), and a larger pool that turns over slowly but is not inert (RSOM). Passive SOM contains both of these conceptual pools of SOM; IOM only contains the former. However, given the close relationship between SOC and IOM, it seems unlikely that the IOM pool in RothC actually represents SOM that is only inert. If IOM were truly inert, a close relationship between IOM and the current stock of SOC in soils would not be expected for soils where previous land management has significantly affected total SOC stocks.

True IOM (ISOM) is related to soil forming conditions, and ancient coals and sediments, and is not part of contemporary SOM turnover processes. RSOM is part of the current turnover process in soils, and is influenced by the factors affecting the turnover of bulk SOM – climate, soil texture, and management, as was described in the CENTURY runs.

The simple sensitivity analysis for IOM and Passive SOM at the field scale showed that CENTURY was much more sensitive to Passive SOM than was RothC to IOM. Predicting IOM within the confidence limits of the regression could lead to significant errors in RothC-predicted SOC, at the field scale. Significant errors in field scale CENTURY SOC modelling studies could result from uncertainties in the size of the Passive SOM pool. This suggests that in terms of field-scale predictive modelling, using the regression to predict IOM for RothC may lead to smaller errors in predicted SOC values compared to uncertainties in CENTURY's Passive SOM, which will have a larger effect on future SOC predictions.

Finally, the simple land management-change scenario for European soils (shown by Smith et al. (1997b, 1998a,c) to produce the largest change in European SOC stock) indicated that uncertainties in IOM estimates may introduce significant errors in RothC-predicted SOC values within the area of land affected by the scenario. These errors are relatively small when considering changes in the SOC stock for Europe as a whole. However, the errors in the predicted change in SOC stocks could be large. The upper 95% confidence interval of the regression for IOM on SOC effected an error of 35.59% of the change in SOC using IOM set with radiocarbon data. It must be stressed that the maximum error presented here was for the outer 95% confidence interval of the regression of IOM on SOC, and for the land-management change scenario most effective in sequestering C in soils. Thus, this represents the maximum error that could be caused by uncertainties in the size of the IOM pool. As most scenarios to be investigated will involve smaller changes in SOC stocks and IOM would normally be set from the actual regression and not the 95% confidence interval, the IOM-related error will be considerably less. Finally, it was shown that the size of the maximum error introduced to RothC-based carbon sequestration estimates by uncertainties in the size of the IOM pool are less than those inherent in previously approaches for carbon sequestration estimates.

4. ESTIMATING PLANT INPUTS OF C TO SOIL

4.1 INTRODUCTION

The Net Primary Production (NPP) and input of plant-derived C to the soil are major driving variables of the soil C balance at scales from field to globe. Hence, their accurate estimation is essential for predictive SOM modelling and terrestrial global change studies (Bolinder et al. 1997). Indeed, as shown by both simulation modelling and experimental data, there is a direct relationship between SOC and C inputs to soil (Paustian et al. 1997a; Buyanovsky and Wagner, 1998). However, C inputs to soil are difficult to measure directly (Jenkinson et al. 1992) and the lack of theory and data regarding soil C inputs make strict SOM model evaluation impossible (Katterer and Andren, 1999).

Estimating C inputs for RothC is a significant step towards using the model in a predictive mode, and contributes to model evaluation (Chapter 5) and regional-scale modelling (Chapter 6). In this chapter, approaches to measuring, estimating, and modelling NPP and plant inputs of C to soil are briefly reviewed. A modelling exercise using RothC with 60 long-term datasets from the Global Change and Terrestrial Ecosystems Soil Organic Matter Network (GCTE-SOMNET) is also presented, which enables default plant input values to be estimated for three major land use classes. The sensitivity of RothC to the magnitude and quality of annual C inputs is investigated for the Geescroft Wilderness Experiment (Poulton, 1995a), at equilibrium and over a timescale of decades to centuries. The results are applied to a simple land management change scenario for the stock of SOC in European arable land, to investigate the effect of errors in C inputs on SOC predictions at the regional scale.

4.1.1 NPP and inputs of C to soil from plants

NPP is the net flux of C from the atmosphere into green plants per unit time. For a site under steady-state conditions, NPP is given by the annual inputs of organic matter to the soil plus the annual removal of organic matter from the site, if any (Jenkinson et al. 1999b). Thus for ungrazed grasslands or undisturbed forests, the

NPP is equal to the annual input of C (Parshotam and Hewitt, 1995). The annual input of C to soil includes above ground and below ground components.

Above ground components consist of litterfall, senescence and branchfall. In agroecosystems, above ground C inputs can be divided in to straw, stubble and surface debris (Bolinder et al. 1997). Surface debris is not important for cereal crops, but is important for perennial forages and natural grasslands. By contrast, in early forest successions, up to two-thirds of C input to soil is leaf fall early in the succession (Jenkinson et al. 1992).

The below ground components of the annual input of C include root growth, dead mycorrhizal fungi, dead animals, organic matter fixed by soil autotrophs (e.g. algae), root exudation and rhizodeposition (Jenkinson and Coleman, 1994; Jenkinson et al. 1999b). Below ground C inputs in agroecosystems include root biomass left in soil after harvest and the extra-root C produced during the growth season (root turnover, exudates and secretions; Bolinder et al. 1997). Below ground residue production for annual crops is typically between 20-50% of total dry matter production (Paustian et al. 1997a; Jenkinson et al. 1992). Although inputs of C to soil may also include external inputs such as organic manures (Paustian et al. 1997a), these components of C input are usually easily quantified and well known. This chapter is thus primarily concerned with the return of plant C to the soil, which is much harder to measure (Jenkinson et al. 1992).

Input of C to soils varies temporally in arable, grassland and forest ecosystems (Paul et al. 1995). Plant inputs of C to the soil are dependent on phenological, climatic, soil and other factors, which influence the input quantity, quality (C:N ratio and lignin content), and its temporal distribution. Important parameters include: plant related factors (species, growing season and age; Breymeyer et al. 1996), climatic factors (rainfall, temperature and soil moisture; Breymeyer et al. 1996), soil physical properties (Parshotam et al. 1995), soil microbial composition, biota and activity (O'Brien, 1984), atmospheric gases and deposition, and fertilisation (Jansen, 1990). Table 4.1 shows the mean annual input of C to soil to 1m depth for the world's major ecosystems their area and SOC stock, calculated from Post et al. (1982) and Jenkinson et al. (1991). This table demonstrates the variability of plant inputs of C between different ecosystem types. For example, deserts have the lowest C inputs since they have a low NPP, whilst tropical forests have the highest C inputs and largest NPP (C input up to 10 t C ha^{-1} y^{-1}; Batjes and Sombroek, 1997). Annual C input to soils from most cereals is typically 1-2 t C ha^{-1} y^{-1}, with grasslands receiving perhaps twice as much (Jenkinson and Rayner, 1977; Paustian et al. 1990).

In agroecosystems, Jenkinson et al. (1992) used RothC to estimate C inputs at Rothamsted, UK of 0.15 t C ha^{-1} y^{-1} for bare soil, 0.2 t C ha^{-1} y^{-1} for unfertilised spring barley, and from 1.9 to 2.95 t C ha^{-1} y^{-1} for an inorganic fertilised all-arable rotation. Paustian et al. (1990) used a C and N budget approach at Kjettslinge, Sweden to estimate annual carbon inputs to soil from barley and leys (1.5-1.8 t C ha^{-1} y^{-1} and 2.6-2.7 t C ha^{-1} y^{-1}, respectively). For predictive SOM modelling studies, it is important to consider the magnitude of the C input flux over the year, the input quality, and its distribution throughout the year (Jenkinson et al. 1992).

Table 4.1 SOC stocks and carbon inputs (to 1m depth) for the Worlds major life zones (after Jenkinson et al. 1991 and Post et al, 1982)

Life Zone	Area (10^{12} ha)	SOC stock (10^{15} g)	SOC stock (t ha^{-1})	Carbon input (t ha^{-1} y^{-1})
Tundra	8.8	191.8	21.80	0.10
Boreal desert	2	20.4	10.20	0.05
Cool desert	4.2	41.6	9.90	0.21
Warm desert	14	19.6	1.40	0.04
Tropical desert bush	1.2	2.4	2.00	0.08
Cool temperate steppe	9	119.7	13.30	0.30
Temperate thorn steppe	3.9	29.6	7.59	0.46
Tropical woodland and savannah	24	129.6	5.40	0.48
Boreal forest moist	4.2	48.7	11.60	0.19
Boreal forest wet	6.9	133.2	19.30	0.68
Temperate forest cool	3.4	43.2	12.71	0.91
Temperate forest warm	8.6	61.1	7.10	0.83
Tropical forest very dry	3.6	22.0	6.11	0.47
Tropical forest dry	2.4	23.8	9.92	0.46
Tropical forest moist	5.3	60.4	11.40	2.49
Tropical forest wet	4.1	78.3	19.10	3.73
Cultivated land	21.2	167.5	7.90	0.48
Wetlands	2.8	202.4	72.29	-
TOTAL	129.6	1395.3	10.77	0.58

4.1.2 Measurements of plant inputs of C

4.1.2.1 Direct measurements

Direct measurements of some components of the annual input of C to soil are physically measurable. For example, it is possible to measure the dry matter production of a crop, grassland or forest by destructively measuring changes in above ground biomass, and to make litterbag measurements to approximate litterfall (e.g.

Paustian et al. 1990). Then (aboveground) litter input to soil can be calculated as total above ground primary production minus the sum of accumulated above ground matter (i.e. harvested biomass, above ground consumption and net annual change in litter and plant standing crops; Paustian et al. 1990).

In order to derive the *total* annual C input, it is also necessary to calculate or estimate the below ground component. Belowground C inputs to soil have been estimated using root core samples (e.g. Barraclough et al. 1988; Paustian et al. 1990), growth chamber experiments using ^{14}C-labelled CO_2 (e.g. Merckx et al. 1987; Liljeroth et al. 1990; Paustian et al. 1990), or simple relationships between dry matter production and the total C input (e.g. Campbell et al. 1991), or between dry matter production and the C input from crop roots as rhizo-deposition (20-50% of dry matter production: O'Brien and Stout, 1978; Jenkinson, 1990). Root C inputs in forests may be a greater proportion of total C inputs to soil than in arable soils, at around 65% of the total C input (Tate et al. 1993). Although direct measurement approaches can provide good estimates of C inputs to soil at the site scale, they are generally too data intensive for large scale spatial modelling studies.

4.1.2.2 Indirect measurements

As discussed in Chapter 1, a change in vegetation from C_3 to C_4 plant cover allows the use of changes in ^{13}C natural abundance for determining the relative contribution and input of C to SOM from the C_3 and C_4 vegetation (e.g. Skjemstad et al. 1990; Lefroy et al. 1993). However, this technique cannot be used to determine carbon inputs to soil where there has not been a significant vegetation change from C_3 to C_4 crops, and so it limited in its scope for studies on a range of land use and management changes.

Harvest index (HI) can also be used to calculate the total above ground biomass produced and estimate the C inputs from straw and stubble. HI is defined as the ratio of the yield of grain to the biological yield excluding roots (Donald and Hamblin, 1976). HI may be used to determine the amount of straw removed and straw left in soil (which can be converted to C using dry matter and C content), thus yielding C inputs from straw. C inputs from roots are usually estimated using the shoot to root ratio (SR) at peak standing crop. The total C input to soil is then assumed to equal the sum of C inputs from roots and straw.

Examples of HI for some Canadian crops are wheat: 37% to 49%, oats: 41%; barley: 56%, triticale: 34%, and of SR wheat: 4.9 to 7.0, oats: 2.5, barley: 2.0, and triticale: 5.4 (Campbell et al. 1991). Extra-root C (i.e. that from root turnover, exudates and secretions) is difficult to determine under field conditions and can represent 50% or more of the C allocated belowground. However, extra-root C can be estimated using ^{14}C labelling studies. There may also be more root allocation in droughty sites (Jenkinson et al. 1992). Since HI and SR vary between sites and C inputs can vary as a function of species, variety, management and climate, accurate estimation is often difficult, especially across large spatial areas.

4.1.3 Models for NPP and plant C inputs

A comprehensive review and inter-comparison of NPP models is given by Cramer et al. (1995). Their model classification is used here, with the inclusion of one extra category for soil process models. Table 4.2 gives an overview of the models reviewed by Cramer et al (1995), with the addition of RothC.

4.1.3.1 Soil process models

Models that simulate SOM decomposition such as RothC have been used to determine the annual input of carbon to soil using data on management, climate, SOC and (where available) radiocarbon measurements. This is done by fitting the modelled SOC values to measured SOC values, whilst iteratively changing the annual input of C. For systems under steady-state conditions, the annual input approximates the NPP. For example, RothC has been used to give reasonable estimates of NPP at field, regional and global scales (e.g. Jenkinson et al. 1991, Jenkinson et al. 1994). Tate et al. (1995) used RothC to derive C inputs in New Zealand grasslands and forests, and estimated higher C inputs in forests than grasslands. Parshotam et al. (1995) estimated C inputs in New Zealand grasslands and forests in at the national scale using RothC. Other authors have used models such as RothC, making arbitrary estimates of C inputs (Wu et al. 1998).

Table 4.2 Selected models for NPP and plant inputs of C (after Cramer et al. 1995)

Model	Temporal resolution	NPP influenced by	No. of vegetation C pools	Reference
Soil Process models				
RothC	1 month	Temp, Soil C, Soil Clay, AET, rainfall	0	Coleman and Jenkinson (1996)
Models for seasonal biogeochemical fluxes				
HRBM 3.0	1 month	NPP=f((Temp, Prec, AET/PET, CO_2, Fert)	0	Esser et al. (1993)
CENTURY 4.0	1 momth	NPP=f(VegC, Dead, Temp, SW, Prec, PET, N, P, S)	8	Metherell et al. (1993)
TEM 4.0	1 month	GPP=f(Srad, Kleaf, Temp, AET/PET, CO_2, N) R_A=f(VegC, GPP, Temp)	1	McGuire et al. (1995)
CARAIB 2.1	1 day	GPP=f(Srad, LAI, Temp, SW, VPD, CO_2, O_2) R_A=f(VegC, LAI, Temp)	2	Nemry et al. (1996)
FBM 2.2	1 day	GPP=f(Srad, LAI, Temp, SW, CO_3) R_A=f(VegC, T)	2	Kindermann et al. (1993)
PLAI 0.2	1 day	GPP=f(Srad, LAI, Temp, SW, CO_3) R_A=f(VegC, Temp)	2	Pochl and Cramer (1995)
SILVAN 2.2	6 days	GPP=f(Srad, LAI, Temp, AET/PET, CO_3) R_A=f(VegC, Temp)	3	Kaduk and Heimann (1996)
BIOME-BGC	1 day	GPP=f(Srad, LAI, Temp, SW, VPD, CO_3, LeafN) R_A=f(VegC, Temp)	4	Running and Hunt (1993)
KGBM	1 day	GPP=f(Srad, LAI, Temp, SW, VPD) R_A=f(GPP)	1	Kergoat (2001)
Models of process and pattern (function and structure)				
BIOME3	1 month	GPP=f(Srad, LAI, Temp, AET/PET, CO_3) R_A=f(LAI, GPP)	0	Haxeltine et al. (1996)
DOLY	1 year	GPP=f(Srad, LAI, Temp, SW, VPD, CO_3, SoilCandN) R_A=f(VegC, T, SoilCandN)	2	Woodward et al. (1995)
HYBRID 3.0	1 day	GPP=f(Srad, FPAR, Temp, SW, CO_3, N) R_A=f(VegC, Temp, VegN)	4	Friend et al. (1997)
Satellite-based models				
CASA	1 month	NPP=f(Srad, fPAR, Temp, AET/PET)	0	Field et al. (1995)
GLO-PEM	10 days	GPP=f(Srad, fPAR, Temp, SW, VPD) R_A=f(VegC, GPP)	2	Prince and Goward (1995)
SDBM		NPP=f(Srad, FPAR, CO_2)	0	Knorr and Heimann (1995)
TURC	1 month	GPP=f(Srad, FPAR) R_A=f(VegC, Temp)	3	Ruimy et al. (1996)
SIB2	12 minutes	GPP=f(Srad, FPAR, LAI, Temp, SW, VPD, CO_3) R_A=f(GPP, Temp, SW)	2	Sellers et al. (1996)

Key: GPP – Gross Primary Production; Srad – Solar radiation; LAI – Leaf Area Index; Temp – Temperature; AET/PET – Actual Evapotranspiration or Potential Evapotranspiration; CO_2 – Carbon dioxide concentration; R_A – autotrophic respiration; SW – Soil Water Content; VPD – Vapour Pressure Deficit; LeafN – Leaf N content; VegC – Vegetation C content; FPAR – Fraction of Photosynthetically Active Light; Dead – Dead Biomass; Prec – Precipitation; N – Nitrogen availability; P – Phosphorus availability; S – Sulphur availability; SoilC - Soil Carbon; SoilN – Soil Nitrogen; VegN – Vegetation Nitrogen Content.

4.1.3.2 Models for seasonal biogeochemical fluxes

These models simulate biogeochemical fluxes on the basis of soil and climate characteristics only, and by using either spatially explicit vegetation datasets or biogeography models to determine potential vegetation distribution. These models simply describe functional changes *within* particular vegetation types. Some biogeochemical models are capable of using satellite data, but only in calibration.

4.1.3.3 Models of process and pattern (function and structure)

These models simulate both changes in ecosystem structure (vegetation distribution and phenology) and ecosystem function (biogeochemistry). The determination of the vegetation types by these models uses rules of process optimisation (maximisation of the NPP according to soils and climate, or maximisation of the leaf area index (LAI) to satisfy the annual moisture and carbon balances).

4.1.3.4 Satellite-based models

These models require satellite data to determine the temporal behaviour of the photosynthetically active component of the canopy. These models can be used to simulate the effect of climate variability on NPP, but the simulation periods for this class of models is restricted to that of the satellite archive (Cramer et al. 1995). Satellite-based models make use of global datasets like the NOAA/AVHRR. These models use the linkage between the Normalised Difference Vegetation Index (NDVI) and the fraction of absorbed photosynthetically active radiation (FPAR; Goward and Huemmrich, 1992; Kumar and Monteith, 1981; Sellers, 1985; Sellers, 1987) to predict the absorbed photosynthetically active radiation (APAR), giving biological productivity (Maisongrande et al. 1995). Production Efficiency Models (PEM) use the concept of light use efficiency (LUE) to convert APAR to biomass, since production is closely related to PAR absorption (Monteith, 1977; Landsberg, 1986).

4.1.4 Introduction to modelling exercises

In order to predict changes in SOC stocks over large areas, a model approach is needed that is capable of a) predicting plant returns of C dynamically under changes in land use and management, and b) of running from the available data at the regional scale. With the exception of RothC and CENTURY, most NPP models (e.g. Table 4.2; see Cramer et al. 1995) were developed for natural ecosystems, and for use at the regional scale or above. Experimental approaches to estimating C inputs to soil are generally limited to intensive site scale studies. As this work aims to predict changes regional SOC stocks under changes in agricultural land use and management, natural ecosystem models for NPP are not appropriate, and would require substantial adjustment for this study. For RothC to be used in predictive studies of the impacts of changes in land use and agricultural practice, a method to estimate plant inputs is needed. Here, a method to estimate plant inputs of C at equilibrium for three major land use types is described, using 60 long-term datasets and the RothC model (Falloon et al. 1998b). The sensitivity of SOC predictions with RothC to the magnitude and quality of annual C inputs is investigated (Falloon et al. 2000) for the Geescroft Wilderness Experiment (Poulton, 1995a). The results are applied to a simple land management change scenario for the stock of SOC in European arable land, to investigate the effect of errors in C inputs on SOC predictions at the regional scale.

4.2 ESTIMATING PLANT INPUTS OF C FOR ROTHC

4.2.1 Methods

As an initial method to estimate the plant input of C to the soil for RothC, 32 sites from SOMNET were modelled to equilibrium using RothC. The data used in Chapter 3.2.1 (Falloon et al. 1998a) were also used, bringing the total number of sites to 60. Soil, land management and weather data were collected from SOMNET dataholders and converted to RothC input format. Default IOM values were estimated using the regression described in Equations 1 and 2, Chapter 3.2.2 (Falloon et al. 1998a) unless radiocarbon data were available to set IOM. The model was run iteratively to fit initial soil C levels in the 0-30 cm soil layer; runs were completed using IOM set using a) the regression (Chapter 3.2.2 equation 2), and b) plus (Chapter 3.2.2 equation 3) or minus (Chapter 3.2.2 equation 4) the standard error of the regression. These data were analysed in GENSTAT, using soil clay content, mean annual rainfall, mean annual temperature, land use type, pH as independent variables in single and multiple linear regressions against annual inputs of C.

4.2.2 Results and Discussion

The modelled data showed great variance between different climates, soil types, and land uses (Table 4.3). Runs plus or minus the standard error of the default IOM indicated that the IOM value used has a relatively small effect on the modelled annual plant input. The modelled data and results for the 60 sites is given in Table 4.3. Simple linear regressions between C inputs and soil environmental parameters are given in Table 4.4.

Soil clay content, FAO soil code and soil pH showed significant correlation with annual C inputs, and mean annual precipitation, SOC and IOM showed highly significant correlation with annual C inputs. Since C inputs had been predicted from SOC and IOM values, SOC and IOM could not be further used in C input estimation. A multiple linear regression including the remaining parameters showing significant correlation with annual C input (mean annual precipitation, FAO soil code, soil clay content and soil pH) was highly significant ($p<0.001$), but only accounted for 28.2% of the variance in annual C inputs. Since the project aim was to model C sequestration under changes in land use and management, this required estimates of annual C inputs under different land uses. The mean plant inputs for different land uses were as follows: 3.55 t C $ha^{-1}y^{-1}$ for arable, 3.72 t C $ha^{-1}y^{-1}$ for grasslands, and 7.09 t C $ha^{-1}y^{-1}$ for forestry (Falloon et al. 1998b). These figures are similar to those suggested in the introduction and literature review, which suggested C inputs of around 1-2 t C $ha^{-1}y^{-1}$ for arable, 2-4 t C $ha^{-1}y^{-1}$ for grasslands (Jenkinson and Rayner, 1977; Paustian et al. 1990), and up to 10 t C $ha^{-1}y^{-1}$ for forestry (Batjes and Sombroek, 1997). The figures presented in Table 4.1 are generally somewhat lower than both of these sets of estimates, perhaps because they were averaged over such large, heterogeneous unmanaged spatial areas for life zones. The estimates from the literature review and our modelling exercise were, by contrast, from relatively homogenous and often managed field-scale sites.

It is concluded that a more precise method of estimating C inputs to soil should be used for future studies. This could include the use of a dynamic NPP model to predict C inputs to soil, or the use of a plant production model from another SOM model, such as CENTURY.

4.3 SENSITIVITY OF ROTHC TO ANNUAL INPUTS OF C

4.3.1 Methods

4.3.1.1 Sensitivity of equilibrium SOC values predicted with RothC to the quantity of annual C inputs

In order to investigate the sensitivity of equilibrium SOC values predicted with RothC to the quantity of annual C inputs, soil, weather and land management data for the Rothamsted Geescroft Wilderness Experiment were collected. All model inputs were identical to those used by the model developers in their simulation of the data (Coleman et al. 1997) for their equilibrium model run, with the exception of the amount of annual C input. The annual input was not distributed throughout the year as in Coleman et al. (1997), but was added in one month, for simplicity. For this investigation, the annual C input was set at 0%, 10%, 25%, 50%, 75%, 100%, 125%, 150%, 200% and 300% of the value of 1.25 t C ha^{-1} y^{-1} used by Coleman et al. (1997), and the model was run to equilibrium. RothC-simulated total SOC values from the different runs were then studied.

4.3.1.2 Sensitivity of SOC values predicted with RothC to the quantity of annual C inputs during the experimental timescale

The Rothamsted Geescroft Wilderness Experiment was also used to study the sensitivity of SOC values predicted with RothC to the quantity of annual C inputs over the experimental timescale. Model inputs were also identical to those used by Coleman et al. (1997), except that the size of the annual C input was varied during the experimental period, after the equilibrium run.

After an equilibrium run using the same C input amount as Coleman et al. (1997), the model was run with annual C inputs during the experimental term set at 0%, 10%, 25%, 50%, 75%, 90%, 95%, 99%, 100%, 101%, 105%, 110%, 125%, 150%, 200% and 300% of the value of 2.85 t C ha^{-1} y^{-1} used by Coleman et al. (1997). The annual C input was not distributed throughout the year as in Coleman et al. (1997), but was added in one month for simplicity. The total SOC values predicted with RothC from these runs were studied.

Table 4.3 Plant inputs to soil at equilibrium, modelled using RothC-26.3

Site	Clay content (%)	Equilibrium land use	SOC to 30 cm (t ha^{-1})	Crop type	Current land use	IOM (t C ha^{-1})	IOM+SD (t C ha^{-1})	IOM-SD (t C ha^{-1})	Total annual precipitation (mm)	Mean annual temperature (degrees C)	pH	Soil type	C inputs (t ha^{-1} y^{-1})	C inputs+sd (t ha^{-1} y^{-1})	C inputs-sd (t ha^{-1} y^{-1})
Askov long term experiments on animal manure and mineral fertilisers -Sandmarken	4	arable	38.57	arable rotation	arable	3.08	10.44	4.58	862	7.7	5.8	Cambisols	2.96	2.33	3.15
Broadbalk Wheat Experiment	25	arable	30.40	wheat	arable	3.80			693	9.1	7.5	Luvisols	1.45		
Calhoun Experimental Forest	15.4	arable	8.89	lobololly pine	forestry	0.77			1170	17	4.3	Arenosols	0.58		
Effect of tillage system on growth, maturity, and yield of corn and soybean	32	prairie	74.49	arable rotation	arable	5.69	18.27	0.42	919	9.13	6.4	Gleysols	7.08	5.70	7.36
Essai permanent	13.5	arable	33.41	arable rotation	arable	2.61	8.65	1.18	767	9.1	6.75	Cambisols	2.63	2.12	2.79
Geescroft Wilderness	21	arable	21.00	woodland	forestry	2.50			734.2	9.1	5.6	Luvisols	1.00		
Hungary long term fertiliser experimental network -Bicserd	33	arable	27.29	arable rotation	arable	2.20	7.52	0.59	647	10.4	5.6	Cambisols	3.18	2.55	3.37
Hungary long term fertiliser experimental network-Iregszemcse	22	arable	43.48	arable rotation	arable	3.73	13.80	1.39	614	10.2	7.4	Cambisols	5.41	4.09	5.83
Hungary long term fertiliser experimental network -Karcag	37	arable	45.10	arable rotation	arable	3.89	14.47	0.86	476	10.4	4.7	Regosols	4.99	3.67	5.35
Hungary long term fertiliser experimental network-Keszthely	26	forest	27.58	arable rotation	arable	2.22	7.62	0.75	623	10	5.9	Cambisols	2.96	2.36	3.17
Hungary long term fertiliser experimental network -Nagyhorcsok	23	arable	65.28	arable rotation	arable	5.93	23.45	1.97	548	9.9	7.2	Cambisols	7.54	5.32	7.93
Hungary long term fertiliser experimental network -Putnok	28	arable	43.59	arable rotation	arable	3.74	13.84	1.09	564	8.7	3.9	Cambisols	5.01	3.74	5.40
L28, Effect of N Fertilisation by different organic fertilization	9	arable	30.49	arable rotation	arable	2.36	7.68	1.62	600	7.7	6	Arenosols	2.91	2.36	3.08
Lancaster rotation, Winsonsin	17	arable	53.69	continuous corn	arable	4.49	16.06	1.49	845	7.8	6.8	Luvisols	4.44	3.38	4.76
Land Use Expt. (D111, Berlin-Dahlem)	4	arable	17.38	arable rotation	arable	1.20	3.45	1.69	547	9.2	5.5	Luvisols	2.00	1.71	2.08
Legume/cereal rotation-red soil	44	arable	42.89	arable rotation	arable	3.34	10.98	0.35	676	17.5	6.9	Vertisols	1.35	1.10	1.43
Long term slurry on grassland	25.4	grassland	133.98	grasland	grassland	12.22	48.50	1.83	872.5	9.08	6	Fluvisols	7.97	5.60	8.56
Long term soil fertility expt., Orja	23	arable	46.51	arable rotation	arable	3.81	13.32	0.96	730	7.9	7	Cambisols	3.37	2.61	3.58
Long term effect of farmyard manure and N on yield of pearl-millet-wheat cropping sequence	22	grassland (native shrubs)	15.13	arable rotation	arable	1.02	2.82	0.25	426	23.8	8.2	Cambisols	1.44	1.26	1.52
Long term static fertiliser experiments in Skierniewice	6	arable	13.35	arable rotation and monocultures	arable	0.95	2.80	1.35	518.9	7.7	6	Luvisols	1.44	1.22	1.52
Morrow plots	25	grassland	71.70	arable rotation and monocultures	arable	6.85			922	11.1	6	Gleysols	6.06		
Muencheberg long term fertiliser experiment V140/00	5	arable	21.62	arable rotation	arable	1.59	4.91	2.08	435	8.5	5.85	Luvisols	2.16	1.79	2.27
Nutrient deficiency experiment (DIV, Thyrow)	3	arable	13.85	arable rotation	arable	1.01	3.10	3.34	514	8.7	5	Podzols	1.45	1.20	1.52

Site	Clay content (%)	Equilibrium land use	SOC to 30 cm (t ha^{-1})	Crop type	Current land use	IOM (t C ha^{-1})	IOM+SD (t C ha^{-1})	IOM-SD (t C ha^{-1})	Total annual precipitation (mm)	Mean annual temperature (degrees C)	pH	Soil type	C inputs (t ha^{-1} y^{-1})	C inputs+sd (t ha^{-1} y^{-1})	C inputs-sd (t ha^{-1} y^{-1})
Old permanent manurial trial	23	arable	3.89	arable rotation	arable	0.25	0.67	0.19	56.2	26.9	8	Cambisols	0.63	0.56	1.01
Palace Leas Meadow hay plots	25.4	grassland	87.59	grassland	grassland	8.17	33.31	2.15	650	9.3	5	Gleysols	5.34	3.64	5.76
Park Grass	23	grassland	92.98	grassland	grassland	8.55			728.4	9.1	5.1	Luvisols	3.75		
Pendleton residue management	18	arable	37.79	arable rotation	arable	3.18	11.49	1.48	415	10.2	6.1	Cambisols	1.70	1.30	1.82
Praha-Ruzyne	28	arable	34.98	arable rotation	arable	2.79	9.46	0.65	464	7.98	6.95	Luvisols	2.48	1.96	2.63
SATWAGL	29	grassland	46.46	arable rotation	arable	3.46	10.77	0.39	529	15.7	4.93	Luvisols	4.35	3.63	4.62
Sanborn Field	20	grassland	43.60	arable rotation	arable	3.54	12.24	1.03	916	12	5.1	Luvisols	5.06	3.94	5.28
Statischer Duengungsversuch Bad Lauchstadt	21	arable	43.01	arable rotation	arable	3.69	13.61	1.44	483	8.7	6.6	Phaeozems	1.66	1.23	1.76
Tillage systems/cropping sequences trial c1	60	arable	35.96	arable rotation	arable	3.01	10.77	0.42	616	13.76	7.15	Vertisols	3.80	2.93	4.06
Triple Cereal Cropping and soil organic matter long term experiment	62.5	arable	44.53	arable rotation	arable	3.55	11.98	0.29	1500	16	5.4	Gleysols	8.85	7.12	9.65
Waite permanent rotation trial	18	grassland	82.23	arable roation	arable	6.62	22.68	1.08	604	16.8	6.2	Luvisols	6.44	5.08	6.79
Woburn ley-arable	63	arable	44.53	arable rotation	arable	3.74	13.47	0.42	647.1	9.5	7	Arenosols	2.07	1.56	2.21
Woburn market garden	9.4	arable	26.84	arable/horticultural rotation	arable/hort.	2.08	6.79	1.57	647.1	9.5	6.5	Cambisols	1.63	1.32	1.72
Woodslee tile runoff rotation experiment	37	forest	233.82	arable rotation	arable	21.78	88.62	1.46	876	8.7	6.1	Gleysols	14.03	9.64	15.16
South Road Tillage Gleysol	15	arable	80.50	arable rotation	arable	7.02	26.38	2.26	866	8.3	6.5	Gleysols	5.30	3.88	5.66
RES drain guage	23	arable	38.00	arable	arable	6.30			717	9.1	7.9	Luvisols	1.73		
Woburn sp. barley	10.3	arable	47.80	arable	arable	3.50			620	9.3	5.6	Arenosols	2.44		
Gleadthorpe EHF	6	arable	44.28	arable	arable	11.50			1642	8.8	6.1	Arenosols	3.39		
Rosemaund EHF	26	arable	50.45	arable	arable	3.37			679	9.1	7.8	Luvisols	3.79		
Bridgets EHF	20.5	arable	31.43	arable	arable	6.13			815.1	9.5	7.8	Cambisols	5.52		
ICARDA wheat/wheat rotation	62.5	arable	18.98	arable	arable	3.45			324	18	8	Cambisols	1.08		
Nairobi National Park	44.5	savannah	48.30	savannah	savannah	4.45			834	20	6.9	Vertisols	4.46		
Machange	24	savannah	24.73	savannah	savannah	2.42			795	23	6.47	Cambisols	1.79		
Mutuobare	18	savannah	28.98	savannah	savannah	1.38			855	25	6.54	Cambisols	2.62		
Lower Tano Basin	45	Secondary rainforest	72.07	Secondary rainforest	Secondary rainforest	4.75			1496	26.1	4.4	Ferralsols	9.60		

Site	Clay content (%)	Equilibrium land use	SOC to 30 cm ($t\ ha^{-1}$)	Crop type	Current land use	IOM ($t\ C\ ha^{-1}$)	IOM+SD ($t\ C\ ha^{-1}$)	IOM-SD ($t\ C\ ha^{-1}$)	Total annual precipitation (mm)	Mean annual temperature (degrees C)	pH	Soil type	C inputs ($t\ ha^{-1}\ y^{-1}$)	C inputs+sd ($t\ ha^{-1}\ y^{-1}$)	C inputs-sd ($t\ ha^{-1}\ y^{-1}$)
Ho-Keta Plains	7	savannah	20.00	savannah	savannah	0.70			1110	27	6.4	Calcisols	2.00		
Station Creek	28	forest	88.17	forest	forest	4.03			1520	9.8	4.6	Podzols	5.11		
Conroy	12	grassland	27.00	grassland	grassland	0.30			350	8.5	5.85	Leptosols	3.40		
Tima	16	grassland	45.00	grassland	grassland	6.10			630	8.5	6	Luvisols	0.90		
Tawhiti	19	grassland	75.00	grassland	grassland	10.00			1100	5.5	4.7	Cambisols	2.50		
Carrick	11	grassland	73.00	grassland	grassland	12.00			1230	3.5	4.6	Podzols	1.80		
Obelisk	5	grassland	164.00	grassland	grassland	29.00			1570	2	4.4	Podzols	3.80		
Kaikoura	24	grassland	98.00	grassland	grassland	9.90			1300	5.5	5.4	Cambisols	3.30		
Bealey	30	Forest	86.00	forest	forest	3.50			1300	6	5.5	Cambisols	2.80		
Maruia I	27	Forest	136.00	forest	forest	8.00			2000	9.8	4.13	Podzols	8.00		
Maruia II	26	Forest	151.00	forest	forest	18.00			2000	9.8	4.13	Podzols	8.20		
Puruki	12	Forest	148.00	forest	forest	15.00			15000	10.5	5.5	Podzols	8.90		

Table 4.4 Simple linear regressions between C inputs and soil and environmental parameters

Parameter	r^2	p	Significance	% variance accounted for
Soil clay %	0.86	0.022	Significant	7.2
FAO soil code	0.90	0.003	Significant	13.0
Mean annual precipitation (mm)	0.90	<0.001	Highly significant	18.0
Mean annual temperature (°C)	0.90	0.319	Not significant	0.0
Soil pH	0.98	0.031	Significant	6.2
SOC (t C ha^{-1})	0.78	<0.001	Highly significant	60.5
IOM (t C ha^{-1})	0.71	<0.001	Highly significant	28.7
Equilibrium land use code	0.88	0.066	Not significant	4.1

4.3.1.3 Sensitivity of equilibrium SOC values predicted with RothC to the quality of annual C inputs

In order to investigate the sensitivity of equilibrium SOC values predicted with RothC to the quality of annual C inputs, soil, weather and land management data for the Rothamsted Geescroft Wilderness Experiment were used. All model inputs were identical to those used by Coleman et al. (1997) for their equilibrium model run, with the exception of the DPM/RPM ratio, which determines the quality (or decomposability) of the annual C input to soil. The DPM/RPM ratio is usually set at 1.44 for arable crops and managed grasslands, 0.67 for natural grasslands and scrub, and 0.25 for forests and woodlands. Hence Coleman et al. (1997) used a value of 1.44 for their equilibrium model run for Geescroft Wilderness, as the site was originally arable land. For this investigation, the DPM/RPM ratio was set at arbitrary values between 0 and 2 (0.1, 0.2, 0.3, 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, 1.0, 1.1, 1.2, 1.3, 1.4, 1.5, 1.6, 1.7, 1.8, 1.9 and 2.0). The DPM/RPM ratio was also set using the three recommended values for arable lands, grasslands and forests (1.44, 0.67 and 0.25, respectively). The annual input was not distributed throughout the year as in Coleman et al. (1997), but was added in one month for simplicity. The model was run to equilibrium, and the total SOC values predicted with RothC from the different runs were studied.

4.3.1.4 Sensitivity of RothC-predicted SOC values to the quality of annual C inputs over the experimental timescale

The Rothamsted Geescroft Wilderness Experiment was also used to study the sensitivity of SOC values predicted with RothC to the quality of annual C inputs over the experimental timescale. Model inputs were also identical to those used by

Coleman et al. (1997), but the DPM/RPM ratio was varied during the experimental period, after the equilibrium run. In their simulation of Geescroft Wilderness, Coleman et al. (1997) used an equilibrium DPM/RPM value of 1.44 to simulate the arable crops present (in previous land use), DPM/RPM value of 1.44 until 1904, and a DPM/RPM ratio of 0.24 after 1904 (by which time trees were well established). After an equilibrium run using the same annual C input amount and DPM/RPM ratio as Coleman et al. (1997), RothC was run with the experimental term DPM/RPM ratio set as follows: a) 1.44 until 1904 and 0.25 thereafter; b) 1.44 until 1904 and 0.67 thereafter; c) 1.44 throughout the experiment; d) 0.67 throughout the experiment, and e) 0.25 throughout the experiment. A value of DPM/RPM=1.44 is usually used for arable crops and managed grassland, DPM/RPM=0.67 for unmanaged grassland and scrub, and DPM/RPM=0.25 for woodlands and forests. Therefore runs a) to e) represent a) arable crops/managed grasslands followed by forest, b) arable crops/managed grassland followed by unmanaged grassland/scrub, c) arable crops/managed grassland throughout, d) unmanaged grassland/scrub throughout, and e) forest/woodland throughout. The annual C input was not distributed throughout the year as in Coleman et al. (1997), but was added in one month for simplicity. The total SOC values predicted with RothC from the different runs were studied.

4.3.1.5 Sensitivity of RothC to C inputs for large spatial scale SOC predictions

The sensitivity of SOC values predicted with RothC to errors in C inputs at large spatial scales was investigated using a simple carbon sequestration scenario. To estimate the maximum error, the scenario used was that most effective in increasing the SOC stocks of European arable land (Smith et al. 1997b, 1998a,c), i.e. afforestation of 30% of surplus arable land. C inputs, weather and land management were based upon the Geescroft Wilderness Experiment (Jenkinson et al. 1991). The change in SOC stocks over 100 y was calculated using model runs with all model inputs identical to those used by Coleman et al. (1997), with the exception of the size of the annual C input. C inputs were set at 0%, 10%, 25%, 50%, 75%, 90%, 95%, 99%, 100%, 101%, 105%, 110%, 125%, 150%, 200% and 300% of their value of 2.85 t C ha^{-1} y^{-1}. The annual input was not distributed throughout the year as in Coleman et al. (1997), but was added in one month for simplicity. The proportionate change in SOC stocks from these values were then applied to the current SOC stock of European arable land (Smith et al. 1997b).

4.3.2 Results and Discussion

4.3.2.1 Sensitivity of equilibrium SOC values predicted with RothC to the quantity of annual C inputs

Table 4.5 and Figure 4.1 show the annual C input values used and results obtained for the model runs investigating the sensitivity of SOC values predicted with RothC to equilibrium annual C inputs. As can be seen from Figure 4.1, the model shows a linear response to annual C inputs, and errors in the estimate of the annual C input have a large effect on SOC predictions. An over- or underestimation of 25% of the annual C input results in an error of more than 20% in predicted SOC values; over- or underestimation of 50% in annual C input could result in an error of more than 44% in predicted SOC values. Table 4.5 also gives the values for the Sensitivity Index (SI; Ng and Loomis, 1984) for these model runs, calculated as:

$$SI = \frac{\%change \quad in \quad input \quad variable}{\%change \quad in \quad output \quad variable}$$

A large value of SI indicates that the model output variable is relatively insensitive to changes in the input variable; a small SI value indicates that the model output variable is sensitive to changes in the input variable. A SI value of 1 indicates that changes of 1% in the input value result in a change of 1% in the output variable. Negative SI values indicate that increasing the input value decreases the output value; positive SI values indicate that increasing the input value increases the output value. The SI ratio confirms that SOC predicted with RothC at equilibrium is highly sensitive to errors in C inputs.

4.3.2.2 Sensitivity of SOC values predicted with RothC to the quantity of annual C inputs during the experimental term

Table 4.6 and Figure 4.2 show the annual plant C input values used and results for the model runs investigating the sensitivity of SOC values predicted with RothC to annual C inputs during the experimental term. The runs using 90%, 95%, 99%, 101%, 105% and 110% of the C input used by Coleman et al. (1997) are not shown in Figure 4.2 for clarity.

