

FINAL REPORT

[16 December 2008]

“REVIEW OF EXISTING INFORMATION ON THE INTERRELATIONS BETWEEN SOIL AND CLIMATE CHANGE”

Contract number 070307/2007/486157/SER/B1

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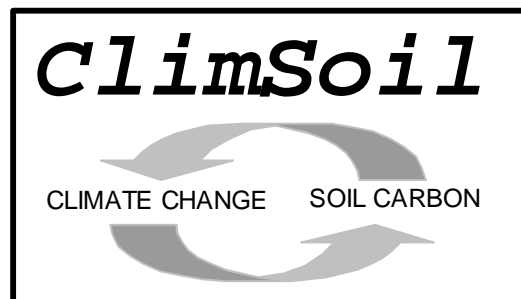
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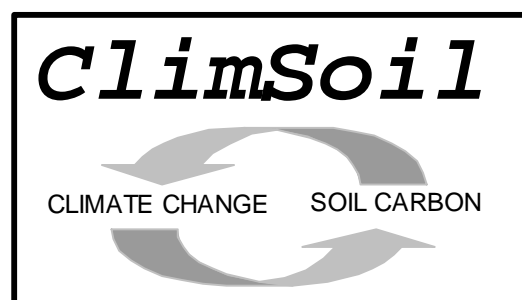
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**“SERVICE CONTRACT: REVIEW OF EXISTING
INFORMATION ON THE INTERRELATIONS BETWEEN SOIL
AND CLIMATE CHANGE”**

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Acronyms and abbreviations

C	Carbon
CAP	Common Agricultural Policy
CO ₂	Carbon Dioxide
CH ₄	Methane.
CDM	Clean Development Mechanism
CRF	Common Reporting Format
COP	Conference of the Parties
DOC	Dissolved Organic Carbon
EEA	European Environmental Agency
ERU	Emission Reduction Unit
EU	European Union
FAO	Food and Agriculture Organization of the United Nations.
GAEC	Good Agricultural and Environmental Condition
GWP	Global Warming Potential
GHCN	Global Historical Climatology Network
GHG	Greenhouse Gases
GPP	Gross Primary Production
IPCC	Intergovernmental Panel on Climate Change
IPPC	Integrated Pollution Prevention and Control
JI	Joint Implementation
JRC	Joint Research Centre
LFA	Less Favoured Area

LULUCF	Land Use, Land-Use Change, and Forestry
MS	Member States
N	Nitrogen
N ₂ O	Nitrous Oxide.
NBP	Net Biome Production
NEP	Net Ecosystem Production
NIR	National Inventory Report
NPP	Net Primary Production
NRCS	National Resources Conservation Service
NSI	National Soil Inventory
NVZ	Nitrate Vulnerable Zones
OC	Organic Carbon
OECD	Organisation for Economic Cooperation and Development
POC	Particulate Organic Carbon
RES	renewable Energy Sources
SD	Standard Deviation
SMR	Statutory Management Requirement
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SPS	Singel Payment System
UNFCCC	United Nations Framework Convention on Climate Change
USDA	United States Department of Agriculture

Key messages

1. *Carbon stock in EU soils* – The soil carbon stocks in the EU27 are around 75 billion tonnes of carbon (C); of this stock around 50% is located in Sweden, Finland and the United Kingdom (because of the vast area of peatlands in these countries) and approximately 20% is in peatlands, mainly in countries in the northern part of Europe. The rest is in mineral soils, again the higher amount being in northern Europe.
2. *Soils sink or source for CO₂ in the EU* – Both uptake of carbon dioxide (CO₂) through photosynthesis and plant growth and loss of CO₂ through decomposition of organic matter from terrestrial ecosystems are significant fluxes in Europe. Yet, the net terrestrial carbon fluxes are typically 5-10 times smaller relative to the emissions from use of fossil fuel of 4000 Mt CO₂ per year.
3. *Peat and organic soils* - The largest emissions of CO₂ from soils are resulting from land use change and especially drainage of organic soils and amount to 20-40 tonnes of CO₂ per hectare per year. The most effective option to manage soil carbon in order to mitigate climate change is to preserve existing stocks in soils, and especially the large stocks in peat and other soils with a high content of organic matter.
4. *Land use and soil carbon* – Land use and land use change significantly affects soil carbon stocks. On average, soils in Europe are most likely to be accumulating carbon on a net basis with a sink for carbon in soils under grassland and forest (from 0 - 100 billion tonnes of carbon per year) and a smaller source for carbon from soils under arable land (from 10 - 40 billion tonnes of carbon per year). Soil carbon losses occur when grasslands, managed forest lands or native ecosystems are converted to croplands and vice versa carbon stocks increase, albeit it slower, following conversion of cropland.
5. *Soil management and soil carbon* – Soil management has a large impact on soil carbon. Measures directed towards effective management of soil carbon are available and identified, and many of these are feasible and relatively inexpensive to implement. Management for lower nitrogen (N) emissions and lower C emissions is a useful approach to prevent trade off and swapping of emissions between the greenhouse gases CO₂, methane (CH₄) and nitrous oxide (N₂O).
6. *Carbon sequestration* – Even though effective in reducing or slowing the build up of CO₂ in the atmosphere, soil carbon sequestration is surely no 'golden bullet' alone to fight climate change due to the limited magnitude of its effect and its potential reversibility; it could, nevertheless, play an important role in climate mitigation alongside other measures, especially because of its immediate availability and relative low cost for 'buying' us time.
7. *Effects of climate change on soil carbon pools* – Climate change is expected to have an impact on soil carbon in the longer term, but far less an impact than does land use change, land use and land management. We have not found strong and clear evidence for either overall and combined positive or negative impact of climate change (atmospheric CO₂, temperature, precipitation) on soil carbon stocks. Due to the relatively large gross exchange of CO₂ between atmosphere and soils and the significant stocks of carbon in soils, relatively small changes in these large and opposing fluxes of CO₂, i.e. as result of land use (change), land management and climate change, may have significant impact on our climate and on soil quality.

8. *Monitoring systems for changes in soil carbon* – Currently, monitoring and knowledge on land use and land use change in EU27 is inadequate for accurate calculation of changes in soil carbon contents. Systematic and harmonized monitoring across EU27 and across relevant land uses would allow for adequate representation of changes in soil carbon in reporting emissions from soils and sequestration in soils to the UNFCCC.
9. *EU policies and soil carbon* – Environmental requirements under the Cross Compliance requirement of CAP is an instrument that may be used to maintain SOC. Neither measures under UNFCCC nor those mentioned in the proposed Soil Framework Directive are expected to adversely impact soil C. EU policy on renewable energy is not necessarily a guarantee for appropriate (soil) carbon management.

Executive summary

The European Commission has recently adopted the Thematic Strategy for soil protection (COM(2006)231 final), with the objective to ensure that Europe's soils remain healthy and capable of supporting human activities and ecosystems. Climate change is identified as a common element in many soil threats. Therefore the Commission intends to assess the actual contribution of the protection of soil to climate change mitigation and the effects of climate change on soil productivity and the possible depletion of soil organic matter as result of climate change. The objective of this study is to provide a state of the art and more robust understanding of interactions between soil under different land uses and climate change than is available now, through a comprehensive literature review and expert judgment.

1 Carbon stock in EU soils

The amount of carbon in European soils is estimated to be equal to 73 to 79 billion tonnes. These estimates are based on applying a common methodology across Europe, the larger estimate was based on a method developed by the Joint Research Centre of the European Commission and the smaller estimate on a soil organic carbon (SOC) map of the United States Department of Agriculture. These two methodologies gave similar estimates for most of the European countries. The estimates were of the same order of magnitude as national estimates based on national methodologies and are therefore deemed reliable.

Carbon in EU27 soils is concentrated in specific regions: roughly 50% of the total carbon stock is located in Sweden, Finland and the United Kingdom (because of the vast area of peatlands in these countries) and approximately 20% of the carbon stock is in peatlands mainly in the northern parts of Europe. The rest of soil C is in mineral soils, again the higher amount being in northern Europe.

2 Soils sink or source for CO₂ in the EU

Uptake of carbon dioxide (CO₂) through photosynthesis and plant growth and loss (decomposition) of organic matter from terrestrial ecosystems are both significant fluxes in Europe. Yet, the net terrestrial carbon fluxes (uptake of CO₂ minus respiration by vegetation and soils) are typically smaller relative to the emissions from use of fossil fuel. The current changes in the carbon pool of the European soils were estimated from different studies using different methods, by land use category using models that simulate carbon cycling in soil. The results of the different studies deviated considerably from each other, and all results were accompanied with wide uncertainty ranges. Some studies on the basis of measurements in UK, Belgium and France on soil carbon over longer periods show losses of carbon especially from cropland; other studies from the UK and from the Netherlands show no change or increases in soil carbon stocks over time.

Grassland soils were found in all studies to generally accumulate carbon. However, the studies differ on the amount of carbon accumulated. In one study, the sink estimate ranged from 1 to 45 million tonnes of carbon per year and, in another study, the mean estimate was 101 million tonnes per year, although with a high uncertainty.

Cropland generally acts as a carbon source, although existing estimates vary highly. In one study, the carbon balance estimates of croplands ranged from a carbon sink equal to

10 million tonnes of carbon per year to a carbon source equal to 39 million tonnes per year. In another study, croplands in Europe were estimated to be losing carbon up to 300 million tonnes per year. The latter is now perceived as a gross overestimation.

Forest soils generally accumulate carbon in each European country. Estimates range from 17 to 39 million tonnes of carbon per year with an average of 26 million tonnes per year in 1990 and to an average of 38 million tons of carbon per year in 2005.

It would seem that on a net basis, soils in Europe are on average most likely accumulating carbon. However, given the very high uncertainties in the estimates for cropland and grassland, it would not seem accurate and sound to try to use them to aggregate the data and produce an estimate of the carbon accumulation and total carbon balance in European soils.

3 Peat and organic soils

The current area of peat occurrence in the EU Member States and Candidate Countries is over 318 000 km². More than 50% of this surface is in just a few northern European countries (Norway, Finland, Sweden, United Kingdom); the remainder in Ireland, Poland and Baltic states. Of that area, approximately 50% has already been drained, while most of the undrained areas are in Finland and Sweden.

Although there are gaps in information on land use in peatlands, it can be estimated that water saturated organic rich soil (peatland) have been drained for:

- agriculture – more than 65 000 km² (20% of the total European peatland area);
- forestry – almost 90 000 km² (28%);
- peat extraction – only 2 273 km² (0.7%).

This is important as the largest emissions of CO₂ from soils are resulting from land use change and related drainage of organic soils and amount to 20-40 tonnes of CO₂ per hectare per year. The emission from cultivated and drained organic soils in EU27 is approximately 100 Mt CO₂ per year. Peat layer have been lost by oxidation during land use, but the estimate derivable from the published data, ca. 18 000 km², is very probably an underestimate.

4 Land use and soil carbon

Monitoring programs, long term experiments and modelling studies all show that land use significantly affects soil carbon stocks. Soil carbon losses occur when grasslands, managed forest lands or native ecosystems are converted to croplands. Vice versa soil carbon stocks are restored when croplands are either converted to grasslands, forest lands or natural ecosystems. Conversion of forest lands into grasslands does not affect soil carbon in all cases, but does reduce total ecosystem carbon due to the removal of aboveground biomass.

The more carbon is present on the soil, the higher the potential for losing it. Therefore the potential losses of unfavourable land use changes on highly organic peat soils are a major risk. The most effective strategy to prevent global soil carbon loss would be to halt land conversion to cropland, but this may conflict with growing global food demand unless per-area productivity of the cropland continues to grow.

5 Soil management and soil carbon

Soil management practices are an important tool to affect the soil carbon stocks. Suitable soil management strategies have been identified within all different land use categories and are available and feasible to implement. These are:

- On **cropland**, soil carbon stocks can be increased by
 - (i) agronomic measures that increase the return of biomass carbon to the soil,
 - (ii) tillage and residue management,
 - (iii) water management,
 - (iv) agro-forestry.
- On **grassland**, soil carbon stocks are affected by
 - (i) grazing intensity
 - (ii) grassland productivity,
 - (iii) fire management and
 - (iv) species management.
- On **forest lands**, soil carbon stocks can be increased by
 - (i) species selection,
 - (ii) stand management,
 - (iii) minimal site preparation,
 - (iv) tending and weed control,
 - (v) increased productivity,
 - (vi) protection against disturbances and
 - (vii) prevention of harvest residue removal.
- On **cultivated peat soils** the loss of soil carbon can be reduced by
 - (i) higher ground water tables.
- On **less intensively / un-managed heathlands and peatlands**, soil carbon stocks are affected by
 - (i) water table (drainage),
 - (ii) pH (liming), fertilisation,
 - (iii) burning
 - (iv) grazing.
- On **degraded lands**, carbon stocks can be increased after restoration to a productive situation.

Given that land use change is often driven by demand and short term economic revenues, the most realistic option to improve soil carbon stocks is to a) protect the carbon stocks in highly organic soils such as peats mostly in northern Europe, and b) to improve the way in which the land is managed to maximise carbon returns to the soil and minimise carbon losses. Increased nitrogen fertilizer use has made a large contribution to the growth in productivity, but further increased use will lead to greater emissions of nitrous oxide (N₂O). Hence future emphasis should be concentrated on the other main driver of productivity, i.e. improved crop varieties.

6 Carbon sequestration

Soils contain about three times the amount of carbon globally as vegetation, and about twice that in the atmosphere. There is a significant and large uncertainty associated with the response of soil carbon (and other pools of biospheric carbon) to future climate changes. Most response are calculated with simulation models with some models

predicting large releases of additional carbon from soils and vegetation under climate change, and others suggesting only small feedback. The maximum possible amount of carbon that soil sequestration could achieve is about one third of the current yearly increase in atmospheric carbon (as carbon dioxide) stocks. This is about one seventh of yearly anthropogenic carbon emissions of 7500 Mt C. In Europe emissions of greenhouse gases amount to approximately 4100 Mt CO₂ (or 1000 Mt C) per year.

Today, soils in Europe are most likely a sink and the best estimate is that they sequester up to 100 Mton C per year. Higher sequestration is possible with adequate soil management. Soil C-sequestration alone is surely no 'golden bullet' to fight climate change but is it realistic to link climate change with soil carbon conservation, as soil carbon sequestration is cost competitive, of immediate availability, does not require the development of new and unproven technologies, and provides comparable mitigation potential to that available in other sectors.

Therefore, given that climate change needs to be tackled urgently if atmospheric carbon dioxide concentrations are to be stabilized below levels thought to be irreversible, soil carbon sequestration or the even more effective conservation of current carbon stocks in soils has a key role to play in any raft of measures used to tackle climate change.

7 Effects of climate change on soil carbon pools

We have not found strong and clear evidence for either an overall combined positive or negative impact of climate change (raised atmospheric CO₂ concentration, temperature, precipitation) on terrestrial carbon stocks. There are suggestions for enhancing soil C stocks at higher atmospheric CO₂ concentration and reducing soil C stocks when temperatures are rising. Most studies have taken moderate changes in temperature increases and sudden and more severe changes in temperature of precipitation have not been considered, as the management of land and soils overrules any impact on soil carbon from climate change.

All of the factors of climate change (raised atmospheric CO₂ concentration, temperature, precipitation) affect soil C, with the effect on soils of CO₂ being indirect (through photosynthesis) and the effects of weather factors being both direct and indirect. Climate change affects soil carbon pools by affecting each of the processes in the C-cycle: photosynthetic C-assimilation, litter fall, decomposition, surface erosion, hydrological transport. Due to the relatively large gross exchange of CO₂ between atmosphere and soils and the significant stocks of carbon in soils, relatively small changes in these large but opposing fluxes of CO₂ may have significant impact on our climate and on soil quality. Therefore, managing these fluxes (through proper soil management) can help mitigate climate change considerably.

8 Monitoring systems for changes in soil carbon

Today, monitoring and knowledge on land use and land use change in EU27 is insufficient, yet land use and land use change are a key source of greenhouse gas emissions in many of the EU27 member states. Soil monitoring in EU27 seems like the Tower of Babel: countries tend to have their own systems, if any, sometimes even more than one system, and the results are not fully compatible across EU27. The few existing systems tend to have been set up for different purposes, often not including that of providing evidence concerning the impact of climate change on soil carbon pools. This

lack of systematic and comparable data gathering and analyses seriously hampers any attempt to provide reliable, EU-wide data on the soil carbon stock and changes therein.

Moreover, the new goal of monitoring stock-changes rather than stock-magnitudes may necessitate significant changes to current soil sampling procedures.

Given the lack of reliable national monitoring systems and without an EU wide harmonized system of monitoring of soil carbon in place, it would be a significant advance if the EU were to ask for a design or initiate implementation of a harmonized EU27 monitoring for land uses and for specific activities that affect soil carbon stocks and emissions of CO₂. Such monitoring would also allow for adequate representation of changes in soil carbon in EU27 in reporting to the United Nations Framework Convention to Combat Climate Change (UNFCCC).

9 EU policies and soil carbon

We have critically reviewed EU policies that are likely to have impacts on soil carbon (C) to assess whether any of those policies might have adverse impacts on soil C in the long term. Policies reviewed were the Common Agricultural Policy (CAP), the Nitrates Directive, the Renewable Energy Sources Directive, the Biofuels Directive, Waste policy and the EU Thematic Strategy for soil protection.

Legislation to encourage the production of arable crops to provide feed stocks for renewable energy is perhaps the legislation most likely to lead to decreases in the overall carbon content of European soils. While studies may indicate much of the demand may be met by imports from outside the EU, and hence may have little impacts on soil C within the EU, there may be serious implications for soil C stocks in those countries which supply renewable energy or their substrates.