Table 4.5 The effect of annual C input quantity on equilibrium SOC values predicted with RothC

Plant input (% of original value)[1]	0%	10%	25%	50%	75%	100%	125%	150%	200%	300%
Annual C input value ($t\ C\ ha^{-1}\ y^{-1}$)	0.0000	0.1250	0.3125	0.6250	0.9375	1.2500	1.5625	1.8750	2.5000	3.7500
Difference in predicted SOC value from that of Coleman et al. (1997) ($t\ C\ ha^{-1}$)	-22.47	-20.22	-16.85	-11.23	-5.62	0.00	5.62	11.23	22.47	44.93
% Difference in predicted SOC value from that of Coleman et al. (1997)	-89.99	-80.99	-67.49	-45.00	-22.50	0.00	22.49	44.99	89.98	179.96
Equilibrium predicted SOC ($t\ C\ ha^{-1}$)	2.5	4.75	8.12	13.73	19.35	24.97	30.58	36.20	47.43	69.90
Sensitivity index (SI)	1.11	1.11	1.11	1.11	1.11	-	1.11	1.11	1.11	1.11

Table 4.6 The effect of C input quantity on experimental SOC values predicted with RothC

Annual C input (% of original value)[1]	0%	10%	25%	50%	75%	90%	95%	99%
Annual C input value ($t\ C\ ha^{-1}\ y^{-1}$)	0.00	0.29	0.71	1.43	2.14	2.57	2.71	2.82
Difference in predicted SOC value from that of Coleman et al. (1997) ($t\ C\ ha^{-1}$)	-47.74	-42.97	-35.81	-23.87	-11.94	-4.78	-2.39	-0.47
% Difference in predicted SOC value from that of Coleman et al. (1997)	-81.50	-73.35	-61.12	-40.75	-20.37	-8.16	-4.08	-0.81
Final predicted SOC, 1995 ($t\ C\ ha^{-1}$)	10.84	15.61	22.77	34.71	46.65	53.80	56.19	58.11
RMSE	73.37	65.84	54.55	35.84	17.58	8.02	6.63	6.77
SI	1.23	1.23	1.23	1.23	1.23	1.22	1.22	1.24
Annual C input (% of original value)[1]	**100%**	**101%**	**105%**	**110%**	**125%**	**150%**	**200%**	**300%**
Annual C input value ($t\ C\ ha^{-1}\ y^{-1}$)	2.85	2.88	2.99	3.14	3.56	4.28	5.70	8.55
Difference in predicted SOC value from that of Coleman et al. (1997) ($t\ C\ ha^{-1}$)	0.00	0.48	2.39	4.78	11.94	23.87	47.74	95.49
% Difference in predicted SOC value from that of Coleman et al. (1997)	0.00	0.81	4.07	8.15	20.37	40.75	81.50	163.00
Final predicted SOC, 1995 ($t\ C\ ha^{-1}$)	58.58	59.06	60.97	63.36	70.52	82.46	106.33	154.07
RMSE	7.02	7.33	9.12	12.07	22.45	40.95	78.54	154.08
SI	-	1.23	1.23	1.23	1.23	1.23	1.23	1.23

Figure 4.1 Sensitivity of SOC values predicted with RothC at equilibrium to annual inputs of C

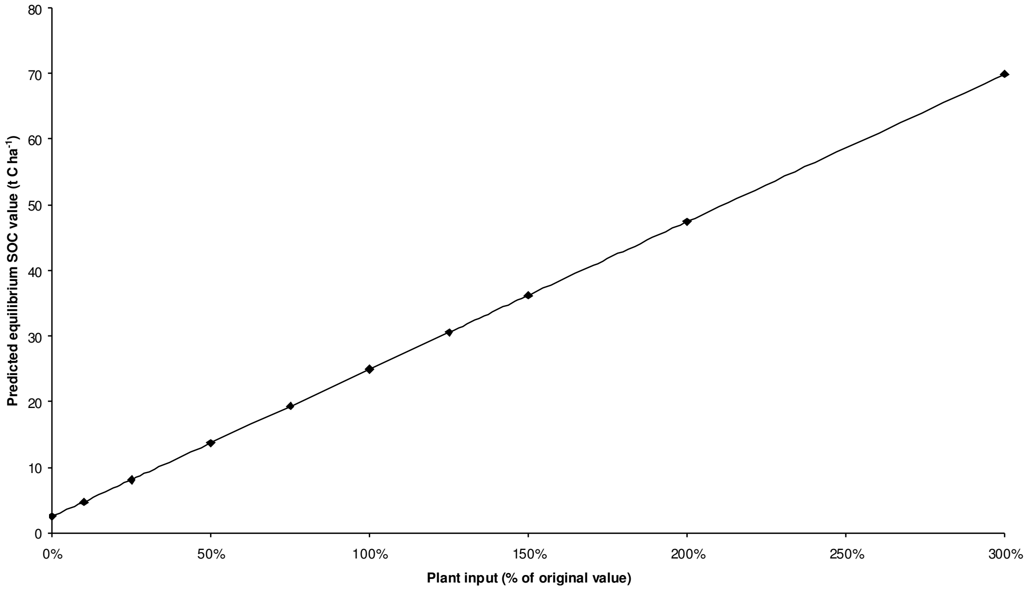


Figure 4.2 Sensitivity of SOC values predicted with RothC to the magnitude of annual C inputs for the Geescroft Wilderness Experiment

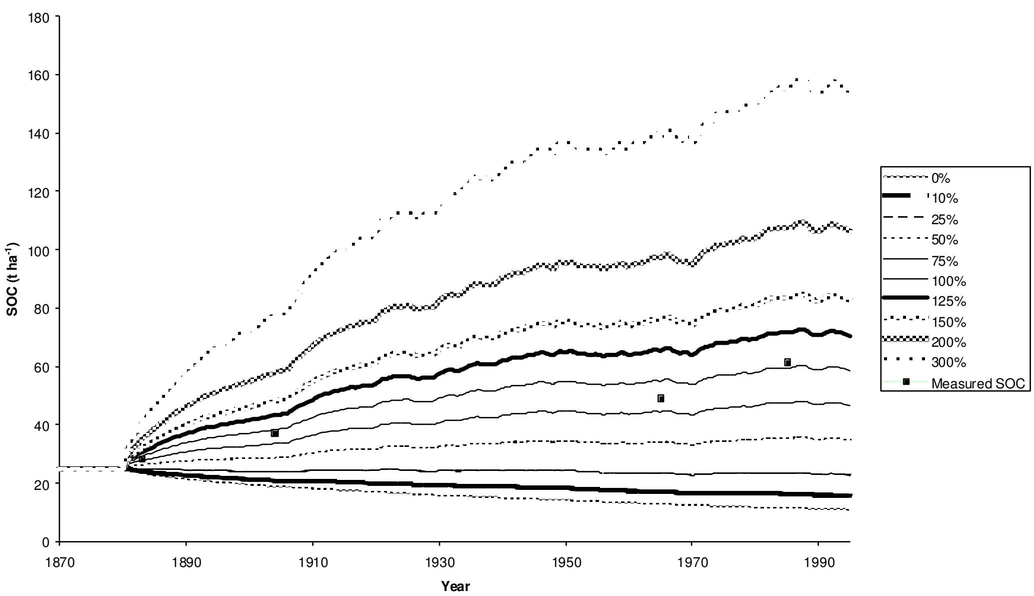


Figure 4.3 Sensitivity of SOC values predicted with RothC at equilibrium to C input quality

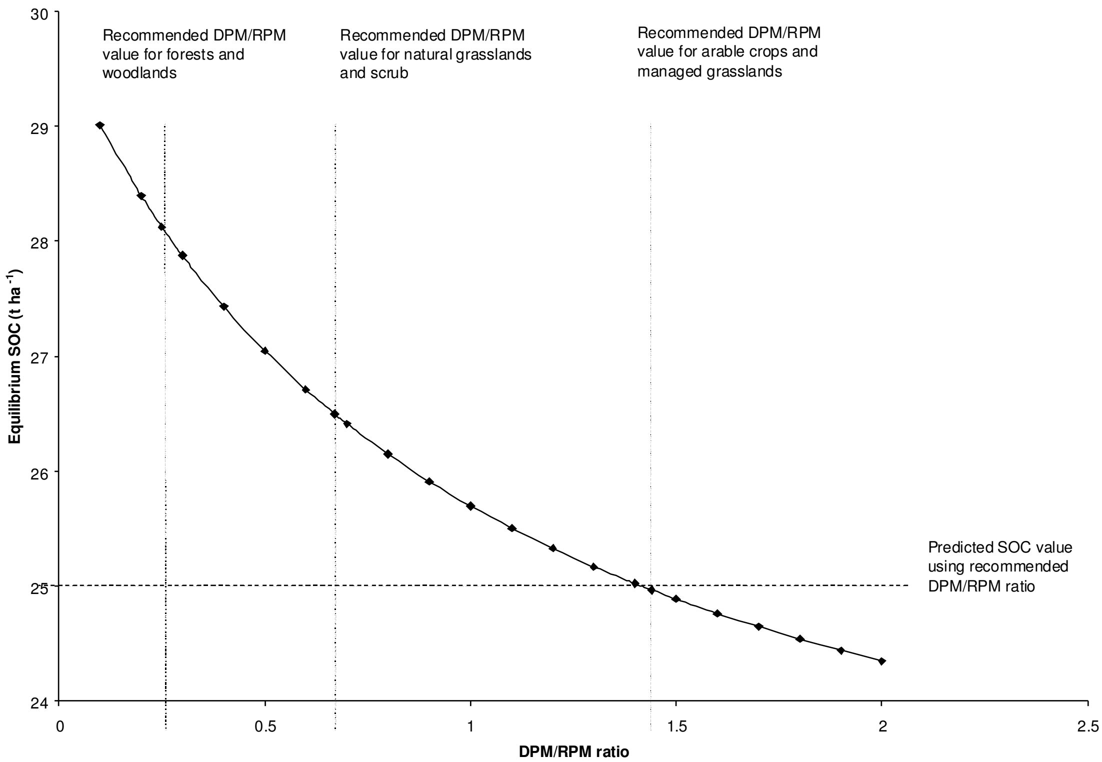
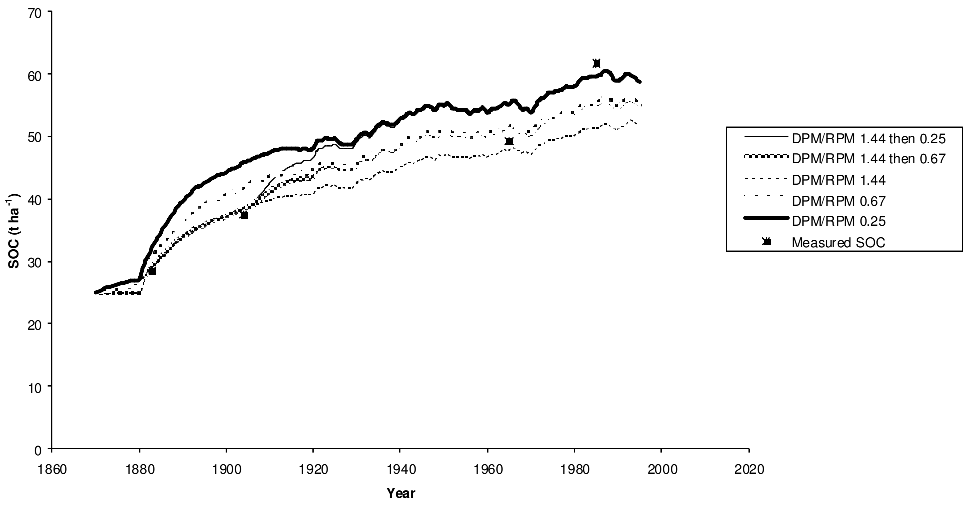


Figure 4.4 Sensitivity of RothC predicted SOC values to C input quality for the Geescroft Wilderness Experiment



As can be seen from Table 4.6 and Figure 4.2, SOC values predicted with RothC are highly sensitive to the size of the annual C input in the experimental term. Whilst the value used by Coleman et al. (1997) gives a reasonable fit to the measured data (RMSE=7.02), the model fit is poor for an error greater than plus or minus 10% in annual C inputs. This is confirmed by the SI of 1.23, which also shows that SOC predicted with RothC in the experimental term is slightly less sensitive to errors in C inputs than at equilibrium.

4.3.2.3 Sensitivity of equilibrium SOC values predicted with RothC to the quality of annual C inputs

Table 4.7 and Fig. 4.3 show the effect of C input quality on SOC values predicted with RothC at equilibrium. As can be seen from Fig. 4.3 and Table 4.7, SOC values predicted with RothC are more sensitive to differences in the DPM/RPM ratio at smaller values of the DPM/RPM ratio, i.e. where the C input has a higher proportion of resistant material. Therefore errors in the DPM/RPM ratio for forests will cause greater differences in SOC predictions than will those for grasslands or arable lands. This is confirmed by the SR values in Table 4.7, which also show that RothC-predicted SOC values are less sensitive to changes in C input quality than C input quantity.

Table 4.7 The effect of C input quality on equilibrium SOC predictions with RothC

DPM/RPM ratio	0.1	0.2	0.25	0.3	0.4	0.5	0.6	0.67	0.7	0.8	0.9	1
Difference in predicted SOC value from that of Coleman et al. (1997) (t C ha^{-1})	4.04	3.43	3.16	2.91	2.47	2.08	1.74	1.53	1.44	1.18	0.94	0.73
% Difference in predicted SOC value from that of Coleman et al. (1997)	16.19	13.74	12.65	11.66	9.87	8.33	6.98	6.13	5.79	4.73	3.78	2.92
Equilibrium predicted SOC (t C ha^{-1})	29.01	28.40	28.13	27.88	27.43	27.05	26.71	26.50	26.41	26.15	25.91	25.70
SI	-5.75	-6.27	-6.53	-6.79	-7.31	-7.84	-8.36	-8.72	-8.88	-9.40	-9.93	-10.45

DPM/RPM ratio	1.1	1.2	1.3	1.4	1.44	1.5	1.6	1.7	1.8	1.9	2
Difference in predicted SOC value from that of Coleman et al. (1997) (t C ha^{-1})	0.54	0.36	0.20	0.06	0.00	-0.08	-0.20	-0.32	-0.43	-0.53	-0.62
% Difference in predicted SOC value from that of Coleman et al. (1997)	2.15	1.45	0.81	0.22	0.00	-0.32	-0.82	-1.28	-1.71	-2.11	-2.48
Equilibrium predicted SOC (t C ha^{-1})	25.50	25.33	25.17	25.02	24.97	24.89	24.76	24.65	24.54	24.44	24.35
SI	-10.97	-11.49	-12.02	-12.54	-	-13.05	-13.58	-14.10	-14.62	-15.15	-15.67

This is important to note when considering afforestation scenarios, as it implies that assuming instantaneous planting of mature trees on arable land (and the C input size and DPM/RPM ratio of 0.25) would not affect the final C sequestration estimate after approximately 60 years. The runs assuming DPM/RPM=1.44 and 0.67 throughout (i.e. arable/managed grassland and unmanaged grassland/scrub, respectively) do not, however, reach the final measured SOC value.

Table 4.8 The effect of C input quality on experimental SOC values predicted with RothC

DPM/RPM value	1.44 to 1904 and 0.25 thereafter	1.44 to 1904 and 0.67 thereafter	1.44 throughout	0.67 throughout	0.25 throughout
Difference in predicted SOC value from that of Coleman et al. (1997) (t C ha^{-1})	0.00	-3.51	-6.80	-3.45	0.12
% Difference in predicted SOC value from that of Coleman et al. (1997)	0.00	-5.98	-11.60	-5.88	0.20
Final predicted SOC, 1995 (t C ha^{-1})	58.58	55.08	51.79	55.14	58.70
RMSE	7.02	7.72	11.95	9.49	12.54

4.3.2.4 Sensitivity of SOC values predicted with RothC to the quality of annual C inputs over the experimental timescale

Table 4.8 and Figure 4.4 show the sensitivity of experimental SOC values predicted with RothC to C input quality. Fig. 4.4 and Table 4.8 demonstrate that RothC is relatively insensitive to errors in C input quality during the experimental term. It is interesting to note that the runs using DPM/RPM values of 1.44 then 0.25 (as used by Coleman et al. 1997), 1.44 then 0.67, and 0.67-throughout all show an acceptable fit to the measured data, with RMSE<10. The run using a DPM/RPM ratio of 0.25 throughout effectively assumes forest C input quality from the beginning of the experiment, and although the model is unable to fit the measured point at 1904, it reaches the same final value as the run using Coleman et al. (1997)'s DPM/RPM values.

4.3.2.5 Sensitivity of SOC values predicted with RothC to C inputs at large spatial scales

Table 4.9 shows differences in SOC, and predicted changes in European SOC stocks due to differences in the size of the annual C input, after 100y afforestation of 30% European arable land. The table shows that an error of 10% in the annual C input could cause an error of 0.3565 Pg C (14.35%) in the predicted change in SOC stocks, or 0.96% in the size of the European SOC stock, for this scenario, over 100y.

Table 4.9 Differences in predicted SOC increase for European arable land, due to the size of the annual C input, assuming 30% afforestation of arable land

C input value used (% of the value of 2.85 t C ha^{-1} y^{-1} used by Coleman et al. 1997)	Initial modelled SOC (t C ha^{-1})	SOC value after 100y (t C ha^{-1})	SOC increase after 100y (t C ha^{-1})	Increase in European C stocks over 100y[1] (Pg)	Predicted total European SOC stock after 100y afforestation of 30% arable land (Pg)
0%	24.94	12.42	-12.52	-1.0812	33.5588
10%	24.94	16.55	-8.39	-0.7247	33.9153
25%	24.94	22.74	-2.20	-0.1899	34.4501
50%	24.94	33.06	8.12	0.7014	35.3414
75%	24.94	43.38	18.44	1.5927	36.2327
90%	24.94	49.57	24.63	2.1275	36.7675
95%	24.94	51.65	26.70	2.3057	36.9457
99%	24.94	53.29	28.35	2.4483	37.0883
100%	24.94	53.70	28.76	2.4840	37.1240
101%	24.94	54.11	29.17	2.5196	37.1597
105%	24.94	55.76	30.82	2.6623	37.3023
110%	24.94	57.83	32.90	2.8405	37.4805
125%	24.94	64.02	39.08	3.3753	38.0153
150%	24.94	74.34	49.40	4.2666	38.9066
200%	24.94	94.98	70.04	6.0492	40.6892
300%	24.94	136.26	111.32	9.6145	44.2545

[1]Assuming the same area of land and scenario as in Smith et al. (1997b, 1998a,c) and using RothC for the Geescroft Wilderness Experiment to predict C increases.

4.4 GENERAL DISCUSSION AND CONCLUSIONS

Estimating C inputs the single most important factor in estimating soil C sequestration under changes in land use and management. However, the importance of accurately estimating the quality of this input depends upon the timescale and the land-use change in question. Although plant inputs of C to soil can be estimated from direct and indirect measurements, these measurements are not routinely performed for the majority of long-term experiments investigated in this project, or, more importantly, over large spatial areas, which are the major focus of this thesis. A modelling approach is more useful, since plant C inputs can be estimated from other parameters (such as climate and soil), and some dynamic NPP models can provide C inputs under changing management and climatic conditions.

This chapter has demonstrated a simple method for estimating plant returns of C to soil from SOMNET data, for the RothC model. The use of this method was shown to be limited in its scope for predictive, regional studies since C inputs showed only poor relationships with environmental parameters, and in effect only provided one value for each of three land use categories (arable, grassland and forest). This regression based method for estimating plant returns of C to soil from long-term experimental data also highlighted the large variability of plant C inputs to soil, and did not provide a satisfactory way to estimate plant C inputs under changes in land use or management.

The sensitivity analysis of RothC to the quantity of C inputs showed that RothC predicted-SOC values are highly sensitive to errors in estimates of the size of annual C inputs, especially at equilibrium. The sensitivity analysis of RothC-predicted SOC values to errors in the quality of C inputs to soil showed that errors in C input quality have smaller effects than errors in the quantity of C inputs. Errors in C input quality are more important for simulation of forests than other land uses at equilibrium. During the experimental timescale, RothC-predicted SOC values are less sensitive to errors in C input quality. For example, with a land-use change assuming instantaneous afforestation of arable land, assuming that the C input quality is that of forest from the time of the land use change has little effect on the final SOC estimate after a period of about 60 years.

Investigation of the impacts of errors in C inputs for RothC-SOC predictions at the European scale indicated that an error of 10% in C inputs could result in an error of 14.35% in SOC stocks, much larger than the resultant error in SOC stocks from a similar proportionate error in IOM estimation (see Chapter 3).

It is concluded that a more precise method of estimating C inputs to soil for RothC should be used in future studies. Linkage of RothC to a dynamic plant production model such as CENTURY would a) provide dynamic estimates of C inputs under changes in land use and management, b) allow application of RothC in a predictive mode both regionally and at the site scale and c) allow a means to equally compare results of simulations from two of the most widely used and validated SOM models.

5. EVALUATING ROTHC AND CENTURY WITH LONG-TERM EXPERIMENTAL DATA

5.1 INTRODUCTION

Few data are available to validate the performance of soil organic matter (SOM) models when used for estimating SOM changes at the regional scale. It is therefore essential that models be thoroughly tested under the environmental conditions of the region to be simulated, and for the management scenarios of interest at the site scale.

Both RothC and CENTURY have previously been extensively evaluated against datasets from site-scale long-term experiments (LTE's) covering arable, forest and grassland management regimes in many of the major climatic regions of the world (see Chapter 2). Chapter 6 of this thesis presents a case study of modelling soil organic carbon (SOC) dynamics at the regional scale, for an area of Central Hungary.

Neither model has been well tested using Hungarian LTE data, nor have they been evaluated for all the management scenarios to be investigated in this study. Chapter 4 showed that without detailed data on plant C inputs to soil, RothC would need to be linked to a combined plant production and SOC model such as CENTURY in order to predict SOC changes in a changing environment. In addition, a comparison of the performance of RothC and CENTURY contributes to the regional scale model comparison presented in Chapter 6.

The aims of this chapter are to:

a) Evaluate the performance of RothC and CENTURY under Hungarian environmental conditions.
b) Evaluate the performance of RothC and CENTURY for the full range of management scenarios to be tested in the regional case study.
c) Test the implications of using CENTURY to make dynamic estimates of C inputs to soil for use with RothC.
d) Compare the performance of two of the most widely used SOM models, using common datasets.

5.2 METHODS

5.2.1 Model input data

Input data for RothC and CENTURY were collected for seven sites in Western Europe and Hungary covering arable, grassland and forest management, and different soil types. Table 5.1 shows the long-term experimental sites and treatments, and their basic environmental conditions. Selected treatments were run for each site, relating to the management scenarios of interest to the regional-scale C sequestration study presented in Chapter 6. Much of the data used were collected from the Global Change and Terrestrial Ecosystems Soil Organic Matter Network (GCTE SOMNET - Smith et al. 1996a,b).

In order to compare the performance of RothC and CENTURY on an equal basis, all data inputs to the model were, where possible, identical. Major input data for RothC and CENTURY are given in Table 5.2. Annual meteorological records (where available) were used for all sites except Headley Hall, where long-term averaged data from a nearby station were used (Mueller, 1982). For some variables, the use of identical input data was possible, for example RothC and CENTURY both use monthly rainfall data and soil textural information. In other cases, different model input parameters were required. To allow close comparison, RothC was always run using evaporation data estimated using the CENTURY model water modules. This requires maximum and minimum temperature data and the latitude for the site. CENTURY also requires the user to set a parameter, fwloss(4), to scale estimated potential evapotranspiration to measured values. The values of fwloss(4) needed at each site are given in Table 5.1.

Plant inputs of C to soil for RothC were set in one of two ways. Firstly, RothC was run using 'inverse modelling' of SOC (or in 'reverse mode': Coleman et al. 1997): by adjusting the annual plant input of C to soil to obtain the best fit to modelled SOC values. In this case, model best fit was assessed using the Root Mean Square Error (RMSE), a measure of the overall percent model fit to measured data (Smith et al. 1996d). Secondly, RothC was run using plant inputs of C to soil estimated by the CENTURY model. CENTURY model plant C inputs to soil were the same as those obtained during the CENTURY model simulations presented in this chapter.

Table 5.1 Summary of long-term experiments selected for model evaluation

Site and Reference	Latitude / Longitude	Country	Land use	Equilibrium land use	Treatments	Fwloss(4)	Soil clay %	Initial SOC (t C ha^{-1} to 20cm)	Mean total annual rainfall (mm)	Mean total annual potential evapotranspiration (mm)	Mean annual temperature (^{O}C)
Martonvasar 2.14 (Sarkadi, 1991)	47.32 N 19.00 E	Hungary	Arable	Forest then arable with FYM	Nil, FYM once every 4 y	0.71	31.0	73.36	453.50	931.00	10.28
Geescroft Wilderness (Poulton, 1995a)	51.50 N 0.50 W	UK	Natural woodland regeneration	Arable	No additions	0.55	21.0	24.94	704.00	452.00	9.29
Park Grass (Poulton, 1995b)	51.50 N 0.50 W	UK	Grassland	Grassland	Unmanured, NPKNaMg	0.53	23.0	78.00	704.00	452.00	9.29
Nagyhorcsok (Kovacs et al. 1995)	46.54 N 18.31 E	Hungary	Arable	Arable rotation	Nil, high inorganic NPK	0.64	23.0	68.00	548.88	921.00	11.27
Headley Hall (Chaney et al. 1985)	53.63 N 1.16 W	UK	Arable	Arable/ley rotation	Direct drill, ploughed	0.98	25.0	86.00	650.00	637.00	9.23
Woburn ley arable (Johnston, 1973)	52.20 N 0.36 W	UK	Ley-arable	Grass then arable rotation	Arable with roots, arable with hay, ley arable	0.48	63.0	32.64	506.68	432.00	9.28
Ultuna (Kirchmann et al. 1994)	60.00 N 17.00 E	Sweden	Arable	Arable	No N, Straw, Sewage sludge	0.49	36.5	42.91	555.00	480.00	6.68

Table 5.2 Major input variables for the RothC and CENTURY models

	RothC	CENTURY
Soil variables	Clay content	Sand content
	Inert Organic Matter content	Silt content
	SOC content	Clay content
		Bulk density
		SOC content
		Active SOM content
		Slow SOM content
		Passive SOM content
Weather variables	Total monthly precipitation	Total monthly precipitation
	Mean monthly temperature	Mean monthly maximum temperature
	Total monthly evaporation	Mean monthly minimum temperature
Management variables	Residue quality factor (DPM/RPM ratio)	Residue lignin/N ratio
	Soil Cover	Plant C and N content
	Residue C input	Atmospheric N deposition
	Manure C inputs	

A FORTRAN program (READSITE) was created to extract evaporation and plant C input data from monthly CENTURY output files, and to create weather and land management files for running the RothC model. This provided an equal basis for comparison on RothC and CENTURY at site and regional scales and provided dynamic estimates of C inputs for RothC, thus allowing use of RothC in a predictive mode at site and regional scales.

Initial pool sizes for RothC and CENTURY were estimated as follows. If both total SOC measurements and soil ^{14}C dates were available, an inverse modelling procedure was used to derive the IOM content of the soil for the RothC model (Geescroft Wilderness and Park Grass; from former model evaluations; Jenkinson et al. 1992). In the absence of soil ^{14}C dates the equation derived in Chapter 3 (Section 3.2.2), relating IOM content to total SOC content (Falloon et al. 1998a), was used to estimate the IOM content of the soil. For CENTURY, initial pool sizes were set based upon a modification of the guidelines given by Metherell et al. (1993). The 'Passive SOM' fraction was first set from total SOC and clay content, using the equation derived in section 3.3.2. As suggested by Metherell et al. (1993), the remainder of total SOC was then divided into active SOM (2-3 times the soil microbial biomass C, or 2-4% of total SOC), and slow SOM (approximately 55% of total SOC).

Model input files were created using the input data collected for each site, and model runs executed to accurately simulate management regimes at each of the sites studied. Model outputs were then studied, model fit to measured data assessed using the RMSE, and plant C inputs to soil recorded.

5.2.2 RothC development for no-till soils and sewage sludge amended soils

Currently, the RothC model is not specifically set up to simulate either no-till soils, or sewage sludge applications to land. No-till agriculture and sewage sludge applications to land were, however, scenarios of land management of interest to the regional scale case study. Therefore a different version of the RothC model was also created (ROTHCX), to allow testing of the RothC model with no-till soils and sewage sludge applications to soil. ROTHCX allows the fixed parameters (which are normally set within the executable code) to be read in as inputs to the model in a file (fix.dat). Organic manure fixed parameters in this file include FFYM(1), the fraction of FYM-C flowing to the DPM pool, FFYM(2), the fraction of FYM-C flowing to the RPM pool, FFYM(3), the fraction of FYM-C flowing to the BIO pool, FFYM(4), the fraction of FYM-C flowing to the HUM pool, and FFYM(5), the fraction of FYM-C flowing to the IOM pool. Other fixed model parameters set in this way are FDEC1(3), the fraction of (BIO-C plus HUM-C) flowing to the BIO pool, FDEC1(5), the fraction of (BIO-C plus HUM-C) flowing to the HUM pool, and the maximum decomposition rate constants for DPM, RPM, BIO and HUM (DECOMP(1), DECOMP(3), DECOMP(3), and DECOMP(5)) .

ROTHCX fixed parameters for testing with no-till soils were set using information taken from the CENTURY model. The standard version of the RothC model was developed for use with tilled arable systems, when used with the decomposable plant material/resistant plant material (DPM/RPM) factor set at 1.44, whilst the CENTURY model was developed for no-till (grassland systems). Hence in CENTURY simulations of no-till soils, the potential maximum decomposition is increased by a certain factor during the month of tillage, dependent on the tillage operation. In order to simulate no-till soils with RothC, the maximum decomposition rate should therefore be reduced overall in comparison with tilled soils.

ROTHCX was developed based on the tillage factors used in the CENTURY model for ploughing operations. CENTURY increases the maximum decomposition rate by a factor of 1.6 during the month of ploughing, versus a factor of 1.1 for a direct-drilled system. Therefore for a no-till soil in ROTHCX, the maximum decomposition rate in the month of tillage would be reduced by a factor of 0.6875 relative to a ploughed soil. This effect was simplified to account for the overall effect of tillage over one year, effectively slowing decomposition by a factor of 0.9739 annually. The maximum decomposition rates for all SOM pools in ROTHCX were therefore slowed by a factor of 0.9739. Table 5.3 shows the difference between maximum annual decomposition rates in RothC and ROTHCX.

ROTHCX was also used to test RothC with sewage sludge amended soils. The standard version of RothC only accounts for additions of organic matter to soils in terms of either plant carbon (straw, roots or other materials) or FYM. Sewage sludge has a higher P content than FYM, and the activity and size of the soil microbial biomass may be reduced in soils that have been subject to sludge additions (McGrath and Brookes, 1986).

Table 5.3 Maximum annual decomposition rates in the RothC and ROTHCX models

SOM Pool	Maximum annual decomposition rate, RothC model y^{-1}	Maximum annual decomposition rate, ROTHCX model y^{-1}
DPM	10	9.739
RPM	0.30	0.292
BIO	0.66	0.64
HUM	0.02	0.019
IOM	0	0

ROTHCX made it possible to test different assumptions about sewage sludge composition in terms of proportions of DPM, RPM and HUM (humus pool) added to the soil. FYM is usually considered to be composed of 49% DPM, 49% RPM and 2% HUM. Arbitrary proportions of 10% DPM, 70% RPM, and 20% HUM were assumed as initial testing values, based on the fact that sewage sludge is often both more decomposed and resistant than FYM (McGrath and Brookes, 1986).

5.2.3 CENTURY model development for sewage sludge amended soils

The current version of CENTURY not specifically set up to simulate sewage sludge applications to land. Sewage sludge application to land was, however, a scenario of land management of interest to the regional scale case study. The CENTURY model was therefore also adapted, in a similar manner to RothC, for testing with sewage sludge amended soils. In CENTURY, organic additions to soil are split into the different SOM pools in the 'partit' routine of the model, which partitions added organic matter between the 'structural' and 'metabolic' litter pools of the model dependent upon the lignin:N ratio of the added organic matter. A separate 'partits' routine was created for the model for use with sewage sludge additions to soil, which runs only for additions of organic materials to soil, and not additions of plant C. For FYM containing 15% lignin, and with a C:N ratio of 15, the split would be around 70% 'metabolic' and 30% 'structural'. An arbitrary split of added sludge into 10% 'metabolic', 60% 'structural' and 30% 'slow SOM' was used to test the model initially, since sewage sludge is often both more decomposed and resistant than FYM (McGrath and Brookes, 1986).

5.3 RESULTS AND DISCUSSION

5.3.1 CENTURY simulations

5.3.1.1 Martonvasar Experiment 2.14, Hungary

Initial SOC pool sizes for the CENTURY model were set using a modification of the values suggested by Metherell et al. (1993), as described in section 5.2.1.

Using these values for the long-term arable experiment at Martonvasar resulted in a large overestimation of observed SOC values. Paustian et al. (1992), in their modelling of the arable site at Ultuna (Sweden) divided total SOC into 60% 'passive SOM', 37% 'slow SOM' and 3% 'active SOM'. Dividing the total SOC in this way for Martonvasar still resulted in overestimation of measured SOC. A reasonable fit to the measured data was obtained using 34% 'passive SOM', 63% 'slow SOM' and 2.6% 'active SOM'.

Large, recent inputs of FYM-C to the soil before the establishment of the experiment (P. Csatho, personal communication) may explain the greater than expected proportion of 'slow SOM' needed to model this site. After these adjustments, the CENTURY model simulated SOC values showed similar patterns to those observed in the measured SOC data (Fig. 5.1). Both treatments in this experiment showed a decline in SOC. The declining SOC concentrations at Martonvasar may also have been due to the sites former land management (historically forestry) and before the experiment was established large amounts of FYM were applied to the soil. A conversion from forest to arable land tends to result in a SOC decline, due to an increase in C input decomposability and a decrease in the amount of C input under arable management. A reduction in FYM inputs to the soil more recently would also reduce total C inputs to soil, and hence SOC.

The FYM treatment showed a higher SOC concentration, due to greater total C inputs to soil, and the CENTURY simulation of the FYM plot showed a 'saw-tooth' pattern of change in SOC (Fig. 5.1). This pattern of change was due to the addition of FYM only once every four years, with SOC increasing shortly after the time of FYM addition, and declining in subsequent years. Average C inputs to soil as estimated by CENTURY were 3.48 t C ha^{-1} y^{-1} for the nil treatment and 3.50 t C ha^{-1}

y^{-1} for the FYM treatment (with 4.4 t C ha^{-1} y^{-1} FYM once every 4 years). Model fit to measured SOC values was good for both treatments (RMSE=5.34 and 7.77 for the nil and FYM treatments, respectively).

5.3.1.2 Geescroft Wilderness, UK

Kelly et al. (1997) formerly used CENTURY to simulate SOC changes during natural woodland regeneration on abandoned arable land at the Geescroft Wilderness Experiment. Their values for the initial distribution of total SOC amongst model pools were used in this simulation. The CENTURY model run for Geescroft included simulation of the early scrub-forest succession. The CENTURY model simulation of Geescroft Wilderness showed a dramatic increase in SOC, close to the measured values (Fig. 5.2). The increase in SOC at Geescroft is attributed to the change in management from arable land to natural woodland regeneration. C inputs to soil under woodland are more resistant to decomposition than those under arable land, and the total amount of C input to soil under woodland may also often be greater than that under arable land. This reduction in decomposability and increase in quantity of C inputs results in increased SOC concentrations. The CENTURY model showed a reasonable fit to the measured SOC values (RMSE=10.66), with a slight overestimation of measured SOC in general. Average C inputs to soil as estimated by the CENTURY model were 3.02 t C ha^{-1} y^{-1}.

5.3.1.3 Park Grass, UK

Kelly et al. (1997) formerly simulated SOC changes in the long-term grassland experiment at Park Grass with CENTURY. Their estimation of the initial SOC pool size distribution for the CENTURY model was used for these runs. CENTURY model simulations were run for two treatments at this site: Plot 3d (unmanured) and Plot 14d (NPKNaMg). In general, the CENTURY model simulations showed similar patterns to the measured SOC values (Fig. 5.3), although there was little overall variation in observed SOC values. The CENTURY model predicted little difference in SOC values between the two treatments, whilst the measurements suggest a slightly greater SOC concentration in the unmanured treatment compared to the NPKNaMg treatment. The reason for this difference is unclear (Coleman et al. 1997). It would normally be expected that the fertilisation of the grassland would lead to a greater grass yield, and hence greater C inputs to soil

Figure 5.1 Measured and CENTURY simulated SOC at Martonvasar

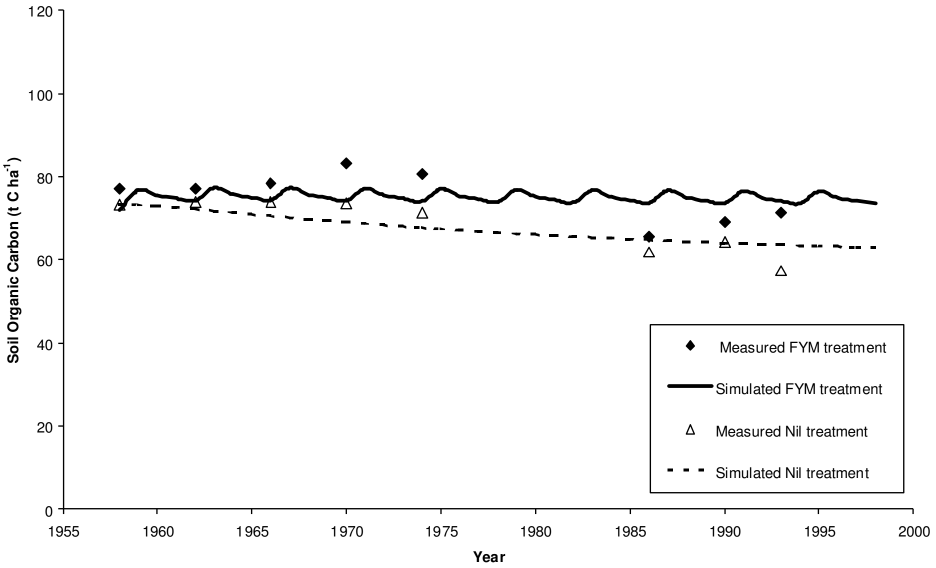
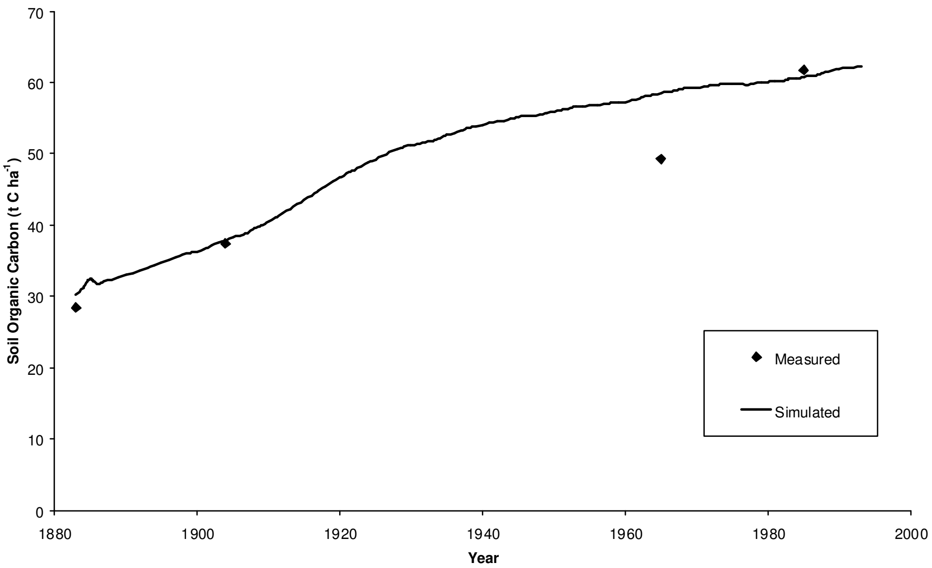


Figure 5.2 Measured and CENTURY simulated SOC at Geescroft Wilderness



Average C inputs to soil as estimated by CENTURY were 3.56 t C ha^{-1} y^{-1} in the unmanured plot and 3.46 t C ha^{-1} y^{-1} in the NPKNaMg treatment. Model fit to the measured SOC values was good for both treatments (RMSE=6.29 and 8.05 for the unmanured and NPKNaMg plots, respectively).

5.3.1.4 Nagyhorksok, Hungary

Two treatments at this long-term arable, winter wheat - maize - maize - winter wheat rotation experiment were chosen: the nil input (1) and high NPK (19) treatments. The measured SOC at this site showed a general decline in SOC values observed for both treatments, and only small differences in SOC between the treatments. The reason for the overall decline in SOC is unclear, since there is little information regarding historical land management at the site, other than that it was under an arable rotation. It is possible that this rotation may have included ley periods, which would have resulted in increased C inputs and SOC values relative to the current day. CENTURY fit to measured SOC values was reasonable (RMSE=16.35 and 8.19 for the nil and high NPK treatments, respectively). Average C inputs to soil as estimated by the CENTURY model were 1.55 t C ha^{-1} y^{-1} for the nil treatment and 2.01 t C ha^{-1} y^{-1} for the high NPK treatment. The observed SOC for the nil treatment begins at a higher value than the high NPK treatment, but eventually falls below the SOC observed in the high NPK treatment. A greater SOC under NPK could be expected due to greater crop yields and C inputs to soil. However, the CENTURY simulation of Nagyhorcsok shows a higher SOC concentration in the nil treatment throughout, despite a lower C input to soil in the nil treatment (Fig 5.4).

5.3.1.5 Headley Hall, UK

The no-till (direct-drilled) and tilled (ploughed) treatments of the Headley Hall arable experiment were simulated with the CENTURY model. The measured SOC data show little temporal variation overall, but a small increase in SOC in the no-till treatment relative to the tilled treatment.

The CENTURY model predicted a similar overall SOC concentration to that observed (Fig. 5.5), but a much smaller difference in SOC between treatments than observed. Nevertheless, the fit between modelled and measured data was good

Figure 5.3 Measured and CENTURY simulated SOC at Park Grass

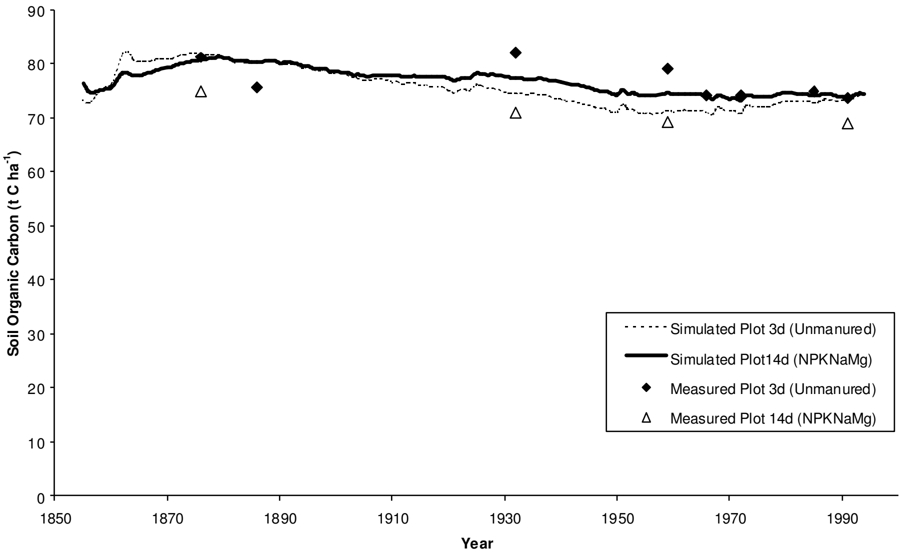
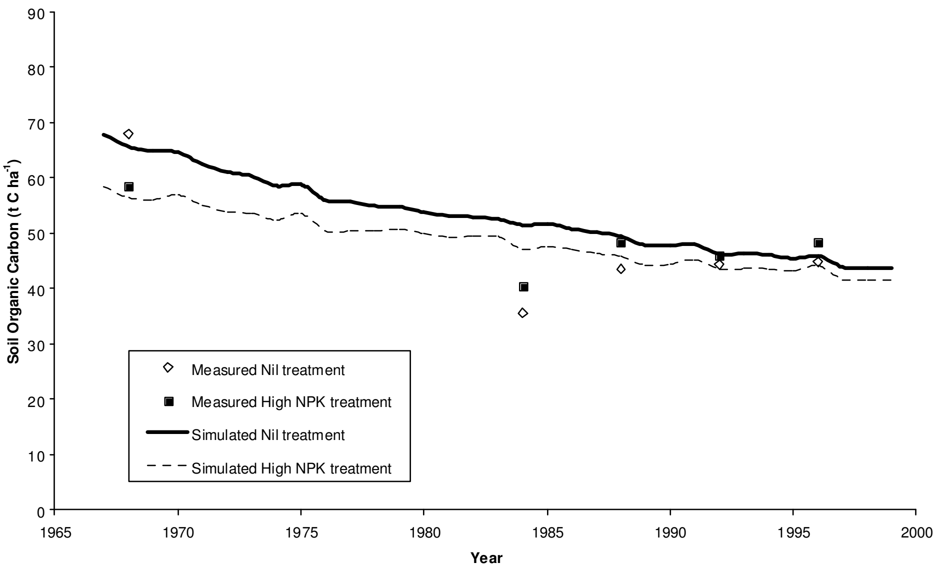


Figure 5.4 Measured and CENTURY simulated SOC at Nagyhorcsok



(RMSE=3.84 and 5.83 for the till and no-till treatments, respectively). Average plant C inputs to soil as predicted by CENTURY were 2.63 t C ha^{-1} y^{-1} and 2.50 t C ha^{-1} y^{-1} for the till and no-till treatments, respectively.

5.3.1.6 Woburn Ley Arable, UK

Three treatments of the Woburn Ley-Arable experiment were simulated with CENTURY (Fig. 5.6), the 'arable with roots, no FYM' (Arable-Roots), 'arable with hay, no FYM' (Arable-Hay), and 'grazed ley, no FYM' (Grazed-Ley) treatments. The Arable-Roots treatment consisted of a five-course rotation of potato-cereal-root followed by potato-barley, sugarbeet-barley, or barley-barley. The five-course rotation in the Arable-Hay treatment was potato-cereal-hay, followed by potato-barley, sugarbeet-barley, or barley-barley. For the Grazed-Ley treatment the five-course rotation was three years of grass clover ley (with inorganic N and grazing) followed by potato-barley, sugarbeet-barley, or barley-barley. The measured SOC showed a slight decrease in SOC under the Arable-Roots and Arable-Hay treatments, and an increase in SOC under the Grazed-Ley treatment.

With the initial distribution of SOC between CENTURY model pools set as described in section 5.2.1, CENTURY was unable to follow the loss of SOC in the Arable-Roots treatment. A better fit to measured data was obtained using 3% 'active SOM', 50% 'slow SOM' and 47% 'passive SOM'. Available information on historical management practices at this site suggest ley periods and FYM applications may have been included in the rotations, which could have increased the size of the 'slow SOM' pool relative to the other model pools.

With this pool size distribution, CENTURY model accurately simulated changes in SOC with an excellent fit to measured data (RMSE=2.23, 1.98 and 1.83 for the Arable-Roots, Arable-Hay and Grazed-Ley treatments, respectively). CENTURY also suggested some details of SOC dynamics not shown in the measured data. The CENTURY simulations of the Arable-Hay and Grazed-Ley treatments both showed a SOC increase during the years when the treatments were under grass, due to the greater quantity, and lower decomposability of C input under grassland compared to arable management. Average C inputs to soil predicted by CENTURY were 1.03, 1.26, and 1.36 t C ha^{-1} y^{-1} for the Arable-Roots, Arable-Hay and Grazed-Ley treatments, respectively. The average C input during grassland management was 1.55 t C ha^{-1} y^{-1}, and during arable management 1.03, 0.96, and 0.73 t C ha^{-1} y^{-1} for the Arable-Roots, Arable-Hay and Grazed-Ley treatments, respectively.

5.3.1.7 Ultuna, Sweden

Three treatments from the Ultuna long-term arable experiment were simulated using CENTURY (Fig. 5.7), the no-N, straw incorporation, and sewage sludge incorporation treatments. The measured SOC data showed a decrease in SOC under no-N, little change in SOC under straw incorporation, and a large increase in SOC under sewage sludge incorporation. All crop residues (with the exception of roots) were removed from the soil following harvest, so the C inputs estimated here represent the contribution of below ground (root-derived) C to the soil. The initial distribution of total SOC amongst CENTURY model pools was set using 60% 'passive SOM', 37% 'slow SOM' and 3% 'active SOM' according to a former CENTURY simulation of several treatments at Ultuna (Paustian et al. 1992). CENTURY accurately simulated changes in the no-N and cereal straw incorporation treatments, giving a good fit to the measured SOC data (RMSE=3.74 and 9.04 for the no-N and cereal straw treatments, respectively). Whilst CENTURY tended to slightly overestimate of SOC increases in the cereal straw treatment, there was a greater underestimate of SOC changes in the sewage sludge treatment (RMSE=14.56), when considering sewage sludge to be similar in composition to FYM.

However, a much better fit to measured SOC data was obtained using the CENTURY model with the additional 'partits' routine for sewage sludge. The best model fit to measured SOC data (RMSE=7.33) was obtained with added sewage sludge split into 10% 'metabolic C', 60% 'structural C', and 30% 'slow SOM', relative to the proportions 70% 'metabolic C' and 30% 'structural C' usually used for FYM. The changes made to the model represent an increase in the proportion of resistant undecomposed material, and in humified organic matter added to the soil.

CENTURY predicted average plant C inputs to soil of 0.66, 0.45, and 0.53 t C ha^{-1} y^{-1} for the no-N, straw incorporation and sewage sludge treatments, respectively.