We conclude that the need to comply with environmental requirements under the Cross Compliance requirement of CAP is an instrument that may be used to maintain SOC. The measures required under UNFCCC are not likely to adversely impact soil C. Nor are there any measures in the proposed Soil Framework Directive that would be expected to lead to decreases on soil C.

1 Introduction

1.1 Background and objective

The European Commission has recently adopted the Thematic Strategy on the protection of soil and its accompanying proposal for a Soil Framework Directive¹. This is a strategy to ensure that Europe's soils remain healthy and capable of supporting human activities and ecosystems. Member States have to identify the areas in their national territory where there is a decisive evidence or legitimate ground for suspicion that the following soil degradation has occurred or is likely to occur: erosion by water or wind, organic matter decline, compaction, salinisation and landslides. Climate change is identified for all of these threats as a common element for the identification of areas at risk.

In the Thematic Strategy, the European Commission has announced that it “will build a robust approach to address the interaction between soil protection and climate change from the viewpoints of research, economy and rural development so that policies in these areas are mutually supportive”. It includes a proposal for a Soil Framework Directive aiming at strengthening, among other things, the role of soil in climate change mitigation. In fact, soil as a carbon pool is explicitly mentioned as a soil function that should be preserved.

It is against this background that the Commission intends to assess the actual contribution of the protection of soil to climate change mitigation and the effects of climate change on soil productivity and the possible depletion of soil organic matter as result of climate change. The objective of this study is to provide a state of the art and more robust understanding of interactions between soil under different land uses and climate change than is available now, through a comprehensive review and expert judgment by European experts. The main information sources were the Intergovernmental Panel on Climate Change (IPCC) 4th Assessment Report and other (supra)national assessment reports, published peer reviewed literature, national and European reports and documents, results from ongoing national and European projects and expert knowledge.

1.2 Soil organic matter

Organic matter is one of the most complex and dynamic components of soils. It is a mixture of plant and animal residues, living and decaying organisms and humic substances. Plant residues are usually roots and stubbles, but also include harvest residues. Animal residues are dead animals, excreta from grazing animals or applied manures from stables. These residues are present in the soil as fresh material, but also in all stages of decomposition. All residues are broken down by the soil organisms (Figure 1), ranging from microscopically small microbes and fungi to the relatively large earthworms.

¹ COM(2006)231 and COM(2006)232, 22.9.2006 (<http://ec.europa.eu/environment/soil/index.htm>).

Soil organisms continually change organic compounds from one form to another. Eventually, the organic compounds become stabilized and resistant to further changes.

Under normal conditions all plant residues are broken down by micro-organisms. However, very wet and anaerobic conditions in soils may hamper the breakdown which leads to a large accumulation of plant material and thus to the formation of peat soils. Therefore, compared to mineral soils, peat soils contain huge amounts of organic matter. Most peatlands were formed in lowlands collecting waters from catchments, but high precipitation and humidity has also led to the formation of bogs on hills and slopes.

The presence of soil organic matter in soils is particularly important to several environmental and ecological functions of soils such as fertility, biological activity and gas exchanges with the atmosphere and leaching losses to water. From a farming perspective, soil organic matter is important for nutrient cycling, water dynamics and soil structure.

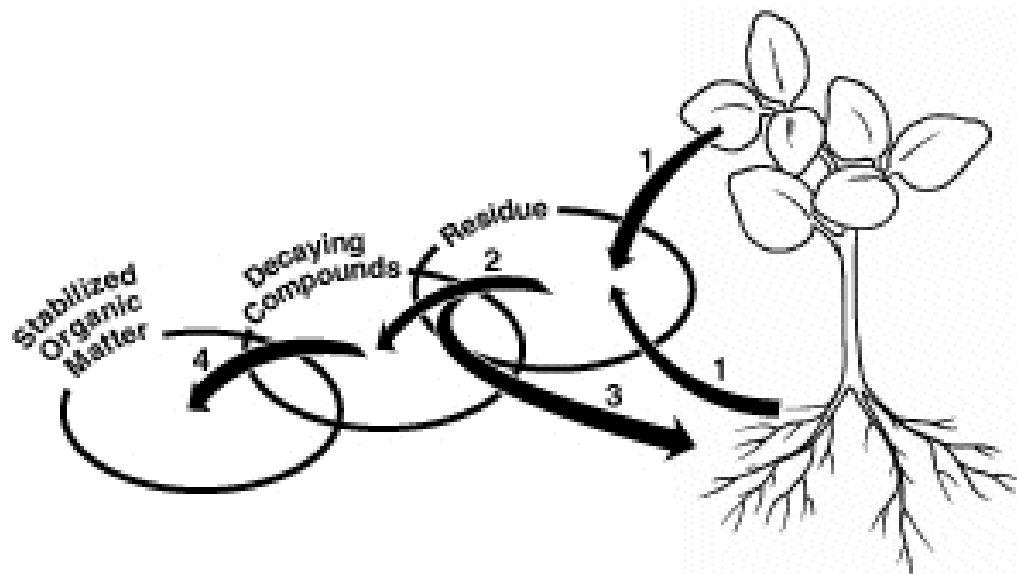


Figure 1 The changing forms of organic matter (University of Minnesota, Organic matter mangement)

The turnover rate of soil organic matter is an important property for the characterization of different types of organic matter. For a better understanding, the various organic matter components in soil are often grouped together in categories with similar breakdown characteristics. Many soil organic matter models either use a two-component approach with a stable and reactive organic matter pool, or a three-component approach with pools representing a fast, intermediate and slow organic matter turnover.

The amount of organic matter in any soil at a given moment is the net result of the addition through plant and animal residues and the loss through decomposition. The major factors affecting this balance are soil management, soil texture, climate and vegetation.

1.3 The global carbon cycle

Organic matter contains approximately 50% of carbon (C). Soils worldwide hold 2500 Gt C, of which 1500 Gt C is found in organic matter (Lal, 2004, Batjes, 1996), the focus of focus of this study. For reference, the atmospheric pool of carbon amounts to 760 Gt and the terrestrial biotic pool to 560 Gt C (Figure 2).

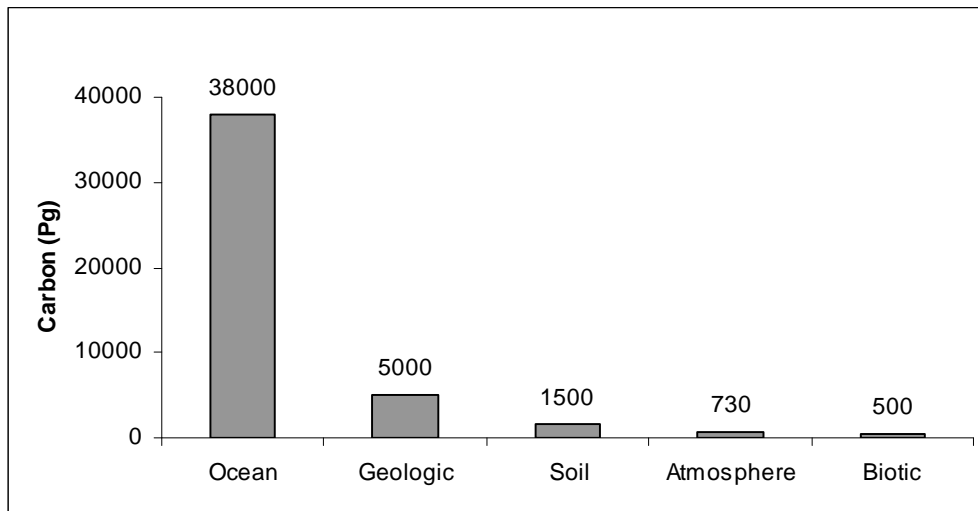


Figure 2 Principal global carbon pools in Pg (1 Pg = 1 Gt = 10¹⁵ g).

Figure 3 (IPCC 2001) presents a schematic diagram of the C cycle, showing the main pools and flows of the natural global C cycle, as well as the human perturbation to the flows of carbon between the pools.

The gross photosynthetic uptake of carbon from the atmosphere to plants growing on land (Gross Primary Productivity [GPP]) is in the order of 120 Pg C y⁻¹ (IPCC 2000a). However, plants respire approximately 50% of GPP, leaving a Net Primary Productivity (NPP) in the order of 60 Pg C y⁻¹ (IPCC 2000a). In turn, all organisms consuming plant material respire carbon dioxide (CO₂), returning 55 Pg C y⁻¹ to the atmosphere. Additionally, fires are responsible for CO₂ release of some 4 Pg C y⁻¹. The size of the pool of Soil Organic Carbon (SOC), 1500 Pg C, is therefore large compared to the annual fluxes of C of 120 Pg C (see Figure 3, top) to and from the terrestrial biosphere (Smith 2004).

During the 1990s, fossil fuel combustion and cement production emitted approximately 5 to 6 Pg C y⁻¹ to the atmosphere, whilst land-use change emitted nearly 2 Pg C y⁻¹ (Schimel *et al.* 2001; IPCC 2001). These C sources led to an increase of atmospheric C of some 3 Pg C y⁻¹. The oceans absorbed another 2 Pg C y⁻¹ and the estimated terrestrial sink was also in the order of 2 Pg C y⁻¹ (Schimel *et al.* 2001; IPCC 2001) (see Figure 3, bottom).

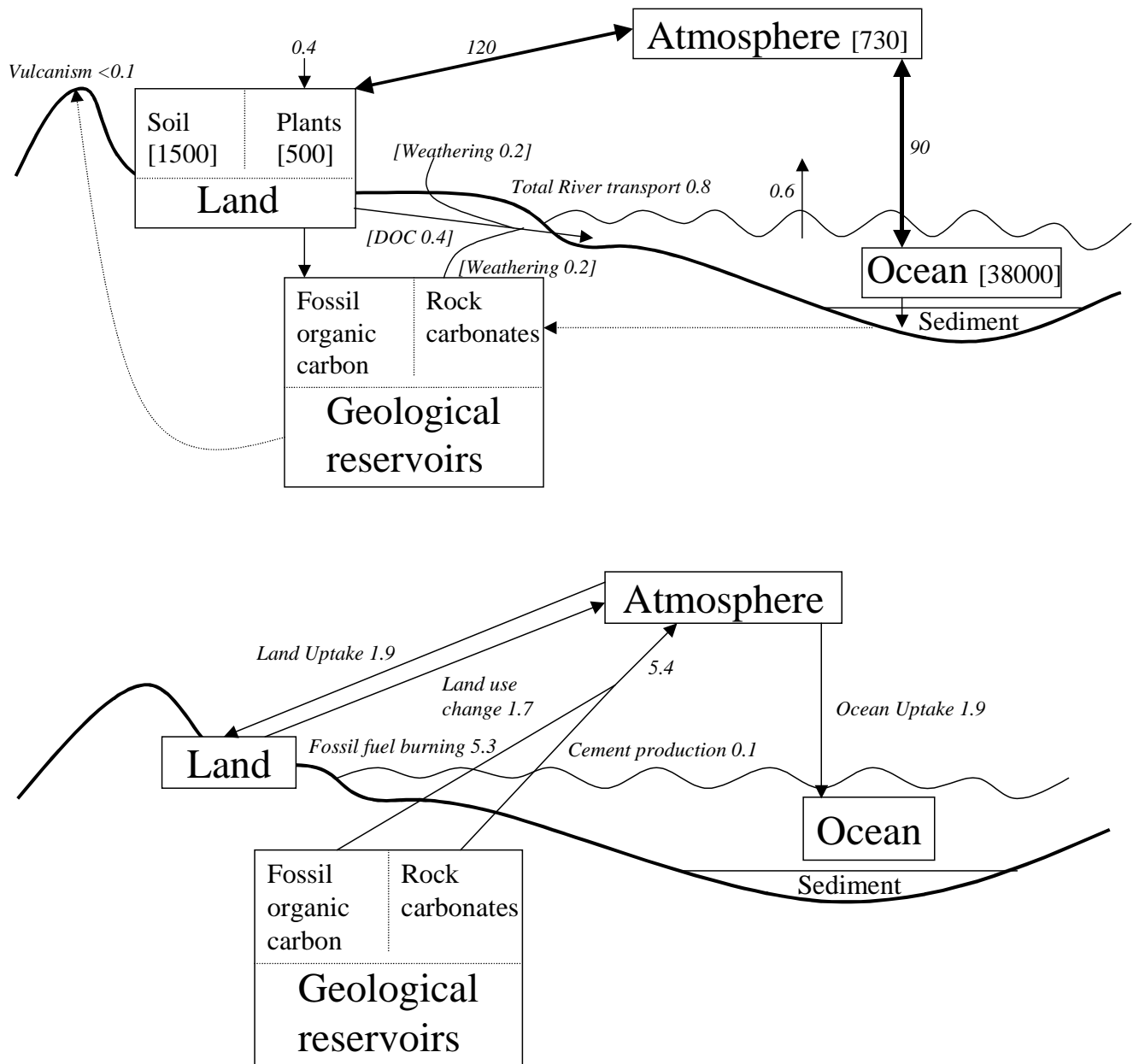


Figure 3 Schematic diagram of carbon cycle, with (above) main pools and flows of the natural global C cycle, and (below) human perturbation to the flows of C (in Pg) between the pools.

1.4 Climate change, land use and soil carbon

The increase in concentration of greenhouse gases in atmosphere and its effects on global warming is currently one of the most debated issues. According to the Kyoto Protocol, the reduction of atmospheric carbon dioxide (CO₂) can be done by

- decreasing emissions or
- by removing CO₂ from the atmosphere.

Thus, the possibility of using terrestrial ecosystems as carbon sinks has been established as one of the strategies to reduce the concentration of greenhouse gases in the atmosphere. It is worth stressing that given the size of this pool the conservation of carbon in soils to prevent emissions of CO₂ is highly relevant to both the climate change debate and to soil protection. Small changes in such significant SOC pool could have dramatic impacts on the concentration of CO₂ in the atmosphere. The response of SOC to global warming is, therefore, of critical importance.

Depending on the local conditions, soil is at the same time a source and a sink of greenhouse gases. This balance between sink and source function is very delicate. Soil not only contains worldwide twice as much carbon as the atmosphere, the flux of CO₂ between soil and the atmosphere is also large and estimated at ten times the flux of carbon dioxide from fossil fuels. Water logged and permafrost soil types hold major stocks of carbon but also are important emitters of methane (CH₄) and nitrous oxide (N₂O).

The store and flux of soil carbon are climate-dependent (Figure 4). For this reason, soil may cause an important feedback to climate change. If carbon is released from soil to the atmosphere or if methane and nitrous oxide emissions increase, climate change will be exacerbated. On the other hand, if more carbon is accumulated in soil and the emissions decrease, climate change will be retarded.

Agriculture and farming activities do approximately contribute 25% of the global greenhouse gas (GHG) emissions, in Europe approximately 10%, excluding emissions due to land use and land use change. However, carbon and nitrogen cycles have been severely altered due to agriculture leading to imbalances in the soil/air/water ecosystem (Batjes, 1996; Janzen, 2004). Thus, organic C content of the soil has been reduced in many areas, while an increase in atmospheric CO₂ has been detected. Soil organic C depletion and emissions of non-CO₂ greenhouse gas emissions related to farming activities can be offset by appropriate soil management practices and organic amendments such as manures. However, these practices influence C and N cycles, nutrients and microbial populations in soil, which are crucial factors in the emission of CO₂, CH₄ and N₂O (Huang et al., 2004; Paillat et al., 2005). Therefore, it is crucial to assess such activities as to its effectiveness and adverse or harmful environmental effects.

Both forestry and agricultural soils may be considered as carbon sinks according to the Kyoto Protocol. However, there are still many uncertainties and unanswered questions related to this issue of carbon sequestration, such as the size of sink, its sustainability and its accounting. This topic is highly relevant across Europe for the soils more susceptible to desertification (i.e. Spain, Italy) and peat soils with large stocks of organic matter in the north (i.e. Finland, Scotland) or the intensively managed soil in major agricultural areas (France, Germany, the Netherlands, the United Kingdom) are places where organic matter is expected and anticipated to further decrease.

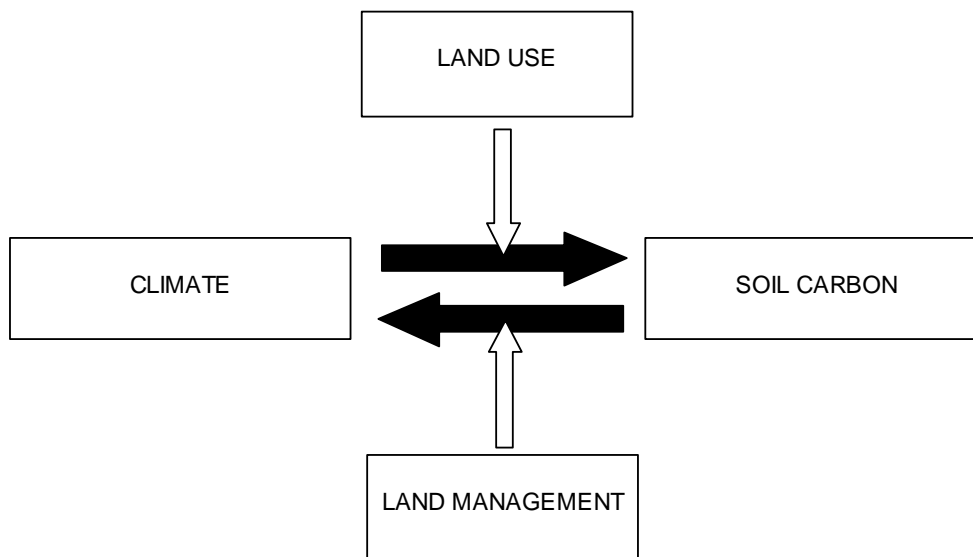


Figure 4 Climate change affects the soil carbon pool and vice versa changes in soil carbon affect the climate. For these relationships, land use and land management are major factors.