Figure 5.5 Measured and CENTURY simulated SOC at Headley Hall

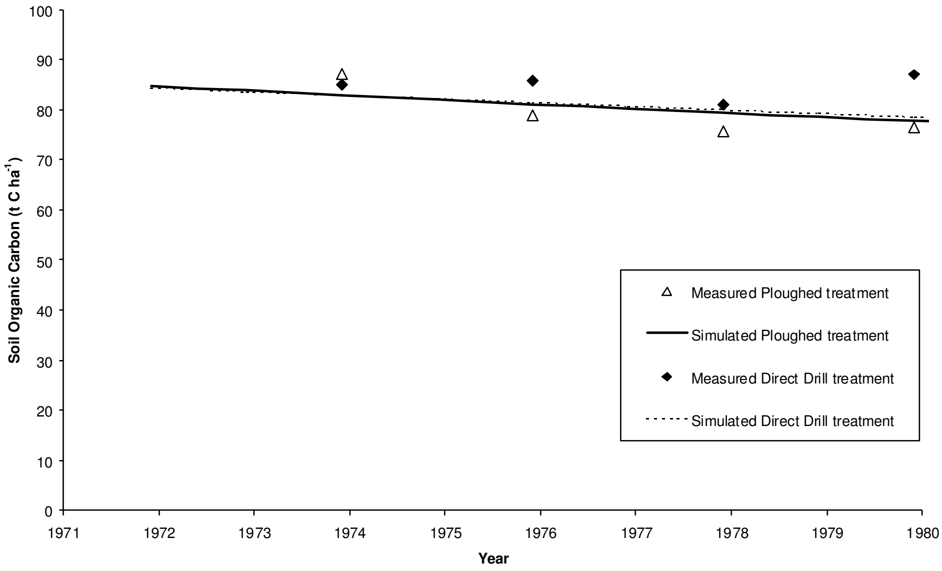


Figure 5.6 Measured and CENTURY simulated SOC at Woburn Ley Arable

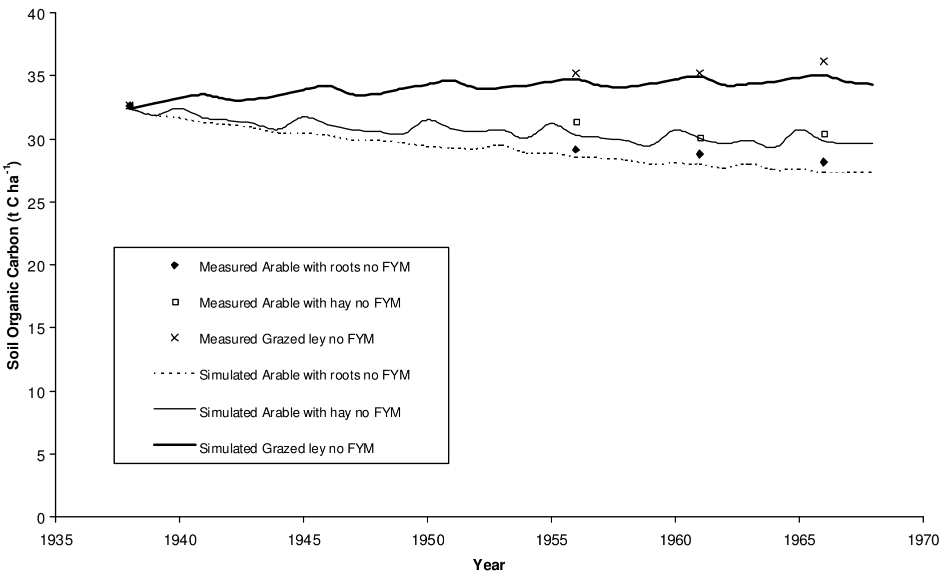
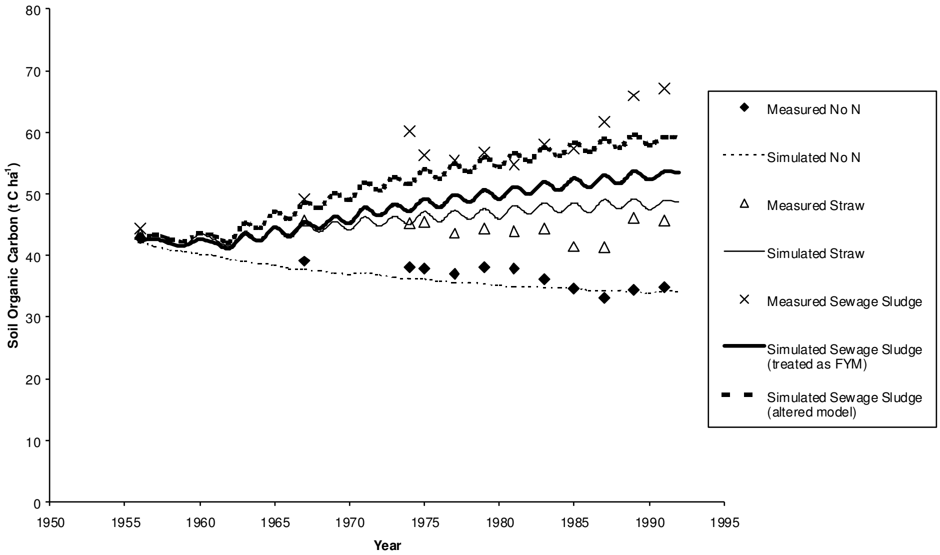


Figure 5.7 Measured and CENTURY simulated SOC at Ultuna



5.3.2 RothC simulations - inverse modelling

5.3.2.1 Martonvasar Experiment 2.14, Hungary

Management treatments and trends in measured SOC data at Martonvasar have been described in section 5.3.1. In the absence of soil ^{14}C dates, the IOM content of the soil was estimated as 6.26 t C ha^{-1} using the equation relating IOM content to total SOC (Falloon et al. 1998a), derived in Chapter 3. RothC gave a good fit to measured data (RMSE=7.33 and 7.96 for the nil and FYM treatments, respectively), and simulated SOC dynamics followed the measured data (Fig. 5.8). RothC simulations showed a slight overall decrease in SOC during the experiment, with an increase in SOC under FYM relative to the nil treatment. As with CENTURY simulations, RothC suggested additional information on SOC dynamics that was not apparent from available SOC measurements. RothC showed a 'saw-tooth' pattern of SOC change for the FYM treatment, whereby SOC increased in the year of FYM application, and decreased for the following three years without FYM application. Average plant inputs of C to soil predicted by RothC were 1.70 and 1.75 t C ha^{-1} y^{-1} for the nil and FYM treatments, respectively.

5.3.2.2 Geescroft Wilderness, UK

SOC dynamics at Geescroft Wilderness have been simulated previously with RothC by Coleman et al. (1997). The simulation presented here uses slightly different input data to their simulation, in using evaporation estimated from the CENTURY simulation of Geescroft Wilderness, to allow maximum comparability between RothC and CENTURY model simulations. The IOM content of the soil was set at 2.5 t C ha^{-1}, as previously estimated using soil ^{14}C dates and SOC measurements (Coleman et al. 1997). RothC predicted a sharp increase in SOC (Fig. 5.9), and gave a good fit to measured SOC data (RMSE=12.76). Average plant inputs of C to soil predicted by RothC were the same as those estimated by Coleman et al. (1997), at 2.85 t C ha^{-1} y^{-1}.

5.3.2.3 Park Grass, UK

Coleman et al. (1997) previously simulated changes in SOC in the Park Grass Experiment using RothC. The simulations presented here differ from their simulation since evaporation data estimated from the CENTURY simulation of Park Grass were used (rather than measured data from the site), to allow maximum comparability between model simulations. The IOM content of the soil was set at 6.9 t C ha^{-1}, as previously estimated using soil ^{14}C dates and SOC measurements (Coleman et al. 1997). The model predicted similar changes in SOC to those measured (Fig. 5.10), with little overall change in SOC through time, and a slightly greater SOC concentration in the unmanured plot (3d) compared to the NPKNaMg treatment (plot 14d). Model fit to measured data was good, with RMSE=5.47 and 1.68 for the unmanured and NPKNaMg treatments, respectively. Average C inputs to soil predicted by RothC were 3.20 and 2.90 t C ha^{-1} y^{-1} for the unmanured and NPKNaMg treatments, respectively. As discussed in section 5.3.1, the reasons for this treatment difference are unclear, and it would normally be expected that inorganic fertilisation would increase herbage yield and thus C input to soil. However, it may be that more C is allocated below ground in the unfertilised plots, relative to fertilised plots (Jenkinson et al. 1992).

Figure 5.8 Measured and RothC simulated SOC at Martonvasar

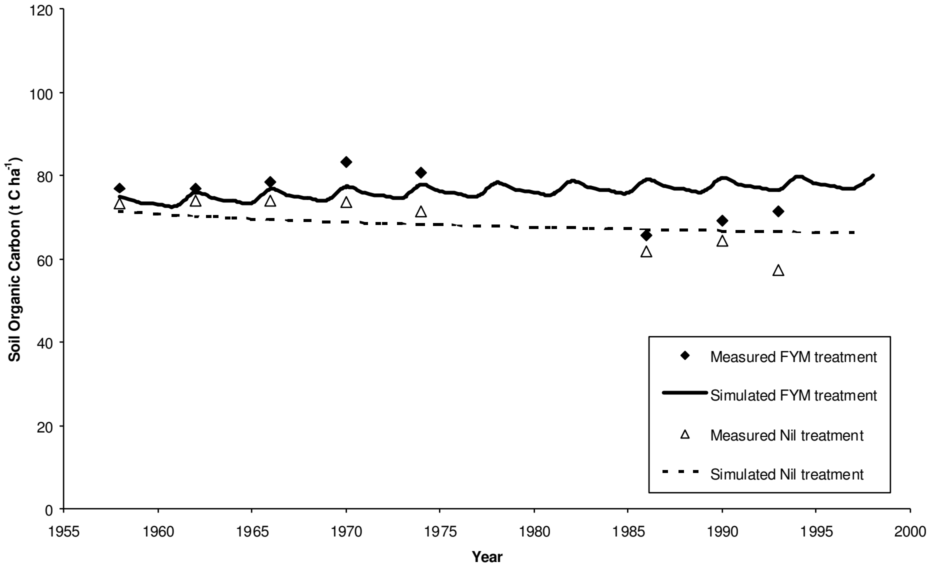


Figure 5.9 Measured and RothC simulated SOC at Geescroft Wilderness

Figure 5.10 Measured and RothC simulated SOC at Park Grass

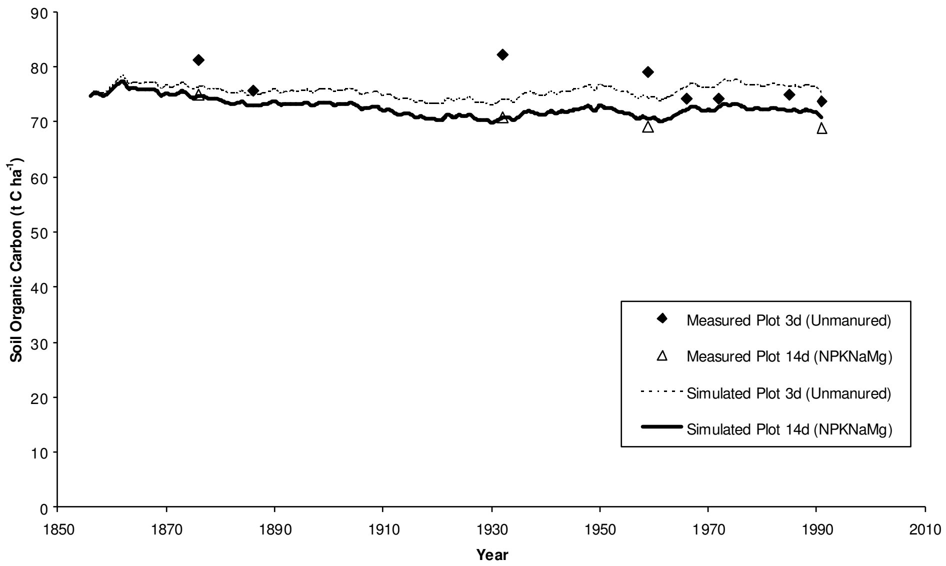
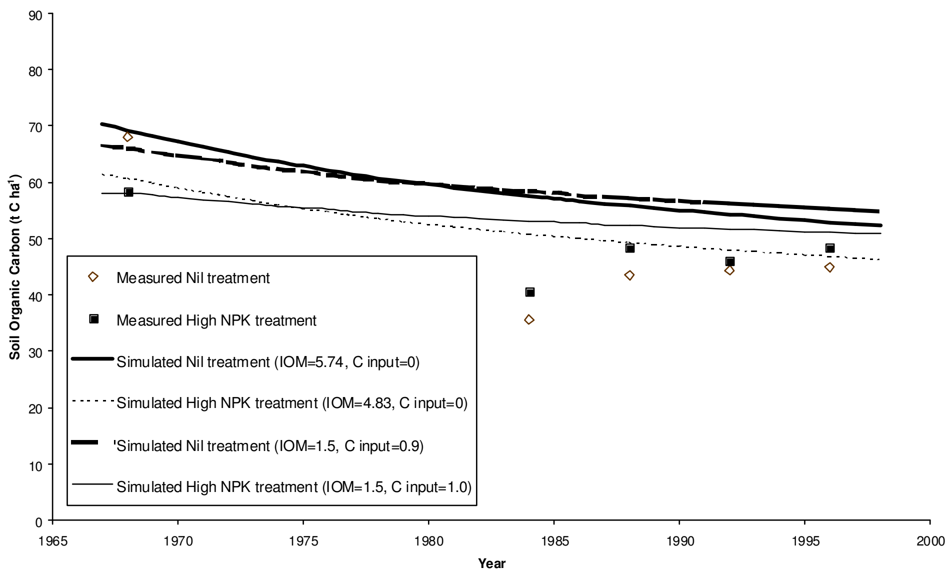


Figure 5.11 Measured and RothC simulated SOC at Nagyhorcsok



5.3.2.4 Nagyhorksok, Hungary

In the absence of soil ^{14}C dates, the IOM content of the soil was estimated using the equation relating IOM content to total SOC (Falloon et al. 1998a), derived in Chapter 3. Separate estimates of IOM for the Nil and NPK treatments of 5.74 t C ha^{-1} and 4.83 t C ha^{-1}, respectively, were used since the starting SOC of the two treatments was quite different. RothC simulation of SOC dynamics at Martonvasar followed general trends in measured SOC data (Fig. 5.11), with an overall decrease in SOC during the experiment.

However, with the default IOM values above, and even with no C input to soil at all (Fig 5.11), RothC was unable to match SOC declines in this experiment. Model fit to measured data was poor (RMSE=26.75 and 10.04 for the Nil and high NPK treatments, respectively). Model runs were then completed with an IOM content of 1.5 t C ha^{-1} (for both treatments) with average C inputs of 0.9 t C ha^{-1} and 1.0 t C ha^{-1} for the Nil and NPK treatments, respectively. Greater plant C inputs to soil would be expected under the high NPK treatment, compared to the Nil treatment, due to increased crop yields and thus returns of C to soil. This still gave a poor simulation of the measured data, (RMSE=29.20 and 13.54 for the Nil and high NPK treatments, respectively), and still tended to overestimate SOC in general.

Since maize often returns 2-5 t C ha^{-1} (Patwardhan et al. 1995) and C inputs unde winter wheat are normally of the order of 0.5-2.0 t C ha^{-1} (Jenkinson et al. 1992), it seemed unreasonable to reduce C inputs further, below an average of 0.9-1.0 t C ha^{-1} for this experiment with a 50% maize, 50% wheat rotation.

5.3.2.5 Headley Hall, UK

In the absence of soil ^{14}C dates, the IOM content of the soil was estimated as 7.57 t C ha^{-1} using the equation relating IOM content to total SOC (Falloon et al. 1998a), derived in Chapter 3. RothC predicted similar overall trends in SOC to the observed SOC data (Fig. 5.12), with a slight decrease in SOC in both treatments during the course of the experiment. RothC gave a good fit to the tilled treatment, with RMSE=4.01 and an average predicted plant C input to soil of 0.5 t C ha^{-1} y^{-1}. However, in order to obtain a reasonable fit to the no-till treatment (RMSE=3.72), it was necessary to assume an average plant C input to soil of 2.65 t C ha^{-1} y^{-1}.

Although there may be some differences in crop yield and thus C returns to soil between tilled and no-tilled soils, a difference of over 300% in C input seemed unreasonable. Available evidence (see literature review, Chapter 1 section 1.3.2) suggests that the major factor influencing SOC increases in no-till soils is not increased C inputs, but reduced decomposition. The ROTHCX model was therefore used with the same C input to soil as used for the tilled treatment (0.5 t C ha^{-1} y^{-1}), but with the maximum decomposition rate constants reduced as described in section 5.2.2. RothC then gave good fit to the measured SOC data for the no-till treatment, with RMSE=3.10. However, RothC predictions still tended to slightly overestimate SOC in the tilled treatment, and to underestimate SOC in the no-till treatment.

5.3.2.6 Woburn Ley Arable, UK

Jenkinson et al. (1987) previously simulated changes in SOC in the Woburn Ley Experiment using RothC. The simulations presented here differ from their simulation since evaporation data estimated from the CENTURY simulation of the Woburn Ley Arable Experiment were used (rather than measured data from the site), to allow maximum comparability between RothC and CENTURY model simulations. In the absence of soil ^{14}C dates, the IOM content of the soil was estimated as 2.49 t C ha^{-1} using the equation relating IOM content to total SOC (Falloon et al. 1998a), derived in Chapter 3.

RothC gave an excellent fit to the measured SOC data (RMSE=1.25, 1.47, and 3.46 for the Arable-Roots, Arable-Hay, and Grazed-Ley treatments, respectively). RothC followed trends in the measured SOC data (Fig. 5.13), with a steady decline in SOC under Arable-Roots, a 'saw-tooth' decline in SOC under Arable-Hay, and a 'saw-tooth' increase in SOC under Grazed-Ley. RothC predicted average plant inputs of C to soil as 1.01, 1.16 and 1.47 t C ha^{-1} y^{-1} under Arable-Roots, Arable-Hay and Grazed-Ley, respectively. The with the average C input during grassland management was 1.95 t C ha^{-1} y^{-1}; average values of C input under arable management were 1.00, 1.00 and 1.20 t C ha^{-1} y^{-1} for the Arable-Roots, Arable-Hay and Grazed-Ley treatments, respectively.

Figure 5.12 Measured and RothC simulated SOC at Headley Hall

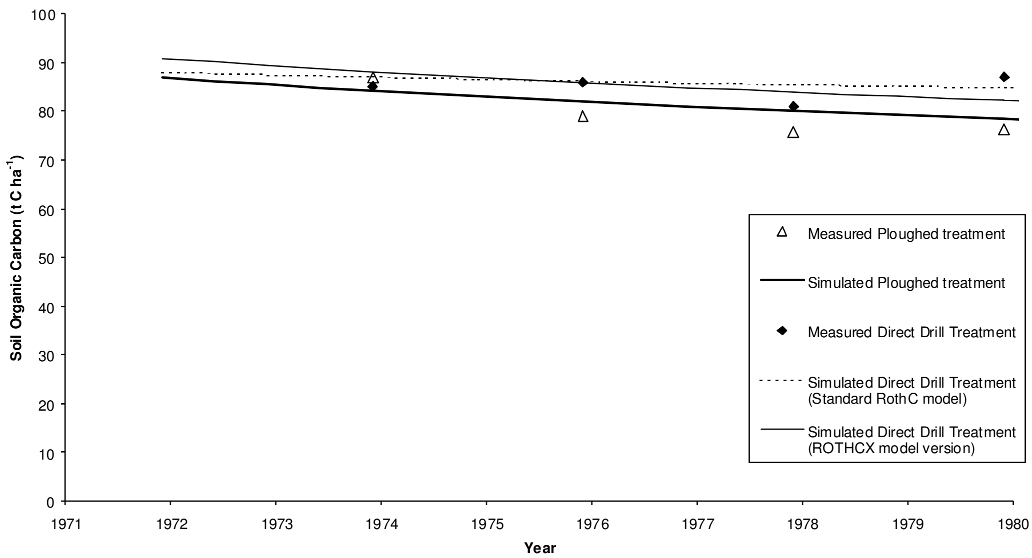


Figure 5.13 Measured and RothC simulated SOC at Woburn Ley Arable

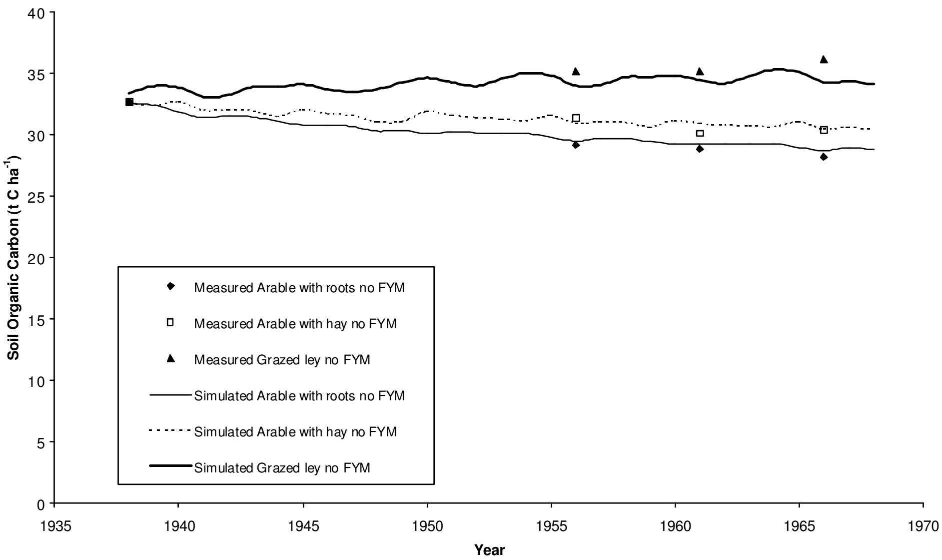
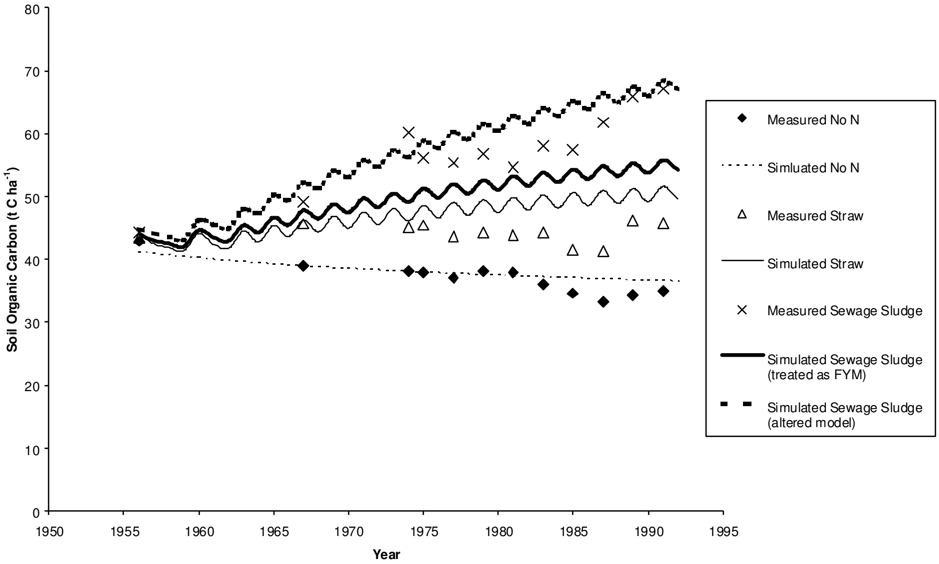


Figure 5.14 Measured and RothC simulated SOC at Ultuna



5.3.2.7 Ultuna, Sweden

In the absence of soil ^{14}C dates, the IOM content of the soil was estimated as 3.40 t C ha^{-1} using the equation relating IOM content to total SOC (Falloon et al. 1998a), derived in Chapter 3. SOC values predicted by RothC followed trends in measured SOC for the No-N and Cereal Straw treatments (Fig. 5.14), with a good fit between measured and modelled data (RMSE=4.83 and 12.54 for the No-N and Cereal Straw treatments, respectively). However, when using the conventional version of RothC, and treating the Sewage Sludge as an input of FYM to soil, RothC considerably underestimated SOC increases in the Sewage Sludge treatment (RMSE=11.67).

The ROTHCX version of the model was then used, assuming that Sewage Sludge was composed of 10% DPM, 70% RPM, and 20% HUM (rather than the values of 49% DPM, 49% RPM and 2% HUM used for FYM).

Running ROTHCX in this way, the modelled SOC changes were in much closer agreement with measured SOC (RMSE=7.46). This assumed that sewage sludge was composed of a greater proportion of resistant plant material and humified organic material, and less decomposable plant material, compared to FYM. Average C inputs to soil as predicted by RothC were 0.55 t C ha^{-1} y^{-1}, 0.35 t C ha^{-1} y^{-1}, and 0.40 t C ha^{-1} y^{-1} for the No-N, Cereal Straw and Sewage Sludge treatments, respectively.

5.3.3 RothC simulations with CENTURY plant C inputs

5.3.3.1 Martonvasar Experiment 2.14, Hungary

When using C inputs to soil derived from the CENTURY simulation of the Martonvasar Experiment, RothC followed general trends in measured SOC (Fig. 5.15), but tended to overestimate measured SOC values overall. RothC did predict a higher SOC concentration in the FYM amended treatment than in the Nil treatment, and predicted a 'saw-tooth' trajectory of SOC as discussed in sections 5.3.1 and 5.3.2. The model fit to measured SOC values was poorer for both treatments than the simulations using either RothC with inverse modelling, or CENTURY alone (RMSE=27.75 and 26.67 for the Nil and FYM treatments).

5.3.3.2 Geescroft Wilderness, UK

The simulation of SOC changes at Geescroft Wilderness using RothC driven by C inputs derived from CENTURY resulted in a similar estimation of measured SOC values (Fig. 5.16) to that estimated by CENTURY or RothC alone. The model predicted a pattern of SOC increase similar to that measured, and the fit of simulated SOC to measured SOC values was good, with RMSE=9.66.

5.3.3.3 Park Grass, UK

In common with observed SOC values, RothC predicted a greater SOC concentration in the Un-manured treatment (3d) at Park Grass relative to the NPKNaMg treatment (14d), when run using C inputs derived from CENTURY (Fig. 5.17). RothC also predicted little temporal variation in either treatment, which was also observed in the measured SOC data. However, RothC did slightly underestimate measured SOC values in general, with a poorer fit to measured SOC in both treatments than obtained using either RothC with inverse modelling, or the CENTURY model alone. RMSE values for these runs were 7.70 for the Nil treatment (3d) and 12.69 for the NPKNaMg treatment (14d).

5.3.3.4 Nagyhorksok, Hungary

SOC values predicted by RothC, with C inputs derived from CENTURY, showed similar patterns to those obtained using CENTURY (Fig 5.18). IOM values of 5.74 t

Figure 5.15 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Martonvasar

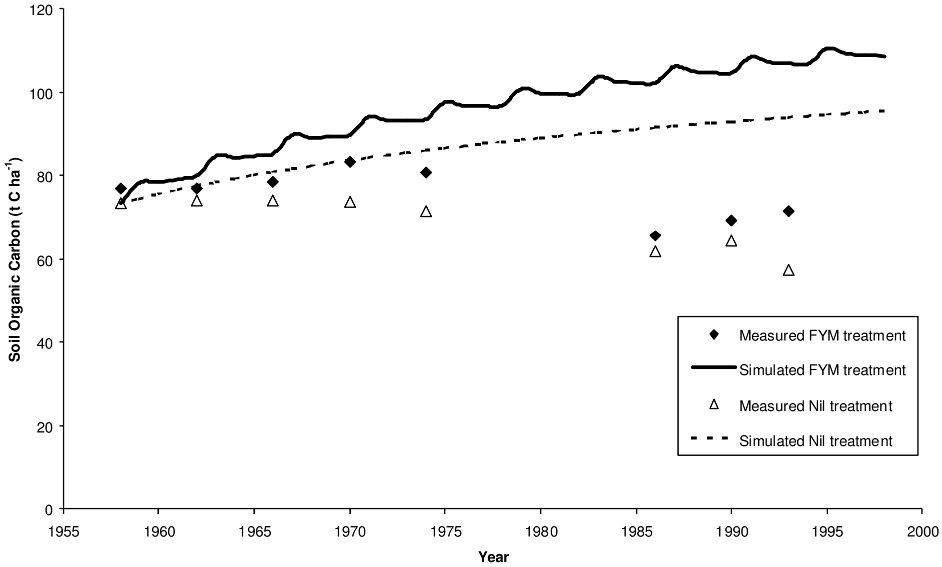


Figure 5.16 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Geescroft Wilderness

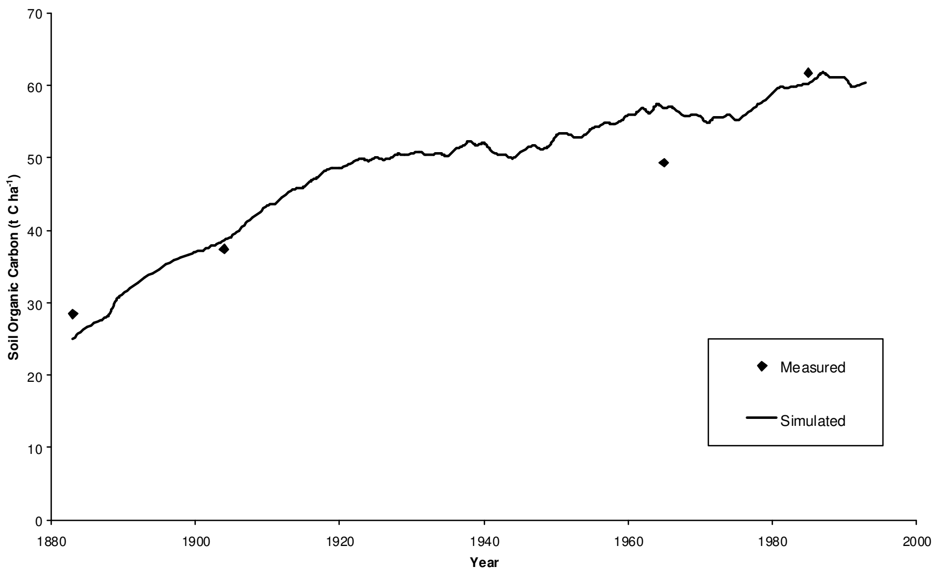


Figure 5.17 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Park Grass

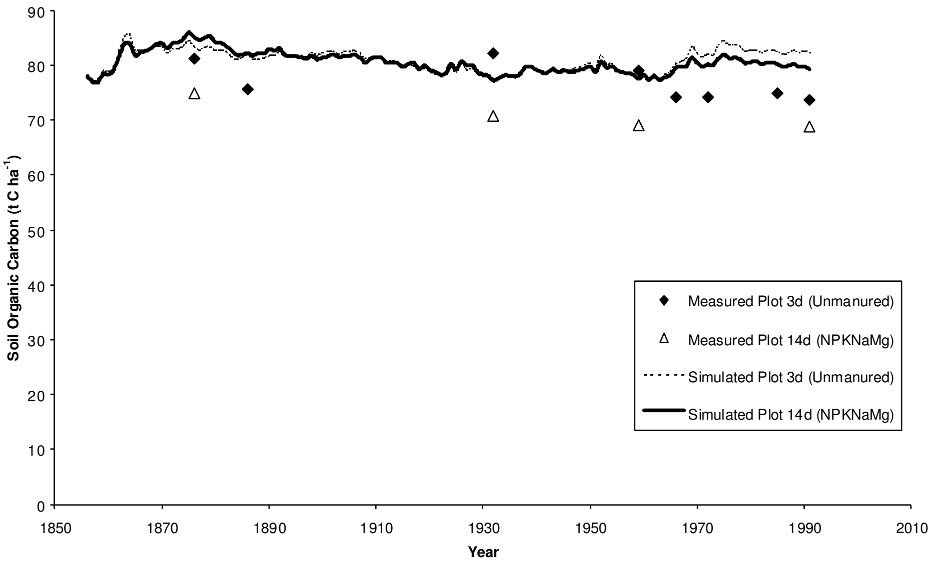
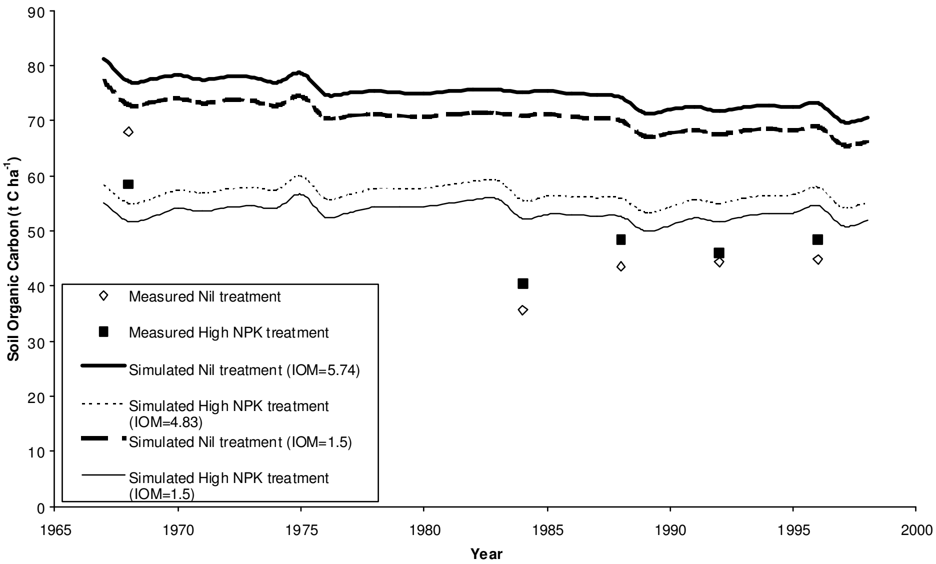


Figure 5.18 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Kezthely



C ha^{-1} and 4.83 t C ha^{-1} (Nil and NPK treatments, respectively) and 1.5 t C ha^{-1} (both treatments) were used, as in section 5.3.2.4. RothC predicted a higher SOC concentration in the Nil treatment compared to the High NPK treatment. In contrast, measured SOC data showed a higher value of SOC in the Nil treatment than in the High NPK treatment, with SOC in the High NPK treatment reaching a slightly higher value than the Nil treatment by the end of the experiment. Reducing the IOM content of the soil did give improve model fit to measured data, but all runs tended to overestimate SOC concentrations. These runs of RothC gave a poorer fit to measured SOC values than runs with RothC and inverse modelling or CENTURY alone.

With the default IOM values, RMSE values for model fit to measured SOC data were 60.93 for the Nil treatment, and 19.99 for the High NPK treatment. With IOM set at 1.50 t C ha^{-1}, RMSE values were 52.60 for the Nil treatment, and 15.24 for the High NPK treatment.

5.3.3.5 Headley Hall, UK

When using C inputs derived from the CENTURY simulation of SOC in the Headley Hall experiment, RothC followed general trends in measured SOC (Fig. 5.19). RothC predicted little difference in SOC values between both treatments. Simulations were completed for the No-Till treatment using both the conventional version of RothC, and the ROTHCX version of the model, with changes to maximum decomposition rate constants as described in sections 5.2.2 and 5.3.2. Model fit to measured SOC values was reasonable for all treatments. RMSE values for these runs were 8.67 for the Tilled treatment, 2.70 for the No-Till treatment (using the conventional version of RothC) and 2.70 for the No-Till treatment (using the ROTHCX version of the model, modified for No-Till soils).

5.3.3.6 Woburn Ley Arable, UK

The simulation of SOC changes in the Woburn Ley Arable experiment using RothC driven by C inputs derived from CENTURY resulted in an overestimation of measured SOC values (Fig. 5.20). RothC did predict patterns of SOC change similar to that measured, with a slight decrease in SOC in the Arable-Roots treatment, little

Figure 5.19 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Headley Hall

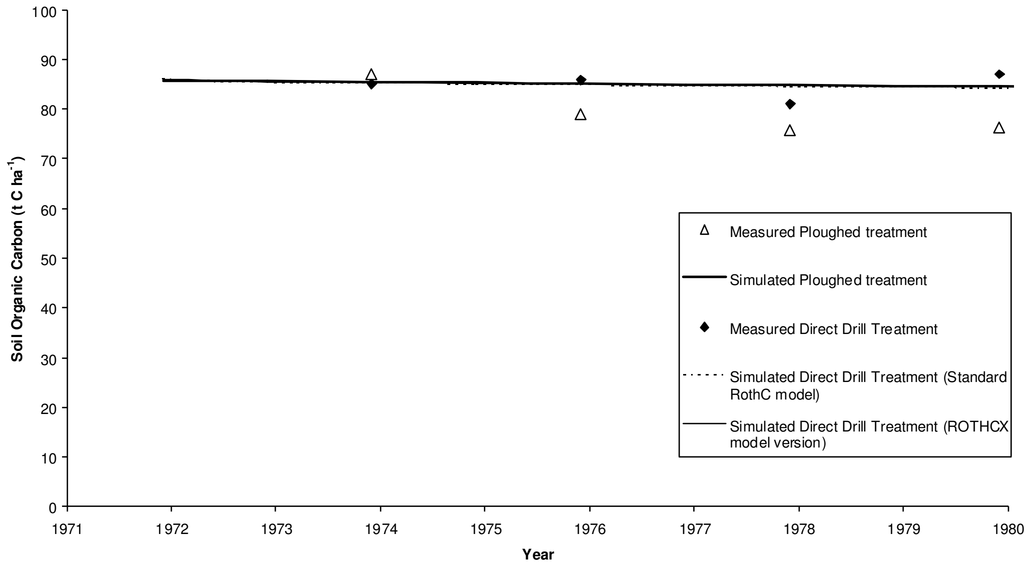


Figure 5.20 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Woburn Ley Arable

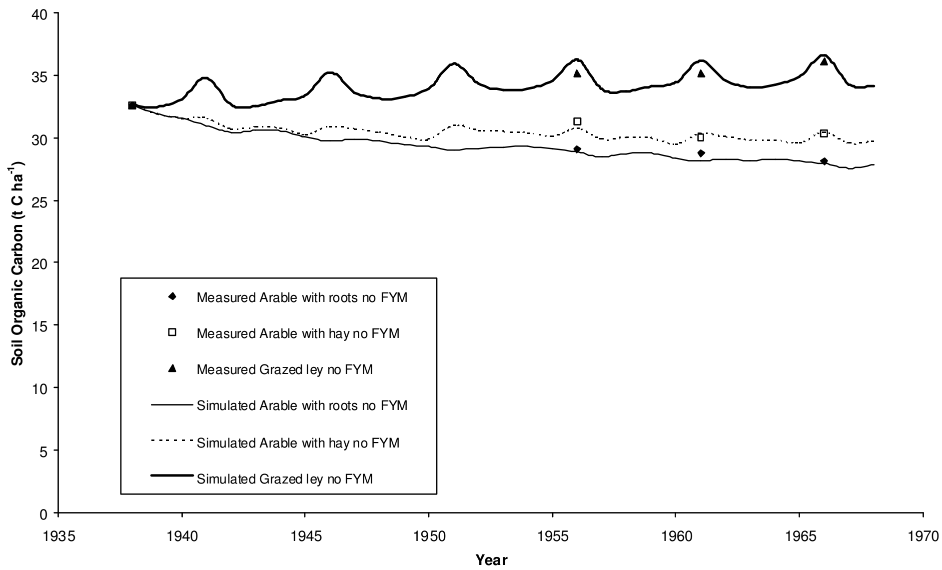
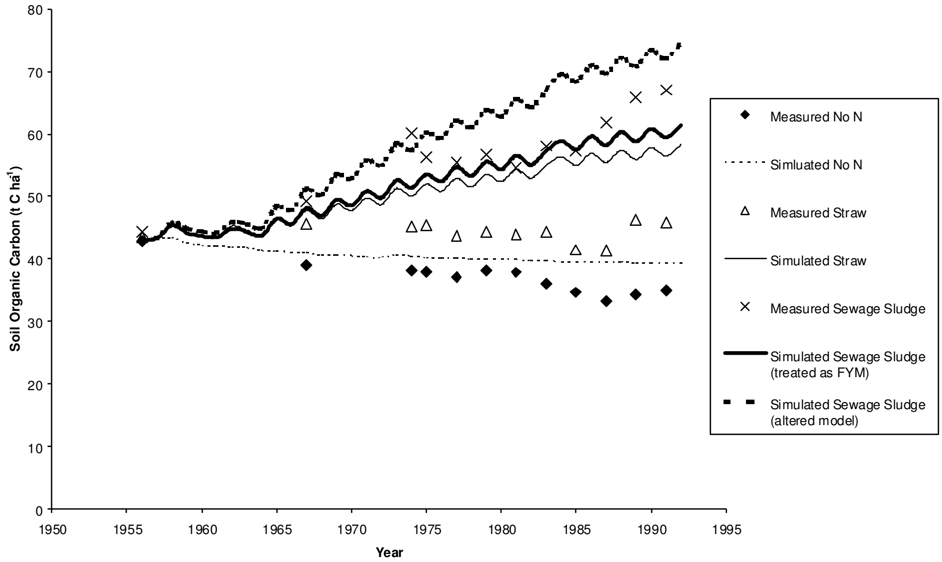


Figure 5.21 Measured and RothC simulated SOC (using CENTURY-derived C inputs) at Ultuna



overall change with a 'saw-tooth' pattern of SOC change in the Arable-Hay treatment, and a 'saw-tooth' pattern of SOC increase in the Grazed-Ley treatment. RMSE values for these runs were 1.28, 1.06 and 2.07 for the Arable-Roots, Arable-Hay, and Grazed-Ley treatments, respectively.

5.3.3.7 Ultuna, Sweden

SOC values predicted by RothC, with C inputs derived from CENTURY, showed similar patterns to those obtained using CENTURY, and to the measured SOC values (Fig. 5.21), but with a general overestimation of measured SOC values. RothC predicted a slight decrease in SOC in the No-N treatment, and a gradual increase in SOC in the Straw incorporation treatment. Using the conventional version of RothC resulted in a steady increase in SOC for the Sewage Sludge treatment similar to that obtained for the Cereal Straw incorporation treatment. The ROTHCX version of the model amended for use with Sewage Sludge incorporation as described in sections 5.2.2 and 5.3.2, predicted a similar increase in SOC with time to that obtained for the Sewage Sludge treatment. These runs of RothC gave a poorer fit to measured SOC values than runs with RothC and inverse modelling (No-N treatment, Cereal Straw treatment, and Sewage Sludge treatment with ROTHCX). A poorer fit to measured SOC values was also obtained in comparison with CENTURY alone (No-N treatment, Cereal Straw treatment, and Sewage Sludge treatment with ROTHCX), with a general underestimation of SOC values. RMSE values for model fit to measured SOC data were 10.06 for the No-N treatment, 20.94 for the Cereal Straw incorporation treatment, 7.66 for the Sewage Sludge treatment (conventional RothC), and 10.97 for the Sewage Sludge treatment (ROTHCX version of the model).

5.4 GENERAL DISCUSSION AND CONCLUSIONS

This chapter has described the evaluation of two SOM models, RothC and CENTURY, for a regional scale model application in Central Hungary. The evaluation exercise used datasets from long-term experiments in Europe and Hungary, covering management regimes of interest to the regional scale study. A program, READSITE, has been created to allow a direct comparison of the two models, by converting evaporation and C input data output from CENTURY model runs to RothC format.

A version of RothC has been created to enable testing with no-till soils and sewage sludge amendments to land. An altered version of CENTURY has also been developed for testing with sewage sludge amendments to land. These new model versions are preliminary, and have only been tested with the datasets modelled here. Further validation and development work is recommended for RothC and CENTURY with sewage sludge amendments to land, and for RothC with no-till soils to test whether the assumptions made here hold with a wider range of datasets.

In general, both models gave a reasonable or good fit to measured long-term SOC data at arable, grassland and forestry management sites. Table 5.4 shows a summary of C inputs and RMSE values from the model runs. RothC did tend to give a slightly better fit to measured SOC data than did CENTURY. This is almost certainly due to the way in which modelled SOC values may be fitted to measured SOC values by iteratively changing the annual C input to soil, or 'reverse mode' or 'inverse' modelling. RothC runs using the inverse modelling procedure tended to require smaller annual C inputs to soil than were predicted by the CENTURY model for the same treatments at the same sites. Possible reasons for this are discussed in detail elsewhere in this section.

Table 5.4 Summary of C inputs to soil and RMSE for CENTURY, RothC and RothC (run using CENTURY C inputs) model evaluations at long-term experimental sites

Experiment		CENTURY C input (t C ha^{-1} y^{-1})	CENTURY RMSE	RothC C input (t C ha^{-1} y^{-1})	RothC RMSE	RothC (CENTURY C inputs) C input (t C ha^{-1} y^{-1})	RothC (CENTURY C inputs) RMSE
Martonvasar	Nil	3.48	5.34	1.70	7.33	3.48	27.75
	FYM	3.50	7.77	1.75	7.96	3.50	26.67
Geescroft Wilderness		3.02	10.66	2.85	12.76	3.02	9.66
Park Grass	3d (nil)	3.56	6.29	3.20	5.47	3.56	7.70
	14d (NPKNaMg)	3.46	8.05	2.90	1.68	3.46	12.69
Nagyhorcsok	Nil (IOM=5.74)	1.55	16.35	0.00	26.75	1.55	60.93
	Nil (IOM-1.50)	-	-	0.90	29.20	-	52.60
	Nigh NPK (IOM=4.83)	2.01	8.19	0.00	10.04	2.01	19.99
	Nigh NPK (IOM=1.50)	-	-	1.00	13.54	-	15.24
Headley Hall	Till	2.63	3.84	0.50	4.01	2.63	8.67
	No till[1]	2.50	5.83	0.50	3.10	2.50	2.70
	No till[2]	-	-	2.35	3.72	-	2.70
Woburn	Arable-Roots	1.03 (1.03)[3]	2.23	1.01 (1.00)[3]	1.25	1.03 (1.03)[3]	1.28
Ley Arable	Arable-Hay	1.26 (0.96)[3]	1.98	1.16 (1.00)[3]	1.47	1.26 (0.96)[3]	1.06
	Grazed-Ley	1.36 (0.73)[3]	1.83	1.47 (1.20)[3]	3.66	1.36 (0.73)[3]	2.17
	Grassland[4]	2.09	-	1.95	-	2.09	-
Ultuna	No-N	0.66	3.74	0.55	4.83	0.66	10.06
	Straw	0.45	9.04	0.35	12.54	0.45	20.94
	Sewage Sludge[5]	0.53	14.56	0.40	11.67	0.53	7.66
	Sewage Sludge[6]	0.53	7.33	0.40	7.46	0.53	10.97

[1]No till runs for standard CENTURY model, and no-till amended (ROTHCX) RothC model
[2]No till runs for un-amended RothC model
[3]Average C inputs to soil under arable phase of the rotation (figures in brackets)
[4]Average C inputs to soil under grassland phase of the rotation
[5]Sewage sludge runs for un-amended RothC and CENTURY models
[6]Sewage sludge runs for sewage sludge amended (ROTHCX) RothC and CENTURY models

Runs of RothC using annual C inputs to soil derived from CENTURY model runs therefore also had a tendency to overestimate measured SOC values, and also generally had the poorest fit to measured SOC values of all model runs.

Annual C inputs to soil estimated by the CENTURY model ranged from 0.45 to 3.48 t C ha^{-1} y^{-1} in arable experiments, 2.09 to 3.56 t C ha^{-1} y^{-1} under grassland, and 3.02 t C ha^{-1} y^{-1} for forestry. RothC (run in reverse mode) estimated annual C inputs to soil of 0.35 to 2.35 t C ha^{-1} y^{-1} for arable experiments, 1.95 to 3.20 t C ha^{-1} y^{-1} for grassland management, and 2.85 t C ha^{-1} y^{-1} for forestry. These values are all within acceptable ranges (see Chapter 4), and reflect the great variability of annual C inputs to soil between different sites and climates.

RothC and CENTURY predicted differences between management treatments similar to those observed. In particular, the development of RothC and CENTURY suggested that sewage sludge could be composed of a greater proportion of resistant plant material and humified organic matter than FYM.

Testing of CENTURY-based hypotheses (namely reduced decomposition rates) in no-till soils with RothC was also encouraging, but for both scenarios, more testing is advisable. This modelling exercise has also shown that, for some sites, using default values for initialising SOM pool sizes in SOM models is not always appropriate, and may not give the most satisfactory results.

However, the lack of detailed land use history information for many sites, or measurable analogues to SOM model pools means that there may be little scientific basis for changing initial SOM pool values away from default values.

In general, this model evaluation exercise has shown that RothC and CENTURY work well under Hungarian arable conditions, and for most of the scenarios of interest to the regional scale SOM modelling exercise. More model development and testing is required in some areas.

The linkage of RothC and CENTURY using the READSITE program has enabled RothC to be run using C inputs derived from CENTURY model runs. This will allow C inputs for RothC to be estimated independently and dynamically, using a process-based plant production model, which will be more reliable than using default or averaged values over large spatial areas. This also allows RothC to be used in a predictive mode at site and regional scales.

Running RothC in this way, with C inputs derived from CENTURY can, however, be expected to give higher estimates of SOC concentrations than comparable CENTURY runs. Possible explanations for the difference in predicted C inputs to soil between RothC and CENTURY could be either a) CENTURY model predicted annual C inputs are too high; b) SOM in CENTURY turns over too quickly (decomposition rates too fast); or c) SOM in RothC turns over too slowly (decomposition rates too slow). Each of these issues is discussed in turn below.

Annual C inputs estimated by CENTURY could be too large if either total plant production was underestimated by CENTURY, or if the allocation of total plant production to C inputs was too great. A large number of simulations at sites in different ecosystems (e.g. Carter et al. 1993; Parton et al. 1993; Myers, 1994; Parton et al. 1994; Patwardhan et al. 1995; Liang et al. 1996) has shown that the model reproduces trends in both SOM and above ground plant production very well. Given this evidence, it seems unlikely that the allocation of C inputs from total plant production is incorrect, since this would lead to errors in the SOM balance. Indeed, C inputs to soil predicted by both RothC and CENTURY are reasonable by comparison with measured and modelled estimates (see Chapter 4).

Tables 5.5 and 5.6 give decomposition rate modifiers for moisture and temperature (and their sums), and modified and unmodified SOC decomposition rates at all modelled sites, calculated by RothC and CENTURY. These values were taken from the final year of simulation. Rate modifiers and their products were directly taken from monthly model outputs. Decomposition rates were calculated from the maximum decomposition rate constants ($k_{pool\ n}$) for each pool (1 to n), total SOC and the SOC concentration in each of the model pools in the final year of simulation as described below.

$$SOC \quad decomposition \quad rate = \sum_{n}^{1} \left(\frac{SOC_{pool\,n}}{total\ SOC} \times k_{pool\,n} \right)$$

$$Modified\ SOC\ decomposition\ rate = SOC\ decomposition\ rate \times rate\ modifier\ product\ sum$$

This comparison between models shows that although the rate modifier products are often similar for particular sites, the rate modifier for temperature is always higher in RothC than in CENTURY, and the rate modifier for moisture is always lower in RothC than in CENTURY. This is demonstrated by the examples given in Figs. 5.22 to 5.27, showing annual time series of rate modifier products, and rate modifiers for moisture and temperature for both models, at Nagyhorcsok and Martonvasar. Differences in the calculation of SOC turnover in RothC and CENTURY will now be discussed in terms of a) how the original pool maximum decomposition rate constants (k values) were set; b) how the rate modifying factors act and were set; and c) how SOC flows between model pools.

Table 5.5 Decomposition rate modifiers and SOC decomposition rates in the final year of simulations - RothC model

Site	Rate modifier product average (average of monthly figures over 12 months)	Rate modifier product sum (sum of monthly figures over 12 months)	Average rate modifier for temperature (average of monthly figures over 12 months)	Rate modifier For temperature sum (sum of monthly figures over 12 months)	Average Rate modifier for moisture average (average of monthly figures over 12 months)	Rate modifier For moisture sum (sum of monthly figures over 12 months)	Unmodified SOC decomposition rate	Modified SOC decomposition rate
Martonvasar	0.223	2.667	1.365	16.383	0.533	6.400	0.265	0.708
Geescroft Wilderness	0.385	4.623	10.38	12.451	0.765	9.184	0.167	0.773
Park Grass	0.352	4.220	1.206	14.472	0.662	7.950	0.149	0.630
Naghorksok	0.364	4.374	1.544	18.527	0.525	6.300	0.063	0.276
Headley Hall	0.398	4.778	1.055	12.658	0.667	8.000	0.042	0.201
Woburn Ley Arable	0.449	5.390	1.049	12.595	0.765	9.184	0.076	0.410
Ultuna	0.285	3.423	0.915	10.979	0.713	8.556	0.142	0.486

Table 5.6 Decomposition rate modifiers and SOC decomposition rates in the final year of simulations - CENTURY model

Site	Rate modifier product average (average of monthly figures over 12 months)	Rate modifier product sum (sum of monthly figures over 12 months)	Average rate modifier for temperature (average of monthly figures over 12 months)	Rate modifier For temperature sum (sum of monthly figures over 12 months)	Average Rate modifier for moisture average (average of monthly figures over 12 months)	Rate modifier For moisture sum (sum of monthly figures over 12 months)	Unmodified SOC decomposition rate	Modified SOC decomposition rate
Martonvasar	0.237	2.843	0.344	4.125	0.772	9.262	0.960	2.730
Geescroft Wilderness	0.262	3.147	0.262	3.149	0.999	11.996	0.733	2.308
Park Grass	0.259	3.108	0.336	4.028	0.854	10.245	0.990	3.077
Naghorksok	0.315	3.784	0.429	5.153	0.852	10.222	0.257	0.971
Headley Hall	0.250	2.996	0.258	3.092	0.976	11.714	0.407	1.218
Woburn Ley Arable	0.196	2.358	0.343	4.120	0.724	8.688	0.522	1.230
Ultuna	0.292	3.506	0.306	3.666	0.978	11.736	0.305	1.073

Figure 5.22 Decomposition rate modifier products in the final year of simulation at Nagyhorcsok - RothC and CENTURY models

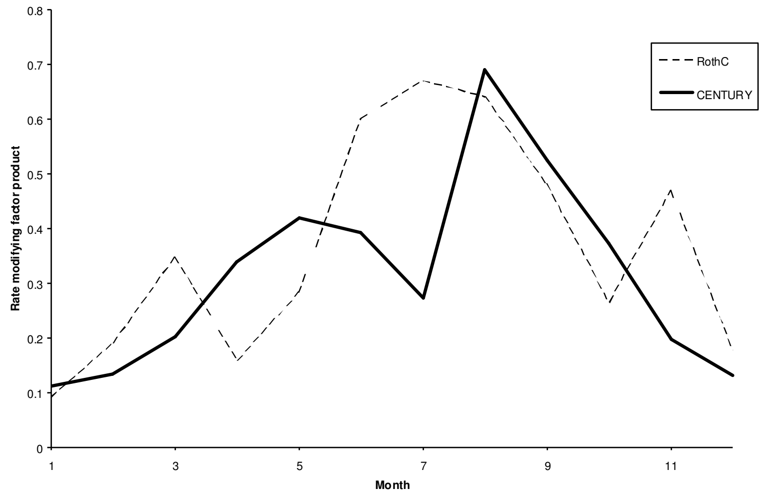


Figure 5.23 Decomposition rate modifier for temperature in the final year of simulation at Nagyhorcsok - RothC and CENTURY models

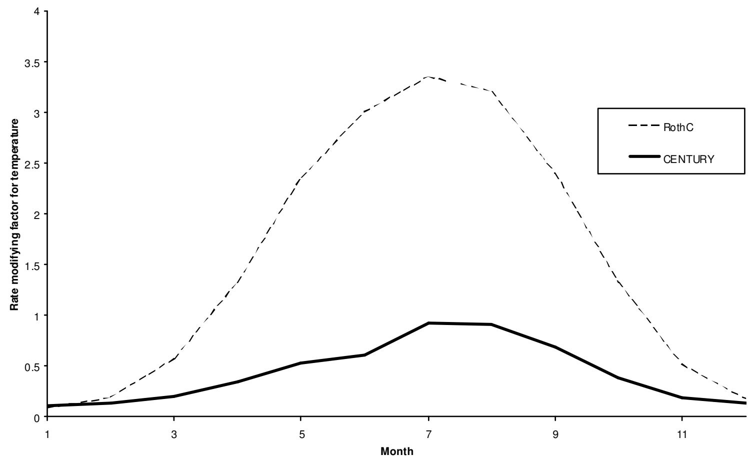


Figure 5.24 Decomposition rate modifier for moisture in the final year of simulation at Nagyhorcsok - RothC and CENTURY models

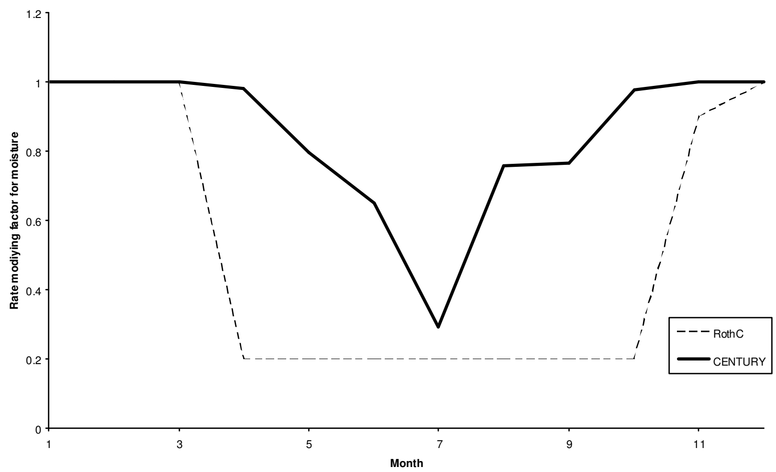


Figure 5.25 Decomposition rate modifier products in the final year of simulation at Martonvasar - RothC and CENTURY models

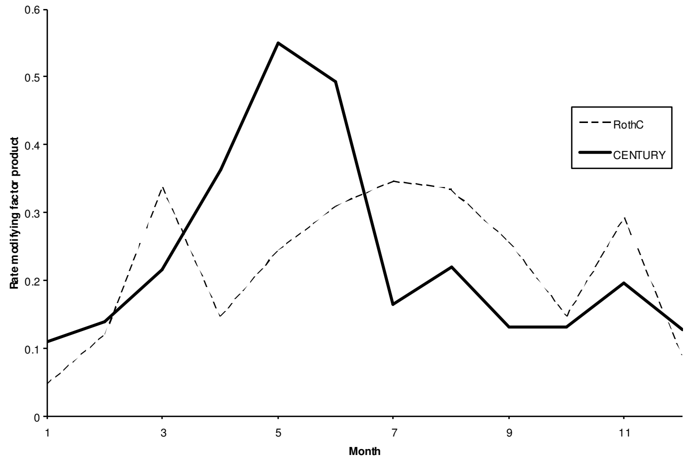


Figure 5.26 Decomposition rate modifier for temperature in the final year of simulation at Martonvasar - RothC and CENTURY models

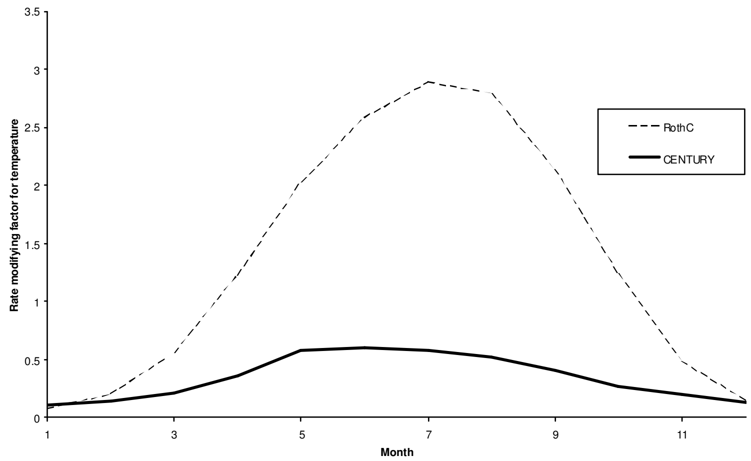
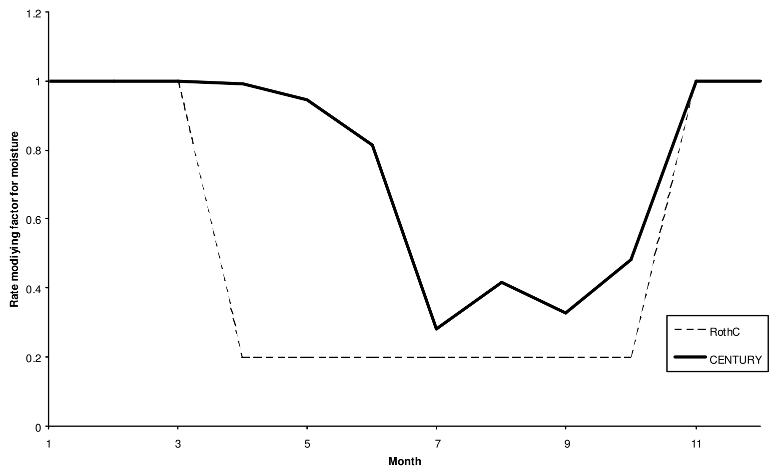


Figure 5.27 Decomposition rate modifier for moisture in the final year of simulation at Martonvasar - RothC and CENTURY models



SOM turnover in CENTURY could be too fast if the decomposition rates predicted by the model were too large. In CENTURY, the maximum decomposition rate constants were set using different sets of data. The k constant for the 'slow SOM' pool was set using 4.5 year labelled cellulose incubations (Parton et al. 1987). The k constant for the 'passive SOM' pool was set using ^{14}C dates of old SOM fractions, and model tuning to total SOM data in North American sites ranging from Colorado to Kansas (Parton et al. 1987). The k constants for the structural and metabolic pools were determined by fitting to twelve one year plant residue incubations, and the k constant for the active pool was determined by an independent fitting process (Parton et al. 1987). Subsequent to this, the model was also tested with a large two-year litter decomposition dataset along a climatic gradient in Hawaii, several five-year Swedish forest litter decomposition datasets, laboratory incubations of ^{14}C labelled plant material, and soil ^{14}C dates (Parton et al. 1994) and shown to perform very well.

The rate modifying factor for temperature in CENTURY was set by fitting to data on the decomposition of labelled cellulose at three different soil temperatures over 90 days (Parton et al. 1987). In contrast to the factor of 1.0 at 9.3 ^{o} C in RothC, the rate modifier for temperature in CENTURY has a value of 1.0 at 29.8 ^{o} C. The rate modifier for temperature in CENTURY is, however, based on soil temperature (as simulated by CENTURY) whilst RothC simply assumes soil and air temperature to be equivalent. The decomposition rate constant modifier for moisture was determined using a 34 year simulation with a daily decomposition model (Parton et al. 1987). The rate modifier for moisture is determined by the ratio of PET to annual precipitation. CENTURY also includes the effect of cultivation on decomposition rates, which can increase decomposition in the month of cultivation by a factor of up to 1.6 (Metherell et al. 1993). This factor is set arbitrarily. Anaerobic conditions may also reduce the decomposition rate by a factor of 0.4 to 1.0.

CO_2 loss on decomposition in CENTURY varies between 30-55%, dependent upon the fraction decomposing (Parton et al. 1994). The proportion of CO_2 released from 'active SOM' depends on sand content, and varies between 25-90%. The proportion of 'slow SOM' formed from 'active SOM' depends on sand and clay content and varies from 10-71.5%. The proportion of 'passive SOM' formed from 'active SOM' depends on clay content and varies from 0.3-3.5%. The proportion of 'passive SOM' formed from 'slow SOM' depends on clay content and varies from 0.3-1.2%. It therefore seems unlikely that SOM turnover in CENTURY is too rapid.

Finally, the turnover of SOM in RothC could be too slow if the predicted decomposition rates were too small. In RothC, the k value for the BIO compartment was set based upon observed measurements of the turnover time of the soil microbial biomass at Rothamsted (Jenkinson, 1990). The k values for the DPM, RPM, and HUM pools were set by fitting to data from experiments in which the decomposition of labelled plant material in different soils was followed for 10 years in the open under Rothamsted conditions (Jenkinson et al. 1987; Jenkinson, 1990). RothC has also been shown to simulate decomposition of ^{14}C-labelled plant material extremely well in 27 experiments worldwide covering a range of soil types, climates, and management regimes (Jenkinson et al. 1991). RothC has been shown to follow the signatures of both total SOC and ^{14}C extremely well (Coleman et al. 1994, 1997; Jenkinson and Coleman, 1994; Jenkinson et al. 1992, 1994).

The rate modifying factor for temperature has a value of 1.0 for the mean annual temperature at Rothamsted (9.2 ^{o}C). The relationship used to calculate the rate modifying factor for temperature was based upon measurements of the rate at which ^{14}C-labelled ryegrass decomposed in the field at Rothamsted or in the Nigerian rainforest (Jenkinson et al. 1987) over a 5-10 year period. The rate-modifying factor for moisture has the effect of allowing decomposition to proceed at at the maximum rate until there is the equivalent of a 20mm soil water deficit in Rothamsted topsoil. The rate modifying factor for plant retainment was based upon measurements of the decomposition of ^{14}C-labelled plant material at Rothamsted in bare soil or under grass (Jenkinson and Rayner, 1977).

In RothC, the ratio between CO_2 released and BIO+HUM formed due to decomposition generally varies between 75-85% CO_2 to 25-15% BIO+HUM (Jenkinson, 1990). This ratio is dependent upon the clay content of the soil, and was based upon experiments following the decomposition of ^{14}C-labelled cellulose in soils of different texture (Jenkinson, 1990) over a 10-day period. The scaling factor for the CO_2: BIO+HUM ratio developed for Rothamsted soils is applied to all soils (Jenkinson, 1990). The ratio between BIO and HUM formed upon decomposition was set arbitrarily to allow 46% BIO and 54% HUM (Jenkinson, 1990). The final proportions of C formed on decomposition in the different pools could therefore vary between 75-85% CO_2, 6.9-11.5% BIO and 8.1-13.5% HUM. It also seems unlikely that SOC turnover in RothC is too slow.

Many of the hypotheses underlying RothC were based on data from the Rothamsted site, and the model was originally designed to simulate SOC turnover under arable experiments and these climatic conditions and later extended to other climatic and management regimes. In contrast, CENTURY was originally developed to simulate the SOM cycle of North American prairie soils and extended to other environments. In both models, some functions have been determined directly from experimental datasets (such as the decomposition rate constant for BIO, the rate constant modifying factors for temperature and plant retainment; the decomposition rate constants for structural and metabolic SOM, and the rate-modifying factor for temperature in CENTURY). Many functions and parameters in both models have been set using tuning or model fitting procedures, and some have been set entirely arbitrarily (e.g. the flows between pools in CENTURY, the split between BIO and HUM formed in RothC).

Given the weight of evidence supporting decomposition rates predicted by both models, and C inputs estimated by CENTURY, it is difficult to determine which, (and indeed, if, either model) is 'correct'. Most of the relationships underlying both models were derived from different datasets, and the models were originally parameterised for different ecosystems. Further, the complex nature of the difference between model structures, rate modifying factors and flows between pools renders making any concrete conclusion almost impossible.

6. COMPARISON OF APPROACHES FOR ESTIMATING CARBON SEQUESTRATION AT THE REGIONAL SCALE USING DYNAMIC SOM MODELS, GEOGRAPHIC INFORMATION SYSTEMS AND LINEAR REGRESSIONS

6.1 INTRODUCTION

6.1.1 Regional estimates of soil carbon sequestration potential

Soil organic matter (SOM) represents a major pool of carbon within the biosphere, estimated at about 1500 to 1550 Pg globally (Batjes, 1996), roughly twice the size of the atmospheric CO_2 pool. The soil can act as both a source and a sink for carbon and nutrients. Changes in agricultural land-use and climate can lead to changes in the amount of carbon held in soils, thus influencing the fluxes of CO_2 to and from the atmosphere. Some agricultural management practices will give rise to a net sequestration of carbon in the soil, others may release soil carbon to the atmosphere. Indeed, radiocarbon dating of soils suggests that up to 60 Gt C could be sequestered globally if agricultural soils across the globe could be engineered back to their original carbon contents (Harrison et al 1993b). Regional estimates of the carbon sequestration potential of agricultural practices are crucial if policy makers are to plan future land-uses with the aim of reducing national CO_2 emissions.

The three methods that have been previously used to estimate changes in regional soil organic carbon (SOC) stocks have been discussed in detail in Section 1.6. Whilst SOM models have been compared and evaluated at the site scale (Smith et al. 1997a), there have been few previous SOM model comparisons at the regional scale (Paustian et al. 1997d; Falloon et al. 1999b, 2001a,b), or comparisons with regression-based approaches. The objective of this chapter is to compare and improve upon methods used for estimating changes in regional SOC stocks.

This chapter presents a comparison of estimates of the carbon sequestration potential of different land management practices using four methods, at the regional scale. The first three methods used are based on the linkage of the dynamic soil organic matter (SOM) models, RothC and CENTURY, to Geographical Information Systems (GIS). These include 1) application of CENTURY, 2) application of RothC using default C inputs to soil and 3) application of RothC using C inputs derived from the CENTURY model runs. The fourth method was the use of regressions based on long-term experimental data (Smith et al 1997b, 1998a,c, 1999, 2000a,b,c,d,e, 2001a,b). The regional scale case study was completed for an area of Central Hungary. The work completed in this chapter has also provided spatial datasets for regional scale model application by linking data layers in a GIS.

6.1.2 Background to the Hungarian Study area

The chosen study area in Central Hungary (24,804 km^2, or approximately 28.7% of the Hungarian land area; Table 6.1) was the same as that used by Falloon et al. (1998b, 1999, 2001a,b).

Table 6.1 Study area

	Lower left of window		Upper right of window	
	X	Y	X	Y
UTM† (34)	325048.59	5183816.64	472979.10	5340133.61
Spherical system	18.708	46.784	20.636	48.213
Hungarian EOV*	624000	160000	768000	320000

† = Universal Terrain Model
= Uniform National Projection of Hungary

6.1.2.1 Land use and cropping

In Hungary, the land use is divided into 50.8% arable, 6.1% plantation crops, 13.1% grassland, 17.9% forests, and 12.1% uncultivated (Nemeth et al. 1997). Major land use changes between 1951-1990 have included a reduction in the arable area (60-51%) and grassland (18-13%), and increases in the areas of plantation crops (3-6%), forests (12-18%), and uncultivated land (7-12%: Nemeth et al. 1997; FAOSTAT, 2000 - see Fig. 6.1). The main crops grown on the arable land are winter wheat (27%) and maize (23%), which make up around 50% of the cultivated area (Nemeth et al. 1997), followed by sunflower (10%), barley (8%) and legumes for silage and forage (6%) (FAOSTAT, 2000). Typical yields for the major crops in 1996 were 5.1 t ha^{-1} for winter wheat, 4.4 t ha^{-1} for winter barley, 5.9 t ha^{-1} for maize, 40 t ha^{-1} for sugarbeet, 2.0 t ha^{-1} for sunflower, and 17.4 t ha^{-1} for potatoes (Nemeth et al. 1997). Dramatic changes in fertiliser applications have occurred in Hungary. Total mineral fertiliser applications (Fig. 6.2; FAOSTAT, 2000) were low in the 1960's at around 60 kg ha^{-1} y^{-1}, rising to around 270 kg ha^{-1} y^{-1} from 1975-1990. After the political changes in 1990, applications fell to around 40 kg ha^{-1} y^{-1}, and have slowly risen

Figure 6.1 Changes in land use in Hungary from 1961-1995 (FAOSTAT, 2000)

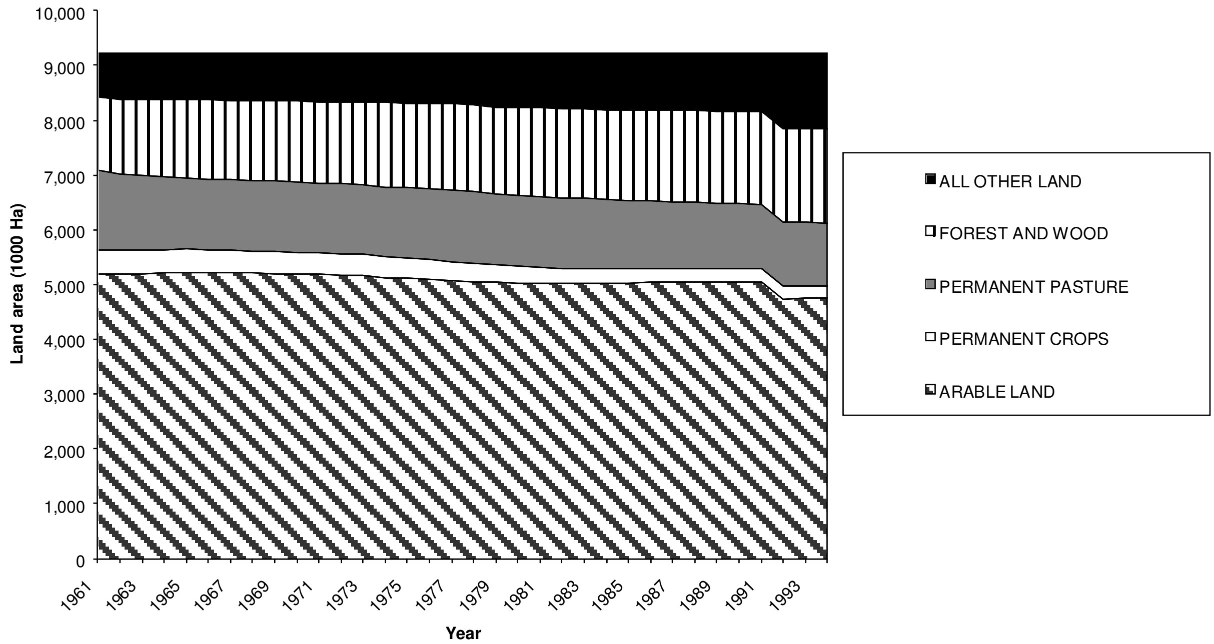
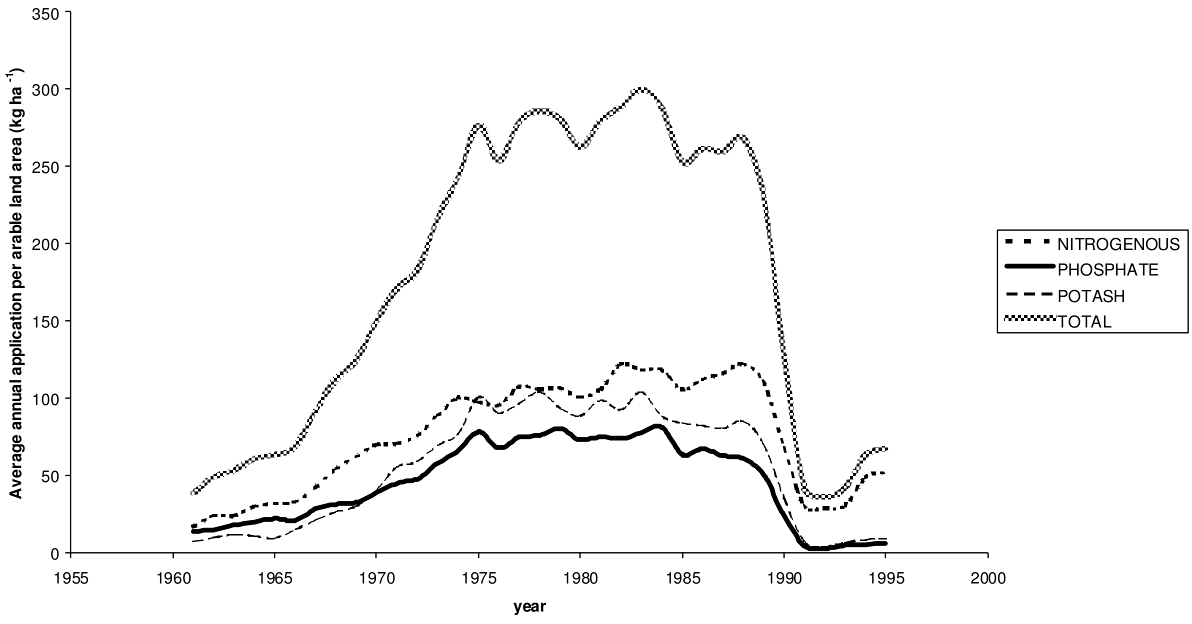


Figure 6.2 Changes in fertilizer applications to arable land in Hungary, 1955-1995 (FAOSTAT, 2000)



since then, to around 70 kg ha^{-1} y^{-1}. The land use of the study area is discussed in more detail in section 6.2.1.2.

6.1.2.2 Soils and climate

The soils and climate of the study region are discussed in greater detail in section 6.2, but general characteristics of Hungarian soils and climate are discussed here. The total annual rainfall in Hungary varies from under 500mm in the Central Hungarian Plain to over 900mm along the Western border. Mean annual temperatures range from 6 ^{o} C in some Northern hilly regions to over 11 ^{o} C in the Southern and Central areas. Total annual potential evapotranspiration ranges from under 650mm in the far Northeast to over 900mm in the Central Plain (Marton et al. 1989). Hungarian soils are very varied and described in detail by Nemeth et al. (1997) and Marton et al. (1989).

6.2 METHODS

6.2.1 Geographic Information System (GIS)

The GIS platforms used were ArcView and ArcINFO.

6.2.1.1 Soils database

Soil data were taken from the 1:500,000 Hungarian HunSOTER database (a country scale database, with an integrated hierarchy of point and polygon layers: Pásztor et al. 1996, Szabó et al. 1996). The study window contained 98 representative soil profiles from a data set of 1361 for the whole of Hungary. The profiles were originally sampled in 1992, and only data for the uppermost horizons were used in this study. This application of this data set differs from that of Falloon et al. (1998b 1999b, 2001a,b) in both the calculation of SOC stocks, and in the treatment of missing data. In this study, SOC stocks were calculated to a standard depth of 20cm (rather than the 30cm used by Falloon et al. (1998b). This depth was used because the CENTURY model simulates the top 20cm of the soil profile. The soil variables used from this data set were organic carbon content (SOC; 0.3-3.1%), sand content (28-97%), silt content (0-16%), clay content (2-60%) and bulk density (0.97-2.08 Mg m^{-3}).

The profile data were linked to the 351 SOTER unit polygons (representing areas with unique soil, land form and lithology characteristics) for the window. Where bulk density data were missing, bulk density was calculated from the SOC value (Howard et al. 1995) and SOC stocks to 20cm depth then calculated from depth, bulk density and SOC%. Soil units with no SOC data were excluded. Two profiles showed very high SOC contents (>150 t C ha^{-1}), and had been sampled from environmentally sensitive areas. These profiles were also excluded from the analysis, since they were unlikely to be relevant to the case study concerning arable land.

This treatment of missing data differs from a former study, where missing data were replaced with a mean over the whole data-set (Falloon et al. 1998b). Finally, these data were used to calculate the soil IOM content for the RothC model in the absence of radiocarbon data using the method described in Chapter 3 (Falloon et al. 1998a), and SOC to 20cm depth.

Tables 6.2 and 6.3 show summary statistics for soil properties from the study area and the full HunSOTER profile dataset, respectively. Figures 6.3 and 6.4 show general soil properties for the study region.

Table 6.2 Summary statistics for soil profiles in the study area

	Mean	Standard deviation	Minimum	Maximum
SOC%	1.33	0.53	0.30	3.10
SOC t/ha	36.09	13.05	8.94	63.00
Sand	58.21	13.24	28.00	97.00
Silt	6.33	3.47	0.00	16.00
Clay	27.32	14.51	2.00	60.00
BD	1.38	0.19	0.97	2.08
pH	7.42	0.84	5.10	10.00

Table 6.3 Summary statistics for all HunSOTER profiles

	Mean	Standard deviation	Minimum	Maximum
SOC%	0.697	0.786	0.00	13.00
SOC t/ha	40.73	33.93	0.00	229.08
Sand	61.65	14.53	0.00	98.00
Silt	5.95	3.77	0.00	47.00
Clay	23.31	11.38	0.00	79.00
BD	1.307	0.485	0.00	3.40
pH	6.921	1.13	0.00	10.30

Figure 6.3 Soil properties in the study region from HunSOTER: a) sand %, b) silt %, c) SOC % and d) clay %

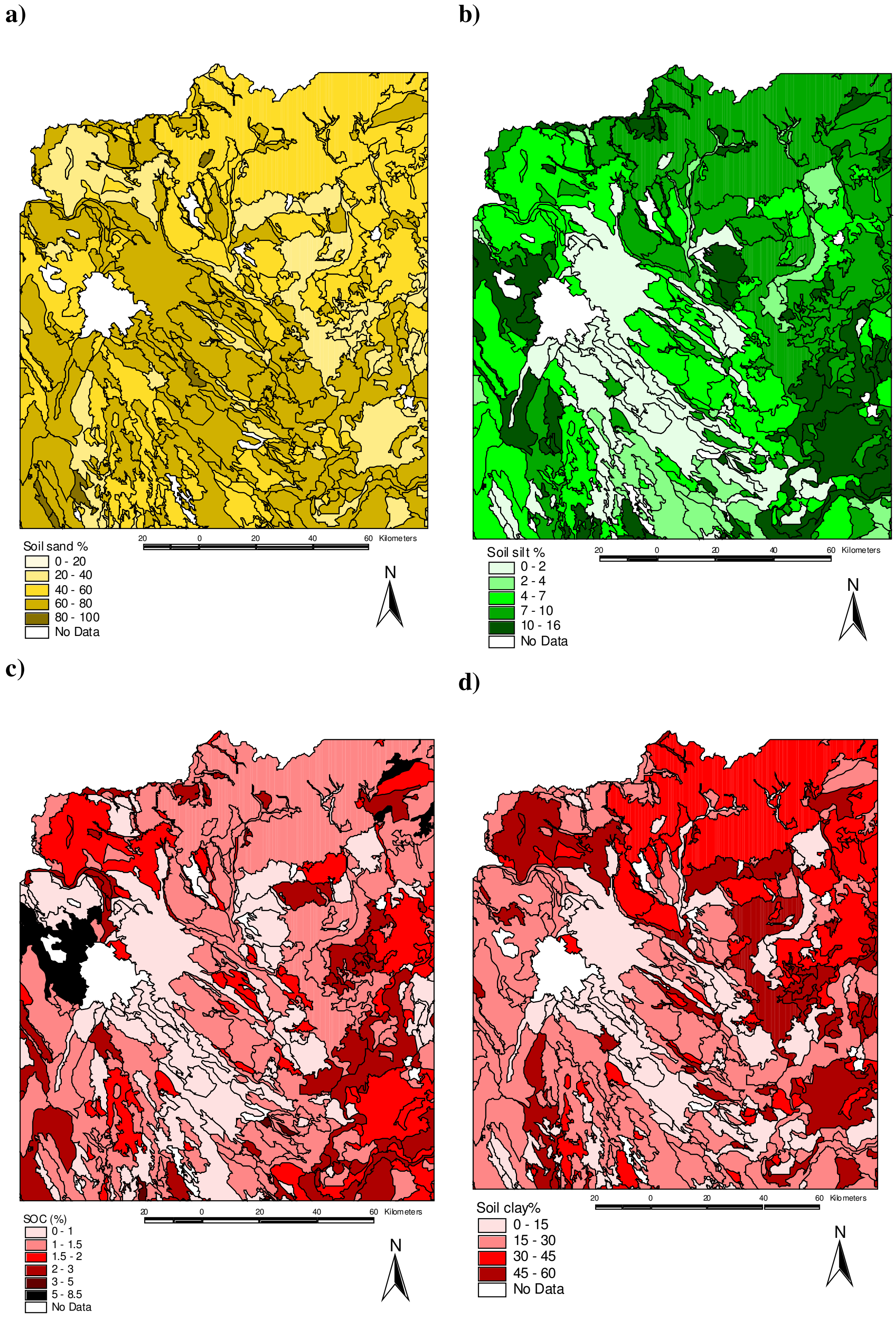
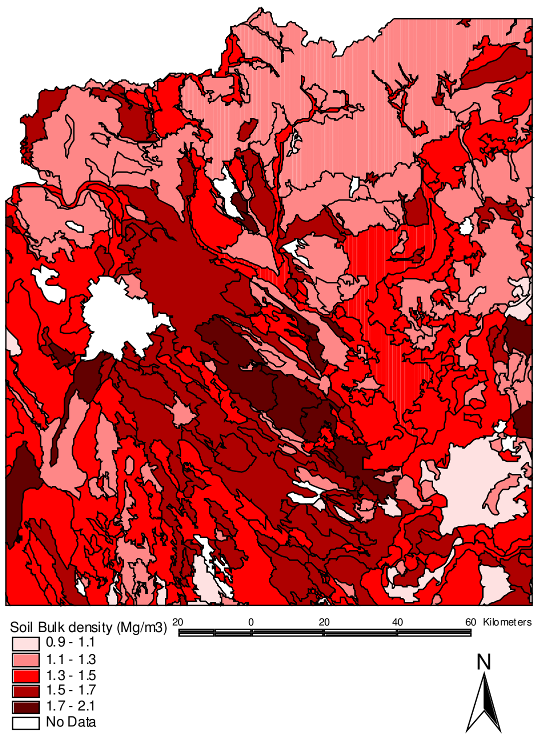
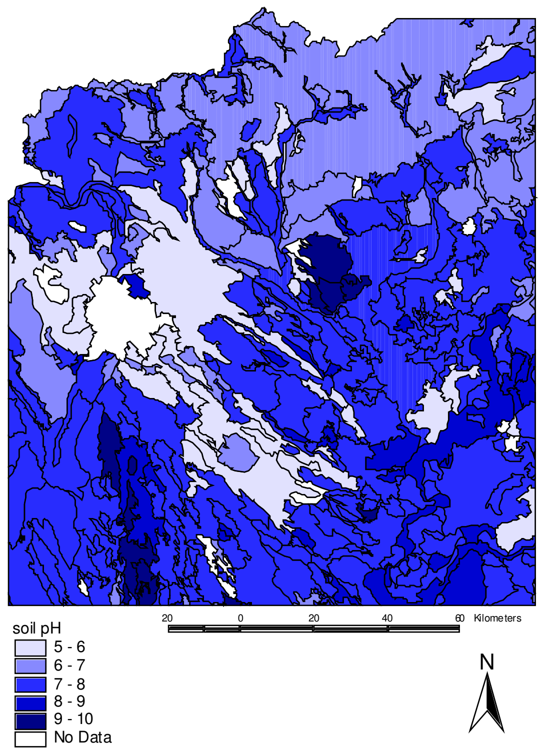


Figure 6.4 Soil properties in the study region from HunSOTER: a) bulk density, b) pH, c) FAO soil type and d) SOC stock

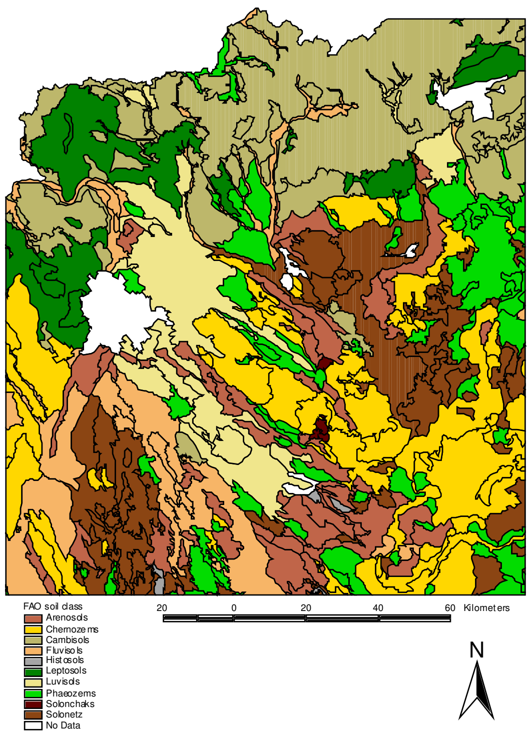
a)



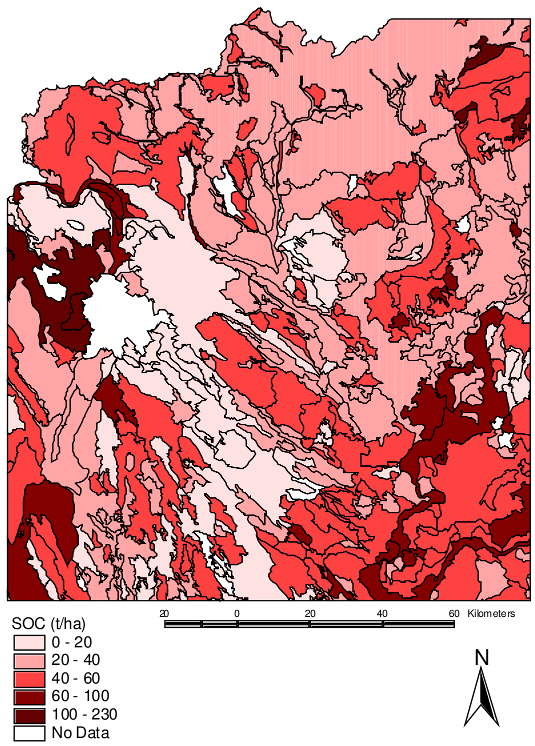
b)



c)



d)



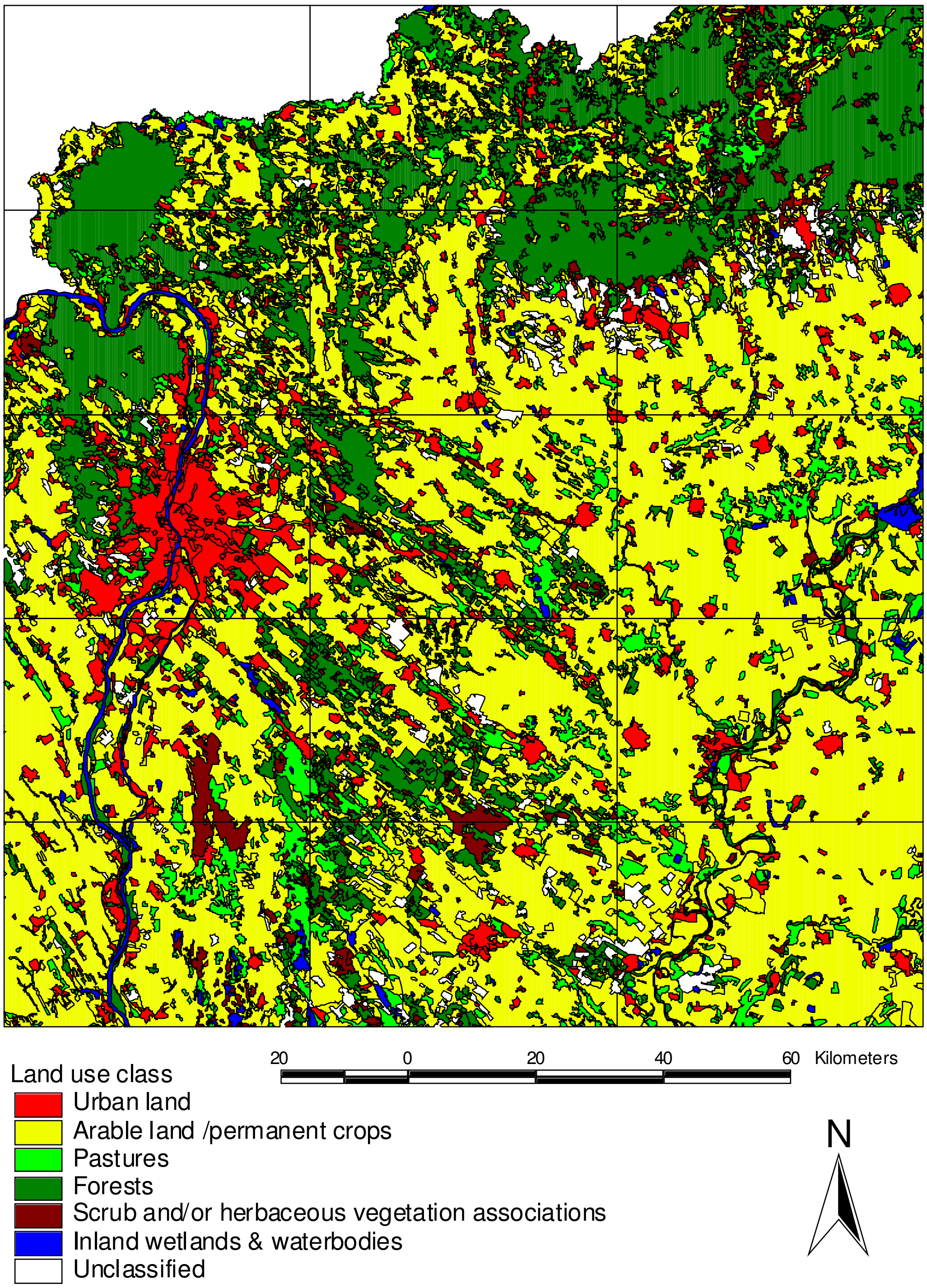
6.2.1.2 Land-use database

Land-use data were taken from the CORINE database for Hungary (scale 1:100,000 - Büttner et al. 1995a,b; Büttner, 1997), for the representative window, with 6470 polygons. Because the original data set contained multi-part polygons, the land-use polygon data were exploded in ArcView using a custom script, giving a total of 7529 polygons. The 44 original land-use codes were rationalised into 4 codes. The rationalised codes were: 1) arable (6 original codes, 21% of total land use), 2) grassland (8 original codes, 8.7% of total land use), 3) forest (6 original codes, 20.6% of total land use) and 4) other uses (including marsh, water bodies, urban areas and so on; 24 original codes, 9.7% of total land use).

6.2.1.3 Meteorological data

In an earlier study (Falloon et al. 1998b), long-term averaged mean monthly temperature (14 stations), rainfall (17 stations), and evaporation (6 stations) data from 1931-1960 (Varga-Haszonits, 1977) were used. For this study, 20 points from the pre-interpolated 1961-1990 gridded surface climatology for Europe were used (Hulme et al. 1995). This data set provided mean monthly temperature and rainfall data. Potential evapotranspiration was estimated for each of these 20 points using the methods in the CENTURY model, and based upon the latitude, and minimum and maximum temperature. The fwloss(4) parameter (a scaling parameter for CENTURY evapotranspiration calculations) was set at 0.7 for these runs, based upon comparison of the estimated total annual potential evapotranspiration values with measured values from the nearest stations in Varga-Haszonits (1977). Using CENTURY-derived evapotranspiration figures allowed maximum comparability between model estimates. Mean annual temperature ranged from 9-10^{o}C over the area, total annual precipitation from 461-590mm, and total annual evaporation from 708-951 mm. The meteorological data was contained within a point layer.

Figure 6.5 Land use classes of the study region from the CORINE database



6.2.2 Models and regressions

The dynamic SOM model - GIS methods made use of the spatial databases described above and the RothC and CENTURY models described in detail in earlier chapters. Estimates derived from these models were also compared with those derived from regression equations based on SOC data from long-term experiments in Europe (Smith et al 1997b, 1998a,c, 1999, 2000a,b,c,d,e, 2001a,b). The scenarios of land management considered in the study are discussed in section 6.2.5.

6.2.3 CO_2 emissions for the area

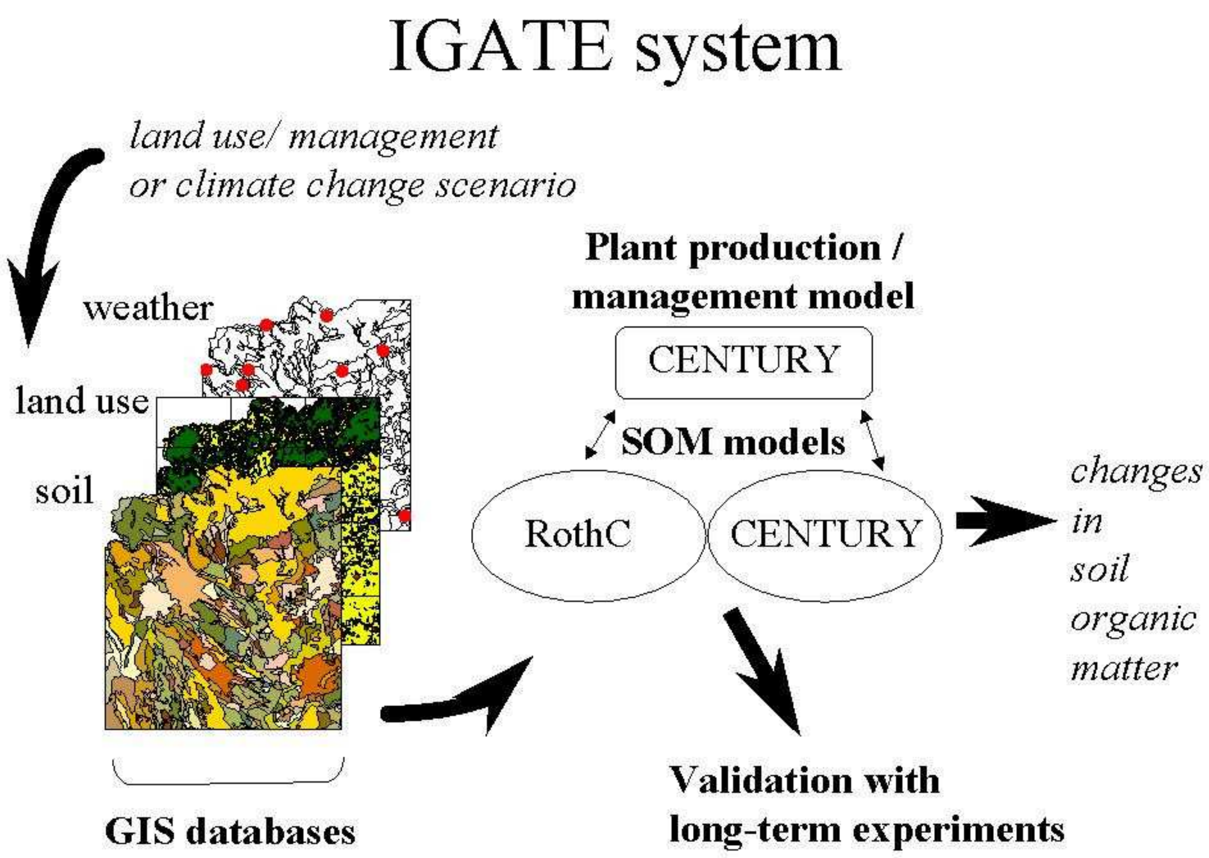
1990 National CO_2 emissions for Hungary of 17.5 Tg C were taken from Marland et al. (1994) since this is the baseline year for Kyoto Protocol reporting. Since the study area was 26.7% of the Hungarian land area the national CO_2 emission figure was scaled to study region (4.7 Tg C), assuming that emissions were equal across the land area of Hungary.