The carbon and greenhouse gas balances of soil are also affected by land use (Figure 4). A change from one land use to another induces changes in the balances. For example, afforestation of an agricultural field usually results in accumulation of soil carbon. In addition, changes in land management practices within the same land use type, such as new forestry practices or changed cultivation methods, cause changes in the carbon and greenhouse gas balances.

Land use management is thus a way to control the carbon and greenhouse gas balance of soil. Land use decisions can be used to combat the adverse effects of climate change or promote the favorable ones. In addition, land management can be an option to mitigate climate change if more carbon can be accumulated in soil or greenhouse gas emissions from soil can be decreased.

2 Effects of climate change on soil carbon

2.1 Introduction

This chapter presents the effects of climate change on soil carbon. We first give an overview of the relevant processes and some issues related to the detection of effects of climate change. In section 2.2, we shall review the effects of changes in atmospheric CO₂ levels, temperature and precipitation. This will include a discussion of the role of modelled scenarios that try to grasp the complexity of all interactions. Following the effects of climate change, we describe in section 2.3 the methods that are available to estimate changes in soil carbon. The implementation of methods to estimate soil carbon in monitoring schemes is discussed in chapter 3.

2.1.1 An overview of processes and their response to climate change

Soil carbon is a mixture of organic compounds with turnover² times ranging from days to millennia. To understand how climate change affects soil carbon and its turnover, we need to know how the underlying processes are affected. The overall change in soil carbon is determined by the balance between carbon inputs from photosynthesis and carbon losses through decomposition and hydrological processes, including erosion (EEA 2003) (Figure 5).

Soil respiration, associated with decomposition and root activity, accounts for two thirds of carbon lost from terrestrial ecosystems (Luo and Zhou 2006). In peat-dominated systems organic carbon dissolved, which may be exported hydrologically, may represent an important pathway for carbon loss (Siemens 2003). Equally, In some heavily managed and degraded systems, the loss of particulate, due to heavy erosion, may also be an important pathway for carbon loss.

The rates of all processes are affected by climate change factors. In this chapter, we review the information on these climate change impacts, and we shall find that the resulting picture is complex, and difficult to quantify. However, in qualitative terms, some process responses to climate change factors are more likely than others. Table 1 shows the most commonly expected impacts.

The table shows how some key elements of environmental change will increase or decrease the rate of processes, and thereby change total soil carbon. The focus of this chapter is on climate change but we include the effect of increased nutrient availability as plays an important role in climate change impact. The primary effect of climate change factors is on the processes that remove carbon from the atmosphere and transfer it to the soil, i.e. plant and litter production. Increased CO₂ and nutrients will stimulate production, but the effects of temperature and water availability will depend on whether the system was below or above optimum to start with. Decomposition of plant material in the soil partly depends on substrate availability, i.e. how much organic material is produced. Therefore, in Table 1, the entries in the columns for production and decomposition are sometimes similar. However, they are not identical because

² Turnover time is the average time it takes to replenish a carbon pool.

decomposition rate is also directly controlled by temperature and water. Erosion rates are expected to increase if climate extremes, in particular for precipitation, become more intense and frequent. The overall effect on soil carbon depends on the relative magnitudes of the different process responses. Note that even for the direction of the soil carbon responses, the uncertainty may be high, as indicated in the rightmost column of Table 1. We stress that Table 1 gives a simplified overview. The process responses and their uncertainties are discussed in more detail in the following sections.

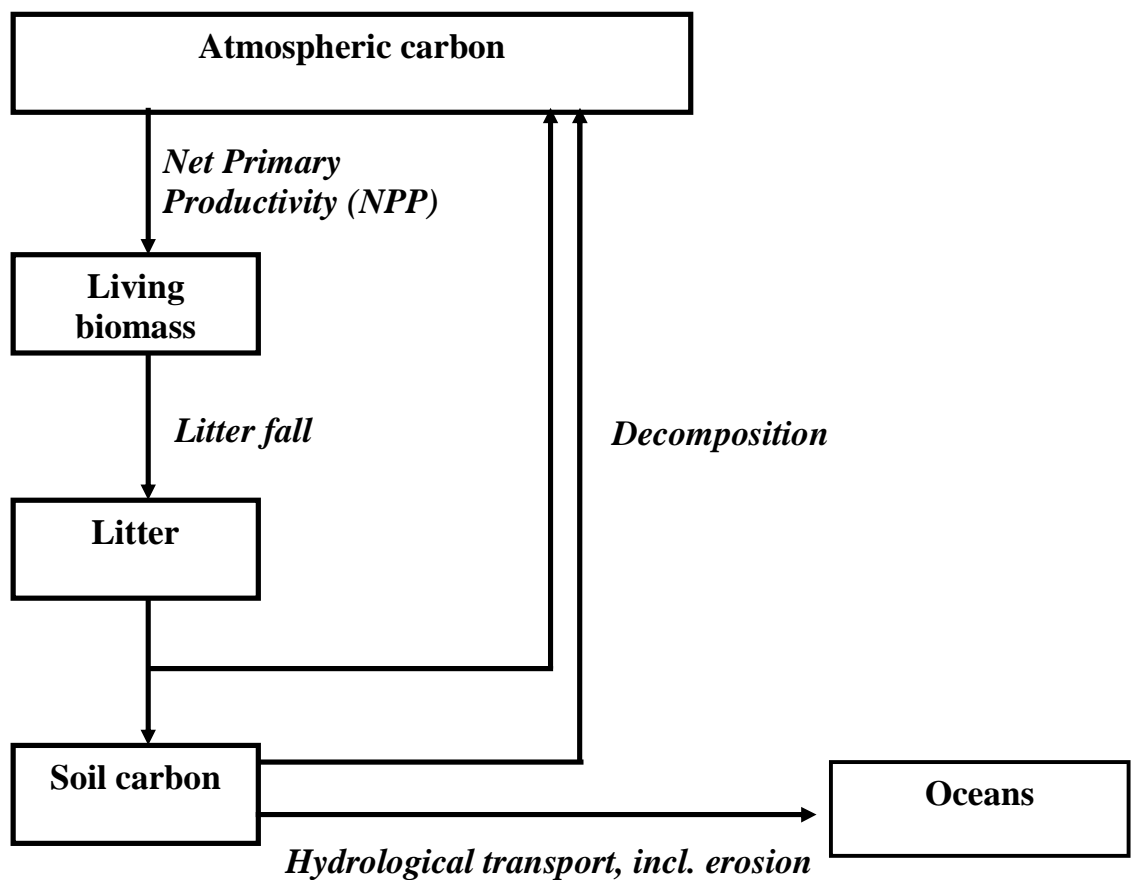


Figure 5 Processes leading to formation and loss of soil carbon.

		Process response			Soil carbon response	
		Plant and litter production	Decomposition	Erosion	Soil carbon	Uncertainty
Environmental change	Increased CO ₂	↗	↘	—	↗	Medium
	Increased temperature	↘	↗	—	↘	High
	Dry spells on mineral soils	↘	↘	—	↘	High
	Dry spells on organic soils	—	↗	↗	↘	Medium
	Heavy rain events	—	—	↗	↘	Medium
	Increased nutrient availability	↗	↘	—	↗	Low

Table 1 Expected responses of soil carbon and the underlying processes to key environmental change factors. (Note: “Uncertainty” refers to the direction of the soil carbon response: uncertainties about magnitudes of change are high throughout.)

2.1.2 Can we detect effects of climate change on soil carbon reliably and accurately?

Table 1 summarises the expected overall qualitative trends. However, once we study the expected impacts in more detail and try to measure the effects, we find that our knowledge is still quite limited. The effects of climate change on soil carbon pools are complex and poorly quantified for reasons associated with detection, attribution and the complexities of ecosystem processes involved. Detection issues arise as changes in soil C are so small on an annual/decadal timescale that they are often beyond detection limits in many monitoring and most experimental studies (e.g. Conen 2003b) and there are also problems concerning consistency of sampling layers due to drainage, erosion and bulk density (Gifford & Roderick 2003). Changes in gaseous, dissolved and particulate carbon fluxes may be more easily detected but inter-annual variability and non-continuous

records can result in problems of interpretation with these approaches (see review in Hyvönen et al. 2007).

Within **monitoring programmes**, to attribute the cause for the change observed is particularly difficult due to

- the diverse nature of the drivers involved (including temperature change, changes in rainfall pattern, elevated CO₂, and changes in extreme events including fire frequency)
- the presence and potential interaction with other drivers such as land use change and atmospheric pollution.

In **experimental approaches**, to attribute the cause for the change to individual drivers is easier. However, ecosystem processes which contribute to changes in carbon fluxes are very complex and they include plant composition, phenology and production, and changes in carbon losses through impacts on decomposition, erosion and hydrological processes. The uncertainty in our current understanding contributes to the widely different results on changes in terrestrial CO₂ sequestration in models (Cox et al. 2000, Friedlingstein et al. 2006, Heimann and Reichstein 2008) and currently prevents the further development of carbon-climate models (Yuo 2007).

The issue of limitations in quality and quantity of our data and models will be a recurring theme of this chapter, in particular sections 2.2 and 2.3.

2.2 Climate change factors and their effects on soil carbon

Scenario studies carried out with models indicate that climate change impacts are likely to affect two crucial stages of the C cycle:

- 1- Decomposition
- 2- Net primary productivity (NPP)

Hereafter, we will concentrate on estimating the different consequences of several climate change drivers into these two stages.

2.2.1 Effects of elevated atmospheric CO₂

Evidence of changes in soil carbon in elevated CO₂ experiments is limited and variable. However a meta-analysis is available which concludes that if results were combined a net increase in soil carbon of about 6% would be observed (Jastrow *et al.*, 2005) indicating an overall positive effect of elevated CO₂ on soil carbon storage.

Elevated CO₂ concentrations in the atmosphere (Van Groeningen *et al.*, 2006) cause that plant production is changed and as a consequence the vegetation and carbon inputs on the soil also change (see reviews in Hyvönen *et al.*, 2006, Fischlin *et al.*, 2007) . However, the plant production and the vegetation composition not only depend on the CO₂ concentrations in the atmosphere but are also determined by nutrient and water limitation. Therefore these factors also have strong interactions with the carbon input from plants to soil, potentially reducing the positive long term responses.

For example, increases in biomass and thus carbon inputs to the soil may not be sustainable if moisture regimes change. Experimental approaches which include multi-factor interaction treatments are therefore a priority for future work (Beier et al. 2004) as current data from the only multi-factorial experiment suggest no overall biomass increases in a grassland system (Shaw et al. 2002). Evidence from single factor studies suggest that impacts on soil carbon losses of plant decomposition are relatively small as soil properties determine the turnover rates of soil carbon and the majority of new carbon inputs do not become long-term soil carbon (Hagedorn et al. 2003, Taneva et al. 2006) although others have suggested the additional litter will form coarse particulate organic matter which initiates aggregation formation (Six et al. 1998). Indeed, some studies suggest that the additional carbon may accelerate decomposition of stable carbon (Fontaine et al. 2007). Decomposition rates of litter *per se* appear little affected by changes in litter quality which will further enhance nutrient limitation of the plant production response to elevated CO₂ (e.g. Norby et al. 2001, Knops et al. 2007).

2.2.2 Effects of temperature

Elevated temperatures have been shown in experimental studies to generally increase the rate of soil respiration and thereby the loss of soil carbon due to increased decomposition rates. This increase ranges from 15 to 45% in different studies across a range of habitats. The loss is thought to be greatest in northern latitudes where current decomposition processes are limited by temperature although experimental studies to date have not always supported this hypothesis.

The effects of elevated temperature have been studied using a variety of approaches and they all provide some evidence for the important controlling effect of temperature on soil carbon fluxes and storage. Increased temperatures may

- (i) cause a stimulation of **soil CO₂ respiration**, which is the dominant pathway for carbon loss from terrestrial ecosystems in response to warming, as reported by many experimental studies via a range of techniques (e.g. Rustad *et al.*, 2001). This same results is not found in all studies due to the possible interference of other environmental constraints (e.g. Emmett *et al.*, 2004). Attributing this increase to increased decomposition and thus true soil carbon loss rather than increased respiration from plant root or additional plant litter is problematic although new methods are demonstrating that both processes are involved. An additional problem, is the observed 'acclimation' of the stimulation of soil respiration in warming experiments whereby the magnitude of the response declines over time, most likely due to limitation of readily available substrate supply (Kirschbaum 1995, 2006). However, plant production may increase due to elevated CO₂ emphasising the need to understand the change in inputs from plants and the relative sensitivity of different soil carbon pools to the different climate drivers and the need for multiple factor experiments. Changes in microbial composition over time to less sensitive communities which are possibly more tolerant to extreme conditions may result in true physiological acclimation. This A shift in microbial communities to less sensitive communities which are possibly more tolerant to extreme conditions has been reported in several studies (e.g.

- Zhang *et al.* 2005). If this occurs it will help to reduce the rate of soil carbon loss due to elevated temperatures.
- (ii) increase **dissolved organic carbon (DOC)** widely in surface waters in North America and northern and central Europe as reported (e.g. Freeman *et al.*, 2001a, Worrall *et al.*, 2003) although other climatic drivers such as elevated CO₂ (Freeman *et al.*, 2004) have been invoked. DOC is an important pathway for carbon loss in peatland systems which are the largest stores of carbon. Most recently, the attribution to climate drivers has been disputed with recovery from acidification and changes in organic matter solubility proposed as the most likely explanation (Monteith *et al.*, 2007).
 - (iii) contribute to changing **topsoil carbon contents**, as reported in England and Wales by Bellamy *et al.* (2005). No attribution work was carried out as such but rather they proposed that the consistency of the patterns observed across the region pointed to a large-scale driver such as climate change. The potential for the magnitude of the loss reported being attributable to climate has more recently been challenged (P. Smith *et al.*, 2007a). Changing land-use and management may have been more important (Kirk and Bellamy, 2008).
 - (iv) cause **a net carbon loss** in combination with extreme drought, as reported in information derived from eddy-covariance studies across Europe in 2003 (Ciais *et al.*, 2005, Reichstein *et al.*, 2006). This illustrates a major concern that it is extreme events and eventually the exceedance of particular thresholds for key processes which may result in a significant shift in soils acting as carbon sinks to sources.

It is expected that the effects of increasing temperatures on decomposition have a higher and more sustained impact on soil C than the effects of temperature on plant production. This is due to the fact that soil respiration is more vulnerable to changes in temperature than photosynthesis and plant respiration, as demonstrated in a review of available laboratory and field studies (excluding moisture limited systems including peats) (Lenton and Huntingford, 2003). This is particularly the case in northern latitudes due to the stronger response of soil respiration at lower temperature (Kirschbaum, 1995). The consequence of this will be a loss of soil carbon and a positive feedback to the climate system in the long term.

However, a wide variety of sensitivities are observed in warming experiments in the field (e.g. Rustad *et al.* 2001, Emmett *et al.* 2004, Davidson and Janssens 2006). Temperature sensitivity of different components of the soil carbon pool has been much studied as even a small increase in the release of stored organic carbon could cause a major positive feedback to the climate cycle. Most models assume a temperature sensitivity coefficient Q_{10} – the quotient of change in respiration caused by a change in temperature of 10°C – of 2 (see review in Yuo 2007) although a wide range of values have actually been reported (Lenton and Huntingford 2003). Much recent work has concentrated on identifying temperature sensitivity of different soil carbon fractions including labile versus stabilised, bulk versus rhizosphere (e.g. Hartley *et al.* 2007) and organic versus deep soil layers (Davidson *et al.* 2006). This proposed variability in sensitivity is reflected in some of the models. However, empirical evidence can be contradictory for reasons poorly understood although this may include other

environmental constraints which can obscure temperature sensitivity of substrate decomposition (Davidson and Janssens 2006). For example, Fierer et al. (2006) found evidence for a greater temperature sensitivity of more labile carbon. Conen et al. (2006a) found no evidence of variable temperature sensitivity of young and old carbon whereas Hakkenberg et al. (2008) reported evidence for greater sensitivity of older carbon pools, and some studies found no sensitivity at all (Giardina and Ryan 2000).

The major area of uncertainty appears to be the controls on stabilised soil organic matter (SOM) as controls on initial stages of decomposition of fresh litter appear better established. Some studies have suggested these pools are not sensitive to temperature (Giardina and Ryan 2000), but Davidson and Janssens (2006) point out that the absence of any sensitivity to temperature is contrary to kinetic theory. Increasingly studies are including other drivers known to influence soil carbon turnover and potentially contributing to the sensitivity to temperature. One study showed that different soil microbial communities are active in different seasons, from which a response of soil biota to warming may be inferred (Monson et al. 2006).

The effects of temperature are complex but generally represented by simple response functions and kinetic sensitivity of photosynthesis and respiration in models (Yuo 2007, Heimann and Riechstein 2008). Most models do not incorporate the stimulatory effect of freeze-thaw cycles which are known to cause pulses in soil respiration rates (Goulden et al., 1998). Effects of temperature on plant production are highly variable depending on species specific factors, vegetation composition, competitive balance and other environmental limitations although most frequently positive responses are reported (e.g. Rustad et al. 2001, Penuelas et al. 2007 and see review by Yuo 2007). This complexity also includes phenological changes which are currently missed in most models. Additional complexity is caused by changes in nutrient supply with increased net nitrogen mineralization, as reported in many studies (e.g. Rustad et al. 2001, Johnson et al 2000) which can enhance plant growth thus removing one of the primary limitations on plant growth response to temperature (and elevated CO₂) although the sustainability of this is likely to be limited (Luo 2004). These changes in plant derived carbon inputs are known to have a major influence on soil carbon decomposition although the effects are poorly quantified.