6.2.4 Layer and model linkage

A custom ArcView script was written to find the centres of all polygons in the land use coverage. The polygon centres were then linked to each of the interpolated weather points, using the nearest point to each polygon. This provided a linked land-use/meteorological layer. The soil layer, based on SOTER units, was overlaid upon the linked land-use/meteorological layer, using a custom ArcView script, giving 16360 polygons representing a unique combination of land-use, soil, and meteorology. From these 5756 polygons were actually used in the modelling exercise, meeting the criteria of a) arable land use, b) valid SOC data and c) SOC content <150 t C ha^{-1} (as discussed in section 6.2.1.1).

Interfaces for the RothC and CENTURY models (the IGATE system: Figure 6.6) were written to read input data from an ASCII file produced by the GIS, to create model schedule and batch files, run RothC or CENTURY, and send results to an

Figure 6.6 Schematic diagram of the IGATE system - linkage of dynamic SOM models to GIS databases



ASCII file. The original source codes of RothC and CENTURY were altered to run within this scheme. The RothCGIS interface was designed to allow runs to be executed in one of two ways. Firstly, RothC could be run to equilibrium by iteratively adjusting the C input to soil to fit measured values from the database and then using default values for different scenarios of land management. Secondly, RothC could be run using C input files derived from CENTURY model runs for different scenarios based on the same data. The files containing results from the models were loaded into Excel, Access and GENSTAT for analysis, and into ArcView for visualisation.

6.2.5 Land management scenarios

Six land management scenarios were chosen for estimation of carbon sequestration potential (Table 6.4), plus one baseline arable management scenario. Areas affected by each land management scenario are shown in Figure 6.7, and only involve current arable land in the study area. All scenarios were applied for 100 years. The scenarios chosen were based on those of Smith et al. (1997b, 1998a,c, 1999, 2000a,b,c,d,e, 2001a,b). The basis for each scenario is discussed below.

6.2.5.1 Natural woodland regeneration on 11.95% (surplus) arable land.

Flaig and Mohr (1994) predicted that there would be 20-30% surplus arable land in Europe by 2010, although this is almost certainly an overestimate (Nemeth et al. 1997; FAOSTAT 2000; Smith et al. 2000b). Smith et al. (2000b) used a more realistic 10% of surplus arable land in their calculations of the carbon mitigation potential of European soils by natural woodland regeneration. An ArcView script was therefore used to randomly select 10% of the arable land in the study area, but since the polygons in the GIS were not of a uniform size, selection of exactly 10% was not possible. Instead, the 11.95% arable land selected by the ArcView script was used. Under this scenario, the surplus arable land area was left to natural woodland regeneration. In addition to accumulating SOC, considerable amounts of C accumulate in the aboveground biomass of trees during natural woodland regeneration. Three methods were used to calculate above ground biomass C. Firstly, Jenkinson (1971) calculated that the standing woody biomass accumulated

Table 6.4 Summary details of land management scenarios applied

Scenario	Area Applied to	% of arable area	Initial SOC (Tg C)	Organic amendments	C/N ratio of organic amendments	Management
Afforestation (1)	178643.93	11.95	6.77			Natural woodland regeneration
Ley arable (2)	178643.93	11.95	6.77			2 year ley in 6 year ley arable rotation
No till (3)	1494655.99	100.00	56.42			Arable Conversion to no-till farming
FYM (4)	330405.47	22.11	11.95	10 t ha^{-1} y^{-1} FYM (0.5 t C ha^{-1} y^{-1})	8.33	Arable Incorporation of 10 t ha^{-1} y^{-1} FYM
Sewage Sludge (5)	1494655.99	100.00	56.42	1 t ha^{-1} y^{-1}Sewage Sludge (0.05 t C ha^{-1} y^{-1})	6.67	Arable Incorporation of 1 t ha^{-1} y^{-1} Sewage Sludge
Cereal Straw (6)	1494655.99	100.00	56.42	4.2629 t ha^{-1} y^{-1}Cereal Straw (1.63 t C ha^{-1} y^{-1})	80.00	Arable Incorporation of 4.2629 t ha^{-1} y^{-1} cereal straw
Current (7)	1494655.99	100.00	56.42			Current arable management

Table 6.5 Yearly percent change in SOC applied for regression-based estimates

Scenario	yearly % change in SOC
Afforestation (1)	1.1700
Ley Arable (2)	1.0200
No-till (3)	0.7300
FYM (4)	0.3260
Sewage Sludge (5)	0.4900
Cereal Straw (6)	0.6670
Baseline (7)	0.0000
Sewage Sludge (5) - based on all experiments	0.1816

Table 6.6 C inputs to soil used for RothC default method

	C input (t C ha^{-1} y^{-1})		
Scenario	Arable	Grassland	Woodland
Afforestation (1)	-	-	2.85
Ley arable (2)	1.01	3.00	-
No till (3)	1.01	-	-
FYM (4)	1.01	-	-
Sewage Sludge (5)	1.01	-	-
Cereal Straw (6)	1.01	-	-
Current (7)	1.01	-	-

Table 6.7 Measured and CENTURY modelled C in tree biomass pools for the forest site at Sifokut (DeAngelis et al. 1981).

C pool	Measured (g C m^{-2})	CENTURY modelled (g C m^{-2})
NPP	507	439
Leaves	151.83	154
Litter	187.61	314.58
Wood	8976.15	9006
Roots	1602.00	1727.00

three times the SOC found in natural woodland regeneration experiments at Rothamsted, UK. This relationship was therefore applied to the predicted SOC stock of each polygon for each of the methods 6.2.6.1 to 6.2.6.4 outlined below, and was used in earlier estimates of C mitigation potential in Europe (Smith et al. 1997b, 1998a,c). Secondly, Smith et al. (2000b) used the mean figure used by the IPCC (1996, p.782; Nabuurs and Mohren, 1993) for the average annual net rate of carbon accumulation in broadleaf forests on former agricultural land (range: 2.2-3.4; mean of these figures 2.8 t C ha^{-1} y^{-1}). This rate was also applied to each method a) to d). Thirdly, estimates of the C accumulation in trees were taken from the CENTURY model run described in section 6.2.6.3 below. The forest production characteristics for CENTURY were based upon model runs at Geescroft Wilderness (UK - section 5.3.1.2), but calibrated to a 60-year old Hungarian woodland site at Sifokut from the International Biosphere Project (IBP; DeAngelis et al. 1981) database. Measured and CENTURY-modelled biomass production and allocation figures at this site are given in Table 6.7. Estimates of tree-C accumulation from the spatial runs of CENTURY were also applied to all methods 6.2.6.1 to 6.2.6.4 below. The total carbon mitigation potential of the natural woodland regeneration scenario was calculated as the sum of the SOC accumulation and the tree biomass C accumulation. It is assumed here that this wood-C would be harvested and used for bioenergy use, thus offering additional C mitigation.

6.2.5.2 Ley-arable farming (agricultural extensification) on 11.95% (surplus) arable land.

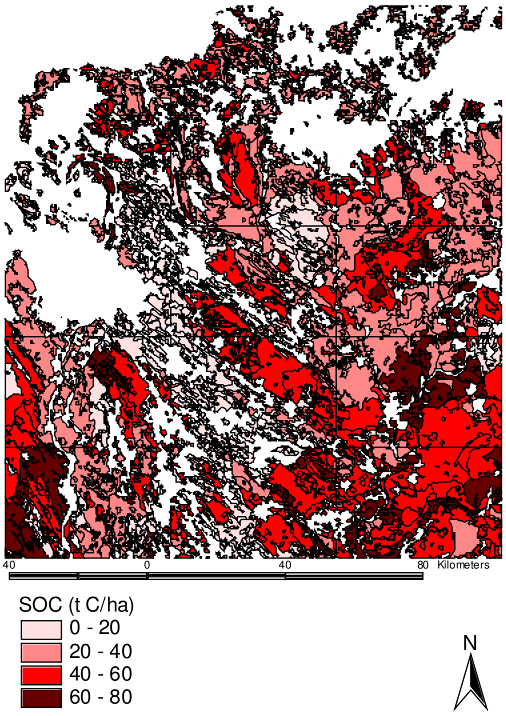
The land area (11.95% arable land) to which this scenario was applied was the same as that used in scenario 6.2.5.1 above, since it also makes use of surplus arable land. The extensification scenario used here involves taking surplus arable land out of intensive agriculture and imposing a ley (grass) arable rotation, with 2 years of a 6-year rotation being under ley (grass) management (Smith et al. 2000b).

6.2.5.3 Conversion of all arable farming to no-till agriculture.

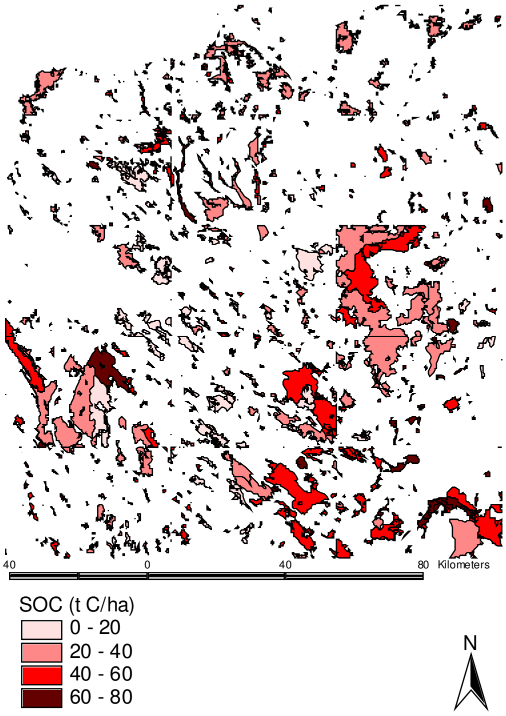
As in Smith et al. (2000b), the area to which this scenario is applied is all arable land. As well as potentially increasing SOC stocks, no-till farming also reduces fossil fuel carbon emissions, even after accounting for the extra herbicides needed (Frye, 1984).

Figure 6.7 SOC stocks from HunSOTER under areas selected for land management scenarios a) all arable land (Scenarios 3, 5, 6 and 7), b) 22.11% of arable land (Scenario 4), c) 11.95% of arable land (Scenarios 1 and 2)

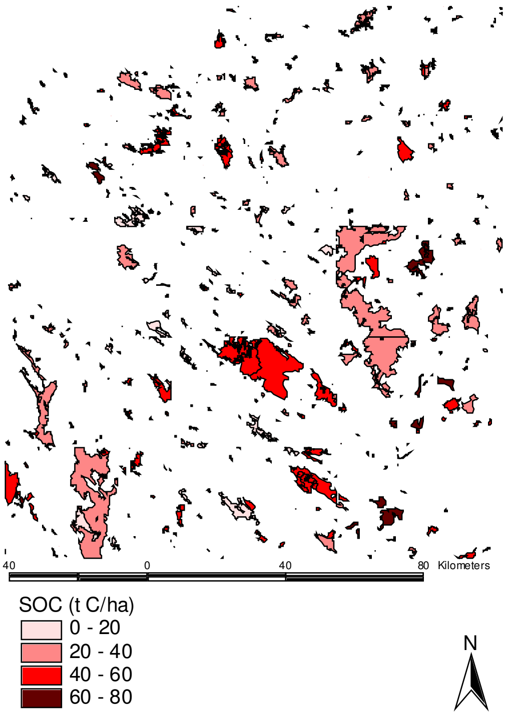
a)



b)



c)



The estimate of fossil fuel C emission reduction used here is 23.8 kg C ha^{-1} (Frye, 1984; Kern and Johnson, 1993). Additional fossil fuel C savings were calculated using this figure for each of methods 6.2.6.1 to 6.2.6.4, and the total carbon mitigation potential calculated as the sum of SOC accumulation and fossil fuel savings. No-till agriculture is much less common in Europe than in North America (Lal et al. 1998; Smith et al. 1998b). Accurate figures on the extent of no till agriculture in Europe are not available (Costa, 1997), but it is known to account for less than 5% of total arable production in many European countries (Costa, 1997), and is practically non-existent in Hungary (Nemeth et al. 1997). The first reason for this is the need for the water storing effect of autumn deep tillage to avoid summer drought stress of row crops. The second reason is avoidance of further drought damage to crops via the mulching effect of no till, which causes soils to warm up more slowly, delaying sowing times (Nemeth et al. 1997). It was therefore reasonable to assume that all benefits of this scenario could be calculated without reference to a baseline area under which no-till currently operates.

6.2.5.4 Incorporation of 10 t ha^{-1} y^{-1} FYM to all arable land.

A rate of 10 t ha^{-1} y^{-1} FYM (0.5 t C ha^{-1} y^{-1}) was chosen since this represents an agronomically beneficial figure (Smith et al. 2000b). The resource availability of FYM was calculated based on estimates of the number of cattle and pigs in Hungary (FAOSTAT, 2000) in the year 2000 (857,000 and 5,335,000, respectively), and average production rates (6.7 t y^{-1} and 1.5 t y^{-1}; MAFF, 1994) at 13,744,400 t. Scaling this to the study area gave a figure of 3,669,754.8 t manure, which could be applied at the rate of 10 t ha^{-1} y^{-1} to 23.904% of the arable land in the case study area. An ArcView script was used to randomly select 23.904% of the arable land in the study area, but since the polygons in the GIS were not of a uniform size, selection of exactly 23.904% was not possible. Instead, the 22.11% arable land selected by the ArcView script was used. Current FYM applications in Hungary are low and falling (<4.9 t ha^{-1} y^{-1}; Csatho and Arendas, 1997, Nemeth et al. 1997), and projecting their figures suggests application rates close to zero for the year 2000.

6.2.5.5 Incorporation of 1 t ha^{-1} y^{-1} sewage sludge to all arable land.

A rate of 1 t ha^{-1} y^{-1} was chosen since it was considered to be an environmentally safe level, based on the lowest recommended maximum rate in Europe (Williams, 1988; Smith et al. 2000b). There is little information available on current sewage sludge application or resource availability in Hungary - it is simply assumed here that there would be enough sewage sludge available to apply to all arable land in the study area as a rate of 1 t ha^{-1} y^{-1}.

6.2.5.6 Incorporation of 4.2629 t ha^{-1} y^{-1} cereal straw to all arable land.

Total cereal straw resource availability was calculated for Hungary from total cereal grain yields in the year 2000 (FAOSTAT, 2000) of 10,657,647 Mt for all of Hungary, with a production area of 2.5 M ha giving an areal production rate of 4.2629 t ha^{-1}. Assuming a harvest index (grain:straw ratio) of 50% (Smith et al. 2000b) indicates a straw resource availability of 4.2629 t ha^{-1} y^{-1} (1.63 t C ha^{-1} y^{-1}). Cereal straw was applied to all the arable land area in the case study, and although this is the maximum achievable amount, current cereal straw incorporation rates in Hungary are very low (Csatho and Arendas, 1997; Nemeth et al. 1997).

6.2.5.7 Baseline arable land management.

This scenario was included for the dynamic model-based estimates since SOC stocks varied widely across the region, and SOC gains and declines could be expected under current management practices, dependent on soil properties. Examples of such SOC changes under arable management are illustrated by the long-term SOC dataset presented in Chapter 5. Therefore calculation of the carbon mitigation potential of land management practices required calculation of changes in SOC stocks under baseline management practices. For each polygon, the change in SOC under the baseline run was subtracted from the change in SOC under a land management scenario to give the net carbon sequestration due to the change in land management from original arable management.

6.2.6 Methods for estimating of C sequestration potential

For each scenario, predictions of carbon mitigation potential were obtained using four methods:

6.2.6.1 Regression-based approach of Smith et al. (1997b, 1998a,c, 1999, 2000a,b,c,d,e, 2001a,b).

Yearly percent changes in SOC under each scenario, calculated from their equations, are given in Table 6.5.

6.2.6.2 RothC default method

Using RothC run to equilibrium, iteratively fitting C inputs to match SOC values from the HunSOTER database. Each land management scenario was then run for 100y, using default C inputs (Table 6.6). The IOM pool was set as described in section 6.2.1.1. C inputs for arable land (1.01 t C ha^{-1} y^{-1}) were set from the mean value required to maintain measured SOC stocks under arable management. C inputs of 2.85 t C ha^{-1} y^{-1} were used for land under natural woodland regeneration, based on runs of RothC at Geescroft Wilderness (Chapter 5 and Coleman et al. 1997); 3.00 t C ha^{-1} y^{-1} was used for grassland management, based on runs of RothC at Park Grass (Chapter 5 and Coleman et al. 1997).

6.2.6.3 CENTURY

Using the CENTURY model. For each scenario, the model was run for 40y under current arable management using SOC values from the HunSOTER database. The purpose of this 40y run was to allow SOC pools to equilibrate (Kelly et al. 1997). Each land management scenario was then run for 100y. Initial values for the SOC pools in the model were set as described in section 5.2.1. Plant production parameters for arable land, grassland and natural woodland regeneration were set based on the CENTURY simulations completed in Chapter 5, and woodland production parameters adjusted to local conditions as described in the details of the natural woodland regeneration scenario above.

6.2.6.4 RothC with CENTURY C inputs

Using the RothC model with C inputs derived from the CENTURY model runs in section 6.2.6.3). For each scenario, the model was run for 40y under current arable management using SOC values from the HunSOTER database. Each land management scenario was then run for 100y. The IOM pool was set as described in section 6.2.1.1.

6.3 RESULTS

6.3.1 C inputs to soil

Average C inputs to soil to maintain measured SOC stocks under arable management, as estimated by the RothC default method were 1.01 t C ha^{-1} y^{-1} (range 0.24-2.54, standard deviation 0.43; Table 6.6). C input values for each scenario as estimated by CENTURY are given in Table 6.8. As would be expected, C inputs to soil predicted by CENTURY were higher than those predicted by RothC. In general, SOC turns over faster in CENTURY than it does in RothC, so a larger net input of C to soil would be required by CENTURY to maintain the same stock of SOC (see discussion, section 5.4). Variations within predicted average C inputs across the study area predicted by each method were quite small.

Table 6.8 C inputs to soil estimated by CENTURY for the Hungarian test area

	Arable				Grassland				Woodland			
Scenario	Mean	Min.	Max.	St. dev	Mean	Min.	Max.	St. dev	Mean	Min.	Max.	St.dev
Afforestation (1)	-	-	-	-	-	-	-	-	3.25	2.29	4.46	0.53
Ley arable (2)	1.43	1.24	1.89	0.12	5.86	5.17	6.99	0.39	-	-	-	-
No till (3)	1.60	1.17	2.20	0.24	-	-	-	-	-	-	-	-
FYM (4)	1.59	1.20	2.20	0.24	-	-	-	-	-	-	-	-
Sewage Sludge (5)	1.60	1.17	2.20	0.24	-	-	-	-	-	-	-	-
Cereal Straw (6)	1.60	1.17	2.20	0.24	-	-	-	-	-	-	-	-
Current (7)	1.60	1.17	2.20	0.24	-	-	-	-	-	-	-	-

6.3.2 Comparison of SOC dynamics predicted by each method

Figures 6.8-6.11 show SOC dynamics under each of the seven scenarios over the 100 y period, for each method (regression, RothC, CENTURY and RothC with CENTURY C inputs, respectively). Each of these figures represents SOC dynamics under the same, typical polygon from the database, with SOC (42.5 t C ha^{-1}) close to the mean value for the region (36.09 t C ha^{-1}). For both the RothC model and the CENTURY model, there was little change in the SOC content of the baseline scenario with time, although SOC predicted by both models decline very slightly,

Figure 6.8 SOC changes for land management scenarios in a typical arable polygon, regression-based estimates

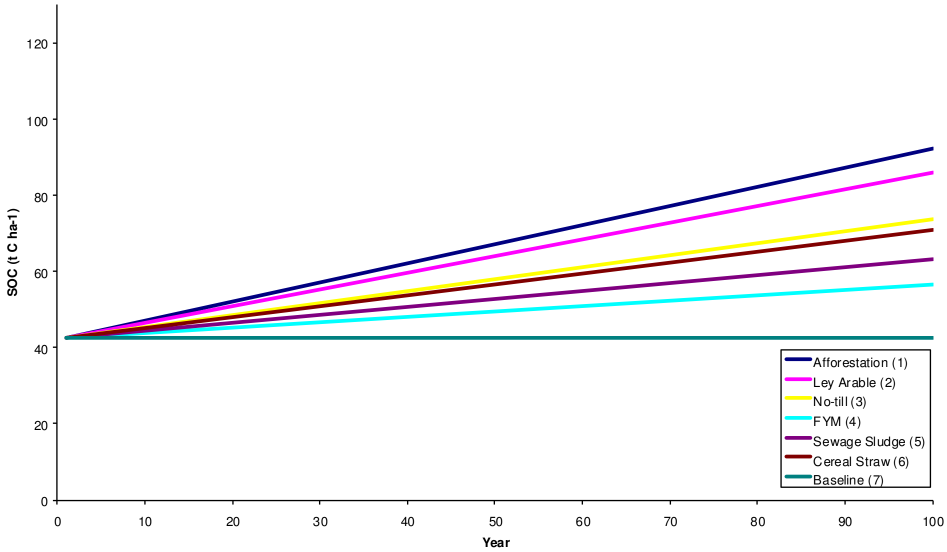


Figure 6.9 SOC changes for land management scenarios in a typical arable polygon, RothC model simulation using default C inputs

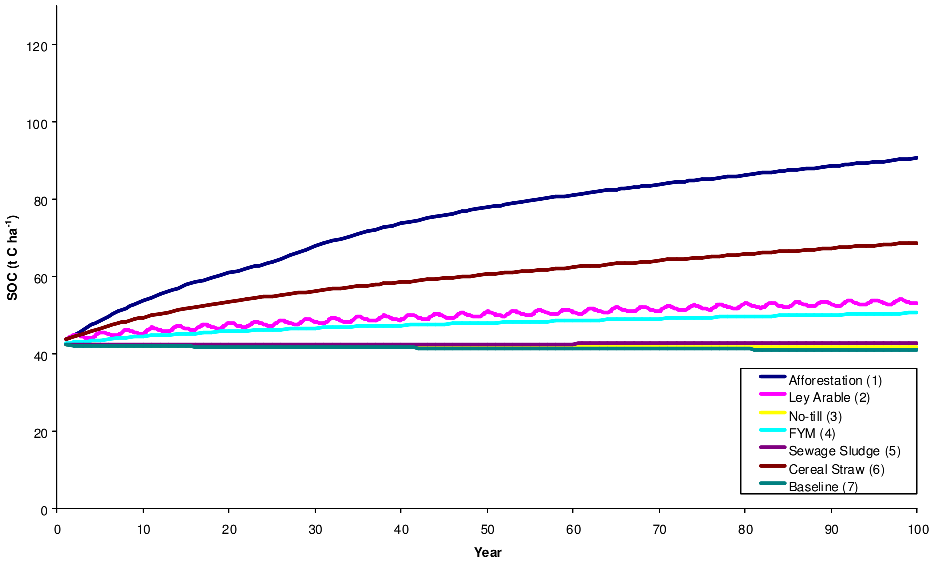


Figure 6.10 SOC changes for land management scenarios in a typical arable polygon, CENTURY model simulation

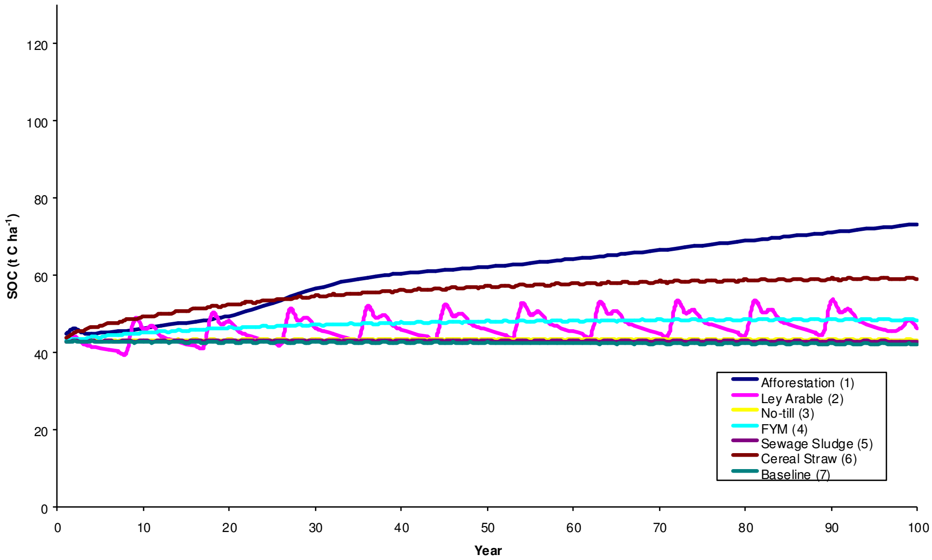
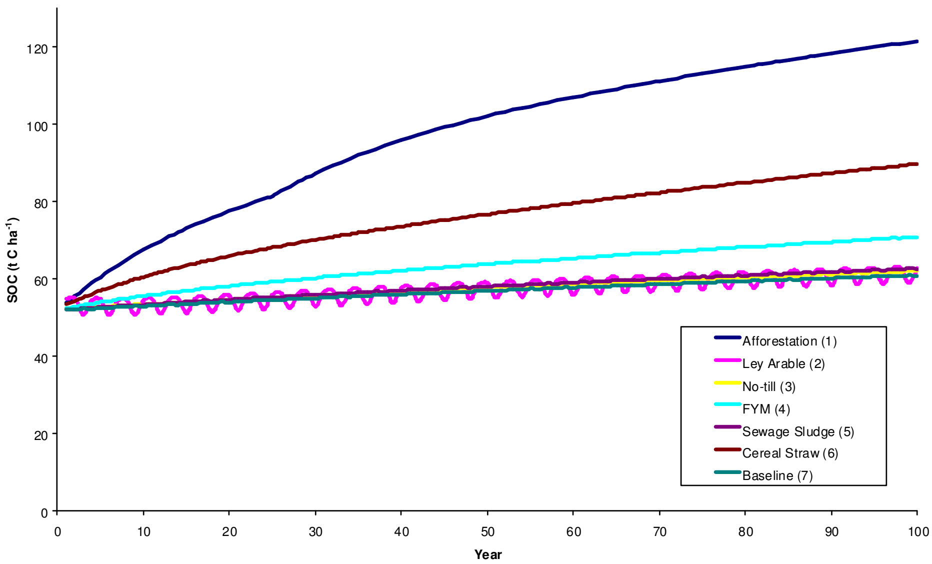


Figure 6.11 SOC changes for land management scenarios in a typical arable polygon, Rothc model simulation using CENTURY C inputs



with a loss of the order of ~1 t C ha^{-1}. When RothC was run using CENTURY C inputs, the baseline SOC tended to increase slightly.

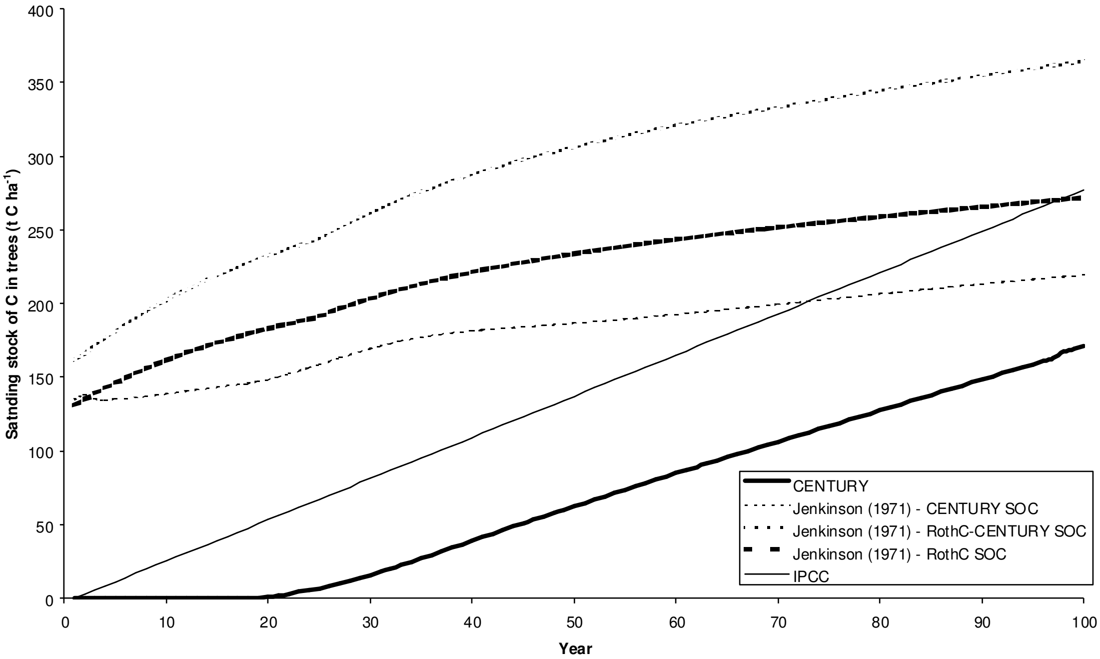
SOC dynamics predicted by the regression were all linear - in contrast with the model-based estimates, which all showed curves approaching a limit towards the end of the 100y period. All methods predicted that the natural woodland regeneration scenario would have the greatest potential in increasing SOC stocks per unit area, with CENTURY, RothC, regression and RothC with CENTURY inputs, in increasing order of magnitude. The order of relative magnitude of SOC increases predicted by RothC and CENTURY were the same, although the regression predicted much greater SOC accumulation under no-till agriculture, ley-arable extensification and sewage sludge incorporation than either model. When RothC was run using CENTURY C inputs, SOC changes tended to be larger than those predicted by RothC or CENTURY alone.

The CENTURY model predicted large variations in SOC stocks under the ley-arable scenario. The reason for this was the predicted additional C inputs from ploughing in the grassland - which were not included in simulations using the RothC default method. Taking the SOC stock under the ley-arable scenario in the final year may be somewhat misleading since there is a large variation in SOC stock between the ley and arable phases of the rotation. It may be more useful to take a running average SOC stock for comparison with baseline runs. Because of this, and because C inputs during the arable phase were slightly lower than those under the baseline run (Table 6.8) SOC under the ley-arable scenario appears to decrease slightly over the 100y time period, as predicted by the RothC model using CENTURY C inputs.

6.3.3 Comparison of approaches for estimating C accumulation in standing above ground biomass for the natural woodland regeneration scenario

Figure 6.12 shows a comparison of the dynamics of above ground biomass C stored in woodland under the natural woodland regeneration scenario. The methods compared are predictions using a) the CENTURY model, b) the IPCC method, c) the method of Jenkinson (1971), based on CENTURY-predicted SOC changes, d) the method of Jenkinson (1971), based on RothC-predicted SOC changes, and e) the method of Jenkinson (1971), based on RothC-predicted SOC changes, with

Figure 6.12 Standing above-ground biomass accumulated in trees under the natural woodland regeneration scenario in a typical arable polygon, predicted by IPCC, Jenkinson (1971), and CENTURY model

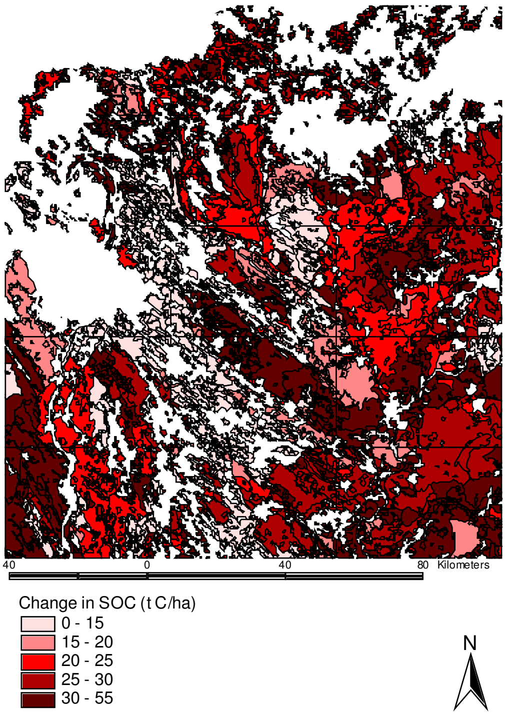


CENTURY C inputs. The comparison is for a typical polygon from the database, with SOC (42.5 t C ha^{-1}) close to the mean value for the region (36.09 t C ha^{-1}).

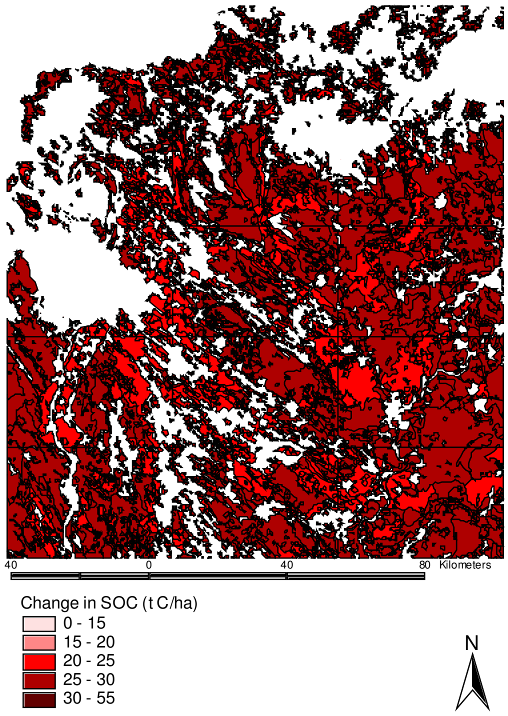
Whilst all estimates are encouragingly of the same order of magnitude, the lowest predicted above-ground biomass increase is that given by the CENTURY model. However, the CENTURY model estimate did use a dynamic tree growth model, calibrated to Hungarian woodland conditions. The CENTURY method also incorporates growth dynamics of the woodland, based upon Kelly et al.'s (1997) simulation of natural woodland regeneration at Geescroft Wilderness. This simulation included the early regeneration period being dominated by herbaceous scrub-like vegetation, hence there is no predicted wood biomass production before year 20 of the scenario. Above-ground biomass C accumulations predicted by Jenkinson's (1971) method depend upon the predicted SOC stock, and as such, predict large (and unrealistic) accumulations of tree C before tree species would dominate the vegetation. The IPCC method predicts a linear increase in aboveground biomass similar in magnitude to the figures predicted using Jenkinson's

Figure 6.13 **SOC changes under the cereal straw incorporation scenario over 100y, predicted by a) regression-based approach, b) RothC model with default C inputs, c) CENTURY model and d) RothC model with CENTURY C inputs**

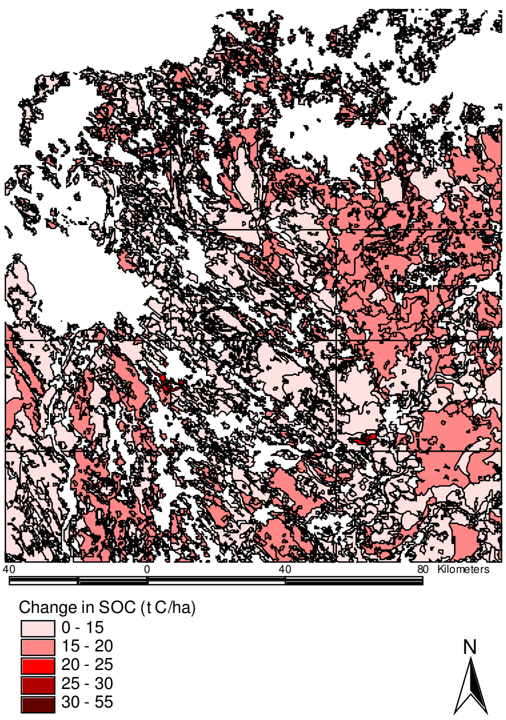
a)



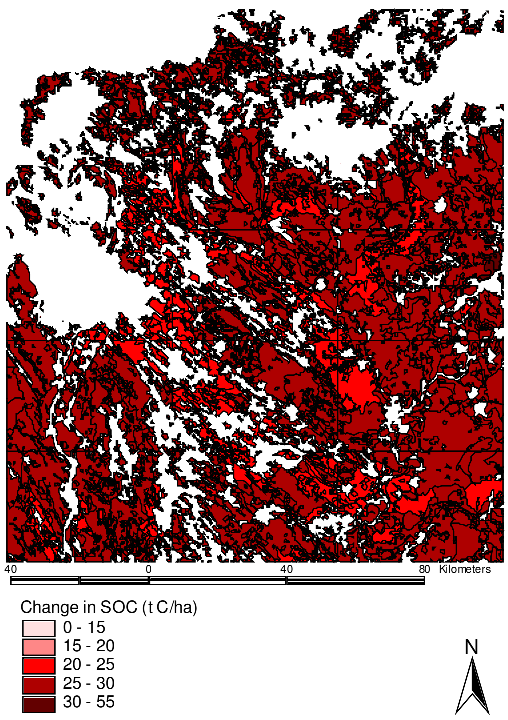
b)



c)



d)



(1971) figures. It is hard to be conclusive about potential forest growth in this region of Hungary since there are limitations to each of the methods presented here. The CENTURY model simulation was only based upon calibration at one site, whilst Jenkinson's (1971) figures were based on a UK woodland, and the IPCC method on average figures for Europe. Taking the upper and lower estimates of above-ground biomass in the IPCC figures (3.4 and 2.2 t C ha^{-1} y^{-1}, respectively) gives figures for this polygon of 333.6 t C ha^{-1} y^{-1} and 217.8 t C ha^{-1} y^{-1} over 100y. These values are close to the range of estimates obtained using CENTURY and Jenkinson's (1971) method.

6.3.4 Spatial variations in SOC sequestration potential

Figure 6.13 shows spatial variations in SOC changes under the cereal straw incorporation scenario over 100y, predicted by a) the regression-based approach, b) the RothC model with default C inputs, c) the CENTURY model and d) the RothC model with CENTURY C inputs. All methods showed some degree of spatial variation in SOC changes. For the regression-based estimate, spatial variations were due to differences in SOC content, since regression estimates were determined from (and therefore directly related to) SOC contents. This implies that soils with larger SOC content have a larger capacity to increase their C storage. This may be explained by soils in the test area with higher SOC contents having higher clay and silt contents (see Figures 6.3 and 6.4), since soil texture is known to be a major factor in determining SOC content in soils. However, this would not be the case for organic soils. In common with the regression-based approach, the three model-based approaches also suggested greater soil carbon sequestration potential in soils with higher clay contents, whilst the SOC sequestration potential of soils in the centre of the area were light textured and showed smaller SOC increases. Other regional scale studies of SOC storage and dynamics for forests in South-western France (Arrouays et al. 1995) and in North American grasslands (Burke et al. 1990) have also showed that clay content was the most important determinant of SOC. The effect of mean annual temperature and total annual rainfall were shown to be less marked. In contrast, at the global scale, the main control on SOC storage is NPP (Burke et al. 1990). The change in SOC over the test area appears to be greater when predicted using the regression, RothC with default C inputs, and RothC with CENTURY C inputs, by comparison with CENTURY alone.

Table 6.9 C mitigation potential of land management scenarios predicted by the regression method

Scenario					Jenkinson (1971)[1]			IPCC[2]			CENTURY[3]		
	Initial total SOC stock (Tg C)	Final total SOC stock (Tg C)	SOC change (Tg C)	SOC change relative to baseline (Tg C)	ACS[4] (Tg C)	TCS[5] (Tg C)	% 1990 ES[6]	ACS (Tg C)	TCS (Tg C)	% 1990 ES	ACS (Tg C)	TCS (Tg C)	% 1990 ES
Afforestation (1)	6.77	14.69	7.92	7.92	44.07	51.99	11.06	50.02	57.94	12.33	23.94	31.86	6.78
Ley arable (2)	6.77	13.68	6.91	6.91	0.00	6.91	1.47						
No till (3)	56.42	97.61	41.19	41.19	3.56	44.74	9.52						
FYM (4)	11.95	15.85	3.90	3.90	0.00	3.90	0.83						
Sewage Sludge (5)	56.42	84.07	27.65	27.65	0.00	27.65	5.88						
Cereal Straw (6)	56.42	94.05	37.63	37.63	0.00	37.63	8.01						
Current (7)	56.42	56.42	0.00	0.00	0.00	0.00	0.00						

Table 6.10 C mitigation potential of land management scenarios predicted by the RothC model (default C inputs)

Scenario					Jenkinson (1971)[1]			IPCC[2]			CENTURY[3]		
	Initial total SOC stock (Tg C)	Final total SOC stock (Tg C)	SOC change (Tg C)	SOC change relative to baseline (Tg C)	ACS[4] (Tg C)	TCS[5] (Tg C)	% 1990 ES[6]	ACS (Tg C)	TCS (Tg C)	% 1990 ES	ACS (Tg C)	TCS (Tg C)	% 1990 ES
Afforestation (1)	6.77	15.54	8.77	8.81	46.61	55.43	11.79	50.02	58.83	12.52	23.94	32.75	6.97
Ley arable (2)	6.77	8.75	1.98	2.13	0.00	2.13	0.45						
No till (3)	56.42	56.29	-0.13	1.04	3.56	4.59	0.98						
FYM (4)	11.95	14.93	2.97	2.97	0.00	2.97	0.63						
Sewage Sludge (5)	56.42	57.67	1.25	2.41	0.00	2.41	0.51						
Cereal Straw (6)	56.42	94.12	37.70	38.67	0.00	38.67	8.23						
Current (7)	56.42	55.24	-1.17	0	0.00	0.00	0.00						

[1] Additional C accumulation in standing woody biomass calculated according to Jenkinson (1971)
[2] Additional C accumulation in standing woody biomass calculated according to IPCC (1996)
[3] Additional C accumulation in standing woody biomass calculated by CENTURY model
[4] ACS= additional carbon sequestration
[5] TCS=Total carbon sequestration
[6] %1990 ES = % 1990 emissions offset

Table 6.11 C mitigation potential of land management scenarios predicted by the CENTURY model

Scenario					Jenkinson (1971)[1]			IPCC[2]			CENTURY[3]		
	Initial total SOC stock (Tg C)	Final total SOC stock (Tg C)	SOC change (Tg C)	SOC change relative to baseline (Tg C)	ACS[4] (Tg C)	TCS[5] (Tg C)	% 1990 ES[6]	ACS (Tg C)	TCS (Tg C)	% 1990 ES	ACS (Tg C)	TCS (Tg C)	% 1990 ES
Afforestation (1)	5.82	10.05	4.23	4.60	30.16	34.76	7.40	50.02	54.62	11.62	23.94	28.54	6.07
Ley arable (2)	5.82	6.08	0.25	0.62	0.00	0.62	0.13						
No till (3)	47.30	45.15	-2.15	1.32	3.56	4.88	1.04						
FYM (4)	10.23	11.55	1.31	1.99	0.00	1.99	0.42						
Sewage Sludge (5)	47.30	44.77	-2.53	0.94	0.00	0.94	0.20						
Cereal Straw (6)	47.30	66.04	18.74	22.21	0.00	22.21	4.72						
Current (7)	47.30	43.83	-3.47	0	0.00	0.00	0.00						

Table 6.12 C mitigation potential of land management scenarios predicted by the RothC model (C inputs from CENTURY)

Scenario					Jenkinson (1971)[1]			IPCC[2]			CENTURY[3]		
	Initial total SOC stock (Tg C)	Final total SOC stock (Tg C)	SOC change (Tg C)	SOC change relative to baseline (Tg C)	ACS[4] (Tg C)	TCS[5] (Tg C)	% 1990 ES[6]	ACS (Tg C)	TCS (Tg C)	% 1990 ES	ACS (Tg C)	TCS (Tg C)	% 1990 ES
Afforestation (1)	7.94	17.24	9.30	8.20	51.71	59.91	12.75	50.02	58.22	12.39	23.94	32.14	6.84
Ley arable (2)	7.94	9.16	1.22	-0.45	0.00	-0.45	-0.10						
No till (3)	66.84	76.90	10.06	1.43	3.56	4.98	1.06						
FYM (4)	14.28	19.50	5.22	3.12	0.00	3.12	0.66						
Sewage Sludge (5)	66.49	78.00	11.51	2.51	0.00	2.51	0.53						
Cereal Straw (6)	66.49	116.11	49.62	40.27	0.00	40.27	8.57						
Current (7)	66.49	75.47	8.98	0.00	0.00	0.00	0.00						

[1]Additional C accumulation in standing woody biomass calculated according to Jenkinson (1971)
[2] Additional C accumulation in standing woody biomass calculated according to IPCC (1996)
[3]Additional C accumulation in standing woody biomass calculated by CENTURY model
[4]ACS= additional carbon sequestration
[5]TCS=Total carbon sequestration
[6]%1990 ES = % 1990 emissions offset

Figure 6.14 Total C mitigation potential of land management scenarios, predicted by regression-based estimates (line represents quantified emission limitation or reduction commitment for Hungary)

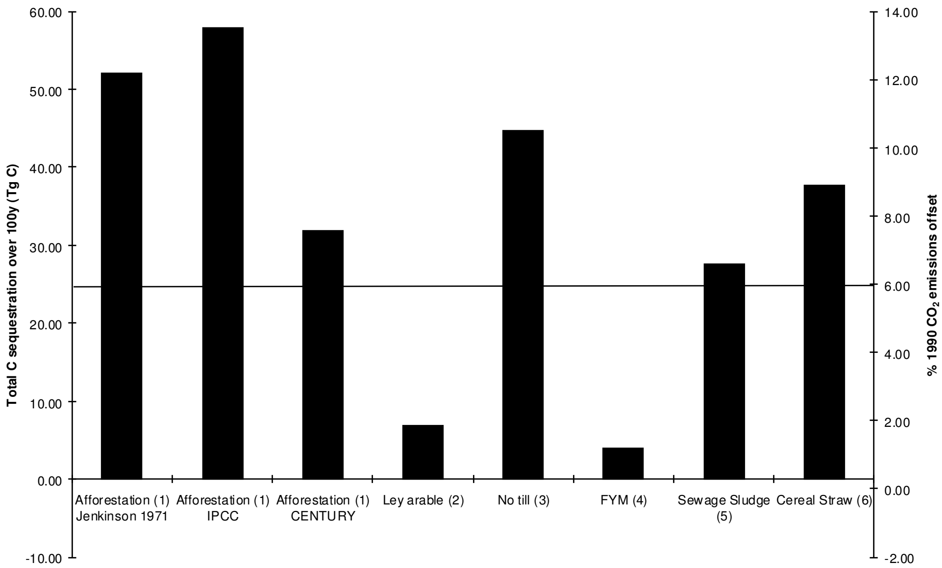


Figure 6.15 Total C mitigation potential of land management scenarios, predicted by RothC with default C inputs (line represents quantified emission limitation or reduction commitment for Hungary)

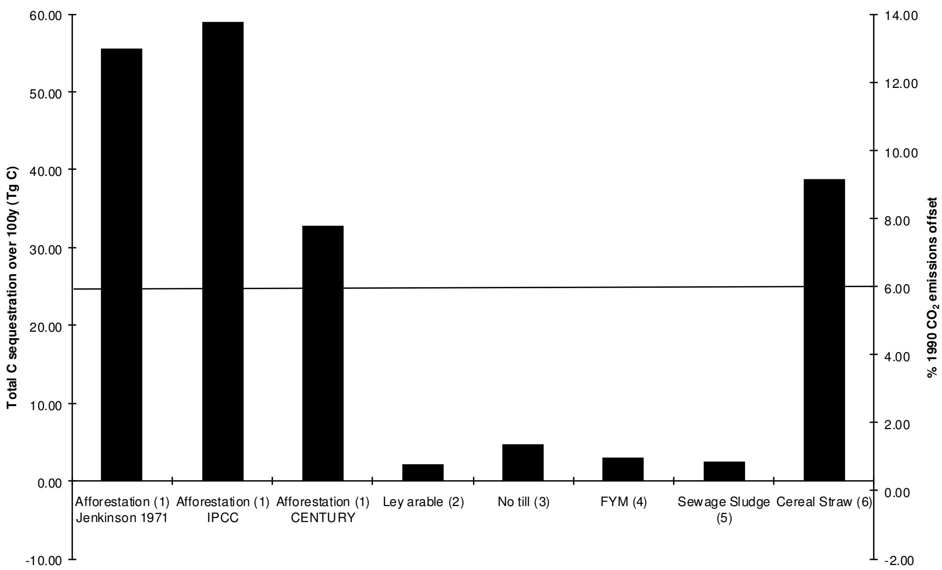


Figure 6.16 Total C mitigation potential of land management scenarios, predicted by CENTURY model (line represents quantified emission limitation or reduction commitment for Hungary)

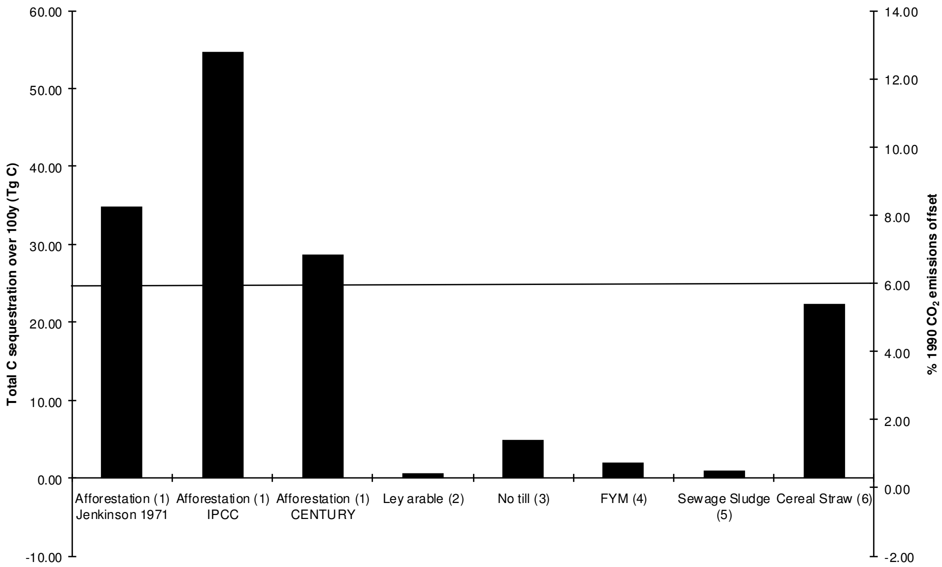
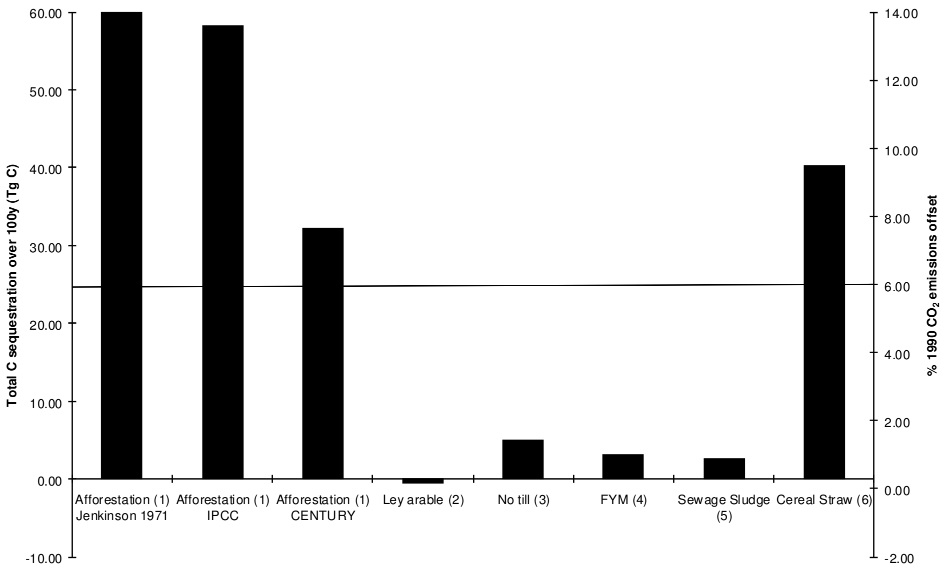


Figure 6.17 Total C mitigation potential of land management scenarios, predicted by RothC model with CENTURY C inputs (line represents quantified emission limitation or reduction commitment for Hungary)



6.3.5 Comparison of total C mitigation potential for each scenario

Tables 6.9-6.12 and Figs. 6.14-6.17 show the C mitigation potential of each of the land management scenarios studied, predicted by each method. The line on each of Figs. 6.14-6.14 line represents the quantified emission limitation or reduction commitment for Hungary, at 6% of 1990 levels. As discussed in section 6.2.1.7, the additional C mitigation potential of the natural woodland regeneration scenario was calculated using three methods, Jenkinson (1971), IPCC (1996) and the CENTURY model.