2.2.3 Effects of changes in precipitation

Unfortunately, describing the response of decomposition to soil moisture changes in models is limited as the direct response of decomposition to changes in water content is less well characterized than for temperature.

The effects of drought are known to be heavily dependent on current hydrological conditions. In water-limited systems, the major effects of increased frequency or severity of drought is likely to be indirect through changes in plant community composition. In wetter systems, there is potential for significant increased carbon loss by soil respiration, with values of +40% reported.

A more extreme hydrological cycle is predicted for many areas which will result in more extreme and frequent periods of soil moisture deficit. This will decrease the rate of decomposition in many systems but will increase rates in waterlogged systems such as

peatlands where much carbon is stored. Various case studies concerning the effects of changes in precipitation are available e.g.

- (i) In a range of European shrublands, the long term effects of repeated summer drought in climate change experiments were observed to either stimulate by 40% or depress soil respiration rates by 30% depending on initial hydrological conditions (Sowerby et al, in press).
- (ii) In a temperate forest, drought significantly decreased soil respiration and the authors suggested there was therefore the potential for an increase in soil carbon storage (Borken et al. 2006).
- (iii) In the Amazon, no change in soil respiration but a large decrease in plant production was observed in a drought experiment indicating a likely net loss of carbon from the ecosystem due to reduced carbon fixation (Brando et al. 2008) which in the long term would lead to reduced soil carbon.
- (iv) Drought was suggested by Schulze and Freibauer (2005) to be the most likely factor to have contributed to the 15% loss of soil carbon stock over the last 20 years reported by the UK national soil monitoring programme (Bellamy *et al.* 2005). However, high temperatures often occur in concert making it difficult to separate their single and interactive effects in monitoring studies such as this.
- (v) Drought in interaction with warming may exacerbate loss of carbon by erosion, with Mediterranean countries having relatively high risks of desertification (EEA 2003).
- (vi) In mountainous areas of central Europe, expected changes in rain event frequency and intensity may increase soil erosion (Sauerborn *et al.*, 1999). Flood events will partly remove eroded carbon from soils but partly lead to redistribution of the carbon across the landscape (e.g. Quinton *et al.*, 2006).

Soil carbon loss via soil respiration is considered to be less sensitive to soil moisture limitation than plant production (Ågren *et al.*, 1996). This results in soil carbon losses exceeding carbon fixation during a period of drought. Of particular concern is if stabilised SOM is somehow made available to microbes due to removal or reduction of environmental constraints such as waterlogging or freezing due to changes in rainfall and temperature which may lead to continued loss beyond that currently predicted from present day measurements. This may happen due to physical cracking of the soil for example. This has been observed in one study where removal of the environmental constraint of waterlogging by repeated experimental droughts caused a large (20-40%) and increasing stimulation of soil respiration which lasted for the duration of the whole non-drought period (Sowerby et al., in press). Increased hydrophobicity due to changes in microbial communities and in some systems fire frequency could also contribute to prolonging the effects of soil drying far beyond the drought period reducing or enhancing soil carbon loss depending on initial conditions. However, as for temperature, shifts in species phenology and composition towards more drought tolerant species such as shrubs will also have major effects on soil carbon through changes in rooting pattern, litter quality and associated soil microbial community changes and even soil physical conditions. For example, earlier onset of senescence with warming was found to increase spring soil moisture (Zavaleta et al., 2003). Repeated drought in successive years and/or increased frequency of severe drought caused by reductions in rainfall below an historic

minimum or threshold, may be critical in causing a change in plant community composition and structure (e.g. Leuzinger *et al.*, 2005) which could fundamentally change the production potential and carbon balance of an ecosystem.

The few available studies are difficult to compare because of differences in measurement methods (Ilstedt *et al.*, 2000). Direct effects are due to an increase in microbial activity with soil water content from a minimum water content (where desiccation stress is observed) to a maximum threshold, usually field capacity, above which a decline in decomposition processes is observed. Indirect effects on diffusivity of substrates are thought to be the primary driver of the sensitivity to soil water content (Grant and Rochette, 1994). However beyond this simple model, variability in sensitivity to water content is observed due to other indirect environmental constraints leading to variable thresholds for different soil types (Ilstedt *et al.* 2000). The combined effect of warming and reduced soil water content is also problematic as whilst diffusion of gases and solutes increases with increasing temperature, drier soils will decrease rates of diffusion of carbon substrates, extracellular enzymes and mobility of microbes. This can result in lower temperature sensitivities during dry periods, which often co-occurs with changes in vegetation-derived substrate supply.

2.2.4 Interactions with nitrogen and phosphorus

Nutrient availability may provide one of the most critical controls on the net balance between plant and soil processes. In systems with low soil nitrogen availability, additional nitrogen from deposition is thought to be one of the major causes of increases in tree growth and soil carbon storage in forests across Europe. Data and evidence for other systems are sparse.

Nutrients such as nitrogen and phosphorus are critical in controlling ecosystem carbon balance as most natural systems are nutrient limited. Thus, the limited availability of nutrients can hamper or constraint the increase of vegetation that higher temperatures or CO₂ atmospheric concentrations would bring about.

Availability of nitrogen in many European ecosystems is now significantly enhanced due to atmospheric nitrogen deposition. This has major consequences for the ecosystem carbon balance. Nitrogen deposition has been estimated to account for approximately 10% of all carbon captured in trees and soil in European forest systems due to the positive effect on tree growth (De Vries *et al.* 2006). Evidence from long-term observations and modeling in Sweden, show that the 10 kg N ha⁻¹ year⁻¹ higher deposition in southern Sweden than in northern Sweden for a whole century could have resulted in 2.0 kg m⁻² more tree C and 1.3 kg m⁻² more SOC in forests in the south Hyvönen *et al.* (2008). These estimates are consistent with differences between south and north in tree C and SOC found by other studies, and 70–80% of the difference in SOC can be explained by different N deposition.

Unfortunately, the effect of N deposition, and nutrient availability in general, on soil organic matter turnover remains largely overlooked by existing models. Assumptions that increased N availability will reduce organic matter decomposition rates (e.g. Fog, 1988; Carreiro *et al.*, 2000; Neff *et al.*, 2002; Hagedorn *et al.*, 2003; Waldrop *et al.*, 2004; Knorr *et al.*, 2005) coexist with the contrary hypothesis (Kirschbaum, 1995). Implications of this are that greatest effects of nitrogen on carbon storage may be

expected in carbon-rich, nutrient poor systems due to both an increase in production and a decrease in the decay rate of an enlarged recalcitrant organic matter pool. The effects are likely to be considerably smaller in agriculturally managed systems, where nitrogen inputs are higher and where regular soil tillage stimulates soil organic matter turnover.

Indirect effects of climate drivers on nutrient availability have also been shown to be important in some ecosystems:

- (i) in **experimental warming**, changes in nitrogen availability have been shown to increase the risk of soil carbon loss. For example, in tundra systems warming results in an increase in the abundance of shrubs (Sturm et al. 2005). This change in vegetation structure causes higher winter soil temperatures and the resulting increase in microbial activity and plant-available nitrogen further promotes shrub abundance and a positive feedback loop.
- (ii) **Drought** has been shown to reduce uptake of phosphorus and other nutrients by trees in a Mediterranean system thus increasing P-limitation of growth (Sardans & Penuelas 2007, Sardans et al., 2008).
- (iii) Following **extreme weather events**, dynamics of nutrient release from litter may also be responsible for oscillations in annual net primary production observed (Haddad et al. 2002).
- (iv) Patterns across **rainfall** gradients indicate that concentrations of extractable/exchangeable nutrients generally decrease with precipitation with a widening of C:nutrient ratios (e.g. Austin and Vitousek, 1998). This suggests an asynchrony of carbon and nutrient dynamics driven by the different sensitivity of photosynthesis and decomposition to temperature and water availability but also the effect of rainfall and temperature on other abiotic and biotic processes specific to individual elements.

2.2.5 Integrated analysis of the combined effects by modelling

Scenario studies carried out with models indicate that climate change is likely to accelerate decomposition and thereby decrease soil carbon stocks, but that effect is counteracted and in certain cases fully compensated by increasing net primary productivity (NPP), changes in land use, soil management technologies in agriculture or changes in age class structure of forests. The regional variation is large. These results underline the complexity of the phenomena.

Therefore, gathering further knowledge and detailed information on a large number of processes and drivers is crucial to improve model projections for the effects of climate change on soil carbon .

The experimental studies analysed in the previous sections demonstrate the complexity of the carbon chain and the manifold interactions between environmental drivers. Only a fraction of this complexity has been represented in models. The general approach in models is to simplify nature by distinguishing only a small number of soil carbon pools, with different levels of stability and therefore different carbon turnover rates. The turnover rates are generally considered to be controlled by substrate supply, temperature and water (see review of soil models by Smith et al. 1997a). However, the degree of control exerted by these factors is assumed to differ between the pools. Although models are simplifications of reality, they are essential in that they are able to

consider combinations of environmental factors that are difficult to establish in experiments, i.e. they can simulate any scenario of climate change. The consequence of this is that models are often the only tool to study climate-change related issues – examples of which are given in this section – but we must evaluate their outcomes with care.

Recent model simulation studies in Europe have shown that changes in land use, soil management technologies in agriculture or changes in age-class structure of forests can be more important than climate change itself. Raising temperatures increase the decomposition in all the soil carbon models leading to the conclusion that soil carbon amounts decrease. However, the effect of rising temperatures can be counterbalanced by other accompanying climatic phenomena, such as droughts. Indeed in certain regions, like in the Mediterranean, drought limits the decomposition. Whether the net change in soil carbon stock is positive or negative depends on the balance between the increased decomposition and possibly increased litter input to soils.

Smith *et al.* (2005) found that increasing litter input and technology improvements balanced the effects of climate in croplands and grasslands. There were regional differences in trends. Smith *et al.* (2007a) carried out a similar scenario study for agricultural mineral soil carbon in European Russia and the Ukraine. The authors conclude that there are large potential losses of carbon from mineral soils under future conditions and that agricultural management can play a major role in adaptation to climate change and mitigation of these losses.

Smith (2006) projected the potential changes in the soil carbon of forests on mineral soil in Europe. According to this simulation study, also in forests, climate change will tend to speed up decomposition, whereas increases in litter input due to increasing NPP and changing age-class structure will slow the loss of SOC. Increases in forest area could further enhance the total soil carbon stock of European forests.

A similar simulation study of the terrestrial carbon stocks in Europe (Zaehle *et al.*, 2007) supported the results above and showed that soil carbon losses resulting from climate warming reduce or even offset carbon sequestration resulting from growth enhancement induced by climate change.

Jones *et al.* (2005) concluded that the projection of a positive feedback between climate and carbon cycle is robust, but the magnitude of the feedback is dependent on the structure of the soil carbon model.

2.2.6 Assessment: Uncertainties and knowledge gaps

The assessment of the effects of climate change on soil carbon requires understanding and quantifiable information on various processes and their drivers involved in the terrestrial carbon cycle. Currently, there are clear gaps in our knowledge. Insufficient understanding of the underlying processes propagates to modelling studies but the magnitude of this uncertainty is difficult to assess. Concentrated efforts should be made to acquire measured information on the critical processes of the carbon cycle in soils. Such efforts could be part of revised soil monitoring schemes – which will be discussed in the last section of this chapter.

Detection, attribution and prediction of the effects of climate change on soil carbon are all subject to numerous uncertainties and knowledge gaps. This applies to all

studies, irrespective of the chosen method: monitoring, experimentation or modeling. With respect to monitoring, of primary importance is the requirement to refine methodologies for measuring both soil carbon stocks and fluxes (gaseous, dissolved and particulate). There is a need to ensure continuation of long term records which hold evidence of inter-annual variability and feedbacks due to changes in vegetation composition and interactions with other environmental drivers. Attribution of observed changes to underlying causes remains a major challenge due to the complexity of drivers and their interactions. Increased understanding of sensitivity of different processes to driving variables and critical thresholds will help us attribute change in the future. A major gap currently is our lack of understanding and quantification of the impacts of freeze-thaw and drought-rewet events on soil carbon. The repeated freeze-thaw events during cold season, freezing of soils in autumn and thawing in spring are typical for the tundra, boreal, and temperate soils. The thawing of soils during winter-summer transitions induces the release of decomposable organic carbon and acceleration of soil respiration. A similar increase in respiration is observed when dry soils are re-wetted.

We need new experiments which involve multiple factors and their interaction to test outputs from ecosystem models in a wide range of ecosystem types to help understand the variability in responses we currently observe. These experiments should involve measurement and manipulation of nutrients other than carbon and nitrogen as many systems are limited by other nutrients. Partitioning of fluxes observed between the autotrophic and heterotrophic community remains a challenge but new isotope techniques are providing at least one new approach to addressing this problem.

There are many sources of uncertainty in model simulations of the climate change impacts on soil carbon and they are handled in different ways. The uncertainty related to future predictions is typically handled by using variable future scenarios spanning a plausible range. Currently most commonly used scenarios are the IPCC-SRES storylines (A1FI, A2, B1, B2). When using different scenarios the aim is not to quantify the uncertainties exactly but to cover roughly the potential future developments. The main challenge for the models used to predict changes due to the changing climate is to describe the most relevant processes and their climate dependencies accurately enough. That can only be tested with the already existing data. Limits to model applicability (soil types, geographical regions, temperature ranges) add further uncertainty to model results, but this is unavoidable as the models are the only tools to extrapolate research findings to new, as yet unobserved, conditions.

2.3 General methodologies to estimate changes in soil carbon

Methods to estimate changes in soil carbon pools can be divided into four categories:

- Statistical analyses of spatially distributed soil samples (repeated measurements or chronosequences),
- Measurements of carbon dioxide fluxes,
- Process-based modeling studies,
- Combinations of monitoring and modeling.

These methods are described in detail in Annex 1.

Among these categories, the best methods to estimate changes in soil carbon pools over larger geographical areas are the statistical analyses of repeated soil carbon pool measurements based on spatially distributed soil samples (soil carbon monitoring), modeling or combinations of these two methods. Measurements of carbon dioxide fluxes, using soil respiration chambers or eddy covariance methods, are less suitable for this purpose because of difficulties in separating plant respiration from decomposition of dead soil organic matter, need of complementary estimates on carbon input fluxes to soil and an insufficient geographical coverage of such measurements.

The main difficulties with soil carbon monitoring are the large amount of work needed and consequently high costs plus the challenge to keep the study methods adequately similar between the monitoring rounds. The amount of work and the costs can be reduced by combining modeling with monitoring. Simulating soil carbon changes to be expected using models helps in designing more effective sampling schemes.

Modeling is a less expensive and an easily applicable method to estimate changes in soil carbon pools but as pointed out in the previous section, the main concern is the reliability of the results. Nevertheless, modeling provides a useful complement to soil sampling, and it is advisable to apply these methods together when monitoring soil carbon pools.

3 Monitoring systems used to estimate changes in soil carbon

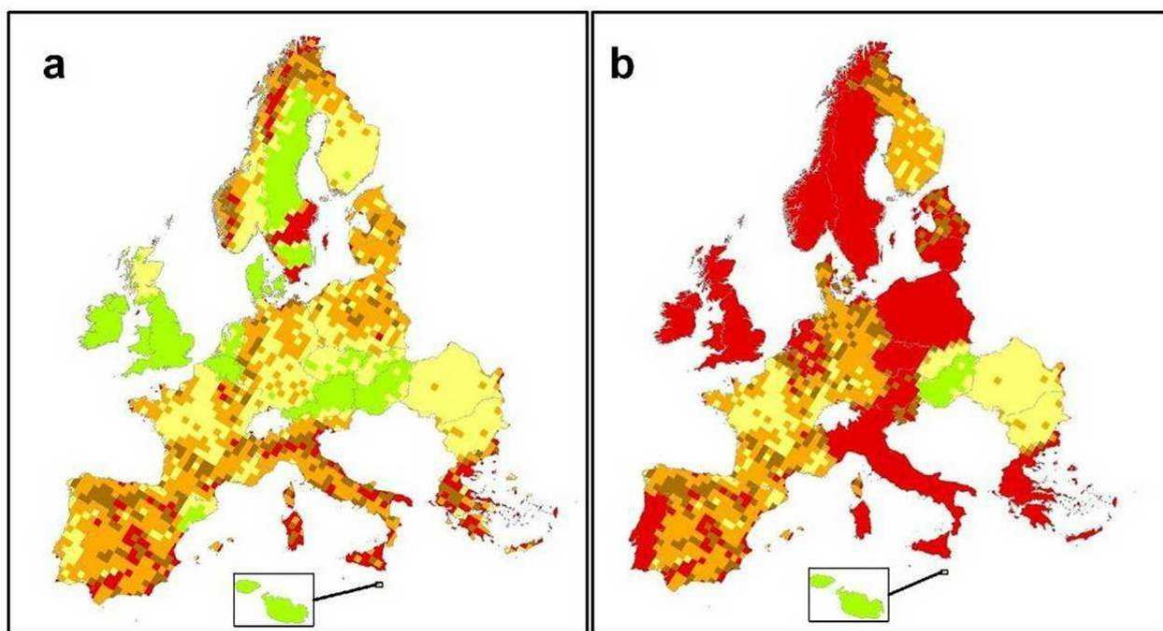
Most EU countries have established soil inventory programmes but basically lack soil monitoring systems. Most of surveying initiatives across Member States of EU cannot be considered as monitoring programmes, since only in very few cases more than one observation in time has been performed. The inventories may serve soil monitoring purposes in the future if the inventories will be repeated in a sufficiently similar way. This maybe hampered due to the fact that soil inventories are usually carried out by a number of different organisations and for many different reasons. For example, monitoring aimed at providing information regarding compliance with the Kyoto protocol (based on IPCC Good Practice Guidelines) could have the limited aim of demonstrating that soil is not a net source of greenhouse gases, which may require less intensive measurements than a survey for other purposes – and may be carried out by a different organisation.

Some inventories are Europe-wide but most are national and some are regional. Often the regional schemes are carried out even without reference or linkage to the other monitoring programmes in the same country. Only European forest soils can be considered to be monitored in a harmonised way (ICP Forest, Forest Focus).

3.1 Description of available monitoring schemes

JRC-IES has compiled a soil organic matter map (see Figure 7) for Europe (Jones *et al.*, 2004) with calculated total soil carbon content (SOC). The soil map is compiled from national soil surveys by means of pedotransfer functions and gives only information on the state of the SOC in EU-25 but not on the change in SOM in time. Information on changes in carbon stocks in the EU-25 are not available as bulkdensity in general has not been a standard measurement. On national level, bulkdensities are sometimes available in databases with old information but more often these are calculated by means of pedotransfer functions. Whether the information in the European soil map is up-to-date remains uncertain as very few national monitoring programs of soil organic matter exist.

A recent assessment of European, national and regional soil monitoring networks, has been undertaken by the EU project ENVASSO (Arrouays and Morvan, 2008, <http://www.envasso.com/>). ENVASSO has concluded that topsoil organic carbon concentration is one of the most widely available indicators in Europe. Measurements of topsoil organic carbon content are available in all countries (Figure 6).



Density of monitoring sites in a 50km cell

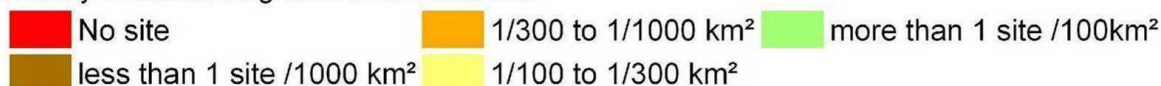


Figure 6 Maps of density of sites at which on the left in a) topsoil organic carbon content is measured and on the right b) topsoil organic carbon stocks can be calculated without necessity of further assumptions for bulk density and/or for calculation of organic C from organic matter. (Source: ENVASSO report, Arrouays & Morvan, 2008).

To generate Figure 6, details of over 80 soil monitoring networks were collated by ENVASSO from all European countries. Table 2 gives a summary of these monitoring networks showing the total number of monitoring sites within each country and of these the sites at which organic carbon or sometimes only organic matter content has been measured. It is important to note that carbon content estimates must be complemented by bulk density estimates to estimate an amount of soil carbon for a specific site. It is for this reason that a national soil carbon estimate may be missing from Table 2 even though this table shows that a country has a national inventory for soil carbon content. Also, the number of sites in Table 2 may deviate from country examples provided in the text on the basis of country reports because the criteria of selecting sites may have been different in the country studies and the ENVASSO project.

Table 2 Total number (N) of actual monitoring sites, number (n) of sites where carbon content (%) is measured, theoretical number (n1) of sites needed to detect a relative decrease of 5% of the national mean of topsoil organic carbon contents according to national statistics on variances, number (n2) of additional sites needed in comparison with n1, number (n3) of additional sites needed in comparison with N (taken from ENVASSO, see Arrouays and Morvan, 2008).

Country	N : actual monitoring sites	n : sites where carbon content is measured	n1 : theoretical number of sites needed to detect a relative decrease of 5%	n2 : additional sites needed in comparison with n	n3 : additional sites needed in comparison with N
Austria	3829	3313	1073	0	0
Belgium	2546	2546	2105	0	0
Bulgaria	436	432	866	434	430
Czech Republic	738	738	1933	1195	1195
Denmark	848	848	1323	475	475
England & Wales	6018	6018	3853	0	0
Estonia	1588	128	2314	2186	726
Finland	1563	1446	2153	707	590
France	1532	1532	2182	650	650
Germany	1380	1254	2079	825	699
Greece	150	146	1230	1084	1080
Hungary	1328	1328	1680	352	352
Ireland	1317	1317	3121	1804	1804
Italy	341	341	1331	990	990
Latvia	127	127	2513	2386	2386
Lithuania	146	146	2849	2703	2703
Luxembourg	6	6	850	844	844
Malta	388	271	34	0	0
Netherlands	531	531	2086	1555	1555
Northern Ireland	582	582	3116	2534	2534
Norway	1057	1057	6988	5931	5931
Poland	895	894	1580	686	685
Portugal	291	290	1540	1250	1249
Romania	952	948	1286	338	334
Scotland	721	721	1255	534	534
Slovakia	432	424	1374	950	942
Slovenia	468	56	850	794	382
Spain	1009	1009	2304	1295	1295
Sweden	4885	4885	1764	0	0
Total	36104	33334	57628	32498	30361

Despite its ubiquitous measurement, a consensus definition of soil organic matter (SOM) is not apparent from literature (Carter, 2001). The main disparities between these definitions (Kibblewith *et al.*, 2008) are:

- i) inclusion/exclusion of living biomass
- ii) inclusion/exclusion of the litter, fragmentation and humification layers
- iii) 'threshold degree' of decomposition.

Also SOM or SOC have been determined by different methods in the studies available. Further, topsoil depth and sampling depth are not well defined. All this makes an assessment of C-stocks across EU Member States problematic.

Very few countries have systematically taken measurements on more than one occasion – particularly at a national scale (see also Annex 2). The only region with 'true' resampling data is England and Wales where 40% of the original sites on a 5x5 km grid were resampled with an interval of 15-25 years (Bellamy *et al.*, 2005). Belgium (Letten *et al.*, 2004) has carried out sampling campaigns over time. Even though these soil profiles have been georeferenced, these sampled sites cannot be considered 'monitoring sites' as defined by ENVASSO as they are not located precisely enough to enable the site to be revisited. Hanegraaf *et al.* (2009) has calculated changes in organic matter on the basis of farm based measurements for the period 1984 – 2004 in the Netherlands. Again, here no real resampling was done and also no measures for bulk density were available and sampling depth was only 0-5 or 0-10 and 0-20 cm for grassland and cropland, respectively.

As a noticeable number of countries do not determine soil bulk density, topsoil carbon stocks cannot be accurately monitored in the United Kingdom, Italy, Portugal, Greece, Poland, Sweden, Norway, Czech Republic, Lithuania, The Netherlands, and in parts of Austria. However, some pedotransfer functions can be used to get estimates of carbon stock changes (see for example Bellamy *et al.*, 2005) from soil carbon concentrations from soil monitoring networks. Nevertheless, as bulk density and organic carbon are correlated, and as changes in bulk density may induce changes in the mineral mass of soil collected down to a given depth, it would be worthwhile to determine bulk density on all sites. Nearly all Member States measure both carbon and nitrogen except for England and Wales, Greece and Malta. Over time, depth of plough layers has changed. This change is hardly recognized in analysis of trends of the stock of organic C in soils.

Of importance in the European context is the Land Use Land Cover Annual Survey (LUCAS, <http://circa.europa.eu/irc/dsis/landstat/info/data/introduction.htm>). LUCAS is an area frame statistical survey that aims at obtaining harmonised data at the EU level on land use, land cover and additional environmental features. The survey consists of ground visits in springtime for sampling about 100 000 points according to a 18x18 km grid. The survey has been carried out in 2001 in the EU13, in 2002 in the UK, Ireland, Estonia, Hungary and Slovenia and in 2003, in the EU15 and Hungary. Such data on land use and land cover are useful and needed in calculations on carbon stock changes in soil.

The EU BioSoil project (<http://biosoil.jrc.it/>), running from 2006 to 2009, monitors the soil carbon content in forest soils across Europe, as a contribution to define the key role of forests in carbon sequestration. The project addresses the suitability of

using existing Level I network for monitoring soils in European forests. The level I network for forests was established in the early 1980s to monitor forest decline across Europe and is a subset of more extensive national forest surveys and includes 300 plots covering the five major tree species (Oak, Beech, Scots pine, Sitka spruce and Norway spruce).

3.2 Evaluation of available monitoring schemes

In several European countries monitoring schemes for soils have been implemented. Annex 3 gives a few examples; this is not a comprehensive list. The purpose is to show that major differences in monitoring activities exist both between countries and within countries for i.e. different land cover and land use. Many of these monitoring schemes have not been designed and implemented for monitoring organic matter or organic carbon in soils. Furthermore, many of the data from this national monitoring across Europe have not been assessed and used for calculating or estimating soil organic carbon stocks. However, these national datasets offer significant potential to create a starting point for calculation of organic carbon stocks in Europe for given land use and land cover. As indicated below, assumptions on i.e. bulk density or transformation of organic matter into organic carbon have to be made. Linking these data with any specific future monitoring scheme for soil organic carbon requires development of a strategy for frequency and distribution of observations that will match the needs of monitoring changes in stocks of soil carbon.

3.2.1 Limitations of existing and proposed monitoring schemes

Soil is a fairly stable medium, with detectable changes occurring only over long time spans; but it is spatially heterogeneous, hence variability in sampling and measurements is often many times larger than variability over time, making stringent standardisation and quality assurance/quality control (QA/QC) procedures mandatory. QA/QC involves quantifying the errors associated with the sampling. An error budget for undisturbed forests has been established in the project CarboInvent (FP 5) and the rules of accounting for errors are described in the Good Practice Guidance of the IPCC. Based on data that will become available from WG1 of COST Action 639 (2006-2010), calculation of error budgets of SOC pool changes for peatlands that include a quantification of error and uncertainty for the SOC pool will be possible.

3.2.2 Costs of soil carbon monitoring

Both JRC (Stolbovoy *et al.* 2007) and COST 639 (Jandl & Olsson 2007) made estimates of the potential cost of a soil sampling programme to detect changes in SOC. Stolbovoy *et al.* (2007) made cost comparisons for the conventional IPCC (IPCC, 2003) and the so-called AFRSS (Area-Frame Randomized Soil Sampling) sampling approaches. The IPCC procedure recommends that nine soil points be tested for each plot, each containing three sampled depths (0-10 cm, 10-20 cm and 20-30 cm). These samples are required to study the spatial variability of the soil parameters for the initial sampling. On the basis of these

data, the number of the soil samples needed for a second sampling was estimated based on the IPCC requirement to detect the changes in the SOC stock with a confidence level of 95%. Stolbovoy et al. (2007) based their estimates on coefficients of variation (CVs) of SOC content for cropland, pasture and forest of 9%, 15% and 23% respectively; these CVs give information on the upper and lower values for SOC stocks and not just the mean estimate for SOC stock and allow a more accurate determination of numbers of samples required for detection of changes of SOC. Their calculations gave requirements of 243, 675 and 1587 samples in total for cropland, pasture and forest respectively. The prices to determine C in commercial laboratories in Europe were reported to vary from €6 to €16 per sample. The laboratory measurement of the C concentration represents only about 20 to 30 % of the total costs of analyzing soil carbon content while the field work and other laboratory practises represent the rest (Mäkipää et al. 2008). The total costs of measuring changes in soil carbon stocks of Finnish forests range from an estimated 35 euros per soil sample if 40 samples are taken from a plot, to 60 euros per soil sample if 10 samples are taken per plot.

The estimate for a single sampling campaign based on the IPCC approach, in which the initial number of samples is 27 (sampling sites containing 3 sampling depths each) was €162 to 432. The verification cost for the second time observation was forecast to be substantially larger due to increase of the number of soil samples to meet a confidence level required by IPCC. In contrast, the cost of a single sampling campaign for a 4 ha agricultural field using the AFRSS method could range from €18 to 48. Hence, on the basis of those calculations, Stolbovoy *et al.* (2007) concluded that the required soil monitoring for a complete periodic assessment of soil C and N stocks throughout Europe, using the IPCC approach, would be prohibitively expensive. As a more cost-effective alternative JRC suggest the AFRSS approach, whereas COST Action 639 suggests that monitoring efforts need to be concentrated at sites and on land management practices where the stock changes are most likely to happen. The latter approach has indeed been shown to decrease the effort needed and thus costs of monitoring changes in soil carbon stocks in Finnish forests (Peltoniemi *et al.* 2007).

When estimating the number of samples needed and the costs of soil carbon monitoring, it is necessary to keep in mind that the reliability of estimating soil carbon changes increases with an increasing number of samples in an asymptotic way, and thus the gain in confidence per sample decreases with the increasing number of samples. As a result of this, a predetermined level of confidence, for instance 95 %, may lead to suggest that a large number of soil samples is necessary, although even a considerable reduction in the number would not seriously impair the reliability of the estimate (Liski 1995).

3.2.3 European harmonisation

In contrast to other monitoring programmes, soil carbon monitoring deals with a global problem. As most countries have not yet chosen or fixed a methodology, there is a considerable scope for harmonization of soil carbon monitoring. A systematic and harmonized monitoring across EU27 would allow for adequate representation of changes in soil carbon in reporting emissions from soils and sequestration in soils to the UNFCCC.

The evaluation of the monitoring schemes carried out in ENVASSO highlighted the problem that the current monitoring networks were all designed for different purposes and in different ways – many countries having several schemes for different reasons. This is why ENVASSO concluded that more harmonisation was required. The lack of harmonisation in the soil monitoring programmes had earlier been noted by the EEA (2003). Across Europe very few countries have designed and implemented monitoring networks for soil organic carbon (SOC) where more than one campaign of sampling has been undertaken. Measurements differ with respect to depth, frequency and across categories of land use. Furthermore, crucial information on e.g. bulk density may not be available. The need for improved methods to account for changes in soil bulk density remains a hindrance to quantification of changes in soil carbon stocks (Izaurrealde and Rice, 2006). ENVASSO concludes that considerable effort would be necessary for some Member States to reach acceptable levels of minimum detectable change for C sequestration accounting. For SOC, a time interval of about 10 years would enable the detection of some simulated large changes in most European countries.

LUCAS has proved its reliability in providing for the first time harmonised and comparable data at EU level. The EEA concludes that LUCAS in observing land use and land cover and their changes, provides fundamental data for indirectly monitoring decline in SOC as well as threats such as erosion, soil sealing, and possibly floods and landslides.

3.3 Recommendations for monitoring schemes

3.3.1 Considerations when making recommendations

Any recommendations concerning monitoring schemes must be targeted to the specific goals of the monitoring. For example, the design of a scheme for mapping carbon stocks is not the same as one optimized for monitoring carbon change (EA, 2008). Moreover, we may wish a programme of soil monitoring not to be limited solely to following changes in soil properties: it should preferably also facilitate studies that increase understanding of the mechanisms behind the observed changes (King *et al.*, 1998). For soil carbon change, this will require measuring environmental factors that affect soil carbon dynamics (see section 3.2), and soil factors that affect the stability of the carbon pools.

Further, new programmes can be continuations, modifications or alternatives to existing monitoring programmes. The existing programmes should be evaluated in the light of the statistical principles of monitoring soil carbon changes. Decisions about the future monitoring programmes should then be based on: (1) the results of this evaluation and (2) benefits obtained when continuing the old programmes even though they may not be statistically optimal. The way forward would be a compromise between (1) and (2), and the optimization problem would be to put weights on the alternatives.

3.3.2 Towards harmonisation of monitoring schemes in Europe

The EEA considered the main problem at present to be the lack of harmonisation in existing soil monitoring programmes (EEA, 2003), and we conclude that this applies also to monitoring soil carbon levels. To achieve a common EU approach to soil monitoring

there needs to be an EU body, such as an EU Soil Conservation Service, to act as a European focal point for soil protection and monitoring. A number of initiatives exist at national and at European level aiming at the collection of basic soil data in the form of inventories accessible in electronic formats. It is crucial that a common approach for the collection of georeferenced soil data is adopted at EU level. There may be a role here for the European Soil Data Centre (ESDAC), based at JRC Ispra. ESDAC has the remit to collate and hold European wide soil data but currently mainly hold metadata (<http://eusoiils.jrc.ec.europa.eu/library/esdac/index.html>).

Once a common baseline is established, soil monitoring could effectively be implemented at European scale. Existing European initiatives (ICP Forest, FOREGS, LUCAS) show that data collection requires a strong harmonisation effort to allow comparability across country borders. The adoption of common standards (ISO, CEN) should be encouraged as far as possible. Currently existing national initiatives are very fragmentary and difficult to harmonise as reported by ENVASSO. Major changes in measurement methods would be required for some of them in order to comply with common ISO or CEN standards. The final recommendations delivered by the Working Group on Monitoring established to support the development of a Thematic Strategy on soil protection (Montanarella *et al.*, 2004) were the following:

1. Establish a common EU wide soil inventory (baseline) containing general soil parameters and specific parameters for each threat to soil as identified in COM 179 (2002).
2. Select a minimum set of common parameters to be monitored which should be part of the existing soil monitoring systems at National level.
3. Promote the adoption for the measurements of the selected common parameters of standardized methods and procedures
4. Organize regular quality-control/quality-assurance procedures including also laboratory ring tests, benchmark sites, etc.
5. Establish a regular reporting procedure (5 years) for the selected parameters from the Member States to the European Commission.
6. Explore the possibility of achieving a stronger EU coordination of soil monitoring activities through an EU Soil Conservation Service.