Firstly, most of the SOC sequestration potential estimates predicted by the regression-based approach are larger than those predicted by dynamic SOM models. This is notably the case for the sewage sludge incorporation and no-till scenarios, and to a lesser extent, the ley-arable extensification scenario. RothC (using default C inputs) predicts a greater SOC accumulation under natural woodland regeneration than does CENTURY. Predictions from RothC using CENTURY C inputs tend to be greater than those predicted using RothC or CENTURY alone.

For predictions made using the regression for sewage sludge application to land, Smith et al. (2000b) linearly interpolated from the experiment with the lowest application rate (6.5 t ha^{-1} y^{-1}) to a rate of 1 t ha^{-1} y^{-1}. This approach was also used here rather than the regression relationship using the whole dataset. However, this particular experiment did show an unusually high annual SOC accumulation rate per tonne of sewage sludge applied, when compared with the other experiments. Taking a mean of the annual accumulation rate for 1 t ha^{-1} y^{-1} sewage sludge over all experiments (Table 6.5) gives a much lower figure. It may be that the regression-based estimate for sewage sludge application to land presented here is still unrealistically high.

The importance of the baseline runs for dynamic model approaches is emphasised by the SOC changes under baseline runs predicted by the three modelling approaches. Both RothC (default C inputs) and CENTURY predicted a small decrease in the total SOC stock of the area under baseline management, whilst RothC (using CENTURY inputs) predicted an increase of total SOC stocks. If the results from the baseline run were not subtracted from the scenario runs, the effect of these changes would be to magnify the SOC sequestration potential of estimates from the RothC model (using CENTURY C inputs) and diminish the SOC sequestration potential of the estimates obtained from RothC (using default C inputs) or CENTURY. As discussed in Chapter 5, the reason for this difference between SOC changes predicted by RothC using default (or fitted) C inputs to soil and RothC with C inputs from CENTURY is the faster overall turnover of SOC in the CENTURY model.

When calculating the total C mitigation potentials, differences between the four approaches are less marked for those scenarios including an additional C saving, such as fossil fuel savings under no-till agriculture, or above-ground biomass production under natural woodland regeneration. There were large differences between the three different approaches used for estimating above-ground biomass C accumulation for the natural woodland regeneration scenario, and this has large effects on the total C mitigation potential thus calculated.

Finally, the total C mitigation potential of some scenarios shows great promise in meeting Hungarian commitments to reduce CO_2 emissions. In the case of the natural woodland regeneration scenario, the emissions reduction target of 6% of 1990 CO_2 emissions is more than met, for all four methods. Predicted total C mitigation potential under the cereal straw incorporation scenario also meets the reduction target when predicted by all methods, with the exception of the CENTURY model.

6.4 DISCUSSION AND CONCLUSIONS

This chapter has presented a regional scale comparison of four methods for estimating C sequestration potential for an arable area of Central Hungary. A GIS database has been compiled, with soil, land use and meteorological layers, to provide regional-scale input data for dynamic SOM models. GIS interfaces have been created for the RothC and CENTURY models, allowing them to be applied and compared at the regional scale. A set of agricultural land management scenarios for carbon sequestration has been developed for the case study area of Hungary, and estimates of carbon mitigation potential obtained using the four different methods. This study has shown that the methods used here to calculate carbon sequestration potential do provide different estimates, although they are largely within the same order of magnitude. Encouragingly, some scenarios alone showed sufficient carbon mitigation potential to meet Hungarian CO_2 emissions reduction commitments.

Some of the scenarios applied may, however, require refinement, whilst others may not be considered practical or beneficial. For example, no-till agriculture is extremely scarce in Hungary, and rarely practised for reasons of soil water conservation (Nemeth et al. 1997). There is little information on application of sewage sludge to agricultural land in Hungary, or on resource availability. For these reasons, the sewage sludge incorporation and no-till scenarios may not be entirely appropriate as presented here. Since the different scenarios were applied to either the entire area of arable land, or a randomly selected portion, no considerations have been made for the land suitability for the different practices applied. This is particularly relevant when considering scenarios applied to surplus arable land, such as natural woodland regeneration or ley arable farming (extensification). It is unlikely that the most productive areas would be taken out of production, and detailed spatial information on land quality, agricultural suitability and environmental sensitivity would be required in order to select the most likely areas to be used for each scenario.

Each of the four approaches applied to estimate regional scale carbon mitigation potential in this study have their own merits and disadvantages. The regression-based approach has very low data requirements, is simple to apply, and calculations could even be made without the use of spatial databases. It is, however, limited by the number of long-term experiments available to derive the regression equations, and is an average approach for European conditions. It is not specific to particular geographic areas, and only represents carbon mitigation potential of the experiments upon which it was based. When applied without spatial databases, the regression approach does not allow spatial identification of geographic areas with the greatest carbon sequestration potential.

All of the modelling approaches used have large data requirements, in terms of both spatial data for regional application, and long-term experimental data for site-scale model validation. Climate data is rarely the limiting factor for dynamic model applications at the regional level (Paustian et al. 1997d). However, since models like CENTURY predict vegetation factors from climate and soil factors, soil data is the often the most limiting data. For many regions, basic soils data (e.g. SOC rather than SOM, water holding capacity and bulk density) are often not directly available in spatial databases. Furthermore the samples held in spatial databases may not provide an appropriate representation of land use and land use history in the region (Paustian et al. 1997d). There is also little information on the relative distribution of SOC between model pools at the regional level, or how differences in pool setting could affect outputs.

The validity of dynamic model approaches ultimately depends on successful model validation over appropriate timescales at experimental sites that pertain to the scenarios of interest and the climatic-geographic conditions of the region in question (Paustian et al. 1997d). Estimates derived from dynamic SOM models are, however, based on mechanistic models that represent state-of-the-art understanding of SOC turnover process. Model-based estimates also account for the effects of climate, soil properties and land use on SOC storage and allow spatial identification of geographic areas with the greatest carbon sequestration potential. Since SOC storage is controlled by a variety of biogeophysical, climatic, and management factors, dynamic models which integrate the main mechanisms governing SOC turnover are the most suitable tools for predicting changes (Paustian et al. 1997d). Model based approaches are also very flexible, and are amenable to a range of different scenarios.

The use of the RothC model with default C inputs effectively considers only the influence of differences in soil properties (SOC stock and texture) and climate on potential SOC changes, since the same value for C input across the study area was used. Spatial information on crops, yields, harvesting, residue treatment, grassland plant biomass C and SOC contents, and forest tree biomass C, litterfall and SOC contents might allow C inputs to be spatial characterised for RothC without the use of a dynamic plant production model such as CENTURY. Such datasets are, however, both scarce and difficult to compile. Comparison of C inputs estimated by RothC at the site (Tate et al. 1995) and national (Parshotam et al. l995) scales for grasslands and forests in New Zealand suggests good agreement, despite a small number of sites and a wide range of climatic and edaphic conditions.

Plant production parameters for the CENTURY model approach in this study were based on the CENTURY model applications at site-scale long term experiments performed in Chapter 5, as well as information on tree growth. The plant production parameters were assumed to be the same across the study area - so differences in production and C input rates were caused by local differences in soil and climate. This enabled a simple consideration of the spatial variation of C inputs to soil. A more in-depth application would require detailed experimental information on plant production characteristics (for crops, grassland and forest) for the region. Kelly et al. (1997) also noted that the CENTURY model was limited in its applicability to forest SOC simulations since it cannot accurately simulate SOM accumulation in a forest with a well developed litter layer. The reason for this is that highly decomposed litter that actually remains on the surface is automatically transferred to the 'slow SOM', sometimes leading to unrealistically high build up of SOM.

The scaling parameter (fwloss(4)) for potential evapotranspiration estimates used in CENTURY was set to 0.7 across the whole region, based on comparison of CENTURY estimates with available measured estimates (Varga-Haszonits, 1977). It may be that setting different values of fwloss(4) across the region would result in better local estimates of evapotranspiration.

Finally, the use of RothC with C inputs from CENTURY tended to predict higher values of SOC sequestration than either RothC (using default C inputs) or CENTURY alone. The reasons for these differences have been discussed in detail in Chapter 5.

7. GENERAL DISCUSSION AND CONCLUSIONS

7.1 REVIEW OF ROTHC AND CENTURY

The review of RothC and CENTURY highlighted differences and similarities between the two models. Both RothC and CENTURY are multi-compartmental dynamic SOM models, with the main difference being that SOM turnover in CENTURY is simulated within the CENTURY general ecosystem model. The CENTURY ecosystem model contains representations of plant growth dynamics and management effects, whilst RothC is purely a soil process model. There are also differences in the maximum decomposition rate constants of the SOM pools in each model, in the relationships that determine how factors such as temperature, moisture and soil texture affect decomposition, and in the flows of C between model pools.

A review of former model validation showed that whilst both models have been extensively validated using datasets from long-term experiments in different climatic zones, there is a lack of validation for particular ecosystems (e.g. forests) and management regimes (e.g. no-till agriculture and sewage sludge applications to land). Whilst RothC and CENTURY have been compared with common datasets at the site scale, RothC has not been tested with C input data derived from CENTURY model simulations at the same location. Finally, RothC and CENTURY simulations have not been compared at the regional scale with common datasets.

The use of RothC and CENTURY in a predictive mode (i.e. for regional scale estimates of SOC sequestration) requires estimation of a) IOM and plant inputs of C for RothC, and b) SOM pool size distributions for CENTURY. Predictive, regional application of RothC and CENTURY also requires model validation using site-scale, long-term datasets relevant to the scenarios and region to be studied.

7.2 THE ROLE OF REFRACTORY SOIL ORGANIC MATTER POOLS IN MODELS

In Chapter 3, the role of the most refractory SOM pools in RothC and CENTURY was investigated. In RothC, this is the IOM pool, and in CENTURY, it is 'passive

SOM'. IOM in RothC is a small and truly inert pool (effectively a constant), whilst 'passive SOM' in CENTURY is a larger pool that turns over very slowly. IOM values for RothC may be set precisely if soil radiocarbon dates are available at the site in question. However, there are few sites where soil radiocarbon measurements have been made, and there are no regional datasets on soil radiocarbon age (Falloon and Smith, 1998, 2000). A statistical analysis of IOM values from 28 sites where radiocarbon dates had been used to set the IOM pool was therefore conducted, against soil, management and climate parameters. This resulted in a simple regression relationship for estimating IOM from total SOC (Falloon et al. 1998a). This novel method contributes to the use of RothC in a predictive mode.

In CENTURY, 'passive SOM' is often either set using default values (e.g. Metherell et al. 1993) or indirectly, from simulations of steady-state SOC content. Long-term CENTURY simulations were used to investigate the influence of soil clay content, total annual rainfall, mean annual temperature, and management practices on 'passive SOM' values. This also yielded an equation for estimating 'passive SOM' values from total SOC content and clay content, therefore contributing to the use of CENTURY in a predictive mode.

Simple sensitivity analyses (Section 3.4) were used to demonstrate the importance of IOM (Falloon et al. 2000) and 'passive SOM' values to studies of SOC sequestration.

7.3 ESTIMATING PLANT INPUTS OF C TO SOIL

As discussed in section 7.1, predictive, regional use of RothC requires estimation of plant C inputs to soil. Information from 60 long-term experiments was collected and used to investigate relationships between the C input needed to maintain SOC stocks at equilibrium predicted by RothC and other factors, such as management, climate and soil type. Great variability between estimated C inputs was observed, and reflected the variation in soil types, climatic regimes, and management practices at the sites modelled. C inputs were found to be significantly correlated with soil clay content, FAO soil code, soil pH, mean annual precipitation, SOC and IOM. However, a multiple linear regression including the most significant factors (mean annual precipitation, FAO soil code, soil clay content and soil pH) only accounted for 28.2% of the variance in C inputs.

Since the project aim was to model C sequestration under changes in land use and management, this required estimates of annual C inputs under different land uses. The mean plant inputs for different land uses were as follows: 3.55 t C $ha^{-1}y^{-1}$ for arable, 3.72 t C $ha^{-1}y^{-1}$ for grasslands, and 7.09 t C $ha^{-1}y^{-1}$ for forestry (Falloon et al. 1998b). These figures are similar to the values suggested in the introduction and literature review, of around 1-2 t C $ha^{-1}y^{-1}$ for arable, 2-4 t C $ha^{-1}y^{-1}$ for grasslands (Jenkinson and Rayner, 1977; Paustian et al. 1990), and up to 10 t C $ha^{-1}y^{-1}$ for forestry (Batjes and Sombroek, 1997). It was concluded that a more precise method of estimating C inputs to soil should be used for future studies. This could include the use of a dynamic NPP model to predict C inputs to soil, or the use of a plant production model from another SOM model, such as CENTURY.

Sensitivity analyses (Section 4.3) showed that although the quality of C inputs to soil did have some effect on predicted total SOC values, the quantity of C input had a far greater effect. Errors in C input to soil could have potentially large effects on regional scale SOC predictions since C input is the main determinant of SOC stocks.

7.4 EVALUATING ROTHC AND CENTURY WITH LONG-TERM EXPERIMENTAL DATA

As discussed in section 7.1, predictive, regional use of RothC and CENTURY requires evaluation of the performance of both models with datasets from long-term experiments relevant to the scenarios of interest to the regional scale study, and relevant to the geographic study region. Datasets from seven long-term experiments were therefore collected and used to evaluate the performance of both models.

Neither model was specifically adapted to simulate the addition of sewage sludge to agricultural land, so modified model versions were tested and developed based on simulation of the dataset at Ultuna, Sweden. The standard version of RothC is not set up to simulate no-till soils, so a modified version was created for testing with no-till soils, with modifications based on the CENTURY model. A program (READSITE) was also developed to create land management and weather files for RothC based on outputs from CENTURY model simulations. This allowed a comparison of three types of model run for each site 1) CENTURY model alone, 2)

RothC model run to fit measured SOC values, by iteratively adjusting C inputs to soil, and 3) RothC model run using C inputs derived from CENTURY runs. This technique facilitated the use of RothC in a predictive mode at both site and regional scales. This also enabled an assessment of the differences between using RothC with fitted C inputs and with C inputs derived from CENTURY.

In general, the performance of both models was good across all datasets, which included two arable sites in Hungary. The runs using RothC (iteratively changing C inputs to fit measured SOC values) tended to have the best fit to model data, since this method involved direct fitting to observed data. C inputs estimated by RothC were, in general, lower than those estimated by CENTURY, since SOC in CENTURY tends to turn over faster than SOC in RothC. The runs using RothC with CENTURY C inputs tended to have the poorest fit of all. This was because CENTURY predicted greater C inputs than were required by RothC to maintain the same SOC content. However, the order of magnitude of predicted C inputs was similar.

The use of default methods for estimating initial SOC pools in RothC and CENTURY may not always be appropriate and may require adjustment for specific sites (e.g. Paustian et al. 1992). The scaling parameter fwloss(4) used in the calculation of potential evapotranspiration with the CENTURY model varied quite widely between sites, from 0.48-0.98. The dynamic SOM model simulations presented here also suggest some details of SOC dynamics not shown by available measured data, and this emphasises the importance of archived soil samples in long-term experiments.

7.5 COMPARISON OF APPROACHES FOR ESTIMATING REGIONAL SCALE CARBON SEQUESTRATION

The regional scale SOC modelling study involved a comparison of four methods of estimating C mitigation potential for a region of Central Hungary. The four approaches used were 1) linear regressions based on long-term experiments in Europe (Smith et al. 1997b, 1998a,c, 1999, 2000a,b,c,d,e, 2001a,b); 2) RothC model with default C inputs (Falloon et al. 1998b); 3) CENTURY model (Falloon et al. 1999b, 2001a,b) and 4) RothC model with C inputs from CENTURY (Falloon et al.

1999b, 2001a,b). This study represents the first comprehensive comparison of two SOM models and regression approaches at the regional scale. A GIS database was created, with soil, land use and climate layers, and this provided input data for the SOM models. GIS interfaces were created for RothC and CENTURY, and C mitigation scenarios for the study area developed.

Estimates of C mitigation potential obtained using the four methods described above were variable, but of the same order of magnitude. The method used to estimate C accumulation in above ground woody biomass for the natural woodland regeneration scenario had large effects on the estimate obtained. In general, the regression approaches provided the largest estimates, whilst using RothC with CENTURY C inputs provided higher estimates of C mitigation potential than RothC or CENTURY alone. Encouragingly, some scenarios showed sufficient C mitigation potential to meet Hungarian CO_2 emission reduction commitments, although some scenarios may be impractical or require refinement for the study area.

7.6 FUTURE WORK

The aim of this section is to discuss future developments which are needed to enhance regional scale estimates of C sequestration, both in terms of dynamic SOM models and regional applications. Potential solutions to some of the issues raised are also discussed.

7.6.1 Dynamic SOM modelling

There are a number of developments that could be made to enhance the applicability of dynamic SOM models such as RothC and CENTURY to regional scale studies. Potential developments could include a) model comparability issues and further model validation, b) estimation of SOM model pool sizes, c) model development for particular scenarios (e.g. no-till soils and sewage sludge applications to land) and d) general model developments, to allow model application to a wider range of systems.

7.6.1.1 Model testing and comparability

Testing of any new model developments may be limited by the lack of reliable, long-term datasets on SOC available for the scenario of interest (e.g. Smith et al. 1997b).

Continued collection of information on long-term experiments in global networks will be invaluable for this purpose (e.g. SOMNET - Smith et al. 1996a,b), although it may be that new experiments need to be established. Rigid model testing should include not only simulation of total SOC values, but where possible, comparison of measured and simulated SOC fractions such as the soil microbial biomass, CO_2 evolution from soil, and comparison of modelled and measured SOC decomposition rates.

A direct comparison of the performance of RothC and CENTURY proved difficult due to a lack of measurements on C inputs to soil and SOC decomposition rates, as well as the complex nature of the difference between the structure of the two models. Some of these difficulties might be overcome by comparing model performance with common decomposition studies using labelled plant (or other organic) materials, since the C input in such studies is known and quantified. Such studies would have added value if they included the measurements of soil microbial biomass, total SOC, and CO_2 evolution for comparison with model results. This would, at the very least, enable a direct comparison of measured and modelled SOC decomposition rates.

7.6.1.2 SOM model pools

Direct comparison of model performance is not only limited by data available for testing and model structure, but also by methods available to set the size of the different SOC pools in RothC and CENTURY. As shown in Chapter 5, default methods for setting initial pool sizes are not always appropriate. There is a need for robust ways to estimate RSOM (and other SOM) pools for models transparently and consistently, and to be able to estimate errors in these calculations. Standardised methods for setting SOM pools in SOM models would also allow more stringent model comparisons (Falloon and Smith, 1998, 2000).

The potential of RSOM pools in soils to act as a very long term CO_2 sinks is of great relevance to studies of global C cycling, and there is considerable experimental evidence for the existence of such a pool in soils. However, current SOM models are somewhat limited in their ability to predict the behaviour of RSOM for a number of reasons. Firstly, in general, the RSOM pools of SOM models are hard to relate directly to measurements made on soils (with the exception of RothC, where preatomic bomb soil samples are available), making validation of model predictions of RSOM behaviour difficult. Secondly, multi-compartmental models in which the RSOM pool is uncoupled from the rest of the SOM system are unable to describe changes in RSOM under different environmental or management conditions since the pool is assumed to be truly inert in these models. It may be possible to infer the influence of other factors such as management, vegetation, soil type and climate on extreme SOM stabilisation using coupled models, as the RSOM pool is linked with the system (Falloon and Smith, 1998, 2000).

Only models that account for the protection of SOM mechanistically are able to give an insight into how extreme stabilisation of SOM occurs (e.g. Hassink and Whitmore 1996). If truly inert OM does exist (e.g. coal, charcoal), models that do not include IOM as a separate component must combine it with SOM that turns over slowly (e.g. CENTURY - Parton et al. 1987).

The equation relating IOM to total SOC developed in Chapter 4 provides a novel method to set IOM where soil radiocarbon measurements are not available, and is an average across a number of sites where ^{14}C measurements have been made. It does not preclude the need for further ^{14}C measurements to be made in soils, and the use of ^{14}C measurements allows accurate estimation of both IOM and C inputs to soil using RothC. The limits of the relationship between total SOC and IOM are, however wide and cannot provide precise prediction. Furthermore, this relationship is not expected to hold for waterlogged, highly organic, or recent volcanic soils, or subsoils. The regression, and indeed, the RothC model, should also not be used for soils containing large amounts of recently added IOM, such as charcoal (Jenkinson et al. 1999b).

The collection and critical analysis of long-term data on changes in ^{14}C and ^{13}C in whole soils, and SOM fractions, is essential for the development of better models of SOM. Such data might also assist in linking measured estimates of pool sizes to model conceptual pools, in estimating pool turnover times and developing standardised ways to initialise models.

7.6.1.3 No-till soils and sewage sludge applications to land

Preliminary modification and testing of RothC (for no-till soils and sewage sludge applications to land) and CENTURY (for sewage sludge applications to land) have been completed. However, testing these new model versions with a wider range of datasets will be necessary to determine if the assumptions made here will hold over a wider range of climates and soil types.

Inclusion of layering in both models would improve model performance, particularly for simulation of no-till soils. Much of the change in SOC that may occur following conversion to no-till agriculture often occurs in the upper layers of the profile. Assuming a homogenous, mixed 20-30cm layer as in the current versions of RothC and CENTURY may be invalid, since a typical profile under no-till agriculture is not usually well mixed in the plough layer. There may, therefore, be differences in not only C inputs to different depths in the soil, but also in the decomposition rates at different depths. The size and turnover rate of the most active pools may need modification, since the microbial biomass under no-till agriculture is often more dominated by fungal species than under a comparable tilled system. Errors may also occur due to differences in water balance and soil temperature under no-till agriculture.

For sewage sludge applications to land, the modifications made to RothC and CENTURY presented here only changed the split of added organic matter between labile and resistant litter pools, and humified organic matter. McGrath et al. (2000) also suggested that the decomposition of sewage sludge added to land could be too rapid when predicted by RothC. It may be, however, that the soil microbial biomass may be suppressed by the increased concentrations of heavy metals in soils to which sewage sludge has been applied (McGrath et al. 2000). Future developments to RothC and CENTURY may need to take account of the effects of the heavy metals applied to soil with sewage sludge on the soil microbial biomass as well as differences between the composition of sewage sludge and FYM.

7.6.1.4 General model developments

There are a number of other situations for which both models have not been developed or tested, or are known to perform poorly. Comprehensive regional scale model applications would, however, require model applicability to all of the systems in a particular region. Potential model development areas could include a) permanently waterlogged soils, b) acidic conditions and the effects of pH; c) highly organic soils and peats, c) allophanic and recent volcanic soils, and d) subsoils, DOC and soil structural effects. Modifications for SOM model simulation of allophanic soils has been discussed in detail in section 3.1.1.6 and Smith et al. (1997c), so is not discussed further here. With the exception of allophanic soils, each of these issues is discussed in turn.

Firstly, although permanently waterlogged soils are not commonly under arable management in the UK, natural systems and pastures may be underlain by waterlogged soils. In other parts of the globe, artificial or natural waterlogging is used in rice-based agricultural systems. Thus regional scale applications including these systems would require model development for waterlogged soils. Both RothC and CENTURY were originally developed for well-drained, agricultural soils (arable and grassland, respectively).

Under anaerobic conditions, decomposition rates may be slowed to around 30% of those in aerobic soils (Jenkinson, 1988; DeBusk and Reddy, 1998; Bergman et al. 1999). Intermittent waterlogging may also increase SOC turnover times, and can impede the decomposition process (Tate et al. 1995), perhaps resulting in more formation of very stable or inert organic matter. Decomposition is also less complete under anaerobic conditions than under aerobic conditions (Jenkinson, 1988). Lignin is not attacked and peat (partly humified organic debris) can accumulate indefinitely (Jenkinson, 1988).

However, the current version of RothC was designed for aerobic soils and should not be used on soils subject to permanent waterlogging (Jenkinson et al. 1999). The rate modifier for moisture in RothC currently assumes no effect of anaerobic conditions on decomposition. It may be that more soil C should be assigned to the IOM compartment for simulating waterlogged soils (Tate et al. 1995), and that the compartmental decomposition rates need to be reduced.

In the CENTURY model, drainage characteristics at a particular site may be accounted for, and the effect of anaerobic conditions in soil causes SOC decomposition to decrease (Metherell et al. 1993). However, there is no published information on how these relationships were derived, or on model testing under waterlogged conditions. Application of RothC and CENTURY to waterlogged soils would require comprehensive, long and short-term datasets on SOC, soil microbial biomass and CO_2 evolution, and soil moisture and temperature, and if possible, ^{14}C and ^{13}C to allow proper model development and testing.

Acidic conditions in soil may have a number of different effects on soil carbon decomposition, and thus may require different parameter sets for current SOM models (Jenkinson et al. 1992). Model development to include the effects of pH would also enable SOM model estimates of the C mitigation potential of liming (e.g Lal et al. 1999). Global and regional statistical analyses have shown that total SOC (Motavalli et al. 1995) and the temporal variability (Wardle et al. 1998) and populations (Coleman, 1983) of biomass C are negatively related to soil pH.

Several authors have shown that increasing soil pH would reduce the rate of microbial turnover rate (Baath et al. 1980a,b; Coleman, 1983; Motavalli et al. 1995; Wardle, 1998; Bergman et al. 1999), thus increasing SOC turnover time. Acidic conditions may also lead to an accumulation of inactive and dead fungi (organic mats: Jenkinson, 1988) in the upper soil layers, with a greater proportion of input C becoming organic residue in acid soils than in neutral or alkaline soils (Amato and Ladd, 1992).

However, as noted by Jenkinson (1988), Jenkinson et al. (1999), Oades (1988) and Motavalli et al. 1995), the initial stages of decomposition (in particular microbial use of labile substrates) are much slower under acid conditions. Over longer time periods, there may be little difference in C turnover (Jenkinson, 1977; Jenkinson, 1988) as would be expected due to hindrance of decomposition by stabilizing mechanisms, such as interactions between calcium ions, clay and organic compounds, found in high pH soils (Oades, 1988).

Jenkinson et al. (1999) suggested decreasing the rate constant of the DPM compartment by a factor of 4, which allowed accurate simulation of C residue decomposition in acidic soils, over a 10y period. Using this modification to the model, only small differences in C input to soil as calculated using RothC for a

Zambian soil were observed (around 5%). Motavalli et al. (1995) showed that combined reductions in the metabolic, structural and active C pool decomposition rates of CENTURY were required to fit the observed incubation decomposition patterns from soils with pH <6.5.

Neither RothC, nor CENTURY, have been developed for, or applied to, highly organic soils. Due to their large C content, organic soils often account for a large proportion of soil C stocks in any geographic area where they are present. For this reason, models that can accurately simulate SOC cycling in organic soils are urgently required for studies of SOC dynamics at the regional scale. There are, however, currently very few such models. Consideration of SOC cycling in highly organic soils would require either development or use of a separate SOC model (e.g. the mangrove model of Chen and Twilley, 1999), or adaptation of RothC or CENTURY.

A number of factors influence the differences between SOC cycling in highly organic and other soils, including a) low pH, b) waterlogging, c) peat formation and stratification, d) CO_2 and methane production. These factors are discussed below, with the exception of pH and waterlogging, which have already been discussed above.

The rate of peat formation depends on climatic factors (rainfall and temperature; Botch et al. 1995). Root C inputs to soil may be more important than litter fall C inputs in peat soils (Chen and Twilley, 1999), thus requiring a model of root distribution and growth. The marked increase of radiocarbon age and total SOC with depth in British acid upland soils (Huang et al. 1996) suggest little disturbance or mixing in the soil profile, in direct contrast to the current versions of RothC and CENTURY, which simulate the top soil as a well mixed layer. Model developments to include distinct soil layers (section 7.6.1.3) would therefore also aid simulation of C cycling in organic soils.

Peat soils are generally sinks for CO_2 (Laine and Minkkinen, 1996), and sources of methane. Drainage may transform these systems into net CO_2 sources (Laine and Minkkinen, 1996). Non-organic soils, however are generally sources of CO_2 and sinks for methane (e.g. Paustian et al. 1997a,b). Neither RothC nor CENTURY currently simulate methane consumption or production in soils.

Methane and CO_2 production in wetlands is mainly the result of anaerobic decomposition of plant remains and exudates from roots of vascular plants (Bergman et al. 1999). Carbon dioxide is also produced under aerobic conditions in wetlands by the decomposition of plant remains and root decomposition (Bergman et al. 1999). Methane production and consumption are most strongly regulated by mean annual temperature (Yavitt et al. 1997), pH and the amount and quality of available substrate (Bergman et al. 1999). Carbon dioxide production in peatlands shows a strongly negative relationship with mean annual temperature and the amount and type of lignin in the peat (which in turn decreases with decreasing mean annual temperature; Yavitt et al. 1997).

RothC and CENTURY currently only simulate SOC turnover in the top 20-30cm of soil. Many global studies of SOC simulate the top 1m of soil and as discussed in Chapters 1 and 3, SOC turnover at depth is commonly much slower than in the topsoil. Simulation of SOC turnover at depth, and simulation of forest, grassland, and no-till systems would also benefit from the introduction of layers (and the transfers between these layers) to models such as RothC and CENTURY (section 7.6.1.3). Whilst CENTURY accounts for DOC processes in soils, the current version of RothC does not, and this is an important process in some systems (Tipping et al. 1999). Finally, neither RothC nor CENTURY account for the effects of soil structure on SOM turnover.

7.6.2 Estimation of C input to soil

C input to soil is the single most important factor determining SOC balances. However, it remains difficult to measure experimentally, especially across large spatial areas. Compilation of databases containing information on C input to soil and components of C input to soil would greatly aid modelling studies. Such a database should include information on, for example, measurements of above and belowground production, litterfall, crop harvest indexes and shoot to root ratios, as well as information on soils, climate and management. Comparison of direct measurements with models, and more ^{14}C dating of soils would allow more accurate model estimates and evaluation.

7.6.3 Estimates of C sequestration potential at the regional scale

Regression-based approaches for estimating carbon sequestration potential at the regional scale are greatly limited by data availability. Collection of a greater number of datasets, across more climates and soil types, would allow a) more confidence in regression derived estimates, and b) perhaps individual relationships to be developed for particular climates and soil types.

All methods of estimating C sequestration potential should pay careful attention that undue credit is not claimed for particular management practices, and this emphasizes the need for monitoring of 'baseline conditions'. For example, if SOC in a particular location was increasing with time under 'baseline' management, the projected increase in SOC over the timescale of a C sequestration project under 'baseline' management should be discounted from increases due to management imposed by the project. In other words - SOC changes claimed under C sequestering activities should be directly due a change in activities.

Further future developments to this approach should also include the application of combined scenarios with variable application rates and estimates of the C mitigation potential of bioenergy crops (Smith et al. 2000b). Inclusion of modelling trace gas fluxes with C fluxes may show the full greenhouse gas mitigation potential (GHGMP) of some scenarios to be reduced, whilst the GHGMP of others could be increased, relative to considering carbon alone (Smith et al. 2000b). Carbon mitigation potential also depends upon which practices or scenarios are practical for a particular region, what the environmental (soil and climatic) limitations are, and the size of the emissions reduction target. For countries such as Hungary, where the emissions reduction target is low, relative to the European target, soil carbon sequestration may be a more promising option.

A more detailed full carbon accounting approach should also include socio-economic factors, and accounting for differences in C stored in plant biomass, as well as just the soil, fuel and wood components considered here. Socio-economic models could also be incorporated in order to predict future land use changes and potential changes in SOC storage. Land suitability aspects should also be incorporated, to cover aspects such as environmental sensitivity and agricultural suitability. These developments would require detailed regional level databases on management factors, but would also improve upon estimates of additional C savings under C sequestration activities.

The use of forecast climatic data from global circulation models (GCM's) could prove useful in predicting the effects of climate change on regional SOC stocks, and the effects of interactions between land management scenarios and climatic change. The fate of eroded SOC in regional and national studies of soil C dynamics has not been studied extensively and is also of great importance (Kern and Johnson, 1993). Most regional studies of SOC dynamics also ignore transfers between spatial units, which are important for processes such as water, soil (erosion) and gaseous element transfers between units (Paustian et al. 1997d).

Uncertainty analysis of the estimates presented here should be completed, and include uncertainty terms related to errors in both measurements and model outputs. There was no information in the spatial soil database used here on the spatial variability of SOC (or indeed other soil parameters) within individual soil unit polygons.

An understanding of spatial variability of soil properties at scales from micro-plot to region would be invaluable in assessing the validity of regional scale predictions. For example, regional estimates of soil C storage can vary by up to 50% for the same region, dependent upon the mapping method used (Homann et al. 1998). Significant differences in SOM storage estimates using different scale databases for the same region were also noted by Burke et al. (1990). The spatial scale of regional studies can be critical, since some processes related to SOC storage have non-linear responses and small areas with high or low values for model driving variables can have a disproportionate effect on results (Burke et al. 1990). Averaging and aggregation of soils data can also have large and disproportionate effects on SOM model outputs (Paustian et al. 1997d).

Finally, it is important to consider how the different approaches for estimating C sequestration potential could be practically used for Kyoto Protocol related activities. For example, it may be questionable whether a) regional scale model applications can capture the great spatial variability of SOC or b) current regional datasets would allow this. Since there are legal aspects to C sequestration commitments, uncertainty analysis and verifiability of model predictions are also of great importance. Whilst RothC and CENTURY are perhaps the most widely evaluated SOM models worldwide, a number of model evaluation exercises using datasets from long-term experiments still show unexpected model behaviour (see, for example, Paustian et al. 1992, Coleman et al. 1997, Kelly et al. 1997, or Chapter 5 of this thesis). Yet the only feasible way to assess certainty in SOM model outputs is by simulating long-term experimental datasets for the region of interest, as presented in Chapter 5.

Validation of model outputs in the true sense would require measurements of SOC to be made across the study region, at points in time after model predictions had been made. Most of the management changes likely to be applied as C sequestration activities may give rise to subtle changes in total SOC, and project lifetimes may be short. This has given rise to a current debate as to whether such changes in SOC can be accurately, cost-effectively measured against a relatively large background stock of SOC (e.g. Smith, 2001), particularly at the regional level. Uncertainty analysis of model predictions would require a) detailed sensitivity analyses of the SOM models to be used, and b) more information on the reliability of database information on soils, weather and land use at the regional level. This highlights the three largest data gaps for regional scale studies: a) information on spatial variation of soil, land use and climatic properties within mapping units, b) information on uncertainty of soil, land use and climatic measurements in databases and c) a lack of long-term experimental datasets for validation. A rational, scientific basis for verifying not only model predictions and uncertainty, but also measured changes in SOC is therefore urgently required.

7.7 CONCLUSIONS

Most former estimates of regional scale C sequestration potential have made use of either linear regressions based on long-term experimental data, whilst some have used dynamic SOM models linked to spatial databases. Few studies have compared these two methods, and none have compared regressions with two different SOM models. This thesis has presented a case study investigation of the potential of different land management practices to sequester carbon in soil in arable land, and preliminary estimates of other potential C savings.

Two dynamic SOM models were chosen for this study, RothC (a soil process model) and CENTURY (a general ecosystem model). RothC and CENTURY are the two most widely used and validated SOM models world-wide. Methods were developed to enhance use and comparability of the models in a predictive mode. These methods included a) estimation of the IOM pool for RothC, b) estimation of C inputs to soil, c) investigation of pool size distributions in CENTURY, and d) creation of a program to allow use of C inputs derived from CENTURY with the RothC model. Sensitivity analyses demonstrated the importance of errors in a) C inputs to soil, and b) refractory SOM pool estimates for predictive SOM modelling.

A site-scale comparison of RothC and CENTURY using datasets from seven European long-term experiments was used to a) verify their ability to predict SOC changes under changes in land use and management relevant to studies of C sequestration potential, b) evaluate model performance under European climatic conditions, and c) compare the performance of the two models. Both models were shown to perform generally well with datasets from long-term experiments, although RothC (with fitted C inputs) did tend to fit measured data better than CENTURY, and RothC (with C inputs from CENTURY) tended to have the poorest fit of all model simulations. Differences in SOC turnover rates predicted by RothC and CENTURY were found. More detailed datasets and more intensive model comparison exercises are required to determine which model is most capable of predicting SOC decomposition in soils.

Finally, linkage of RothC and CENTURY to regional-scale spatial datasets was achieved by a) assembling a Geographic Information System (GIS) containing soil, land use and climate layers for a case study region in Central Hungary and b) creation of GIS interfaces for the RothC and CENTURY models. This allowed a comparison of estimates of the C sequestration potential of different land management practices obtained using the two models and using regression-based estimates.

Although estimates obtained by the different approaches were of the same order of magnitude, differences were observed. Encouragingly, some of the land management scenarios studied here showed sufficient C mitigation potential to meet Hungarian CO_2 reduction commitments. Simulation of trace gas fluxes, full system-

C accounting, uncertainty analysis, land suitability, and socio-economic modelling may help further refine estimates of C sequestration potential at the regional scale. More detailed datasets (in particular on land management and C input to soil) are required to improve upon these estimates. Adoption of the techniques for estimating C sequestration at the regional scale presented here ultimately depends on a) detailed model evaluation and comparison exercises, b) more detailed information on variability of soil, land use and climatic properties and c) improvements in measurement techniques to validate model predictions.

REFERENCES

AGREN, G.I., KIRSCHBAUM, M.U.F., JOHNSON, D.F. and BOSATTA, E. (1996) Ecosystem physiology – soil organic matter. In: *Global Change: Effects on Coniferous Forests and Grasslands*, pp. 207-228. (Eds. A.I. Breymeyer, D.O. Hall, J.M. Mellilo and G.I. Agren) John Wiley and Sons Ltd.

AJWA, H.A., RICE C.W., and SOTOMAYOR, D. (1998) Carbon and nitrogen mineralization in tallgrass prairie and agricultural soil profiles. *Soil Science Society of America Journal*, 62, 942-951.

ALEMU, G., UNGER, P.W. and JONES, O.R. (1997) Tillage and cropping system effects on selected conditions of a soil cropped to grain sorghum for twelve years. *Communications in Soil Science and Plant Analysis*, 28, 63-71.

ALLEN, L.H. (1990) Plant responses to rising carbon dioxide and potential interactions with air pollutants. *Journal of Environmental Quality*, 19, 15-34.

ALMENDINGER, J.C. (1990) The decline of soil organic matter, total-N, and available water capacity following the late Holocene establishment of jack-pine on sandy mollisols, North-Central Minnesota. *Soil Science*, 150, 680-694.

AMATO, M. AND LADD, J.N. (1992) Decomposition of 14C-labelled glucose and legume material in soils: properties influencing the accumulation of residue C and microbial biomass C. *Soil Biology and Biochemistry*, 24, 455-464.

ANDERSON, D.W. and PAUL, E.A. (1984) Organomineral complexes and their study by radiocarbon dating. *Soil Science Society of America Journal*, 48, 298-301.

ANDERSON, J.M. (1991) The effects of climate change on decomposition processes in grassland and coniferous forests. *Ecological Applications*, 1, 326-347.

ANDRÉ, M., THIERY, J.M., and COURMAC, L. (1992) ECOSIMP model: prediction of CO_2 concentration changes and carbon status in closed ecosystems. COSPAR.

ANDRÉN, O. and KÄTTERER, T. (1997) ICBM - the Introductory Carbon Balance Model for exploration of soil carbon balances. *Ecological Applications*, 7, 1226-1236.

ANDREUX, F., CERRI, C., VOSE, P.B. and VITORELLO, V.A. (1990) Potential of stable isotope, ^{15}N and ^{13}C, methods for determining input and turnover in soils. In: *Nutrient cycling in terrestrial ecosystems: field methods, application and interpretation,* pp. 259-275. (Eds. A.F. Harrison, P. Ineson and O.W. Heal) Elsevier, London.

ANGERS, D.A. and GIROUX, M. (1996) Recently deposited organic matter in soil water-stable aggregates. *Soil Science Society of America Journal*, 60, 1547-1551.

ANGERS, D.A., BISSONNETTE, N., LEGERE, A. and SAMSON, N. (1993) Microbial and biochemical changes induced by rotation and tillage in a soil under barley production. *Canadian Journal of Soil Science*, 73, 39-50.

ARROUAYS, D., VION, I. and KICIN, J.L. (1995) Spatial analysis and modelling of topsoil carbon storage in temperate forest humic loamy soils of France. *Soil Science*, 159, 191-198.

ARSLANOV, K.A., GERASIMOV, I.P., ZUBKOV, A.L., KOZYREVA, M.G. and RUBLIN, Y.V. (1970) Determination of the age of chernozem by means of the radiocarbon dating method. *Soviet Soil Science*, 2, 629-633.