We concur with most of these recommendations. We do see the need for a caveat regarding the last of the EEA recommendations (more effective coordination of existing initiatives rather than the establishment of new soil monitoring systems): it is important to realize that the goal of monitoring SOC changes may require significant changes to current soil sampling procedures.

Given the many purposes for which soils need to be monitored, it is recommended that a flexible design will be adopted. A grid based scheme is more flexible than a scheme designed to produce results at a regional scale for different landuses, although the number of points to visit will be higher to get the same confidence in the results. The only way to estimate how many points need to be resampled to estimate change is to carry out a pilot study based on sites already measured to provide the required information on the variability of change. As noted earlier in this chapter, England and Wales are the only countries with real resampled data on a national scale across Europe. We recommend that countries resample as many sites as possible using the original sampling methods – to estimate the variability of change in their own country.

The recommendations from the ENVASSO project were that harmonisation and co-ordination are required to enable a harmonised system to be set up that allows comparison of the data provided by monitoring networks and geographical databases. Creating a minimum coverage of one site per 300 km² is the least that should be accepted, together with an intensive programme of cross validation to permit valid spatial and temporal comparisons both within and between Member States. ENVASSO has shown that, if all 50×50 km cells were to have a site density equal to this median coverage, 4100 new sites would be required, mainly located in Italy, Spain, Greece, parts of Poland, Germany, the Baltic countries, Norway, Finland and France.

CarboEurope stressed the importance of long term monitoring sites with a reasonable frequency of measurements to assess the contribution of land use change to emissions. Non-invasive soil C measurement techniques (tritium probe; multi-spectral RS and infra-red analysis) should be developed to make the technologies usable to improve monitoring and verification networks. IPCC WGIII also concludes that development of remote sensing, new spectral techniques to measure soil carbon, and modelling offer opportunities to reduce costs but will require evaluation (Smith et al., 2007b). Technical Working Group V, established in preparation of the Thematic Strategy for Soil Protection, concluded that peatlands require a specific monitoring programme. Detection of small changes in soil C stocks requires great sampling efforts. They therefore proposed specific sampling schemes for the detection of subtle soil C stock changes with a large impact on greenhouse gas budgets.

4 Carbon storage and trends in Europe

4.1 Introduction

Knowledge on both the amount of carbon stored in the European soils and current trends in the soil carbon stock is the basis for any considerations of the importance of the soils in the European carbon budget.

In this chapter we compare Europe-wide estimates of the soil carbon pool, based on (i) a methodology from the Joint Research Centre (JRC) and (ii) a method derived from a soil organic carbon map of the United States Department of Agriculture (USDA) with national estimates where available.

Trends in soil carbon pools depend on changes in environmental conditions including climate change and land management in a soil-type-specific way. Since monitoring systems for these changes in soil carbon are rare in Europe, we present trends in soil carbon that are based on different model calculations. These modelled estimates are compared with estimates from available case studies from different regions in Europe.

Organic soils contain large amounts of approximately 20% of the total European soil carbon and thus are of paramount importance for the carbon balance of European soils. Therefore, the estimates of carbon stocks and trends of peat soils are treated in a chapter 5.

4.2 Carbon storage and trends

4.2.1 Carbon pool estimates

- *Global level*

Estimates of SOC quantities with global coverage have been provided by several studies (Batjes, 1996; Carter and Scholes, 2000; Jobbagy and Jackson, 2000). To be useful at European level or at larger scales the global estimates are limited by the lack of spatial detail.

- *National, regional level*

Also available in the literature are estimates at national level and for specific sectorial activities at regional level (e.g. Howard *et al.*, 1995; Liski & Westman 1997, Smith *et al.*, 2000; Arrouays *et al.*, 2001). The SOC estimates given at national level are based on very diverse methodologies, which apart from not being generally available with pan-European coverage, hamper any attempts of integrating the results into a harmonized dataset.

- *EU level*

Given the lack of a comprehensive layer of SOC content with pan-European coverage and with a geometry adequate for integrating additional thematic data layer the Joint

Research Centre (JRC) of the European Commission produced a spatial dataset of SOC content (%) estimated for topsoils to a depth of 30 cm across Europe (Jones *et al.*, 2005). The estimates of SOC content were intended to form a basis for improving estimates of the organic carbon stocks in the soils of Europe. A spatial layer of estimates of SOC stocks was generated as an experimental product.

European SOC content map

A map with European coverage depicting estimated SOC contents in the surface horizon is shown in Figure 7. The estimates were computed using the components of the European Soil Database (King *et al.*, 1995; Heineke *et al.*, 1998) (http://eussoils.jrc.it/ESDB_Archive/ESDBv2/index.htm) and complementing the database with ancillary data. The European Soil Database comprises a Soil Geographic Database of Eurasia (SGDBE) and a database of Pedo-Transfer Rules (PTRs) (Van Ranst *et al.*, 1995). The ancillary spatial layers were used for the parameters for land cover (CORINE land cover data set) and accumulated average annual temperature data (Global Historical Climatology Network (GHCN), Easterling *et al.*, 1996). The temperature function was developed based on the measured data of the Soil Profile Analytical Database of Europe (SPADE/M) database of the European Soil Database (Hiederer *et al.*, 2006; <http://eussoils.jrc.it/projects/spade/spadeM.html>).

The distribution of SOC contents (Figure 7) shows areas in southern Europe with a SOC content in the top soil layers between 0 and 1% appear in the expected places and the organic soils with high SOC contents in northern Europe are clearly highlighted.

The modeled values for SOC content in topsoil were compared with data from soil samples taken in England & Wales and Italy. Data from England and Wales originate from the National Soil Inventory (NSI) made during the period 1979-1983 (McGrath and Loveland, 1992). For the UK data an aggregation of the results over administrative areas (NUTS 2) gave a good correlation with an x -coefficient of 1.01 for a linear regression with a coefficient of determination of over 0.9. For Italy the field data do not permit calculating a meaningful coefficient of correlation between observations and modeled values, because the sample sites were restricted to agricultural areas. Yet, the data are very useful for calibrating the temperature correction function for areas with low SOC in southern Europe.

Studies with national scope generally assign data collected at a limited number of sample locations, deemed to be representative of a particular soil type, to the polygons on a soil map, or generate maps by interpolating properties between sample locations. While both approaches are relatively straightforward since the use of ancillary information is very limited their disadvantage is that organic carbon contents can vary greatly within pedologically defined soil units (Batjes, 1996, 1997).

By contrast, the method developed at the JRC uses a sophisticated PTR and applies the rule to the Soil Geographic Database of Eurasia, most detailed (1:1,000,000 scale) and harmonized spatial data set that currently exists for Europe. While the results of generating a European topsoil OC map were validated with ground data from the SPADE/M database and national surveys performed in England & Wales and Italy (agricultural land), further validations should be performed using measured data from other areas in Europe and for the whole range of land cover types. There is scope for

further refining the definition of parameters used for the temperature correction to areas with accumulated annual average temperatures <2000 deg C. There may also be some merit in adding a correction, based on precipitation data, to account for the effect of soil moisture on organic carbon.

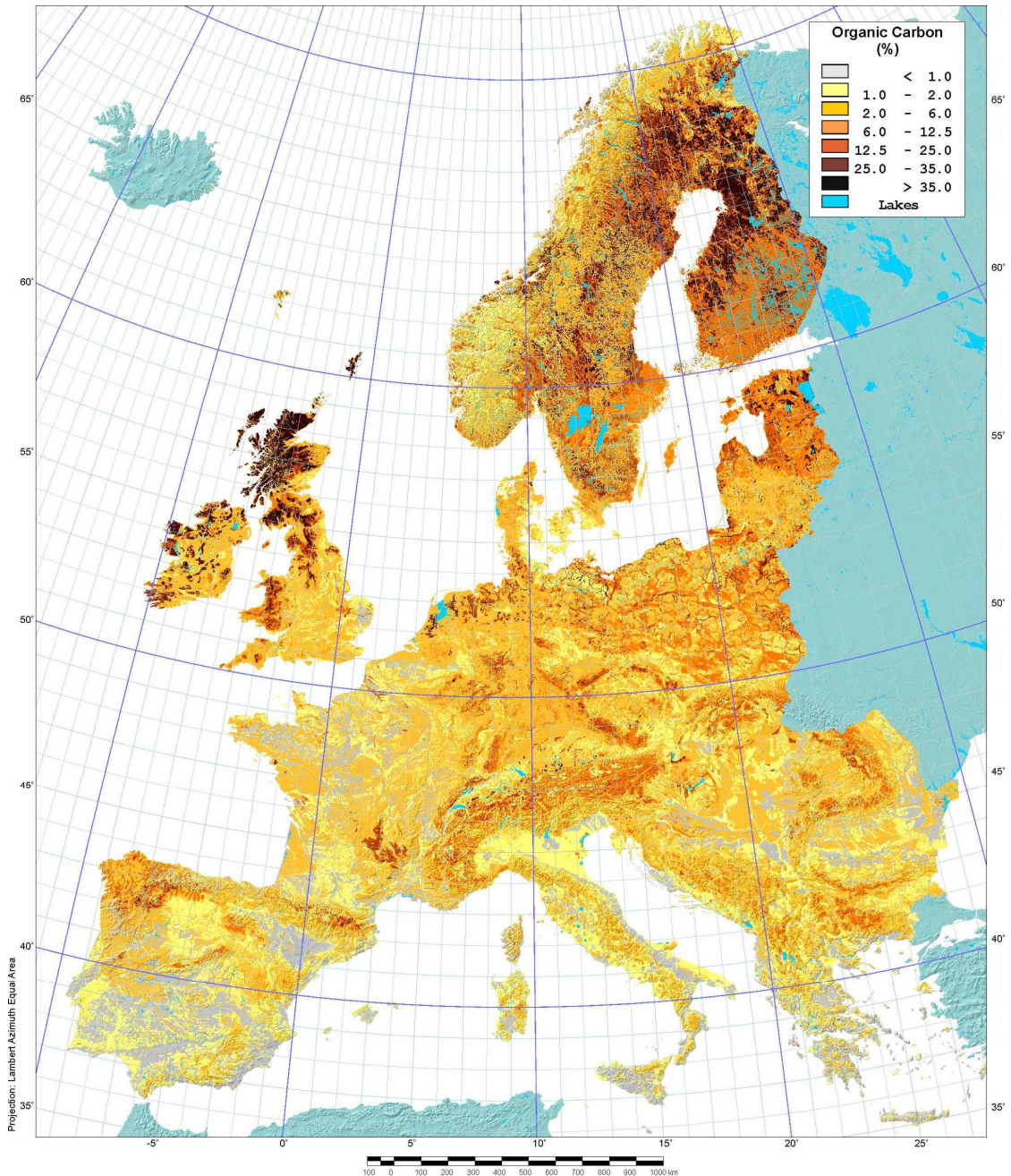


Figure 7 Soil Organic Carbon Content Estimates for Europe

Table 3 Soil Organic Carbon Stock Estimates from JRC pan-European Spatial Layer, USAD NRCS SOC Map and national estimates; the available national figures are all based on observations and measurements on soil organic matter or soil organic carbon and use pedo-transfer rules to calculate stocks of SOC.

Country	JRC	JRC	NRCS	Other	Reference for other estimate
	Pg	%	Pg	Pg	
Sweden	13.5	16.9	6.2		Ahlholm & Silvola 1990, Liski & Westman 1997
Finland	12.1	15.2	10.7	7.5	
United Kingdom	7.1	8.9	6.6	9.8; 4.6	Milne & Brow, 1997; Bradley et al., 2005
Germany	6	7.5	5.4		
Poland	5.7	7.2	5.1		
Norway	5.4	6.8	4.6		
France	5.3	6.6	6.7	3.1	Arrouays, 2001
Spain	3.7	4.6	5.6		
Romania	2.4	3.0	3.1		
Italy	2	2.5	3.9	1.9	
Latvia	1.8	2.3	1.6		
Ireland	1.7	2.1	1.2	2	Tomlinson, 2005
Estonia	1.5	1.9	1		
Austria	1.2	1.5	1.4	1.2	Ginrich et al., 2007
Czech Republic	1.2	1.5	0.9		
Bulgaria	1.1	1.4	1.2		
Bosnia and Herzegovina	1	1.3	0.7		
Hungary	1	1.3	1.1		
Lithuania	1	1.3	1.5		
Netherlands	0.7	0.9	1	0.3	Kuikman et al., 2003
Slovak Republic	0.7	0.9	0.7		
Switzerland	0.7	0.9	0.6	0.5	Bollinger
Denmark	0.6	0.8	0.8	0.6	Krogh et al., 2003
Greece	0.6	0.8	0.9		
Croatia	0.5	0.6	0.7		
Portugal	0.5	0.6	1.1		
Belgium	0.3	0.4	0.4	0.3	Letkens et al., 2004
Albania	0.2	0.3	0.3		
Slovenia	0.2	0.3	0.3		
Cyprus	*		0.1		
Iceland	*		1.2		
Luxembourg	<0.1		<0.1		
TOTAL	79.7	100	75.3**		

* Not estimated in JRC data.

** For areas common with JRC coverage.

European SOC stock map

In the course of the JRC study above, SOC stocks were also estimated at country level (Hiederer *et al.*, 2004).

Table 3 includes a comparison between 3 different types of estimates of SOC stocks in different countries. These are:

- estimates SOC stocks in the top 30cm as estimated in the JRC spatial layer
- estimates derived from the SOC map of the USDA Natural Resources Conservation Service (USDA, 2000). The map data are based on a reclassification of the FAO-UNESCO Soil Map of the World and the estimates were obtained by a combination with a soil climate map. The SOC quantity is estimated to a depth of 100 cm.
- national or regional estimates

Comparison of the different estimates

The pool of organic carbon in the European soils is equal to 79.7 Pg according to the JRC estimate. The estimate derived from the USDA data base is 6 % smaller, 75.3 Pg. The European soils represent about 5 % of the total soil organic carbon pool worldwide equal to about 1,500 Pg (Jobbagy & Jackson 2000). The soil carbon pool is nearly ten-fold the size of another major terrestrial carbon pool in Europe of 8 Pg in forest biomass (Liski *et al.*, 2003).

The estimates of SOC stocks between the JRC and the USDA NRCS maps are not too far apart from each other for most of the European countries when one takes account of the variability of figures quoted in literature. However, the difference is larger than 1 Pg for Finland, France, Italy and Spain. A particular case is Sweden, where the JRC estimates are more than 7 Pg higher than those of the USDA NRCS map.

National estimates of the SOC pools for all land cover types were available for nine European countries (Table 3). When comparing the estimates of the JRC and USDA maps to the national estimates, it appears that the former are more similar to each other than they are to the national estimates. Even within one country, as in the case of the UK, where two national estimates were found in literature these were different to each other. Keeping in mind uncertainty associated also with the national estimates, this comparison did not provide more information on the validity of the Europe-wide estimates.

While the JRC and NRCS estimates are not too far apart, it should be considered that the JRC estimates relate to topsoil to a depth of 30cm only whereas the NRCS estimates cover the soil horizon to a depth of 100cm. One would expect the JRC estimates for the smaller volume to be lower than those to a depth of 100cm. According to the global soils database held at ISRIC in Wageningen, The Netherlands, for most mineral soils about the same amount of carbon is held in the 30-100cm layer as in the 0-30cm layer. Smith *et al.* (2000) fitted a quadratic equation to data from 22 soils from the global soils database of Batjes (1996) to derive this estimate. If the deeper soil layers contained substantial quantities of carbon in Europe, it may at first sight be surprising that the JRC estimates were comparable to the estimates derived from the NRCS map or the national estimates.

Considerable variations for estimates of SOC stock are not uncommon. Jobbagy & Jackson (2000) estimated the error of the mean of their global figure as 320 Pg C for 1,502 Pg C for depth layer 0-100cm and ± 1 Standard Deviation (corresponds to an interval with 68% confidence). Similarities between figures do not necessarily give credence to an estimate because the source of the data and the methodology used to generate the global estimates can often be traced to a single source (FAO data, use of PTR for SOC). Since the SOC estimates of the JRC and the NRCS were obtained from independent datasets and produced by very different methods and still produce similar estimates, this gives a strong support to the validity of the numbers in Table 3.

Sources of variation in regional SOC stock estimates

Because the JRC estimates of SOC content were confirmed by data from soil samples (England & Wales, Italy (agricultural land) and the SPADE/M sites) the source of variations in SOC quantity between the JRC and the USDA maps, comes from the parameters of the transfer of SOC content to SOC quantity. SOC quantity (mass per area, kg m^{-2}) is estimated from SOC density (mass per volume, kg m^{-3}) by a function for a depth of 100cm with the function parameters being SOC content, bulk density, volume of stones and soil depth. Indeed, the variations in estimates suggest that the seemingly simple transfer from SOC content to SOC quantity contains one or several indeterminate factors. The factors that can explain the differences between the estimates are:

- SOC content in the subsoil does not increase to the same degree as SOC quantity. Analysis of the SPADE/M dataset indicates that the SOC content in the 30-100cm subsoil layer is approximately 30% of the SOC content in the topsoil layer. In a stone-free horizon, SOC density does not decrease to the same extent due to the increase in bulk density with soil depth.
- the relatively large difference in SOC stock estimates between the JRC and the NRCS for the Finland, Sweden and the UK may come from the estimates of bulk density used to convert estimates of SOC content to stocks, in particular for organic soils and peat. The JRC used as bulk density for peat values found in the literature, but no differentiation of peat types could be performed.
- A factor reducing the volume of soil is the presence of stones in the soil layer. For the JRC data the volume of stones was estimated from the corresponding European Soil Database PTR. The rule can be considered incomplete in this respect, because the maximum class gives a volume of 20% and important areas for SOC, such as Scotland, are not covered. When estimating volume to a depth of 100cm the increase in the volume of stones with depths would be an important factor.
- The layer volume is reduced since the soil layer does not universally reach a depth of 100cm and can be as shallow as 20cm or even less. This circumstance directly reduces the volume over which the SOC quantity is estimated.

Another, more intrinsic factor not included in the parameters for computing SOC stock from SOC content is the definition of the output classes in the PTR. For example, the original PTR of the European Soil Database only covers mineral soils and the output of the highest class contains all soils with a SOC content of more than 6% and includes

not only high SOC mineral soils but all organic and peat soils together in one class. This definition of size classes and the mean SOC content of a class can indeed bias the final estimate of SOC content and subsequent SOC quantity. In the revised PTR the output was extended to specifically cover not only mineral soils but also organic soils and peat to reflect the range of SOC contents of more than 6% and differences between soil types more accurately.

Conclusions

Considering the uncertainties in the data used for estimating SOC quantities the variation in estimates is hardly surprising. When interpreting the results it should not be forgotten that the parameter measured in soil samples is SOC content and that SOC quantity is a derived value on the basis of parameters such as bulk density, volume of stones and the depth of the soil layer. Therefore, improving the information about these latter parameters is necessary in order to allow to make significant progress and improve the accuracy of the estimates of SOC stocks in Europe.

4.2.2 Carbon trends

Trends in soil carbon are estimated by measurement at different scales and by models, or combinations of both. We first discuss carbon trends observed in measurement campaigns. These measurements give a direction for the effect of land use on carbon trends, but are insufficient to arrive at estimates for carbon trends at higher scales. Therefore, we use the results of models to estimate the effects of land use on carbon trends in the EU as a whole.

As carbon trends are directly related to land use, this section is also relevant for the later section on the effect of land use on C sequestration (section 6.2). However, this section aims to give an estimate of observed trends in Europe, whereas the discussion on land use in section 6.2 mainly aims to give direction on future land use to sequester carbon.

The carbon balance of European soils is the sum of the balances of individual soils responding to changes in environmental conditions and land use and management in a soil-type-specific way. When estimating the carbon balance of the soils at the European scale, all the details of the different soils cannot be accounted for. Rather, the challenge is to account for the effects of the most important ones. Results of detailed case studies are very useful to learn about the processes and describe the most essential ones in models. In addition, the case studies provide information that can be used to test the validity of the large-scale estimates.

Here, we summarize studies carried out on specific sites, soils or regions for grasslands, croplands and/or forests and for land use changes (section 4.2.2.1). The results are used to evaluate the validity of the European estimates described above. Furthermore, the effects of major factors affecting the carbon balance of soils, land-use change and fertilizer use, are discussed based on the case studies for England and Wales, Great Britain, Belgium and France (section 4.2.2.2).

4.2.2.1 Evaluation of estimates on carbon stock changes from experiments

Grassland vs Croplands

Case studies on long-term changes in soil C in agricultural systems in Europe and those from other parts of the world demonstrate that grasslands contain greater soil C stocks than arable systems, and that conversion from grasslands to arable cropping reduces soil C (Soussana *et al.*, 2004). Higher grassland SOC is a result of many factors including absence of soil disturbance, greater return of plant residues and/or higher production and the return of dung during grazing (Rees *et al.*, 2005). Drawing on European case studies, on a Swedish farm with known land use since 1850, SOC concentrations were 1.6 times higher (4% C vs 2.5 % C) in fields under permanent grassland than fields under cereal cultivation since 1880 (Katterer *et al.*, 2008). In addition, clear temporal responses to land use change were detected: SOC fell upon conversion from grassland to arable, and increased when the land was moved back into permanent grassland (+0.6% C in 32 years).

Land use change and rotations

The effect of a land use change is also dependent upon the initial soil C content; soils with high initial C contents are more prone to losses than soils with already low C (Katterer *et al.*, 2004; here 'high' soil C at 2-3.4% and 'low' soil C at < 2%;). The long-term fall in SOC content in English arable soils has been attributed to movement away from grassland in mixed farming rotations into permanent arable cropping (King *et al.*, 2005). Post and Kwon (2000) based on another analysis of literature values, estimated that land use change from arable cropping to grassland results in increases in soil C of 33 g C m⁻² yr⁻¹, although rainfall and the species sown in the new pasture can affect the rate substantially.

Crop rotations

Crop rotation also affects soil C: complex rotations can maintain higher C contents than monocultures (Morari *et al.*, 2006), although this is not always the case (Persson *et al.*, 2008). Enhancing rotation complexity (e.g. changing from monoculture to continuous rotation, changing from crop-fallow to rotation, or increasing the number of crops in a rotation system) sequestered 15±11 g C m⁻² yr⁻¹, with a new equilibrium reached in 40-60 years. To summarise the current state of scientific knowledge, global analyses of all available literature reports on specific studies is required. West and Post (2002) performed such an analysis on data from 67 long-term experiments, to quantify the effect of tillage and crop rotations on SOC. They too give valuable error values and ranges for their estimates for changes in C contents in response to management and crop rotations. Going from conventional tillage (CT) to no tillage (NT) can sequester 57±14 gC m⁻²yr⁻¹ and with increasing complexity in crop rotations may sequester 20±12 gC m⁻²yr⁻¹ with some exceptions where no change in SOC content may be expected. The rates of change of SOC content may peak in 5-10 years and the new equilibrium of higher SOC may typically be reached in 15-20 years assuming continuation of management started.

Land use history

Although individual case studies are useful, differing results are common due to the complexities of individual site histories and soils, specific management practices, and soil sampling methods. Land management prior to the start of the controlled experiments may have a continuing impact on soil C well into the experimental period, and the assumption that soil C content was in equilibrium with C inputs and outputs prior to a land management change (and is therefore simply responding to new management practices) may often be wrong. Any continued disturbance via ploughing is likely to reduce soil C in soils which have not yet reached an equilibrium state, whilst soils which have been cultivated for longer periods have likely reached a new (low-C) equilibrium and the effects of continued ploughing will be slight (Katterer *et al.*, 2004).

Cropland

The numerous factors affecting the carbon balance of croplands illustrate the difficulty in modelling the carbon balance of these soils at the European scale. They help also to understand the large uncertainty associated with the current Europe-wide estimates obtained using models. As there are so many affecting factors, it is not easy to find meaningful counterparts among the case studies to compare to the European scale mean estimate of the cropland soils, which suggests that they are a carbon source equal to $92 \text{ g C m}^{-2} \text{ year}^{-1}$ (Janssens *et al.*, 2003).

A change in management from conventional tillage to no-till practices sequesters $57 \pm 14 \text{ g C m}^{-2} \text{ yr}^{-1}$ (mean $\pm 95\%$ CI), with the exception of wheat-fallow systems, where no C is sequestered when changing to no-till. Carbon sequestration rates will likely peak after 5-10 years of no-till, and new equilibrium SOC contents will be reached after 15-20 years.

Grasslands

Under existing management conditions, most grasslands in temperate regions are considered to be C sinks (Jones and Donnelly, 2004). Measured current C sequestration rates in grasslands are in the range $0\text{-}640 \text{ g C m}^{-2} \text{ yr}^{-1}$; whereas results derived from direct measurements of soil C suggest more realistic sequestration of $45\text{-}80 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Jones and Donnelly, 2004). The average rate of carbon accumulation in the grassland soils of Europe of $67 \text{ g C m}^{-2} \text{ year}^{-1}$ (Janssens *et al.*, 2003) is in the middle of this range.

C stocks in grassland can be increased by using appropriate management, including irrigation, addition of fertiliser (both mineral and organic), and changes to grazing practice. In a meta-analysis of 87 data points, grassland fertilisation and management with appropriate grazing levels led to increased C sequestration in soils of 30 and $35 \text{ g C m}^{-2} \text{ yr}^{-1}$, respectively (Conant *et al.*, 2001).

Forests

There are two main approaches for examining changes in C held in forests:

- i) using data on standing biomass and forest growth from forest inventories in conjunction with soil C models:

with this approach combinations of Scandinavian forest biomass inventories and models suggest that Swedish and Finnish forest soils currently sequester $7\text{-}12 \text{ g C m}^{-2} \text{ yr}^{-1}$ (de Wit *et al.*, 2006; Liski *et al.*, 2006; Agren *et al.*, 2007).

ii) using soil samples taken from specific forests over time:

with this approach an analysis for three forests in Sweden suggested that C sequestration in the forest floor is $18 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Berg *et al.*, 2007).

Combining both approaches at 64 forest sites in S. Finland, Peltoniemi *et al.* (2004) established a good agreement between model estimates of C stock ($7.0 \pm 0.6 \text{ kg m}^{-2}$) and measurements ($6.8 \pm 2.5 \text{ kg m}^{-2}$), with average sequestration of $4.7 \pm 1.4 \text{ g C m}^{-2} \text{ yr}^{-1}$.

This combination of field data and modelling demonstrates that models give similar estimates to field-based measurements, and gives confidence that model outputs – where no corroborative field evidence is available – can be seen as good estimates of stock and change.

The management of forests is clearly important in both the above- and belowground C balance. Though much is known about the effects of management on growth and biomass and on organic (litter) layers in soil, relatively little is known about management effects on the deeper mineral layers of soils (Jandl *et al.*, 2007). Here are a few examples:

- In three Norway Spruce forests in Denmark, thinning (down to 50% of unthinned basal area) resulted in $600\text{--}1100 \text{ g C m}^{-2}$ lower C stocks in litter in the organic O horizons compared to unthinned areas; this was attributed to the changed forest floor microclimate (soils become warmer and possibly wetter) which stimulates decomposition (Vesterdal *et al.*, 1995).
- Reductions in litter inputs after thinning will also contribute to altered soil C pools. Harvesting the trees in a forest will lead to a reduction in litter in the O horizon C due to the cessation of new aboveground litter inputs and changes in microclimate (Lal, 2005); meta-analysis of literature data suggests that the effect on the mineral soil depends on the method of harvesting with sawlog harvesting (i.e. only tree trunks removed) increasing mineral soil C ($18 \pm 9\%$, mean $\pm 99\%$ CI) but whole-tree harvesting reducing mineral soil C ($-6 \pm 6\%$) (Johnson and Curtis, 2001).
- Disturbance by beetle infestation and subsequent forest dieback in Norway spruce forests in Germany led to reduced C after 25 years.

Sofar, the integrated effects of forest management on soil C have not been systematically taken into account in the calculation of changes of soil C stocks across European forests.

4.2.2.2 *Evaluation of estimates on carbon stock changes from case studies*

Soil Carbon stock changes in UK

Bellamy *et al.* (2005) reported on soil organic carbon changes in UK and Wales over the interval 1978 – 2003 on the basis of data from the two samplings of the NSI (Annex 5). The results are:

- peat soils lost carbon an order of magnitude faster than brown soils
- man-made soils, and bogs and upland grass lost carbon an order of magnitude faster than lowland heath
- some soils, i.e. lowland heath, gained carbon
- no statistically significant relations between rate of change and land use, rainfall class or soil textural class were found
- the rate of loss increased with initial organic C content.

Using this relationship it was estimated that carbon was lost from soils across England and Wales over the survey period 1978 – 2003 at a mean rate of 0.6% yr⁻¹ (relative to the existing soil carbon content in 1978). This estimate was based on the soil carbon content of the top 15cm of soil. Converting this to carbon stocks (using a pedotransfer function to estimate bulk density) it was estimated that the soils of England and Wales were losing carbon at the rate of 4.44 Tg C yr⁻¹.

A second assessment of carbon stock change in Britain comes from The Countryside Surveys of Great Britain (GB); these are ongoing ecological assessments of the non-urban land in GB (Annex 5) (Firbank *et al.*, 2003). The surveys have taken place in 1978, 1984, 1990, 1998 and 2007. Average topsoil C concentrations across GB in 1978 and 1998 were 128.8±17.5 and 138.5±17.6 g C kg⁻¹ (mean±95% CI), respectively. The increase of 9.7±6.0 g kg⁻¹ over the 20 years (0.5±0.3 g kg⁻¹ yr⁻¹) was significant (P<0.01). Significant increases in soil C concentration were observed in fertile and infertile grasslands, upland woodlands, and heath and bog habitats, and were in the range 0.2-2.1 g kg⁻¹ yr⁻¹. Taken together, these results suggest that GB topsoil C concentration increased slightly in the period 1978-98, although changes differed between soil type and land use.

Soil Carbon stock changes in Belgium

In Belgium, a comprehensive national soil survey was carried out between 1950 and 1970 (Annex 5) and this survey was resampled in 1989. Van Meirvenne *et al.* (1996) identified an increase in C stocks in permanent arable fields of 930 g C m⁻² between 1950 and 1990 (a rate of 23 g C m⁻² yr⁻¹). Sleutel *et al.* (2006) then extended this time-series with a further sampling of some of the locations in 2003-4, and observed a decrease in soil C stock of 250 g C m⁻² (-19 g C m⁻² yr⁻¹) since 1990.

When all data were included in the analyses, the patterns of soil C change suggest that arable soils have lost C since the original survey at a rate of 3-114 g C m⁻² yr⁻¹. Grasslands were reported either to be sequestering C in soils at rates of 22 or 44 g C m⁻² yr⁻¹ (Lettens *et al.*, 2005a; Goidts and van Wesemael, 2007, respectively), or losing C at 90 g C m⁻² yr⁻¹ (Lettens *et al.*, 2005b). Similar differences in trends in soil C stocks are reported for forests, which are either gaining C at a rate of 73 g C m⁻² yr⁻¹ (Lettens *et al.*, 2005a), or losing C at a rate of 23 g C m⁻² yr⁻¹ (Stevens and van Wesemael, 2008)

Soil Carbon stock changes in France

In France, INRA has measured and reported on measured carbon stocks in the top 0-30 cm layer. All data between 1970 and 2000 for different land uses have been pooled and could be used as an average value for 1990 stock of C (Arrouays *et al.*, 2001). The stocks vary from 15 – 40 ton C ha⁻¹ in mid France to 40 – 50 ton C ha⁻¹ in the richer and more intensive mostly cropping areas in the north and south-west, up to 70 ton C ha⁻¹ in permanent grassland and forest and >90 ton C ha⁻¹ in more mountainous areas and wetlands in the upper 30 cm of soils in France (INRA, 2001; see IFEN (Institut Francais de L'Environnement, 121, 2007). The highest values are reported in organic soil at 350 ton C ha⁻¹. Soils that are under forest, grassland or pasture always have higher organic carbon stocks than identical soils under arable land. IFEN (2007) reports losses of carbon for

soils in some regions and increases of soil carbon in other regions for agricultural soils in France. On average, IFEN (2007) reports that stocks of C in agricultural soils in France are loosing up to 6 Mton C yr⁻¹ or -0.2% of the current stock of soil C in the periods 1990-1995 and 1999-2004 (see also Saby *et al.*, 2008). Forest soils, however, gain carbon and the carbon stock in French forest soils increases with 0.7 Mton C yr⁻¹ or +1.7% of the current stock (INRA, 2002; Arrouays *et al.*, 2002).

4.2.2.3 Comparison of the case studies

As regards the trend

The above cited studies into changes of soil carbon stocks in UK and Belgium are all three based on repeated sampling and report not just different but contradicting results. Bellamy *et al.* (2005) observed a mean loss of topsoil soil organic carbon (SOC) of 0.6% yr⁻¹ between 1978 and 2003 in England and Wales. Also IFEN (2007) reports losses of carbon from agricultural soils in France of 0.2% per year. This figure is based on both gains and losses of soil organic carbon in specific areas and for specific land use and management (IFEN, 2007). These results contradict the results from the Country Side Survey in Britain and also contradict the evidence that the UK and Europe as a whole are a net CO₂ sink (Janssens *et al.*, 2003; 2005). As for non-agricultural areas, it also contradicts data from another long term study of topsoil SOC in British woodlands (Kirkby *et al.*, 2005) and for French forests (IFEN, 2007). Kirkby *et al.* (2005) sampled, in 1971 and in 2000-2003, 1648 plots randomly located in 103 woods; their findings suggest no significant change in SOC over 30 years (slight increase of +0.38% over 30 years; ~+0.01% y⁻¹). IFEN (2007) report carbon gains for forest soils in France. Also, the studies from repeated sampling across Europe show contrasting results with some showing loss of SOC (e.g. for Flemish cropland soils; Sleutel *et al.*, 2003), attributed to changing manure application practices, and others showing no loss of SOC (in Danish croplands; Heidmann *et al.*, 2002 and in Austrian soils; Dersch & Boehm, 1997).

As regards the causes

Bellamy *et al.* (2005) concluded that the observed loss of topsoil soil organic carbon (SOC) of 0.6% yr⁻¹ between 1978 and 2003 in England and Wales was likely caused for up to 60% by higher temperatures and changes in rainfall pattern in the latter decade of the last century and thus attributed to climate change. Smith *et al.* (2007), using two soil carbon models, suggested that only 10-20% of the loss of C from soils in England and Wales reported by Bellamy *et al.* (2005) could be due to climate change. Recent work (Kirk and Bellamy, 2008) has shown that it is likely that past changes in land use history and land management were dominant reasons for the loss of C rather than higher temperatures and changes of precipitation as result of climate change. Also further recent work (Hopkins *et al.*, in press) on soil carbon contents in long-term experimental grassland plots across UK questions whether losses in SOC in recent decades such as reported by Bellamy *et al.* (2005) can be attributed to widespread environmental change i.e. climate change. Changes in bulk density over time or precision and success rate of

actual resampling soils are more likely factors that dominate the observed changes of soil carbon.