AUGRIS, N., BALESDENT, J., MARIOTTI, A., LARGEAU, C. and DERENNE, S. (1998) First evidence of the occurrence of insoluble, non-hydrolysable organic matter in forest and crop soils. Chemical structure, origin and turnover. *16th World Congress of Soil Science, August 20-26 1998, Montpellier, Summaries on CD-ROM*. ISSS, Montpellier. Registration No. 851, Symposium 7.

AYANABA, A. and JENKINSON, D.S. (1990) Decomposition of carbon-14 labelled ryegrass and maize under tropical conditions. *Soil Science Society of America Journal,* 54, 112-115.

BAATTH, E., BERG, B., LOHM, U., LUNDGREN, B., HUNDVIST, H., ROSSWALL, T., SODDERSTROM, B. and WIREN A. (1980a) Soil organisms and litter decomposition in a Scots pine forest - effects of experimental acidification. In: *Effects of acid precipitation on terrestrial ecosystems,* pp. 375-380 (Eds. T.C. Hutchinson and M. Havas) Plenum Press, New York.

BAATTH, E., BERG, B., LOHM, U., LUNDGREN, B., HUNDVIST, H., ROSSWALL, T., SODDERSTROM, B. and WIREN A. (1980b) Effects of experimental acidification and liming on soil organisms and litter decomposition in a Scots pine forest. *Pedobiologia,* 20, 85-100.

BALDOCK, J.A., OADES, J.M., WATER, A.G., PENG, X., VASSALLO, A.M. and WILSON, M.A. (1992) Aspects of the chemical structure of soil organic materials as revealed by solid-state ^{13}C NMR spectroscopy. *Biogeochemistry*, 16, 1-42.

BALESDENT, J. (1987) The turnover of soil organic fractions estimated by radiocarbon dating. *Science of the Total Environment*, 62, 405-408.

BALESDENT, J. (1996) The significance of organic separates to carbon dynamics and its modelling in some cultivated soils. *European Journal of Soil Science*, 47, 485-493.

BALESDENT, J. and MARIOTTI, A. (1996) Measurement of soil organic matter turnover using ^{13}C natural abundance. In: *Mass Spectrometry of Soils*, pp. 83-111. (Eds. A. Barrie, S.J. Prosser, T.W. Boutton and S.I. Yamasaki) Marcel Dekker, New York.

BALL-COELHO, B., TIESSEN, H., STEWART, J.W.B., SALCEDO, I.H., and SAMPAIO, E.V.S.B. (1993) Residue management effects on sugarcane yield and soil properties in Northeastern Brazil. *Soil Science Society of America Journal*, 85, 1004-1008.

BARRACLOUGH, P. B., WEIR, A. H., and KUHLMANN, H. (1988) Factors affecting the growth and distribution of winter wheat roots in the field. *International Society of Root Research, Proceedings of the Uppsala Symposium*, Sweden, August 1988.

BATJES, N.H. (1996) Total carbon and nitrogen in the soils or the world. *European Journal of Soil Science*, 47, 151-163.

BATJES, N.H. (1998) Mitigation of atmospheric CO_2 concentrations by increased carbon sequestration in the soil. *Biology and Fertility of Soils*, 27, 230-235.

BATJES, N.H. and SOMBROEK, W.G. (1997) Possibilities for carbon sequestration in tropical and subtropical soils. *Global Change Biology*, 3, 161-173.

BAZZAZ, F.A. (1990) The response of natural ecosystems to the rising global CO_2 levels. *Annual Reviews of Ecological Systems*, 21, 167-196.

BECKER-HEIDMANN, P. (1996) Requirements for an international radiocarbon soils database. *Radiocarbon*, 38, 177-180.

BECKER-HEIDMANN, P. and SCHARPENSEEL, H.W. (1989) Carbon isotope dynamics in some tropical soils. *Radiocarbon*, 31, 672-679.

BECKER-HEIDMANN, P. and SCHARPENSEEL, H.W. (1992a) Studies of soil organic matter dynamics using natural carbon isotopes. *Science of the Total Environment*, 117, 305-312.

BECKER-HEIDMANN, P. and SCHARPENSEEL, H.W. (1992b) The use of natural ^{14}C and ^{13}C in soils for studies on global climate change. *Radiocarbon*, 34, 535-540.

BECKER-HEIDMANN, P., LIANG-WU, L. and SCHARPENSEEL, H.W. (1988) Radiocarbon dating of organic matter fractions of a Chinese Mollisol. *Zeitschrift fur Pflanzenernahrung und Bodenkunden*, 151, 37-39.

BERGMAN, I., LUNDBERG, P. and NILSON, M. (1999) Microbial carbon mineralisation in an acid surface peat: effects of environmental factors in laboratory incubations. *Soil Biology and Biochemistry*, 31, 1867-1877.

BESNARD, E., CHENU, C., BALESDENT, J., PUGET, P. and ARROUAYS. (1996) Fate of particulate organic matter in soil aggregates during cultivation. *European Journal of Soil Science*, 47, 495-503.

BIRD, M.I., CHIRAS, A.R. and HEAD, J. (1996) A latitudinal gradient in carbon turnover times in forest soils. *Nature*, 381, 143-146.

BOLINDER, M.A., ANGERS, D.A. and DUBUC, J.P. (1997) Estimating shoot to root ratios and annual carbon inputs in soils for cereal crops. *Agriculture, Ecosystems and Environment*, 63, 61-66.

BOSATTA, E. and AGREN, G.I. (1985) Theoretical analysis of decomposition of heterogeneous substrates. *Soil Biology and Biochemistry*, 17, 601-610

BOSATTA, E. and AGREN, G.I. (1991) Dynamics of carbon and nitrogen in the organic matter of the soil: a generic theory. *The American Naturalist*, 138, 227-245.

BOTCH, M.S., KOBAK, K.I., VINSON, T.S. and KOLCHUGINA, T.P. (1995) Carbon pools and accumulation in peatlands of the former Soviet Union. *Global Biogeochemical Cycles*, 9, 37-46.

BOUMA, J., VARALLYAY, G. and BATJES, N.H. (1998) Principal land use changes anticipated in Europe. *Agriculture, Ecosystems and Environment*, 67, 103-119.

BOUTTON, T.W. (1996) Stable carbon isotope ratios of soil organic matter and their use as indicators of vegetation and climate change. In: *Mass Spectrometry of Soils*, pp. 47-82. (Eds. A. Barrie, S.J. Prosser, T.W. Boutton and S.I. Yamasaki) Marcel Dekker, New York.

BRAGATO, G. and PRIMAVERA, F. (1998) Manuring and soil type influence on spatial variation of soil organic matter properties. *Soil Science Society of America Journal*, 62, 1313-1319.

BREYMEYER, A.I., BERG, B., GOWER, S.T., and JOHNSON, D. (1996) Carbon budget: temperate coniferous forests. In: *Global change: effects on coniferous forests and grasslands, Scope 56*. (Eds. A.I. Breymeyer, D.O. Hall, J.M. Melillo, and G.I. Agren) Wiley, Chichester.

BURINGH, P. (1984) Organic carbon in soils of the world. Chapter 3. In: *The role of terrestrial vegetation in the global carbon cycle: measurement by remote sensing*, pp. 91-109. (Ed. G.M. Woodwell) Scope, John Wiley and Sons, Chichester.

BURKE, I.C., SCHIMEL, D.S., YONKER, C.M., PARTON, W.J., JOYCE, L.A. and LAUENROTH, W.K. (1990) Regional modelling of grassland biogeochemistry using GIS. *Landscape Ecology*, 4, 45-54.

BURKE, I.C., YONKER, C.M., PARTON, W.J., COLE, C.V., FLACH, K. and SCHIMEL, D.S. (1989) Texture, climate and cultivation effects on soil organic matter content in U.S. grassland soils. *Soil Science Society of America Journal*, 53, 800-805.

BÜTTNER, G. (1997) *Land Cover - Hungary; Final Technical Report to Phare*, FOMI , Budapest (manuscript)

BÜTTNER, G., CSATHÓ, E. and MAUCHA, G. (1995a) The CORINE Land Cover - Hungary project. In: *17th International Cartographic Conference, Barcelona,* pp. 1813

BÜTTNER, G., CSATHÓ, E. and MAUCHA, G. (1995b) The CORINE Land Cover - Hungary project. In: *Proceedings, International Conference on Environment and Informatics, Budapest*, pp. 54-61.

BUYANOVSKY, G.A. and WAGNER, G.H. (1998) Changing role of cultivated land in the global carbon cycle. *Biology and Fertility of Soils*, 27, 242-245.

BUYANOVSKY, G.A., ASLAM, M. and WAGNER, G.H. (1994) Carbon turnover in soil physical fractions. *Soil Science Society of America Journal*, 58, 1167-1173.

CAMBARDELLA, C.A. and ELLIOTT, E.T. (1992) Particulate soil organic matter changes across a grassland cultivation sequence. *Soil Science Society of America Journal*, 56, 777-783.

CAMPBELL, C.A., LAFOND, G.P., ZENTNER, R.P., and BEIDERBECK, V.O. (1991) Influence of fertilizer and straw baling on soil organic matter in a thin black chernozem in Western Canada. *Soil Biology and Biochemistry*, 23, 443-446.

CAMPBELL, C.A., PAUL, E.A., RENNIE, D.A., and MCCALLUM, K.J. (1967) Applicability of the carbon dating method of analysis to soil humus studies. *Soil Science*, 104, 217-224.

CAMPBELL., C.A., BIEDERBECK, V.O., MCCONKEY, B.G., CURTIN, D. and ZENTNER, R.P. (1999) Soil quality – effect of tillage and fallow frequency. Soil organic matter quality as influenced by tillage and fallow frequency in a silt loam in South-western Saskatchewan. *Soil Biology and Biochemistry*, 31, 1-7.

CAMPELL, C.A., MCCONKEY, B.G., BIEDERBECK, V.O., ZENTNER, R.P., CURTIN, D., and PERU, M.R. (1998b) Long-term effects of tillage and fallow frequency on soil quality attributes in a clay soil in semiarid South-western Saskatchewan. *Soil and Tillage Research*, 46, 135-144.

CAMPELL, C.A., THOMAS, A.G., BIEDERBECK, V.O., MCCONKEY, B.G., SELLES, F., SPURR, D., and ZENTNER, R.P. (1998a) Converting from no-tillage to pre-seeding tillage: influence on weeds, spring wheat grain yields and N, and soil quality. *Soil and Tillage Research*, 46, 175-185.

CAO, M., GREGSON, K., MARSHALL, S., DENT, J.B., and HEAL, O.W. (1996) Global methane emissions from rice paddies. *Chemosphere*, 33, 879-897.

CARTER, M.R., PARTON, W.J., ROWLAND, I.C., SCHULTZ, J.E. and STEED, G.R. (1993) Simulation of soil organic carbon and nitrogen changes in cereal and pasture systems of Southern Australia. *Australian Journal of Soil Research*, 31, 481-491.

CERRI, C.C., BERNOUX, M. and BLAIR, G.J. (1994) Carbon pools and fluxes in Brazilian natural and agricultural systems and the implications for the Global CO_2 balance. *International Soil Science Society*, pp. 399-406.

CHAN, K.Y., ROBERTS, W.P. and HEENAN, D.P. (1992) Organic carbon and associated soil properties of a red earth after 1o years of rotation under different stubble and tillage practices. *Australian Journal of Soil Research*, 30, 71-83.

CHANEY, K., HODGSON, D.R. and BRAIM, M.A. (1985) The effects of direct drilling, shallow cultivation and ploughing on some soil physical properties in a long-term experiment on spring barley. *Journal of Agricultural Science, Cambridge*, 104, 125-133.

CHARMAN, D.J., ARAVENA, R. and WARNER, B.G. (1994) Carbon dynamics in a forested peatland in North-Eastern Ontario, Canada. *Journal of Ecology*, 82, 55-62.

CHEN, R. and TWILLEY, R.R. (1999) A simulation model of organic matter and nutrient accumulation in mangrove wetland soils.

CHERKINSKY, A.Y. and GORYACHKIN. (1992) Geographic regularities of accumulation and renewal time of organic carbon in soils of the Russian European North. *Pochvovdeniye*, 24, 126-132.

CHERTOV, O.G., and KOMAROV, A.S. (1996) SOMM - A model of soil organic matter and nitrogen dynamics in terrestrial ecosystems. In: *Evaluation of soil organic matter models using existing, long-term datasets*, pp. 231-236. (Eds. D.S. Powlson, P. Smith, and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Berlin Heidelberg New York.

CHICHAGOVA, O.A. (1995) Composition, properties, and radiocarbon age of humus in palaeosols. *GeoJournal*, 36, 207-212.

CHICHAGOVA, O.A. (1996) Modern trends in radiocarbon studies of organic matter of soils. *Eurasian Soil Science*, 29, 89-100.

CHRISTENSEN, B.T. (1996) Matching measurable soil organic matter fractions with conceptual pools in simulation models of carbon turnover: revision of model structure. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 143-160 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

COLE, C.V., BURKE, I.C., PARTON, W.J., SCHIMEL, D.S., OJIMA, D.S., and STEWART, J.W.B. (1988) Analysis of historical changes in soil fertility and organic matter levels of the North American Great Plains. In: *Proceedings of the International Conference on Dryland Farming: Challenges in Dryland Agriculture, A Global Perspective,* pp. 436-438. Amarillo, Texas.

COLE, C.V., DUXBURY, J., FRENEY, J., HEINEMEYER, O., MINAMI, K., MOSIER, A., PAUSTIAN, K., ROSENBERG, N., SAMPSON, N., SAUERBECK, D. and ZHAO, Q. (1997) Global estimates of potential mitigation of greenhouse gas emissions by agriculture. *Nutrient Cycling in Agroecosystems*, 49, 221-228.

COLEMAN, D.C. (1983) The impacts of acid deposition on soil biota and C cycling. *Environmental and Experimental Botany*, 23, 225-233.

COLEMAN, K. and JENKINSON, D.S. (1996) RothC-26.3- A Model for the turnover of carbon in soil. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 237-246 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

COLEMAN, K., BRADBURY, N.J. and JENKINSON, D.S. (1994) Calculating the input of inorganic matter to soil in three wooded sites. In: *7th International CIEC Symposium 'Agroforestry and Land Use Change in Industrialised Nations' proceedings*, pp. 491-499. (Eds E. Welle, I. Szabolcs and R.F. Hettle) ISCF, Budapest.

COLEMAN, K., JENKINSON, D.S., CROCKER, G.J., GRACE, P.R., KLIR, J., KÖRSCHENS, M., POULTON, P.R. and RICHTER, D.D. (1997) Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. *Geoderma*, 81, 29-44.

CONDRON, L.M. and NEWMAN, R.H. (1998) Chemical nature of soil organic matter under grassland and recently established forest. *European Journal of Soil Science*, 49, 597-603.

CONNIN, S.L., VIRGINIA., R.A., and CHAMBERLAIN, C.P. (1997) Carbon isotopes reveal soil organic matter dynamics following arid land shrub expansion. *Oecologia*, 110, 374-386.

COSTA, J. (1997) Farmer acceptance and public support for no-tillage - conservation farming in Spain. In: *Experience with the Applicabiility of No-Tilage Crop Production in the West-European Countries,* pp. 127-135, (Eds. F. Tebrugge F and A. Bohrnsen) Proceedings of EC-Workshop III, Evora, April 1996. Wissenschaftlicher Fachverlag, Giessen.

CRAMER, W., COLINET, G., COLLATZ, J., EMANUEL, W., ESSER, G., FIELD, C., FISCHER, A., FRIEND, A., HAXELTINE, A., HEIMANN, M., HOFFSTADT, J., JUSTICS, C., KERGOAT, L., KICKLIGHTER, D., KNORR, W., KOHLMAIER, G., LURIN, B., MAISONGRANDE, P., MARTIN, P., MCKEOWN, R., MEESON, B., MOORE, B., OLSON, R., OTTO, R., PARTON, W., PLEOCHL, M., PRINCE, S., RANDERSON, J., RASOOL, I., RIZZO, B., RUIMY, A., RUNNING. S., SAHAGIAN, B., SAUGIER, B., SCHLOSS, A., SCURLOCK, J., STEFFEN, W., WARNANT, P., and WITTENBERG, U. (1995) *Net primary productivity model intercomparison activity (NPP)*. IGBP/GAIM Report Series, Report No. 5. Eds. K. Hibbard and D. Sahigan. IGBP.

CSATHÓ, P., and ÁRENDÁS, T. (1997) The effect of soil organic matter content on crop responses to N given in mineral or organic forms. *Agrokémia és Talajtan,* 46, 63-76.

CURRAN, P.J., FOODY, G.M., LUCAS, R.M., and HONZAK, M. (1996) The mystery of the missing carbon. *GIS Europe*, 5, 38-41.

CURTIN, D., SELLES, F., WANG, H., CAMPBELL, C.A. and BIEDERBECK, V.O. (1998) Carbon dioxide emissions and transformation of soil carbon and nitrogen during wheat straw decomposition. *Soil Science Society of America Journal*, 62, 1035-1041.

DAI, A. and FUNG, I.Y. 1993. Can climate variability contribute to the "missing" CO_2 sink? *Global Biogeochemical Cycles*, 7, 599-609.

DEANGELIS, D.L., GARDNER, R.H., and SHUGART, H.H. (1981) Productivity of forest ecosystems studied during the IBP: the woodlands dataset. In: *Dynamics* *of Forest Ecosystems*, pp. 572-672. (Ed. D.E. Reichle) Cambridge University Press, Cambridge.

DEBUSK, W.F. and REDDY, K.R. (1998) Turnover of detrital organic carbon in a nutrient-impacted everglades marsh. *Soil Science Society of America Journal*, 62, 1460-1468.

DONALD, C.M. and HAMBLIN, J. (1976) The biological yield and harvest index of cereals as agronomic and plant breeding criteria. *Advances in Agronomy*, 28, 361-405.

DONIGAN, A.S., BARNWELL, T.O., JACKSON, R.B., PATWARDHAN, A.S., WEINRICH, K.B., ROWELL, A.L., CHINNASWAMMY, R.V., and COLE, C.V. (1994) *Assessment of alternative management practices and policies affecting soil carbon in agroecosystems of the central United States*. EPA/600/R-94/067. Environmental Research Laboratory, Athens, GA

DORAN, J.W., ELLIOTT, E.T. and PAUSTIAN, K. (1998) Soil microbial activity, nitrogen cycling, and long-term changes in soil organic carbon pools as related to fallow tillage management. *Soil and Tillage Research*, 49, 3-18.

DOUGLAS, C.L. (JR.) and RICKMAN, R.W. (1992) Estimating crop residue decomposition from air temperature, initial nitrogen content and residue placement. *Soil Science Society of America Journal*, 56, 272-278.

DRAKE, B.G., GONZALEZ-MELER, M.A., and LONG S.P. (1997) More efficient plants: a consequence of rising atmospheric CO_2? *Annual Reviews of Plant Physiology and Plant Molecular Biology*, 48, 609-639.

DRURY, C.F., OLOYA, T.O., MCKENNEY, D.J., GREGORICH, E.G., TAN, C.S. and VANLUYK, C.L. (1998) Long-term effects of fertilization and rotation on denitrification and soil carbon. *Soil Science Society of America Journal*, 62, 1572-1579.

DYKE, G.V, GEORGE, B.J., JOHNSTON, A.E., POULTON, P.R. and TODD, A.D. (1983) The Broadbalk Wheat Experiment 1968-1978: Yields and Plant Nutrients in Crops Grown continuously and in Rotation. *Rothamsted Experimental Station Report for 1982, Part 2*, pp. 5-44. Rothamsted Experimental Station, Harpenden.

ELLIOTT, E.T., PAUSTIAN, K. and FREY, S.D. (1996) Modelling the measurable or measuring the modellable: a hierarchical approach to isolating meaningful soil organic matter fractionations. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 161-180 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

EMANUEL, W.R., KILLOUGH, G.G., POST, W.M. and SHUGART, H.H. (1984) Modeling terrestrial ecosystems in the global carbon cycle with shifts in carbon storage capacity by land-use change. *Ecology*, 65, 970-983.

ESSER, G. and LAUTENSCHAGER, M. (1994) Estimating the change of carbon in the terrestrial biosphere from 18000 BP to present using a carbon cycle model. *Environmental Pollution*, 83, 45-53.

FALLOON, P.D. and SMITH, P. (1998) The role of refractory soil organic matter in soil organic matter models. *Mitteilungen der Deutschen Bodenkundlichen Gesellschaft,* 87, 253-264.

FALLOON, P.D. and SMITH, P. (2000) Modelling refractory organic matter. *Biology and Fertility of Soils,* 30, 388-398.

FALLOON, P.D., SMITH, J.U. and SMITH, P. (1999a) A Review of Decision Support Systems for fertiliser application and manure management. *Acta Agronomica Hungarica,* 47, 227-236

FALLOON, P.D., SMITH, P., COLEMAN, K. and MARSHALL, S. (1998a) Estimating the size of the inert organic matter pool for use in the Rothamsted carbon model. *Soil Biology and Biochemistry*, 30, 1207-1211.

FALLOON, P.D., SMITH, P., COLEMAN, K. and MARSHALL, S. (2000) How important is inert organic matter for predictive soil carbon modelling using the Rothamsted carbon model? *Soil Biology and Biochemistry,* 32, 433-436.

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (2001b) Comparing estimates of regional C sequestration potential using GIS, dynamic SOM models, and simple relationships. *Advances in Soil Science* (in press)

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (2001a) SOM sustainability and agricultural management - predictions at the regional level. Chapter 2.1 In: *'Sustainable Management of Soil Organic Matter' BSSS99 Conference Proceedings*, pp. 54-59. (Eds. R.M. Rees, B.C. Ball, C.D. Campbell and C.A. Watson) CABI, Wallingford, UK.

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (1999b) Linking GIS and dynamic SOM models: estimating the regional C sequestration potential of agricultural management options. *Journal of Agricultural Science, Cambridge*, 133, 341-342.

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (1998b) Regional estimates of carbon sequestration potential: linking the Rothamsted Carbon Turnover model to GIS databases. *Biology and Fertility of Soils*, 27, 236-241.

FAOSTAT (2001) FAOSTAT - *an international statistical database, online at <http://apps.fao.org/>*. FAO, Rome.

FIELD, C., RANDERSON, J. and MALMSTROM, C. (1995) Global net primary production: combining ecology and remote sensing. *Remote Sensing of the Environment*, 51, 74-88.

FLAIG, H. AND MOHR, H. (Eds.) (1994) *Energie Aus Biomasse - Eine Chance Fuel die Landwirtschaft*. Springer, Berlin.

FRANKO, U., OELSHAGEL, B., and SCHENK, S. (1995) Simulation of temperature, water and nitrogen dynamics using the model. In: *CATENA-Verlag, Modelling of Geo-Biosphere Processes.*

FRIEND, A.D., STEVENS, A.K., KNOX, R.G. and CANNELL, M.G.R. (1997) A process-based, terrestrial biosphere model of ecosystem dynamics (Hybrid v. 3.0). *Ecological Modelling*, 32, 249-287.

FRINK, D.S. (1995) Application of the oxidisable carbon ratio dating procedure and its implications for pedogenic research. In: *Pedological Perspectives in Archaeological Research*, pp. 95-106. SSSA Special Publication 44, SSSA, Madison, Wisconsin.

FRYE, W.W. (1984) Energy requirement in no-tillage. In: *No-tillage agriculture. Principles and Practices,* pp. 127-151. (Eds R.E. Phillips and S.H. Phillips). Van Nostrand Reinhold, New York.

GERASIMOV, I.P. (1974) The age of recent soils. *Geoderma*, 12, 17-25.

GIFFORD, R.M., LUTZE, J.L. and BARRETT, D. (1996) Global atmospheric change effects on terrestrial carbon sequestration: Exploration with a global C- and N-cycle model (CQUESTN). *Plant and Soil*, 187, 369-387.

GIJSMAN, A.J. (1996) Soil aggregate stability and soil organic matter fractions under agropastoral systems established in native savanna. *Australian Journal of Soil Research*, 34, 891-907.

GOH, K.M., STOUT, J.D. AND RAFTER, T.A. (1977) Radiocarbon enrichment of soil organic matter in New Zealand soils. *Soil Science*, 123, 385-391.

GOLCHIN, A., OADES, J.M., SKJEMSTAD, J.O. and CLARKE, P. (1995) Structural and dynamic properties of soil organic matter as reflected by ^{13}C natural abundance, pyrolysis mass spectrometry and solid-state ^{13}C NMR spectroscopy in density fractions of an oxisol under forest and pasture. *Australian Journal of Soil Research*, 33, 59-76.

GOULDING, K.W.T., BAILEY, N.J., BRADBURY, N.J., HARGREAVES, P., HOWE, M., MURPHY, D.V., POULTON, P.R. and WILLISON T.W. (1998) Nitrogen deposition and its contribution to nitrogen cycling and associated soil processes. *New Phytologist*, 139, 49-58.

GOWARD, S., and HUEMMRICH, K. (1992) Vegetation canopy PAR absorptance and the normalized difference vegetation index to study atmosphere biosphere exchange of CO_2. *Remote Sensing of the Environment*, 39, 119-140.

GRACE, J., LLOYD, J., MCINTYRE, J., MIRANDA, A.C., MEIR, P., MIRANDA, H.S., NOBRE, C., MONCREIFF, J., MASSHEDER, J., MALHI, Y., WRIGHT, I. and GASH, J. (1995) Carbon dioxide uptake by an undisturbed tropical rainforest in Soutwest Amazonia. *Science*, 270, 778-780.

GRACE, P.R., LADD, J.N. (1995) *SOCRATES v2.00 User Manual*. Co-operative Research Centre for Soil and Land Management, Glen Osmond, South Australia.

GRANT, R.F. (1995) Dynamics of water, carbon, and nitrogen in agricultural ecosystems: Simulation and experimental validation. *Ecological Modelling*, 81, 169-181

GRISI, B., GRACE, C., BROOKES, P.C., BENDETTI, A. and DELL'ABATE, M.T. (1998) Temperature effects on organic matter and microbial biomass dynamics in temperate and tropical soils. *Soil Biology and Biochemistry*, 30, 1309-1315.

GUGGENBERGER, G., ZECH, W., HAUMAIER, L. and CHRISTENSEN, B.T. (1995) Land-use effects on the composition of organic matter in particle-size separates of soils: II. CPMAS and solution ^{13}C NMR analysis. *European Journal of Soil Science*, 46, 147-158.

GUNAPALA, N., VENETTE, R.C., FERRIS, H. and SCOW, K.M. (1998) Effects of soil management history on the rate of organic matter decomposition. *Soil Biology and Biochemistry*, 30, 1917-1927.

GUPTA, R.K., and RAO, L.L.N. (1994) Potential of wastelands for sequestering carbon by afforestation. *Current Science,* 66, 378-380.

HADAS, A., PARKIN, T.B. and STAHL, P.D. (1998) Reduced CO_2 release from decomposing wheat straw under N-limiting conditions: simulation of carbon turnover. *European Journal of Soil Science*, 49, 487-494.

HARKNESS, D.D. and HARRISON, A.F. (1989) The influence of afforestation on upland soils: the use of 'bomb ^{14}C ' enrichment as a quantitative tracer for changes in organic status. *Radiocarbon*, 31, 637-643.

HARKNESS, D.D., HARRISON, A.F. and BACON, P.J. (1991) The potential of bomb C^{14} measurements for estimating soil organic matter turnover. In: *Advances in soil organic matter research: the impact on agriculture and the environment*, pp. 239-251. Proceedings of a symposium, Colchester, 3-4 Sept 1990.

HARRISON, A.F. and HARKNESS, D.D. (1993) Potential for estimating carbon fluxes in forest soils using ^{14}C techniques. *New Zealand Journal of Forestry Science*, 23, 367-379.

HARRISON, K.G, BROECKER, W.S. and BONANI, G. (1993a) A strategy for estimating the impact of CO_2 fertilisation on soil carbon storage. *Global Biogeochemical Cycles*, 7, 69-80.

HARRISON, K.G., BROECKER, W.S. and BONANI, G. (1993b) The effect of changing land use on soil radiocarbon. *Science*, 262, 725-726.

HASSINK, J. (1994) Effects of soil texture and grassland management on soil organic C and N and rates of C and N mineralization. *Soil Biology and Biochemistry*, 26, 1221-1231.

HASSINK, J. and WHITMORE, A.P. (1997) A model of the physical protection of organic matter in soils. *Soil Science Society of America Journal*, 61, 131-139.

HAXELTINE, A., PRENTICE, I.C. and CRESSWELL, I.D. (1996) A coupled carbon and water flux model to predict vegetation structure. *Journal of Vegetation Science*, 7, 651-666.

HAYNES, R.J. and TREGURTHA, R. (1999) Effects of increasing periods under intensive arable vegetable production on biological, chemical and physical indices of soil quality. *Biology and Fertility of Soils*, 28, 259-266.

HIRSCHEL, G., KORNER, C., and ARNONE, J.A. (1997) Will rising atmospheric CO_2 affect leaf litter quality and in situ decomposition rates in native plant communities? *Oecologia*, 110, 387-392.

HOGENBOOM, G., JONES, J.W., HUNT, L.A., THORNTON, P.K. and TSUJI, G.Y. (1994) An integrated decision support system for crop model applications. *Proceedings of the Amercian Socitey of Agricultural Engineers*, Kansas, June 1994, 23pp.

HOMANN, P.S., SOLLINS, P., FIORELLA, M., THORSON, T. and KERN, J.S. (1998) Regional soil organic carbon storage estimates for Western Oregon by multiple approaches. *Soil Science Society of America Journal*, 62, 789-796.

HOUGHTON, R.A., BOONE, R.D., FRUCI, J.R., HOBBIE, J.E., MELILLO, J.M., PALM, C.A., PETERSON, B.J., SHAVER, G.R., WOODWELL, G.M., MOORE, B., SKOLE, D., and MYERS, N. (1987) The flux of carbon from terrestrial ecosystems to the atmosphere in 1980 due to changes in land use: geographic distribution of the flux. *Tellus,* 39B, 122-139.

HOUGHTON, R.A., HOBBIE, J.E., MELILLO, J.M., MOORE, B., PETERSON, B.J., SHAVER, G.R. and WOODWELL, G.M. (1983) Changes in the carbon content of terrestrial biota and soils btween 1860 and 1980: a net release of CO_2 to the atmosphere. *Ecological Monographs,* 45, 235-262.

HOWARD, P.J.A., LOVELAND, P.J., BRADLEY, R.I., DRY, F.T., HOWARD, D.M. and HOWARD, D.C. (1995) The carbon content of soil and its geographical distribution in Great Britain. *Soil Use and Management,* 11, 9-15.

HSIEH, Y.P. (1992) Pool size and mean age of stable organic carbon in cropland. *Soil Science Society of America Journal*, 56, 460-464.

HSIEH, Y.P. (1993) Radiocarbon signatures of turnover rates in active soil organic carbon pools. *Soil Science Society of America Journal*, 57, 1020-1022.

HSIEH, Y.P. (1994) Radiocarbon methods for dynamics of soil organic matter. In: *Humic substances in the global environment and their implications on human health*, pp. 305-310. (Eds N. Senesi, and T.M. Miano) Elsevier Science, B.V.

HSIEH, Y.P. (1996) Soil organic carbon pools of two tropical soils inferred by radiocarbon signatures. *Soil Science Society of America Journal*, 60, 1117-1121.

HSIEH, Y.P. (2001) Soil radiocarbon in the North American grasslands: comparison of interpretations. *Soil Science Society of America Journal* (in press).

HUANG, Y., BOL, R., HARKNESS, D.D., INESON, P. and EGLINTON, G. (1996) Post-glacial variations in distributions, ^{13}C and ^{14}C contents of aliphatic hydrocarbons and bulk organic matter in three types of British acid upland soils. *Organic Geochemistry*, 24, 273-287.

HULME, M., CONWAY, D., JONES, P.D., JIANG, T., BARROW, E.M. AND TURNEY, C. (1995) Construction of a 1961-1990 European climatology for climate change impacts and modelling applications. *International Journal of Climatology*, 15, 1333-1364

HYVONEN, R., AGREN, G.I. and BOSATTA, E. (1998) Predicting long-term soil carbon storage from short-term information. *Soil Science Society of America Journal*, 62, 1000-1005.

IPCC (1996) *Climate Change 1995. Impacts, Adaptations and Mitigation of Climate Change*. Scienfitic-Technical Analyses, Cambridge University Press, Cambridge, UK, 878 pp.

JANSEN, H.H. (1990) Deposition of nitrogen into the rhizosphere by wheat roots. *Soil Biology and Biochemistry*, 22, 1155-1160.

JANSSEN, B.H. (1984) A simple method for calculating decomposition and accumulation of "young" soil organic carbon. *Plant and Soil*, 76, 297-304.

JASTROW, J.D., BOUTTON, T.W. and MILLER, R.M. (1996) Carbon dynamics of aggregate-associated organic matter estimated by Carbon-13 natural abundance. *Soil Science Society of America Journal*, 60, 801-807.

JENKINSON D.S., BRADBURY N.J. and COLEMAN K. (1994) How the Rothamsted Classical Experiments have been used to develop and test models for the turnover of carbon and nitrogen in soil. In: *Long-term experiments in agricultural and ecological sciences,* pp. 117-138. (Eds. R. A. Leigh and A.E. Johnston) CAB International.

JENKINSON D.S., MEREDITH J., KINYAMARIO J.I., WARREN G.P., WONG M.T.H., HARKNESS, D.D., BOL, R., and COLEMAN K. (1999a) Estimating net primary production from measurements made on soil organic matter. *Ecology*, 80, 2762-2733.

JENKINSON, D. S. (1977) Studies on the decomposition of plant material in soil V. The effects of plant cover and soil type on the loss of carbon from ^{14}C labelled ryegrass decomposing under field conditions. *Journal of Soil Science*, 28, 424-434.

JENKINSON, D.S. (1970) Radiocarbon dating of soil organic matter. In: *Rothamsted Experimental Station Report for 1970, Part 1,* pp. 74-75. Lawes Agricultural Trust, Harpenden.

JENKINSON, D.S. (1971) The accumulation of organic matter in soil left uncultivated. In: *Rothamsted Report for 1970, Part 2*, pp. 113-137. IACR-Rothamsted, Harpenden, Herts.

JENKINSON, D.S. (1973) Radiocarbon dating of soil organic matter. In: *Rothamsted Experimental Station Report for 1973*, *Part 1,* pp. 75-76. Lawes Agricultural Trust, Harpenden.

JENKINSON, D.S. (1988) Soil organic matter and its dynamics. In: *Russell's soil conditions and plant growth, 11th edition,* pp. 504-607. (Ed. A. Wild) Longman, London.

JENKINSON, D.S. (1990) The turnover of organic carbon and nitrogen in soil. *Philosophical Transactions of the Royal Society, London B*, 329, 361-368.

JENKINSON, D.S. and COLEMAN, K.C. (1994) Calculating the annual input of organic matter to soil from measurements of total organic carbon and radiocarbon. *European Journal of Soil Science*, 45, 167-174.

JENKINSON, D.S. and RAYNER, J.H. (1977) The turnover of soil organic matter in some of the Rothamsted classical experiments. *Soil Science*, 123, 298-305.

JENKINSON, D.S., ADAMS, D.E. and WILD, A. (1991) Model estimates of CO_2 emissions from soil in response to global warming. *Nature (London)*, 351, 304-306.

JENKINSON, D.S., HARKNESS, D.D., VANCE, E.D., ADAMS, D.E. and HARRISON, A.F. (1992) Calculating net primary production and annual input of organic matter to soil from the amount and radiocarbon content of soil organic matter. *Soil Biology and Biochemistry*, 24, 295-308.

JENKINSON, D.S., HARRIS, H.C., RYAN, J., MCNEILL, A.M., PILBEAM, C.J. and COLEMAN, K. (1999b) Organic matter turnover in a calcareous clay soil from Syria under a two-course cereal rotation. *Soil Biology and Biochemistry*, 31, 687-693.

JENKINSON, D.S., HART, P.B.S, RAYNER, J.H. and PARRY, L.C. (1987) Modelling the turnover of organic matter in long-term experiments at Rothamsted. *INTECOL Bulletin*, 15, 1-8.

JENSEN, C., STOUGGAARD, B., and OSTERGAARD, H.S. (1994) Simulation of nitrogen dynamics in farmland areas of Denmark (1989-1993). *Soil Use and Management*, 10, 111-118

JOHNSTON, A.E. (1973) The effects of ley and arable cropping systems on the amounts of soil organic matter in the Rothamsted and Woburn Ley-arable experiments. In: *Rothamsted Experimental Station, Report for 1972, part 2*, pp. 131-159. IACR-Rothamsted, Harpenden, Herts.

KADUK, J., and HEIMANN, M. (1996) A prognostic phenology scheme for global terrestrial carbon cycle models. *Climate Research*, 6, 1-19.

KAISER, K., GUGGENBERGER, G. and ZECH, W. (1998) Formation of refractory soil organic matter by sorption. *Mitteilungen der Deutschen Bodenkundlichen Gesellschaft*, 87, 211-224.

KALEMBASA, S.J. and JENKINSON, D.S. (1973) A comparative study of titrimetric and gravimetric methods for the determination of organic carbon in soil. *Journal of the Science of Food and Agriculture*, 24, 1085-1090.

KAN, N.A. and KAN, E.E. (1991) Simulation of soil fertility. *Physiolical Biochemistry of Cultivated Plants*, 23, 2-16 (In Russian)

KANDELER, E., TSCHERKO, D. and SPIEGEL, H. (1999) Long-term monitoring of microbial biomass, N mineralisation and enzyme activities of a Chernozem under different tillage management. *Biology and Fertility of Soils*, 28, 343-351.

KATTERER, T. and ANDREN, O. (1999) Long-term agricultural field experiments in Northern Europe: analysis of the influence of management on soil carbon stocks using the ICBM model. *Agriculture, Ecosystems and Environment*, 72, 165-179.

KAUPPI, P.E., MIELIKAINEN K. and KUUSELA. (1992) Biomass and budget of European forests, 1971 to 1990. *Science*, 256, 70-74.

KEELING, R.F. and SHERTZ, S.R. (1992) Seasonal and interannual variations in atmospheric oxygen and implications for the global carbon cycle. *Nature*, 358, 723-727.

KELLY, R.H., BURKE, I.C. and LAUENROTH, W.K. (1996) Soil organic matter and nutrient availability responses to reduced plant inputs in shortgrass steppe. *Ecology*, 77, 2516-2527.

KELLY, R.H., PARTON, W.J., CROCKER, G.J., GRACE, P.R., KLIR, J., KORSCHENS, P.R., POULTON, P.R and RICHTER, D.D. (1997) Simulating trends in soil organic carbon in long-term experiments using the CENTURY model. *Geoderma*, 81, 75-90.

KERGOAT, L. (2001) A model of hydrologic equilibrium of leaf area index at the global scale. *Journal of Hydrology (in press).*

KERN, J.S. and JOHNSON, M.G. (1993) Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Science Society of America Journal*, 57, 200-210.

KINCHESH, P., POWLSON, D.S. and RANDALL, E. (1995) ^{13}C NMR studies of organic matter in whole soils: II. A case study of some Rothamsted soils. *European Journal of Soil Science*, 46, 139-146.

KINDERMANN, J., WURTH, G., KOLMAIER, G.H., and BADECK, F.W. (1996) Interannual variation of carbon exchange fluxes in terrestrial ecosystems. *Global Biogeochemical Cycles*, 10, 737-756.

KING, A.W., POST, W.M. and WULLSCHLEGER, S.D. (1997) The potential response of terrestrial carbon storage to changes in climate and atmospheric CO_2. *Climatic Change*, 35, 199-227.

KIRCHMANN, H., PERSSON, J. and CARLGEN, K. (1994) *The Ultuna Long-term Soil Organic Matter Experiment.* Swedish University of Agricultural Sciences, Department of Soil Sciences, Reports and Dissertations 17, 55pp.

KNORR, W. and HEIMANN, M. (1995) Impact of drought stress and other factors on seasonal land biosphere CO_2 exchange studied through and atmospheric tracer transport model. *Climate Research*, 47, 471-489.

KOBAK, K.I. and KONDRASHEVA, N.Y. (1988) Rate of organic carbon cycling in soils. *Geochimiya*, 8, 1090-1096.

KÖRSCHENS, M. (1997) Long-term datasets from Germany and Eastern Europe. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 69-80 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

KOTTO-SAME, J., WOOMER, P.L., APPOLINAIRE, M. and LOUIS, Z. (1997) Carbon dynamics in slash-and-burn agriculture and land use alternatives of the humid forest zone in Cameroon. *Agriculture, Ecosystems and Environment*, 65, 245-256.

KOUTIKA, L.S., CHONE, T., ANDREUX, F., BURTIN, G. and CERRI, C. 1999. Factors influencing carbon decomposition of topsoils from the Brazilian Amazon Basin. *Biology and Fertility of Soils*, 28, 436-438.

KOVACS, G., NEMETH, T., and RITCHIE, J. (1995) Testing simulation models for the assessment of crop production and nitrate leaching in Hungary. *Agricultural Systems*, 49, 385-397.

KUMAR, M., and MONTEITH, J. (1981) Remote sensing of crop growth. In: *Plants and the daylight spectrum*, pp. 133-144. (Ed. H. Smith) Academic Press, New York.

LAINE, J. and MINKKINEN, K. (1996) Effect of forest drainage on the carbon balance of a mire: a case study. *Scandinavian Journal of Forest Research*, 11, 307-312.

LAL, R., FOLLETT, R.F., KIMBLE, J. and COLE, C.V. (1999) Managing US cropland to sequester carbon in soil. *Journal of Soil and Water Conservation*, 54, 374-381.

LAL, R., KIMBLE, J., LEVIN, E. and WHITMAN, C. (1995) World soils and greenhouse effect: an overview. In: *Soils and Global Change*, pp. 1-7. (Eds. R. Lal, J.M. Kimble, E. Levine and B.A. Stewart) CRC Press, Boca Raton.

LAL, R., KIMBLE, J.M., FOLLET, R.F. and COLE, C.V. (1998) *The potential of US cropland to sequester carbon and mitigate the greenhouse effect*. Ann Arbor Press, Chelsea, MI, 128pp.

LANDSBERG, J. (1986) *Physiological Ecology of Forest Production.* Academic Press, London. 198pp.

LARIONOVA, A.A., YERMOLAYEV., A.M., BLAGODATSKY, S.A., ROZANOVA, I.N., YEDOKIMOV, I.V. and ORLINSKY, D.B. (1998) Soil respiration and carbon balance of grey forest soils as affected by land use. *Soil Biology and Biochemistry*, 27, 251-257.

LAVELLE, P., PASHANISHI B., CHARPENTIER, F.C., ROSSI, J.P., DEROUARD, L., ANDRE, J., PONGE, J.F. and BERNIER, N. (1997) Large-scale effects of earthworms on soil organic matter and nutrient dynamics. In: *Proceedings of the International Symposium on Earthworm Ecology*, Columbus, Ohio, 1996.

LEAMY, M.L., SMITH, G.D., COLMET-DAAGE, F. and OTTOWA, M. (1980) The morphological characteristics of andisols. In: *Soils with variable charge,* pp. 17-34 (Ed. B.K.G. Theng) NZSS, Auckland.

LEAVITT, S.W. and LONG, A. (1988) Stable carbon isotopic chronologies from trees in the southwestern United States. *Global Biogeochemical Cycles*, 2, 189-198.

LEAVITT, S.W., FOLLETT, R.F. and PAUL, E.A. (1996) Estimation of slow- and fast-cycling soil organic carbon pools from 6N HCl hydrolysis. *Radiocarbon*, 38, 231-239.

LEE, J.J., PHILLIPS, D.L. and LIU, R. (1993) The effect of trends in tillage practices on erosion and carbon content of soils in the US Corn Belt. *Water Air and Soil Pollution*, 70, 389-401

LEFROY, R.D.B. and BLAIR, G.J. (1994) The dynamics of soil organic matter changes resulting from cropping. In: *Transactions of the 15th World International Congress of Soil Science, Acapulco, Mexico. 10th-16th July, 1994, Volume 9*. International Soil Science Society. pp. 235-245.

LEFROY, R.D.B., BLAIR, G.J. and STRONG, W.M. (1993) Changes in soil organic matter with cropping as measured by organic carbon fractions and ^{13}C natural isotope abundance. *Plant and Soil*, 155/156, 399-402.

LI C., FROLKING, S. and HARRIS, R. (1994a) Modelling carbon biogeochemistry in agricultural soils. *Global Biogeochemical Cycles*, 8, 237-254

LI, C., NARAYANAN, V. and HARRISSS, R.C. (1996) Model estimates of nitrous oxide emissions from agricultural lands in the United States. *Global Biogeochemical Cycles*, 10, 297-306.

LI. C, FROLKING, S.E., HARRISS, R.C. and TERRY, R.E. (1994b) Modeling nitrous oxide emissions from agriculture: a Florida case study. *Chemosphere*, 28, 1401-1415.

LIANG, B.C., GREGORICH, E.G. and MACKENZIE, A.F. (1996) Modeling the effects of inorganic and organic amendments on organic matter in a Quebec soil. *Soil Science*, 161, 109-114.

LIANG, B.C., GREGORICH, E.G., MACKENZIE, A.F., SCNITZER, M., VORONEY, R.P., MONREAL, C.M. and BAYAERT, R.P. (1998) Retention and turnover of corn residue carbon in some Eastern Canadian soils. *Soil Science Society of America Journal*, 62, 1361-1366.

LIANGWU, L. (1996) Radiocarbon dating of vertisols in China. *Pedosphere*, 6, 147-153.

LILJEROTH, E., VAN VEEN, J. A. and MILLER, H. J. (1990) Assimilate translocation to the rhizosphere of two wheat lines and subsequent utilization by rhizosphere micro-organisms at two soil nitrogen concentrations. *Soil Biology and Biochemistry*, 22, 1015-1021.

LIN, C., LIU, T.S. and HU, T.L. (1987) Assembling a model for organic residue transformation in soils. *Proceedings of the National Council (Taiwan) Part B,* 11, 175-186.

LOMANDER, A., KATTERER, T. and ANDREN, O. (1998) Modelling the effects of temperature and moisture on CO_2 evolution from top- and subsoil using a multi-compartment approach. *Soil Biology and Biochemistry*, 30, 2023-2030.

LUGO, A.E. and BROWN, S. (1992) Tropical forests as sinks of atmospheric carbon. *Forest Ecology and Management*, 54, 239-255.

MAFF (1994) *Fertilizer recommendations for Agricultural and Horticultural Crops (RB209)* (6th Edition). HMSO, London, 112pp.

MAHIEU, N., POWLSON, D.S. and RANDALL, E.W. (1999) Statisitical analysis of published carbon-13 CPMAS NMR spectra of soil organic matter. *Soil Science Society of America Journal*, 63, 307-319.

MAISONGRANDE, P., RUIMY, A., DEDIEU, G. and SAUGIER, B. (1995) Monitoring seasonal and interannual variations of gross primary productivity, net primary productivity and net ecosystem productivity using a diagnostic model and remotely sensed data. *Tellus Series B – Chemical and Physical Meteorology*, 47B, 178-190.

MANN, I.K. (1986) Changes in soil carbon storage after cultivation. *Soil Science*, 142, 279-288.

MARLAND, G., ANDRES, R.J. and BODEN, T.A. (1994) Global, regional and national CO_2 emissions. In: *Trends '93: a compendium of data on Global Change,* pp. 505-584 (Ed. T.A. Boden) ORNL/CDIAV-65, Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, Oak Ridge, TN.

MARTEL, Y.A. and LASALLE, P. (1977) Radiocarbon dating of organic matter from a cultivated topsoil in Eastern Canada. *Canadian Journal of Soil Science*, 57, 375-377.

MARTEL, Y.A. and PAUL, E.A. (1970) An example of a process model: the carbon turnover for use in ecosystem studies in grasslands. In: *Grassland Ecosystems: reviews of research,* pp. 179-189. (Eds R.T. Coupland and G.M. Van Dyne) Range Science Department Scientific Series no 7, Colorado State University, Fort Collins, Colorado.

MARTEL, Y.A. and PAUL, E.A. (1974a) Effects of cultivation on the organic matter of grassland soils as determined by fractionation and radiocarbon dating. *Canadian Journal of Soil Science*, 54, 419-426.

MARTEL, Y.A. and PAUL, E.A. (1974b) The use of radiocarbon dating of organic matter in the study of soil genesis. *Soil Science Society of America Proceedings*, 38, 501-506.

MARTIN, C.W. and JOHNSON, W.C. (1995) Variation in radiocarbon ages of soil organic matter fractions from Late Quaternary buried soils. *Quaternary Research*, 43, 232-237.

MARTIN, P.H. (2001) Soil carbon and climate perturbations: using the analytical biogeochemical cycling (ABC) scheme. *Environmental Science and International Policy Research* (in press)

MARTON, P., BASSA, L., BELUSZKY, P., BERENYI, I., BORAI, A., FUSI, L., KERESZTESI, Z., KOTA, A., MAROSI, S., PAPP-VARY, A. and SZILADI, J. (Eds.) (1989) *National Atlas of Hungary*. Cartographia, Budapest.

MASCIANDARO, G., CECCANTI, B. and GALLARDO-LANCH, J.F. (1998) Organic matter properties in cultivated versus set-aside arable soils. *Agriculture, Ecosystems and Environment*, 67, 267-274.

MCCANE, R.B., RASTETTER, E.B., MELILLO, J.M., SHAVER, G.R., HOPKINSON, C.S., FERNANDES, D.N., SKOLE, D.L. and CHOMENTOWSKI, W.H. (1995) Effects of global change on carbon storage in tropical forests of South America. *Global Biogeochemical Cycles*, 9, 329-350.

MCCARTY, G.W., LYSSENKO, N.N. and STARR, J.L. (1998) Short-term changes in soil carbon and nitrogen pools during tillage management transition. *Soil Science Society of America Journal*, 62, 1564-1571.

MCCASKILL, M. and BLAIR, G.J. (1990) A model of S, P, and N uptake by a perennial pasture: II Calibration and prediction. *Fertiliser Research*, 22, 173-179

MCCOWN, R.L., HAMMER, G.L., HARGREAVES, J.N.G., HOLZWORTH, D.P. and FREEBAIRN, D.M. (1996) APSIM: a novel software system for model development, model testing and simulation in agricultural systems research. *Agricultural Systems*, 50, 255-271.

MCGILL, W.B. (1996) Review and classification of ten soil organic matter (SOM) models. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 111-132 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

MCGRATH, S.P. and BROOKES, S.P. (1986) Effects of long-term sludge additions on microbial biomass and microbial processes in soil. In: *Factors influencing sludge utilization practices in Europe*, pp. 80-89. (Eds. R.D. Davis, H. Haeni and P. L'Hermite) Elsevier, London.

MCGRATH, S.P., ZHAO, F.J., DUNHAM, S.J., CROSLAND, A.R. and COLEMAN, K. (2000) Long-term changes in the extractability and bioavailability of zinc and cadmium after sludge application. *Journal of Environmental Quality*, 29, 875-883.

MCGUIRE, A., MELILLO, J. and KICKLIGHTER, D. (1995) Equilibrium responses of soil carbon to climate change – empirical and process based estimates. *Journal of Biogeography*, 22, 785-796.

MCGUIRE, A.D., MELILLO, J.M., KICKLIGHTER, D.W. and JOYCE, L.A. 1995. Equilbrium responses of soil carbon to climate change: empirical and process-based estimates. *Journal of Biogeography*, 22, 785-796.

MCGUIRE, A.D., MELILLO, J.M., KICKLIGHTER, D.W., PAN, Y., XIAO, X., HELFRICH, J., MOORE III, B., VOROSMARTY, C. and SCHLOSS, A. (1997) Equilibrium responses of global net primary production and carbon storage to doubled atmospheric carbon dioxide: sensitivity to changes in vegetation nitrogen concentration. *Global Biogeochemical Cycles*, 11, 173-189.

MELILLO, J.M., MCGUIRE, A.D., KICKLIGHTER, D.W., MOORE III, B., VOROSMARTY, C.J. and SCHLOSS, A. (1993) Global climate change and terrestrial net primary production. *Nature*, 363, 234-240.

MERCKX R., DIJKSTRA A., DEN HARTOG A. and VAN VEEN J. A. (1987) Production of root derived material and associated microbial growth in soil at different nutrient levels. *Biology and Fertility of Soils*, 5, 126-122.

METHERELL, A.K., HARDING, L.A., COLE, C.V. and PARTON, W.J. (1993) *CENTURY Soil Organic Matter Model environment. Technical Documentation, Agroecosystem Version 4.0*. Great Plains System Research Unit Technical Report No. 4. USDA-ARS, Fort Collins, Colorado.

MIN, Z., LIANG-WU, L. and ZI-TONG, G. (1993) Age and some genetic characteristics of vertisols in China. *Pedosphere*, 3, 81-88.

MOLINA, J.A.E. and SMITH, P. (1998) Modelling carbon and nitrogen processes in soils. *Advances in Agronomy*, 62, 253-298.

MOLINA, J.A.E., CLAPP, C.E., SHAFFER, M.J., CHICHESTER, F.W., LARRSON, W.E. (1983) NCSOIL, a model of nitrogen and carbon transformations in the soil: description, calibration and behaviour. *Soil Science Society of America Journal*, 47, 85-91

MONREAL, C.M. and JANZEN, H.H. (1993) Soil organic carbon dynamics after 80 years of cropping a dark brown chernozem. *Canadian Journal of Soil Science*, 73, 133-136.

MONTEITH, J. (1977) Climate and the efficiency of crop production in Britain. *Royal Society of London, Philosophical Transactions,* 281, 277-294.

MOORHEAD, D.L. and REYNOLDS, J.F. (1991) A general model of litter decomposition in the northern Chihualahuan Desert. *Ecololgical Modelling*, 56, 197-219

MOSIER, A.R. (1998) Soil processes and global change. *Biology and Fertility of Soils*, 27, 221-229.

MOTAVALLI, P.P., PALM, C.A., PARTON, W.J., ELLIOT, E.T. and FREY, S.O. (1995) Soil pH and organic C dynamics in tropical forest soils: evidence from laboratory and simulation studies. *Soil Biology and Biochemistry*, 27, 1589-1599.

MOTAVALLI, P.P., PALM, C.A., PARTON, W.J., ELLIOTT, E.T. and FREY, S.D. (1994) Comparison of laboratory and modelling simulation methods for estimating soil carbon pools in tropical forest soils. *Soil Biology and Biochemistry*, 26, 935-944.

MUELLER, M.J. (1982) *Selected climatic data for a global set of standard stations for vegetation science. Tasks for Vegetation Sciences 5*. Dr W. Junk Publishers, The Hague.

MYERS, R.J.K. (1994) Modelling of soil organic matter dynamics. In: *Soil organic matter management for sustainable agriculture. A workship held in Ubon, Thailand, 24th-26th August 1994*, pp. 140-148 (Eds R.D.B. Lefroy, G.J. Blair and E.T. Craswell). ACIAR, Canberra.

N'DAYEGAMIYE, N. and ANGERS, D.A. (1993) Organic matter characteristics and water-stable aggregation of a sandy loam soil after 9 years of wood-residue applications. *Canadian Journal of Soil Science*, 73, 115-122.

NABUURS, G.J. and MOHREN, G.M.J. (1993) *Carbon fixation through forestation activities. IBN Research Report 93/4.* Institute for Forestry and Nature Research (IBN-DLO), Wageningen, The Netherlands.

NADELHOFFER, K.J., EMMETT, B.A., GUNDERSEN, P., KJONAAS, O.J., KOOPMANS, C.J., SCHLEPPI, P., TIETEMA, A. and WRIGHT, R.F. (1999) Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests. *Nature*, 398, 145-148.

NEMETH, T., CSATHO, P. and ANTON, A. (1997) Soil carbon dynamics in relation to cropping systems in principal ecoregions of Eastern Europe, with particular regard to Hungarian experiences. In: *Management of carbon sequestration in soil*, pp. 255-283 (Eds. R. Lal, J.M. Kimble, R.F. Follett and B.A. Stewart) CRC Press, Boca Raton.

NEMRY, B., FRANCOIS, L., WARNANT, P., ROBINET, F. and GERARD, J. (1996) The seasonality of the CO_2 exchange between the atmosphere and the land biosphere: a study with a global mechanistic vegetation model. *Journal of Geophysical Research*, 101, 7111-7125.

NG, E. and R.S. LOOMIS. (1984) *Simulation of growth and yield of potato crop*. Pudoc Wageningen, pp. 62-65.

NICOLARDOT, B., MOLINA, J.A.E. and ALLARD, M.R. (1994) C and N fluxes between pools of organic matter: model calibration with long-term field experimental data. *Soil Biology and Biochemistry*, 26, 245-251.

O'BRIEN, B.J. (1984) Soil organic carbon fluxes and turnover rates estimated from radiocarbon enrichments. *Soil Biology and Biochemistry*, 16, 115-120.

O'BRIEN, B.J. (1986) The use of natural and anthropogenic ^{14}C to investigate the dynamics of soil organic carbon. *Radiocarbon*, 28, 358-362.

O'BRIEN, B.J. and STOUT, J.D. (1978) Movement and turnover of soil organic matter as indicated by carbon isotope measurements. *Soil Biology and Biochemistry*, 10, 309-307.

OADES, J.M., GILLMAN, G.P., UEHARA, G., HUE, N.V., VAN NOORDWIJK, M., ROBERTSON, G.P. and WADA, K. (1989) Chapter 3 – Interactions of soil organic matter and variable-charge clays. In: *Dynamics of soil organic matter in tropical ecosystems*, pp. 69-95 (Eds. D.C. Coleman, J.M. Oades, and G. Uehara) NifTAL Project, University of Hawaii Press, Honolulu, Hawaii, USA.

ODELL, R.T., MELSTED, S.W. anD WALKER, W.M. (1984) Changes in organic carbon and nitrogen of Morrow Plot soils under different treatments, 1904-1973. *Soil Science*, 137, 160-171.

O'LEARY, G.J. (1994) *Soil water and nitrogen dynamics of dryland wheat in the Victoria Wimmera and Malle*. PhD Thesis, University of Melbourne.

PARFITT, P. R., THENG B.K.G., WHITTON J.S. and SHEPHERD, T.G. 1997. Effects of clay minerals and land use on organic matter pools. *Geoderma*, 75, 1-12.

PARFITT, R.L. (1990) Allophane in New Zealand - a review. *Australian Journal of Soil Research*, 28, 343-360.

PARSHOTAM, A. (1996) The Rothamsted soil-carbon turnover model – discrete to continuous form. *Ecological Modelling*, 86, 283-289.

PARSHOTAM, A. and HEWITT, A.E. (1995) Application of the Rothamsted carbon turnover model to soils in degraded semi-arid land in New Zealand. *Environment International*, 21, 693-697.

PARSHOTAM, A., TATE, K.R. and GILTRAP, D.J. (1995) Potential effects of climate and land-use change on soil carbon and CO_2 emissions from New Zealand's indigenous forests and unimproved grasslands. *Weather and Climate*, 15, 3-12.

PARTON, W.J. (1984) Predicting soil temperatures in a shortgrass steppe. *Soil Science,* 138, 93-101.

PARTON, W.J., COLE, C.V., STEWART, J.W.B., OJIMA, D.S. and SCHIMEL. D.S. (1989b) Simulating regional patterns of soil C, N and P dynamics in the US central grasslands region. In: *Ecology of arable land*, pp. 99-108. (Eds. M. Clarholm and L. Bergstrom) Kluwer Academic Publishers.

PARTON, W.J., HARTMAN, M., OJIMA, D. and SCHIMEL, D. (1998) DAYCENT and its land surface submodel: description and testing. *Global and Planetary Change*, 19, 35-48.

PARTON, W.J., OJIMA, D.S., COLE, C.V. and SCHIMEL, D.S. (1994) A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. In: *Quantitative modelling of soil forming processes, SSSA Special Publication 39*, pp. 147-167. SSSA, Madison, Wisconsin.

PARTON, W.J., SANFORD, R.L., SANCHEZ, P.A., STEWART, J.W.B., BONDE, T.A., CROSLEY, T.A., VAN VEEN, H. and YOST, R. 1989a. Chapter 6 – Modelling soil organic matter dynamics in tropical soils. In: *Dynamics of soil organic matter in tropical ecosystems*, pp. 69-95 (Eds. D.C. Coleman, J.M. Oades, and G. Uehara) NifTAL Project, University of Hawaii Press, Honolulu, Hawaii, USA.

PARTON, W.J., SCHIMEL, D.S., COLE, C.V. and OJIMA, D.S. (1987) Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal*, 51, 1173-1179.

PARTON, W.J., SCURLOCK, J.M.O., OJIMA, D.S., GILMANOV, T.G., SCHOLES, R.J., SCHIMEL, D.S., KIRCHNER, T., MENAUT, J.C., SEASTEDT, T., GARCIA MOAY, E., KAMNALRUT, A. and KINYAMARIO, J.I. (1993) Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Biogeochemical Cycles*, 7, 785-809.

PARTON, W.J., STEWART, J.W.B. and COLE, C.V. (1988) Dynamics of C, N, P and S in grassland soils: a model. *Biogeochemistry* 5,109-131

PÁSZTOR, L., SZABÓ, J. and VÁRALLYAY, G. (1996) Digging deep for global soil and terrain data. *GIS Europe* 5, 32-34

PATWARDHAN, A.S., CHINNASWAMY, R.V., DONIGAN, A.S. JR., METHERELL, A.K., BLEVINS, R.L., FRYE, W.W. and PAUSTIAN, K. (1995) Chapter 32: Application of the CENTURY soil organic matter model to a field site in Lexington, KY. In: *Soils and Global Change*, pp. 385-394 (Eds. R. Lal, J. Kimble, E. Levine and B.A. Stewart) CRC Press.

PAUL, E.A., FOLLETT, R.F., LEAVITT, S.W., HALVORSON, A., PETERSON, G.A. and LYON, D.J. (1997) Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. *Soil Science Society of America Journal*, 61, 1058-1067.

PAUL, E.A., HORWATH, W.R., HARRIS, D., FOLLETT, R., LEAVITT, S.W., KIMBALL, B.A. and PREGITZER, K. (1995) Chapter 24: Establishing the pool sizes and fluxes in CO_2 emissions from soil organic mater turnover. In: *Soils and Global Change*, pp. 297-305 (Eds. R. Lal, J. Kimble, E. Levine and B.A. Stewart) CRC Press.

PAUSTIAN, K. (1994) Modelling soil biology and biochemical processes for sustainable agricultural research. In: *Soil biota: management in sustainable farming systems*, pp. 182-193 (Eds. C.E. Pankhurst, B.M. Doube, V.V.S.R. Gupta and P.R. Grace) CSIRO, Melbourne, Australia.

PAUSTIAN, K., ANDREN, A., CLARHOLM, M., HANSSON, A.C., JOHANSSON, G., LAGERLOF, J., LINDBERG, T., PETTERSSON, R. and SOHLENIUS, B. (1990) Carbon and nitrogen budgets of four agro-ecosystems with annual and perennial crops, with and without N fertilization. *Journal of Applied Ecology*, 27, 60-84.

PAUSTIAN, K., ANDREN, O., JANZEN, H.H., LAL, R., SMITH, P., TIAN, G., TIESSEN, H., VAN NOORDWIJK, M. and WOOMER, P.L. (1997b) Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use and Management*, 13, 230-244.

PAUSTIAN, K., COLLINS, H.P. and PAUL, E.A. (1997a) Management controls on soil carbon. In: *Soil organic matter in temperate agroecosystems: long-term experiments in North America*, pp. 15-49 (Eds. E.A. Paul, K. Paustian, E.T. Elliott and C.V. Cole) CRC Press, Boca Raton.

PAUSTIAN, K., ELLIOTT, E.T. and KILLIAN, K. (1997c) Modeling soil carbon in relation to management and climate change in some agroecosystems in Central North America. In: *Soil Processes and the Global Carbon Cycle*, pp.

459-471 (Eds. R. Lal, J.M. Kimble, R.F. Follett and B.A. Stewart) CRC Press, Boca Raton.

PAUSTIAN, K., LEVINE, E., POST, W.M. and RYZHOVA, I.M. (1997d) The use of models to integrate information and understanding of soil C at the regional scale. *Geoderma*, 79, 227-260.

PAUSTIAN, K., PARTON, W.J. and PERSSON, J. (1992) Modeling organic matter in organic-amended and nitrogen-fertilized long-term plots. *Soil Science Society of America Journal*, 56, 476-488.

PENG, C. and APPS, M.J. (1998) Simulating carbon dynamics along the Boreal Forest Transect Case Study (BFTCS) in central Canada. 2. Sensitivity to climate change. *Global Biogeochemical Cycles*, 12, 393-402.

PENG, C., APPS, M.J., PRINCE, D.T., NALDER, I.A., and HALLIWELL, D.H. (1998) Simulating carbon dynamics along the Boreal Forest Transect Case Study (BFTCS) in central Canada. 1. Model testing. *Global Biogeochemical Cycles*, 12, 381-392.

PERRUCHOUD, D. (1996) *Modelling the dynamics of non-living organic carbon in a changing climate: a case study for temperate forests*. D.O. Perruchoud (1996) PhD Thesis. Diss ETH No 11900, 196pp. (Copies available for borrowing at ETH Library in ETH, Zurich, 8092 Zurich).

PLENTINGER, M.C. and PENNING DE VRIES, F.W.T (Eds) (1996) *CAMASE, register of agro-ecosystems models, version II*. DLO research Insitut for Agrobiology and soil fertility (AB-DLO). AB-DLO, Wageningen.

PLOCHL, M. and CRAMER, W. (1995) Coupling global models of vegetation structure and ecosystem processes. *Tellus*, 47B, 240-250.

POST W.M., EMANUEL W.R., ZINKE P.J. and STANGENBERGER A.G. (1982) Soil carbon pools and world life zones. *Nature*, 289, 156-159.

POST, W.M., A.W. KING and S.D. WULLSCHLEGER. (1996) Soil organic matter models and global estimates of soil organic carbon. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 201-222 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

POULTON, P.R. (1995a) Geescroft Wilderness, 1883-1995. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 385-390 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

POULTON, P.R. (1995b) Park Grass, 1856-1995. In: *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*, pp. 377-384 (Eds. D.S. Powlson, P. Smith and J.U. Smith) NATO ASI Series I, Vol.38. Springer-Verlag, Heidelberg.

PRENTICE, K.C. and FUNG, I.Y. (1990) The sensitivity of terrestrial carbon storage to climate change. *Nature*, 346, 48-51.

PRINCE, S. and GOWARD, S. (1995) Global net primary production: a remote sensing approach. *Journal of Biogeography*, 22, 815-835.

PUGET, P., CHENU, C. and BALESDENT, J. (1995) Total and young organic matter distributions in aggregates of silty cultivated soils. *European Journal of Soil Science*, 46, 449-459.

QUIROGA, A.R., BUSCHIAZZO, D.E. and PEINEMANN, N. (1996) Soil organic matter particle size fractions in soils of the semiarid Argentinian pampas. *Soil Science*, 161, 104-108.

RAPALEE, G., TRUMBORE, S.E., DAVIDSON, E.A., HARDEN, J.W. and VELDHUIS, H. (1998) Soil carbon stocks and their rates of accumulation and loss in a boreal forest landscape. *Global Biogeochemical Cycles*, 12, 687-701.

RHYZOVA, I.M. (1993) Analysis of soil-vegetation systems to variations in carbon turnover parameters based on a mathematical model. *Eurasian Soil Science*, 25, 43-50.

RICHARDS, J.F. (1990) Chapter 10: Land transformation. In: *The Earth as Transformed by Human Action - Global and Regional Changes in the Biosphere over the past 300 years*, pp. 163-178. (Eds. B.L. Turner II, W.C. Clark, R.W. Kates, J.F. Richards, J.T. Matthews and W.B. Meyer) Cambridge University Press.

RICHTER, D.D., MARKEWITZ, D., DUNSCOMB, J.K., HEINE, P.R., WELLS, C.G., STUANES, A., ALLEN, H.L., URREGO, B., HARRISON, K. and BONANI, G. (1995) Carbon cycling in a loblolly pine forest: implications for the missing carbon sink and for the concept of soil. In: *Carbon Forms and Functions in Forest Soils: proceedings of the 5th North American Forest Soils Conference, Gainesville Florida,* pp. 231-251. Soil Science Society of America, Madison, Wisconsin.

RIJTEMA, P.E. and KROES, J.G. (1991) Some results of nitrogen simulation with the model ANIMO. *Fertiliser Research*, 27, 189-198

ROBELS, M.D. and BURKE, I.C. (1998) Soil organic matter recover on conservation reserve program fields in South-Eastern Wyoming. *Soil Science Society of America Journal*, 62, 725-730.

ROMANYÁ, J., CORTINA, J., FALLOON, P., COLEMAN, K. and SMITH, P (2000) Modelling soil organic matter changes after planting fast growing Pinus radiata D. Don on Mediterranean agricultural soils. *European Journal of Soil Science*, 51, 627-641.

RUIMY, A., DEDIEU, G. and SAUGIER, B. (1996) TURC: a diagnositc model of continental gross primary productivity and net primary productivity. *Global Biogeochemical Cycles*, 10, 269-286.

RUNNING, S.W. and HUNT, E.R. (1993) Generalization of a forest ecosystem process model for other biomes, BIOME-BGC and an application for global-scale models. In: *Scaling Physiological Processes: Leaf to Globe*, pp. 141-158. (Eds. J. Ehleringer and C. Field) Academic Press, San Diego.

RUSTAD, L.E. and FERNANDEZ, I.J. (1998) Soil warming: consequences for foliar litter decay in a spruce-fir forest in Maine, USA. *Soil Science Society of America Journal*, 62, 1072-1080.

SAETRE, P., BRANDTBERG, P.O., LUNDKVIST, H. and BENGTSSON, J. (1999) Soil organisms and carbon, nitrogen and phosphorus mineralisaion in Norway and mixed Norway spruce – birch stands. *Biology and Fertility of Soils*, 28, 382-388.

SAGGAR S., TATE, K.R., FELTHAM, C.W., CHILDS, C.W. and PARSHOTAM, A. (1994) Carbon turnover in a range of allophanic soils amended with ^{14}C-labelled glucose. *Soil Biology and Biochemistry*, 26, 1263-1271.

SARKADI, J. (1991) Szerves- es mutragyak hatasa a buza es Kukorica termesere. *Agrokemia es Talajtan*, 40, 87-96.

SCHARPENSEEL, H.W., BECKER-HEIDMANN, P., NEUE, H.U. and TSUTSUKI, K. (1989) Bomb-carbon, ^{14}C-dating and ^{13}C-measurements as tracers of organic matter dynamics as well as of morphogenetic and turbation processes. *The Science of the Total Environment*, 81/82, 99-110.

SCHIFF, S.L., ARAVENA, R., TRUMBORE, S.E., HINTON, M.J., ELGOOD, R. and DILLON, P.J. (1997) Export of DOC from forested catchments on the Precambrian Shield of Central Ontario: clues from ^{13}C and ^{14}C. *Biogeochemistry*, 36, 43-65.

SCHIMEL, D.S., BRASLWELL, B.H., HOLLAND, E.A., MCKEOWN, R., OJIMA, D.S., PAINTER, T.H., PARTON, W.J. and TOWNSEND A. R. (1994) Climatic, edaphic and biotic controls over storage and turnover of carbon in soils. *Global Biogeochemical Cycles*, 8, 279-293.

SCHLESINGER, W.H. (1986) Changes in soil carbon storage and associated properties with disturbance and recovery. In: *The changing carbon cycle, a global analysis,* pp. 194-220. (Eds. J.R. Trabalka and D.E. Reichle) Springer-Verlag, New York.

SCHLESINGER, W.M. (1995) An overview of the carbon cycle. In: *Soils and Global Change,* pp. 9-25 (Eds. R. Lal, J. Kimble, E. Levine and B.A. Stewart, B.A.) CRC Press, Boca Raton.

SCHLESINGER, W.M. (2000) Carbon sequestration in soils. *Science*, 284, 2095.

SCHWARTZ, D., MARIOTTI, A., LAFRANCHI, R. and GUILLET, B. (1986) $^{13}C/^{12}C$ ratios of soil organic matter as indicators of vegetation changes in the Congo. *Geoderma*, 39, 97-103.

SELLERS, P. (1985) Canopy reflectance, photosynthesis and transpiration. *International Journal of Remote Sensing*, 6, 1335-1372.

SELLERS, P. (1987) Canopy reflectance, photosynthesis and transpiration II: the role of biophysics in the linearity of their inter-depedence. *Remote Sensing of the Environment*, 21, 143-183.

SELLERS, P., RANDALL, D. and COLLATZ, G. (1996) A revised land surface parameterization (SiB2) for atmospheric GCMs. Part I: Model formulation. *Journal of Climatology*, 9, 676-705.

SHANG, C. and TIESSEN, H. (1998) Organic matter stabilization in two semiarid tropical soils: size, density, and magnetic separations. *Soil Science Society of America Journal*, 62, 1247-1257.

SHEVSTOVA, L.K. and MIKHAILOV, B.G. (1992) *Control of soil humus balance based on statistical analysis of long-term field experiments database*. Manuscript from All Russian Institute for Fertilisers and Agricultural Soil Science, Moscow (in Russian)

SIEGENTHALER, U. and OESCHAGER, H. (1987) Biospheric CO_2 emissions during the past 200 year reconstructed by deconvolution of ice core data. *Tellus*, 39B, 140-154.

SIROTENKO, O.D. (1991) The USSR climate-soil-yield simulation system. *Meteorologia i Gidrologia,* 4, 67-73 (in Russian).

SIX, J., ELLIOTT, E.T., PAUSTIAN, K. and DORAN, J.W. (1998) Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Science Society of America Journal*, 62, 1367-1377.

SKJEMSTAD, J.O., CLARKE, P., TAYLOR, J.A., OADES, J.M. and MCCLURE, S.G. (1996) The chemistry and nature of protected carbon in soil. *Australian Journal of Soil Research*, 34, 251-271.

SKJEMSTAD, J.O., LE FEUVRE, R.P. and PREBBLE, R.E. (1990) Turnover of soil organic matter under pasture as determined by ^{13}C natural abundance. *Australian Journal of Soil Research*, 28, 267-276.

SMITH, G. (2001) Case study of cost versus accuracy of measuring carbon stock in a temperate forest carbon sequestration project. *Advances in Soil Science* (in press)

SMITH, J.U., BRADBURY, N.J. and ADDISCOTT, T.M. (1995) SUNDIAL: Simulation of nitrogen dynamics in arable land. A user friendly, PC based version of the Rothamsted Nitrogen Turnover Model. *Agronomy Journal*, 88, 38-43

SMITH, J.U., SMITH, P. and ADDISCOTT, T.M. (1996d) Quantitative methods to evaluate and compare soil organic matter (SOM) models. In *Evaluation of soil organic matter models using existing long-term datasets,* pp. 183-202. (Eds. D.S. Powlson, P. Smith and J.U. Smith), NATO ASI Series I, Vol. 38, Springer-Verlag, Heidelberg

SMITH, K.A. (1999) After the Kyoto Protocol: Can soil scientists make a useful contribution? *Soil Use and Management*, 15, 71-75.

SMITH, P. and POWLSON, D.S. (2000) Considering manure and carbon sequestration. *Science*, 287, 428-429.

SMITH, P., ANDREN, O., BRUSSARD, L., DANGERFIELD, M., EKSCHMITT, M., LAVELLE, K. and TATE, K. (1998b) Soil biota and global change at the ecosystem level: describing soil biota in mathematical models. *Global Change Biology*, 4, 773-784.

SMITH, P., FALLOON, P., COLEMAN, K., SMITH, J.U., PICCOLO, M., CERRI, C.C., BERNOUX, M., JENKINSON, D.S., INGRAM, J.S.I., SZABÓ, J. and PÁSZTOR, L. (1999) Modelling soil carbon dynamics in tropical ecosystems. In: *Global Climate Change and Tropical Soils (Advances in Soil Science),* pp. 341-364. (Eds R. Lal, J.M. Kimble, R.F. Follett and B.A. Stewart). CRC Press, Boca Raton.

SMITH, P., FALLOON, P.D., POWLSON, D.S. and SMITH, J.U. (2001a) Carbon Mitigation Options in Agriculture: Improving our Estimates for Kyoto. Chapter 4.13 In: *Sustainable Management of Soil Organic Matter, BSSS99 Conference Proceedings*, pp. 324-329. (Eds. R.M. Rees, B.C. Ball, C.D. Campbell, and C.A. Watson) CABI, Wallingford, UK.

SMITH, P., GOULDING, K.W., SMITH, K.A., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K.C. (2001b) Enhancing the carbon sink in agricultural soils: including trace gas fluxes in estimates of carbon mitigation potential. *Nutrient Cycling in Agroecosystems* (in press)

SMITH, P., GOULDING, K.W., SMITH, K.A., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K.C. (2000c) Including trace gas fluxes in estimates of carbon mitigation potential of UK agricultural land. *Soil Use and Management,* 16, 251-259.

SMITH, P., MILNE, R., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K.C. (2000d) Revised estimates of the carbon mitigation potential of UK agricultural land. *Soil Use and Management,* 16, 293-295.

SMITH, P., POWLSON D.S., SMITH, J.U. and ELLIOTT, T.E. (Eds.) (1997a) Evaluation and comparison of soil organic matter models using long-term datasets. Special Issue of *Geoderma*, 81, 1-255.

SMITH, P., POWLSON, D. S., GLENDINING, M. J. and SMITH, J. U. (1998c) Opportunities and limitations for C sequestration in European agricultural soils through changes in management. In: *Management of carbon sequestration in soil*, pp. 143-152 (Eds. R. Lal, J. Kimble, R. F. Follett and B. A. Stewart) CRC Press, Boca Raton, USA.

SMITH, P., POWLSON, D.S., GLENDINING, M.J. and SMITH, J.U. (1997b) Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. *Global Change Biology,* 3, 67-80

SMITH, P., POWLSON, D.S., GLENDINING, M.J. and SMITH, J.U. (1998a) Preliminary estimates of the potential for carbon mitigation in European soils through no-till farming. *Global Change Biology*, 4, 679-685.

SMITH, P., POWLSON, D.S., SMITH, J.U. and GLENDINING, M.J. (1996a) The GCTE SOMNET A global network and database of soil organic matter models and long-term datasets. *Soil Use and Management,* 12, 104

SMITH, P., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K. (2000a) Meeting the UK's Climate Change Commitments: Options for carbon mitigation on agricultural land. *Soil Use and Management,* 16, 1-11.

SMITH, P., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K. (2000b) Meeting Europe's Climate Change Commitments: Quantitative Estimates of the Potential for Carbon Mitigation by agriculture. *Global Change Biology,* 6, 525-539.

SMITH, P., POWLSON, D.S., SMITH, J.U., FALLOON, P., COLEMAN, K. and GOULDING, K. (2000e) Agricultural carbon mitigation options in Europe: Improved estimates and the Global perspective. *Acta Agronomica Hungarica,* 48, 209-216.

SMITH, P., SMITH, J.U. and POWLSON, D.S. (1996b) *Soil Organic Matter Network (SOMNET): 1996 Model and Experimental Metadata*. GCTE Report 7, GCTE Focus 3 Office, Wallingford, Oxon, 259pp.

SMITH, P., SMITH, J.U. and POWLSON, D.S. (1996c) Moving the British cattle herd. *Nature*, 381, 15.

SMITH, P., WILLISON, T. and HAKAMATA, T. (1997c) Modelling changes in the organic carbon content of Japanese soils using the Rothamsted Carbon Model. In: *Proceedings of the International Workshop on the Evaluation of Soil Processes on Carbon Cycling in Terrestrial Ecosystems and Their Modelling*. NIAES, Tsukuba, Japan, March 1997.

SMITH, T.M. and SHUGART, H.H. (1993) The transient repsonse of terrestrial carbon storage to a perturbed climate. *Nature*, 361, 523-526.

SPARLING, G., VOJVODIC-VUKOVIC, M. and SCHIPPER, L.A. (1998) Hot-water soluble C as a simple measure of labile soil organic matter: the relationship with microbial biomass C. *Soil Biology and Biochemistry*, 30, 1469-1472.

STEINER, R.A. and HERDT, R.W. (Eds.) (1993) *A Global Directory of Long-term agronomic experiments. Vol 1.: Non-European Experiments*. The Rockefeller Foundation, New York, N.Y.

STEMMER, M., GERZABEK, M.H. and KANDELER, E. (1998) Organic matter and enzyme activity in particle-size fractions of soils obtained after low-energy sonication. *Soil Biology and Biochemistry*, 30, 9-17.

STOUT, J.D. and GOH, K.M. (1980) The use of radiocarbon to measure the effects of earthworms on soil development. *Radiocarbon*, 22, 892-96.

STRAIN, B.R. and CURE, J.D. (Eds) (1985) *Direct effects of increasing carbon dioxide on vegetation.* DOE/ER-0238, National Technical Information Service, Washington, D.C.

SZABÓ. J., VÁRALLYAY, G., PÁSZTOR, L. (1996) HunSOTER, a digitised database for monitoring changes in soil properties in Hungary. In: *Proc of Soil monitoring in the Czech Republic*, pp. 150-156. Brno.

TANS, P.P., FUNG, I.Y. and TAKAHASHI., T. (1990) Observational constraints on the global atmospheric CO_2 budget. *Science*, 247, 1431-1438.

TATE, K.R. (1992) Assessment, based on a climosequence of soils in tussock grasslands, of soil carbon storage and release in response to global warming. *Journal of Soil Science*, 43, 697-707.

TATE, K.R., GILTRAP, D.J., PARSHOTAM, A., HEWITT, A.E., ROSS, D.J., KENNY, G.J. and WARRICK, R.A. (1996) Impacts of climate change on soils and land systems in New Zealand. In: *Greenhouse 1994 – Coping with Climate Change*, pp190-204. (Eds. W. Bouma, G. Pearman, and M.R. Manning) CSIRO Press, Adelaide.

TATE, K.R., PARSHOTAM, A. and ROSS, D.J. (1995) Soil carbon storage and turnover in temperate forests and grasslands - a New Zealand perspective. *Journal of Biogeography*, 22, 695-700.

TATE, K.R., ROSS, D.J., O'BRIEN, B.J. and KELLIHER, F.M. 1993. Carbon storage and turnover, and respiratory activity, in the litter and soil of an old-growth southern beech (Nothofagus) forest. *Soil Biology and Biochemistry*, 25, 1601-1612.

THENG, B.K.G., TATE, K.R. and BECKER-HEIDMANN, P. (1991) Towards establishing the age, location, and identity of the inert soil organic matter of a Spodosol. *Zeitschrift für Pflanzenernahrung und Bodenkunde*, 155, 181-184.

THORNLEY, J.H.M., CANNELL, M.G.R. (1994) Prediction of the effects of climate and management change on temperate grassland. *Journal of Agricultural Science*, 123, 151-152.

TIESZEN, H. and STEWART, J.W.B. (1983) Particle-size fractions and their use in studies of soil organic matter: II. Cultivation effects on organic matter composition in size fractions. *Soil Science Society of America Journal*, 47, 509-514.

TIETEMA, A. and VAN DAM, D. (1996) Calculating microbial carbon and nitrogen transformations in acid forest litter with ^{15}N enrichment and dynamic simulation modelling. *Soil Biology and Biochemistry*, 28, 953-965.

TIPPING, E., WOOF, C., RIGG, E., HARRISON, A.F., INESON, P., TAYLOR, P., BENHAM., D., POSKITT., J., ROWLAND., A.P., BOL., R. and HARKNESS, D.D. (1999) Climatic influences on the leaching of dissolved organic matter from upland UK moorland soils, investigated by a field manipulation experiment. *Environment International*, 25, 83-95.

TRACY, P.W., WESTFALL, D.G., ELLIOTT, E.T., PETERSON, G.A and COLE, C.V. (1990) Carbon, nitrogen, phosphorus and sulfur mineralization in plow and no-till cultivation. *Soil Science Society of America Journal*, 54, 457-461.

TRUMBORE, S.E. (1993) Comparison of carbon dynamics in tropical and temperate soils using radiocarbon measurements. *Global Biogeochemical Cycles*, 7, 275-290.

TRUMBORE, S.E. and ZHENG, S. (1996) Comparison of fractionation methods for soil organic matter ^{14}C analysis. *Radiocarbon*, 38, 219-229.

TRUMBORE, S.E., BONANI, G. and WOLFLI, W. (1990) The rates of carbon cycling in several soils from AMS ^{14}C measurements of fractionated soil organic matter. In: *Soils and the greenhouse effect*, pp. 407-414 (Ed. A.F. Bouwman) John Wiley and Sons, Chichester.

TRUMBORE, S.E., SCHIFF, S.L., ARAVENA, R. and ELGOOD, R. (1992) Sources and transformation of dissolved organic carbon in the Harp Lake forested catchment: the role of soils. *Radiocarbon*, 34, 626-635.

TRUMBORE, S.E., VOGEL, J.S. and SOUTHON, J.R. (1989) AMS ^{14}C measurements of fractionated soil organic matter: an approach to deciphering the soil carbon cycle. *Radiocarbon*, 31, 644-654.

TSUTSUKI, K., SUZUKI, C., KUWATSUKA, S., BECKER-HEIDMANN, P. and SCHARPENSEEL, P. 1988. Investigation on the stabilization of the humus in mollisols. *Zeitschrift fur Pflanzenernahrung und Bodenkunde*, 151, 87-90.

VAN DAM, D. and VAN BREEMEN, N. (1995) NICCE - a model for cycling of carbon and nitrogen isotopes in coniferous forest ecosystems. *Ecological Modelling*, 79, 255-275.

VAN DEN POL-VAN DASSELAAR, A., CORRE, W.J., PRIEME, A., KLEMEDTSSON, A.K., WESLIEN, P., STEIN, A., KLEMEDTSSON, L. and OENEMA, O. (1998) Spatial variability of methane, nitrous oxide, and carbon dioxide emissions from drained grasslands. *Soil Science Society of America Journal*, 62, 810-817.

VAN GINKEL, J.H. and GORISSEN, A. (1998) In situ decomposition of grass roots as affected by elevated atmospheric carbon dioxide. *Soil Science Society of America Journal*, 62, 951-958.

VANCLOOSTER, M., VEREEKEN, H., DIELS, J., HYUSMANS, F., VERSTRAETE, F., FEYEN, J. (1992) Effect of mobile and immobile water in predicting nitrogen leaching from cropped soils. *Modelling of Geo-Biosphere Processes*, 1, 23-40

VARGA-HASZONITS, Z. (1977) *Agrometeorologia*. Mezogazdasagi Kiado, Budapest. 224 pp.

VERBERNE, E.L.M., HASSINK, J., DE WILLIGEN, P., GROOT, J.J.R., VAN VEEN, J.A. (1990) Modelling soil organic matter dynamics in different soils. *Netherlands Journal of Agricultural Sciences*, 38, 221-238

WANDER, M.M., BIDART, M.G. and AREF, S. (1998) Tillage impacts on depth distribution of total and particulate organic matter in three Illinois soils. *Soil Science Society of America Journal*, 62, 1704-1711.

WANG, Y.P. and POLGLASE, P.J. (1995) Carbon balance in the tundra, boreal forest and humid tropical forest during climate change: scaling up from leaf physiology and soil carbon dynamics. *Plant, Cell and Environment*, 18, 1226-1244.

WARDLE, D.A. (1998) Controls of temporal variability of the soil microbial biomass: a global synthesis. *Soil Biology and Biochemistry*, 30, 1627-1637.

WHITBREAD, A.M. (1994) Soil organic matter: its fractionation and role in soil structure. In: *Soil Organic Matter Management for Sustainable Agriculture*, pp. 124-131. (Eds. R.D.B. Lefroy, G.J. Blair and E.T. Creswell) ACIAR, Canberra.

WHITEHOUSE, M.J. and LITTLER, J.W. (1984) Effect of pasture on subsequent wheat crops on a black earth soil of the Darling Downs. II. Organic C, nitrogen and pH changes. *Queensland Journal of Agricultural and Animal Sciences*, 41, 13-20.

WHITMORE, A.P. (1995) Modelling the mineralization and leaching of nitrogen form crop residues during three successive growing seasons. *Ecological Modelling*, 81, 233-241

WILLIAMS, J.H. (1988) *Guidelines, recommendations, rules and regulations for spreading manures, slurries and sludges in arable and grassland.* Commission of the European Communities, Luxembourg, 65pp.

WILLIAMS, J.R. and RENARD, K.G. (1985) Assessment of soil erosion and crop productivity with process models (EPIC). In: *Soil erosion and crop productivity*, pp. 67-103. (Eds. R.F. Follett and B.A. Stewart) ASA/CSSA/SSSA, Madison, Wisc,

WOODWARD, F.I., SMITH, T.M. and EMANUEL, W.R. (1995) A global land primary productivity and phytogeography model. *Global Biogeochemical Cycles*, 9, 471-490.

WOOMER, P.L. (1993) Modelling soil organic matter dynamics in tropical ecosystmes: model adoption, uses and limitations. In: *Soil organic matter dynamics and sustainability of tropical agriculture*, pp. 279-294. (Eds. K. Mulongoy and R. Merckx,) IITA/K.U.Leuven/Wiley-Sayce.

WU, J., O'DONNELL, A.G., SYERS, J.K., ADEY, M.A. and VITYAKON, P. 1998. Modelling soil organic matter changes in ley-arable rotations in sandy soils of Northeast Thailand. *European Journal of Soil Science*, 49, 463-470.

YAVITT, J.B., WILLIAMS, C.J. and KELMAN WIEDER, R. (1997) Production of methane and carbon dioxide in peatland ecosystems across North America: effects of temperature, aeration, and organic chemistry of peat. *Geomicrobiology Journal*, 14, 299-316.

YOST, R., LOAGUE, K. and GREEN R. (1993) Reducing variance in soil organic carbon estimates: soil classification and geostatistical approaches. *Geoderma*, 57, 247-262.

ZOGG, G.P., ZAK, D.R., RINGELBERG, D.B., MACDONALD, N.W., PRETZIGER, K.S. and WHITE, D.C. (1997) Compositional and functional shifts in microbial communities due to soil warming. *Soil Science Society of America Journal*, 61, 475-481.

PUBLICATIONS

FALLOON, P.D. and SMITH, P. (1998) The role of refractory soil organic matter in soil organic matter models. *Mitteilungen der Deutschen Bodenkundlichen Gesellschaft,* 87, 253-264.

FALLOON, P.D. and SMITH, P. (2000) Modelling refractory organic matter. *Biology and Fertility of Soils,* 30, 388-398.

FALLOON, P.D., SMITH, J.U. and SMITH, P. (1999a) A Review of Decision Support Systems for fertiliser application and manure management. *Acta Agronomica Hungarica,* 47, 227-236

FALLOON, P.D., SMITH, P., COLEMAN, K. and MARSHALL, S. (1998a) Estimating the size of the inert organic matter pool for use in the Rothamsted carbon model. *Soil Biology and Biochemistry*, 30, 1207-1211.

FALLOON, P.D., SMITH, P., COLEMAN, K. and MARSHALL, S. (2000) How important is inert organic matter for predictive soil carbon modelling using the Rothamsted carbon model? *Soil Biology and Biochemistry,* 32, 433-436.

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (2001b) Comparing estimates of regional C sequestration potential using GIS, dynamic SOM models, and simple relationships. *Advances in Soil Science* (in press)

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (2001a) SOM sustainability and agricultural management - predictions at the regional level. Chapter 2.1 In: *'Sustainable Management of Soil Organic Matter' BSSS99 Conference Proceedings*, pp. 54-59. (Eds. R.M. Rees, B.C. Ball, C.D. Campbell and C.A. Watson) CABI, Wallingford, UK.

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (1999b) Linking GIS and dynamic SOM models: estimating the regional C sequestration potential of agricultural management options. *Journal of Agricultural Science, Cambridge*, 133, 341-342.

FALLOON, P.D., SMITH, P., SMITH, J.U., SZABO, J., COLEMAN, K. and MARSHALL, S. (1998b) Regional estimates of carbon sequestration potential: linking the Rothamsted Carbon Turnover model to GIS databases. *Biology and Fertility of Soils*, 27, 236-241.

ROMANYÁ, J., CORTINA, J., FALLOON, P., COLEMAN, K. and SMITH, P (2000) Modelling soil organic matter changes after planting fast growing Pinus radiata D. Don on Mediterranean agricultural soils. *European Journal of Soil Science*, 51, 627-641.

SMITH, P., FALLOON, P., COLEMAN, K., SMITH, J.U., PICCOLO, M., CERRI, C.C., BERNOUX, M., JENKINSON, D.S., INGRAM, J.S.I., SZABÓ, J. and PÁSZTOR, L. (1999) Modelling soil carbon dynamics in tropical ecosystems. In: *Global Climate Change and Tropical Soils (Advances in Soil*

Science), pp. 341-364. (Eds R. Lal, J.M. Kimble, R.F. Follett and B.A. Stewart). CRC Press, Boca Raton.

SMITH, P., FALLOON, P.D., POWLSON, D.S. and SMITH, J.U. (2001a) Carbon Mitigation Options in Agriculture: Improving our Estimates for Kyoto. Chapter 4.13 In: *Sustainable Management of Soil Organic Matter, BSSS99 Conference Proceedings*, pp. 324-329. (Eds. R.M. Rees, B.C. Ball, C.D. Campbell, and C.A. Watson) CABI, Wallingford, UK.

SMITH, P., GOULDING, K.W., SMITH, K.A., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K.C. (2001b) Enhancing the carbon sink in agricultural soils: including trace gas fluxes in estimates of carbon mitigation potential. *Nutrient Cycling in Agroecosystems* (in press)

SMITH, P., GOULDING, K.W., SMITH, K.A., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K.C. (2000c) Including trace gas fluxes in estimates of carbon mitigation potential of UK agricultural land. *Soil Use and Management,* 16, 251-259.

SMITH, P., MILNE, R., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K.C. (2000d) Revised estimates of the carbon mitigation potential of UK agricultural land. *Soil Use and Management,* 16, 293-295.

SMITH, P., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K. (2000a) Meeting the UK's Climate Change Commitments: Options for carbon mitigation on agricultural land. *Soil Use and Management,* 16, 1-11.

SMITH, P., POWLSON, D.S., SMITH, J.U., FALLOON, P.D. and COLEMAN, K. (2000b) Meeting Europe's Climate Change Commitments: Quantitative Estimates of the Potential for Carbon Mitigation by agriculture. *Global Change Biology,* 6, 525-539.

SMITH, P., POWLSON D.S., SMITH, J.U., FALLOON, P., COLEMAN, K. and GOULDING, K. (2000e) Agricultural carbon mitigation options in Europe: Improved estimates and the Global perspective. *Acta Agronomica Hungarica,* 48, 209-216.