A major criticism of the papers cited above is the use of a pedotransfer function to estimate bulk density in the absence of any measures on the sites to allow estimation of carbon stocks. This problem highlights the need to measure bulk density in any future monitoring of soil carbon.

Conclusions

From these repeated sampling studies no clear conclusion as regards to soil being a sink or a source can be drawn; the results of various studies are inconsistent and methodological constraints and omission of relevant data, i.e. bulk densities, may well produce too much noise.

In Annex 5, we describe the three out of four national studies discussed above in detail to illustrate crucial questions related to estimating changes in soil carbon stocks based on repeated soil inventories. These cases are:

- England and Wales, National Soil Inventory
- Great Britain, countryside survey
- Belgium

These three are the only studies found that presented country wide data for determining changes in soil carbon stocks based on repeated soil inventories and measurements. As for the French national study, we are not sure about actual repeated soil measurements and therefore the crucial questions discussed in Annex 5 do not necessarily apply.

4.2.2.4 Estimate on carbon stock changes for Europe per land use

Trends in SOC have been estimated for the main ecosystem types (land cover) in Europe, i.e. grassland, cropland and forest³ (Table 4). The trends in peatlands are covered in chapter 4. The trend estimates covering the whole of Europe are based on modeling because measurements to calculate these trends are not available or incomplete, as indicated in the previous section.

Grassland

The soil of grasslands is estimated to accumulate organic carbon across Europe. The carbon sink estimates for European grassland range from low values between 1 and 45 Tg year⁻¹ (Smith *et al.*, 2005b) to as high mean values as 101 Tg year⁻¹ (Janssens *et al.*, 2003). The high mean estimate is associated with a large standard deviation equal to 133 Tg year⁻¹. This means that there is a considerable probability that the grassland soils in Europe were actually losing carbon despite of the high mean estimate. Janssens *et al.* (2005) give a predicted calculated range of -50 gC m⁻² to 170 gC m⁻².

³ See Annex 4 for methods and their reliability.

Cropland

In European croplands, the size of the soil organic carbon pool is estimated to be decreasing. Again - as with the soil of grasslands - there is a lot of variability in the estimates of the loss rate. These estimates range actually from a small sink value equal to 10 Tg year⁻¹ to a relatively small source value equal to 39 Tg year⁻¹ (Smith *et al.*, 2005b) or to a large source figure equal to 300 Tg year⁻¹ (Janssens *et al.*, 2003). The standard deviation of the high source estimate is large, 186 Tg year⁻¹ (Janssens *et al.*, 2003).

Table 4 Estimated changes in soil carbon pool under different land uses in Europe. Positive figures mean increase in the pool, negative ones decrease; sd stands for standard deviation.

Land use	Change in soil carbon pool (Tg year ⁻¹)	Source
Grasslands	+1 to +45 +101 (sd 133)	Smith <i>et al.</i> , 2005 Janssens <i>et al.</i> , 2003
Croplands	-39 to +10 -300 (sd 186)	Smith <i>et al.</i> , 2005 Janssens <i>et al.</i> , 2003
Forest	+17 to +39	Liski <i>et al.</i> , 2002

In their analysis of the European carbon budget, Janssens *et al.* (2003) concluded that there was a large soil organic carbon (SOC) loss to the atmosphere from croplands. This loss was based on extrapolation from an earlier model study with simple assumptions about crop yield and farmer practice (Vleeshouwers and Verhagen, 2002). In fact, the large and widespread increase in crop yield observed everywhere in Europe during recent decades, does not seem to have entrained a parallel increase in soil carbon stocks (Arrouays *et al.*, 2002). Regional inventories and two out of the three models indicate that croplands are a net source of CO₂ to the atmosphere, but this source is 5 times smaller than the large positive flux ($90 \pm 50 \text{ gC m}^{-2} \text{ yr}^{-1}$) given by Janssens *et al.* (2003; 2005), based upon output from the CEASR model of Vleeshouwers and Verhagen (2002).

Forests

European forest ecosystems are currently sequestering carbon, due to changes in tree age structure and management (Nabuurs *et al.*, 2003). Many forests were planted in the last 100 years and are still actively growing, thus sequestering C in biomass. Additionally, changes in management in recent decades have led to trees standing for longer and less material being removed from forests as a proportion of total biomass. Forest soils are also considered to be gaining C, mainly due to increasing inputs of litter from larger more productive trees (Liski *et al.*, 2002).

At the pan-European level, combination of forest inventories and a soil C model suggest that the total forest soil C sink was 26 Tg yr⁻¹ (range of 17-39 Tg) in 1990, with forecasts suggesting that the sink will increase to 38 Tg C per year (range 26 to 54) in

2005 and to 44 Tg yr⁻¹ (range 28-65 Tg) in 2040 mainly due to increased litter inputs from older trees.

In 1990, the soils accounted for 24 or 32% of the total forest sink while in 2040 the share of soils is calculated to be 38 or 41% (Liski et al., 2002). Carbon in soils in forest were least important in southern Europe where the soil carbon sink was less than 25% compared to the tree carbon sink. Until 2040, the soil carbon sink will become relatively larger than the tree carbon sink and is between 61 to 69% in 2040 (Liski *et al.*, 2002)

4.2.2.5 Qualitative extrapolation of total carbon stock change at EU level

It is not appropriate to try and calculate a total carbon balance for the combined European grassland, cropland and forest soils based on the estimates in Table 4 because of the considerable uncertainty for each land cover class. At their best, the estimates allow for drawing only a rather qualitative conclusion about the carbon balance of the European soils.

Forest soils are a sink of carbon equal to a few tens of Tg per year. Croplands are probably losing carbon but the estimates of this carbon source vary a lot from small values to figures as large as a few hundreds Tg per year. Grasslands are most likely accumulating carbon; the sink estimates range from very low figures close to zero up to values for a sink equal to about 100 Tg per year or even higher.

Soils in Europe- sink or source?

The total effect – sink or source – of the European soils to the atmospheric carbon levels is the sum of two large and opposite fluxes. These fluxes – uptake of CO₂ (photosynthesis and plant growth) and loss (decomposition) of organic matter from terrestrial ecosystems – are significant fluxes in Europe compared to fluxes from other sources of CO₂ to the atmosphere. Is it important to recognize the importance of this as due to the relatively large gross exchange of CO₂ between atmosphere and soils and the significant stocks of carbon in soils, relatively small changes in these large and opposing fluxes of CO₂ may have significant impact on our climate and on soil quality. It is thus relevant to assess and to know the impact of regional differences across Europe, land use and land management and impact of environmental conditions and climate change on these fluxes of CO₂ and soil carbon stocks. Based on the estimates presented here for carbon sinks in grassland and forest and carbon sources in cropland across Europe (Table 4), we estimate that the European soils accumulate carbon and are a sink for CO₂ and this sink is in the order of 1-100 million tons of CO₂ per year.

4.2.3 Conclusions

Carbon pool

The soil carbon stock in the EU27 is around 70 to 80 Pg C. Roughly 50% is located in Sweden, Finland and the United Kingdom, because of the larger share of peat soils in

these countries. Two independent estimates give similar values despite using different approaches. Proper error estimates for the pool value can be obtained from extensive analyses of the statistical aspects of the methods behind the estimates such as sampling design and data analysis. Geostatistical methods help to improve the reliability of the pool estimates in those cases where the soil carbon data are spatially correlated. The potential of these methods has not been fully explored yet.

Methods to estimate the pool sizes would best be harmonized as much as possible between categories of soils, land use types, soil types or countries, as this will improve the comparability of the pool estimates and reduce uncertainty and error. The critical variables to harmonize include soil layers and methods to estimate soil bulk density and carbon concentration.

The amount of coarse fractions in soil, stones and boulders, is a serious source of error in carbon pool estimates for stony soils. An easy method to measure the amount of these fractions in soils in a reliable way has not yet been developed. Fortunately, estimates of changes in soil carbon pools are disturbed less by these coarse fractions because their quantities remain usually unchanged.

Carbon trends

Systems to monitor changes in soil carbon are still very rare in Europe and elsewhere if any exist at all. For this reason, the estimates of changes and trends in soil carbon are not only based on measures and observations but on (different) calculation models. Although current model-based methods give generally correct estimates of the order of magnitude of soil carbon trends as compared to the results of soil sampling studies for specific areas across Europe, Europe-wide estimates for croplands and grasslands are particularly uncertain and model-dependent. This is likely because these land use categories consist of highly heterogeneous soils and the land use history may not be well documented. Other reasons come from methodological constraints.

Models used are usually dynamic models describing carbon cycling in soil at the process level. As the need for reliable information on soil carbon budgets is increasing, the issue of transparency of the development and of the application is required. A protocol on application of soil carbon models would be helpful.

It is difficult to estimate the reliability of the model-based carbon trend estimates more accurately than by comparing them broadly with smaller-scale case studies. Error in parameter values of the soil carbon models is usually not well known and the models must be applied to conditions where the models have not been calibrated or tested for in order to obtain estimates that cover the whole of Europe. To improve the estimates of soil carbon changes across Europe, it would be of great help if Europe-wide monitoring systems would be developed and established and if existing data on carbon would be analysed more extensively. The number of case studies is still low and could be increased to further support calculation or estimation of soil carbon changes at larger regions.

Model-based approaches will – in the near future – remain the most important means to estimate soil carbon changes from land use and management. Such model based approaches are useful to estimate the *current* changes, yet they are the only way to estimate *future* changes and they are very useful to study causes of changes and trends in soil carbon stocks.

The reliability of the model-based soil carbon change estimates can be improved if more attention would be given to obtaining proper statistical estimates of uncertainty for the model-calculated values and parameters. Further, model calculations yield better results if measurements from as wide a range of conditions and soils as possible is used for calibration of the models and testing their validity in order to avoid extrapolating the models to area for which they have not been developed for.

5 Peat soils

5.1 Introduction

Peat soils contain large amounts of carbon of 15-20 Pg of C or 20% of the carbon in European soils. This justifies a prominent place for peat in the climate change-carbon debate. The significance of managed peat soils has been emphasized recently, especially as they are sources of CO₂ (carbon dioxide) but also of the non-CO₂ greenhouse gases CH₄ (methane) and N₂O (nitrous oxide).

Drainage of peats for forestry and agriculture has led to loss of peat and peat soils and a net release of CO₂ from these soils. Most fertile peatlands have been utilised for agriculture, with increased emissions of CO₂ and N₂O. This loss of CO₂ and N₂O from agricultural use of peatlands is estimated at approximately 100 Mt CO₂ equivalents.

Even after abandonment of management, the N₂O emissions may continue, accompanied with a strong release of CO₂ (Maljanen *et al.*, 2007). Those emissions from forest on peat soils put these soils in a special position in national greenhouse gas inventories. Whereas forests on mineral soils appear as CO₂ sinks, the soil emissions of CO₂ and of non CO₂ greenhouse gases methane and nitrous oxide significantly decrease or even offset the C sink in forest biomass on peat soils. Therefore, afforestation of peat soils cannot be considered an effective means of sequestering carbon.

Furthermore, peat extraction for use as fuel or substrate in horticulture releases the C in peat that accumulated during the thousands of years before and this is true for all peat that is lost, no matter the reason or method of extraction. The extraction rate of peat in Europe is stable and amounts to 10-15 Mton of peat per year.

In the following sections, we first describe how peat is formed and where in Europe it can be found. Special attention is paid to peat extraction and the agricultural use of peat soils. We then continue with the greenhouse gas losses from drained peat soils and conclude with the impact of land use and soil management on those losses.

5.2 Peat formation

Peat is the accumulated remains of dead organic material, and it forms in growing peatlands (mires) where the activity of decomposing organisms is suppressed in waterlogged conditions (Lappalainen 1996). Peat may consist of remains of mosses and sedges in arctic, subarctic and boreal regions, reed/sedges and woody litter in temperate regions (Gore 1983). Peatlands were formed during the Holocene in places where the supply of moisture either from precipitation or adjoining watercourses is adequate, and the soil beneath has a low permeability for infiltrating water. Most peatlands were formed in lowlands collecting waters from the catchment, but high precipitation and humidity may also support the formation of bogs on hilltops and slopes. Also alpine environments with adequate water supply can support topographically restricted peat accumulation e.g. in sites with exfiltration of ground water or riparian areas.

Two basic mechanisms of peat formation have been distinguished (Gore 1983). Terrestrialization is a gradual overgrowth and filling of a water body or riparian system by the litter of mosses and aquatic helophytes. Primary paludification occurs when the

principal vegetation community is a peat forming one, while secondary paludification of forest soil follows from a change in local hydrology favoring the peat forming species. Paludification may also occur in association with flooding when the water transported materials sediment as a barrier to slow down to prevent the escape of the excess water.

After the initial development of peats, autogenic processes such as the responses of vegetation communities to ecohydrology may take control of the further development of the peatland (Anderson *et al.*, 2003). Under influence of groundwater flow from upland soils, minerotrophic mires, or fens are sustained. When the thickness of the peat layer increases, the living vegetation may detach from its groundwater-fed nutrient supply, and further nutrients to the so formed ombrotrophic mires (i.e. bogs) are obtained solely from precipitation and this limits growth of the peat layers.

The many factors controlling the peatland formation and development show geographical differences in their distribution, leading to regional differences in peatland types across Europe. Because of the partly autogenic nature of peat accumulation in aged peatlands, no single climatic or geographical factor alone is probably responsible for the development of the peat deposit, the current rate of peat accumulation, or the future prospects concerning the fate of the peat deposit or its increase. However, climate warming (IPCC, 2007) may cause substantial changes to the balance and annual distribution of precipitation and evapotranspiration, which have been shown to induce marked disturbances on the hydrological cycle annual net balance of gross productivity and decomposition in pristine (e.g. Alm *et al.*, 1999) or forested, managed peatlands (Trettin *et al.*, 2006).

The facts that peat and peatlands have been defined differently depending on country, scientific discipline or even due to linguistic problems in translation of many peat-related terms (Joosten and Clarke, 2002) have added to uncertainties in the reviews and soil databases and maps, e.g. FAO-UNESCO (1974), FAO-UNESCO-ISRIC (1990), FAO (1998). For the definition of organic soil and Histosols according to FAO (1998), see Annex 7. As a consequence of slightly different definitions for peat, estimates of peat areas found in literature differ depending on the definition and ancillary data used to define peat and rather large ranges for estimates are given. Furthermore, wide ranges for estimates are due to the fact that the presence and extent of peat cannot always be established with certainty.

The problems associated with the range of definitions of peat and peat-forming ecosystems have been elaborated by Montanarella *et al.* (2006). They assessed information of topsoil organic content from the Map of OC (organic carbon) Topsoils (Jones *et al.*, 2004) and the European Soil Database (King *et al.*, 1994), and amended the derived soil attribute results using CORINE plant cover data and Historical Climatology Network (GHCN, Easterling *et al.*, 1996). The analysis of the results derived from the European databases revealed a difference of the order of 9 % in the Map of OC Topsoils with OC \geq 25% as compared with data obtained in inventories made in Great Britain (Burton 1996) and Northern Ireland (Shrier 1996). The estimate concerning Finnish total peatland area had similar accuracy. As a conclusion they suggest to use the OC 25 % Map of Topsoils to estimate the distribution of peat and peat-topped soils in Europe.

As peatlands are different in their ecological functions due to differences in their development and current ecohydrology, they also respond differently to management and climate change. The usability of databases for a more uniform view on peatlands should

increase. We may expect that when the revision of land cover map (CORINE 2000) and the extension of the European Soil Database are finished, the information on European peatlands becomes less scattered. Still, there is for example a category in the CORINE 2000 database called “Peatbogs” that includes all types of minerotrophic mires, ombrotrophic mires, and peat extraction areas, adding uncertainty in e.g. an estimation of methane emissions from European wetlands (Saarnio *et al.*, in press).

5.3 Occurrence of peat in the European Union

Most peatlands in the EU are found in the Nordic Countries that are located mostly in the boreal zone, but also in the temperate zone in Ireland and UK, especially in highlands with maritime climate, and in the Baltic countries and Belarus in northern Central Europe with increasing continentality in climate towards the east. The peatlands become sparse from Central towards Southern Europe, where the peatlands are largely confined to river valleys and in geomorphologically suitable depressions in the alpine areas (Lappalainen, 1996).

Peatlands cover approx. 3% of the global land surface (Strak, 2008). Quite disproportional to the area covered they represent a major terrestrial C store, estimated at 20-30% of the global soil stock (Moore, 2002; Turunen *et al.*, 2002). In Europe over 85% of the peatlands are located within European Russia, Fennoscandia and the British Isles (Byrne *et al.*, 2004). They represent approximately 20% of the total European carbon stock.

Where the peatlands are most abundant, they have also been drained for agriculture, forestry, or the extraction of peat. For their economical importance, peat reserves have been inventoried in many countries, because of their usability as energy source or growing media among other uses. Lappalainen (1996) described the global distribution of peatlands and peat resources, and Bragg and Lindsay (2003) those of Central Europe. Peat soils are predominantly found in northern ecosystems, especially abundant in continental boreal and sub-arctic regions (Figure 8). In Europe, the former extent of pristine mires may have been 617 000 km². The area was reduced by approximately 50 % (Joosten and Clarke, 2002), predominantly due to agriculture, forestry, urbanization, inundation or erosion. Due to land use changes, some of the original peatland area has been lost entirely, but also difficulties in recognizing the origin of e.g. forested abandoned peatlands (Turunen 2008) may also be a reason for peatland loss.

Table 5 summarizes the information obtained from the areas of peatlands drained and undrained in the EU Member States and Candidate Countries. The occurrence of peat in the EU is based on recently published information whenever available, but using Lappalainen (1996) and the estimates of Montanarella *et al.* (2006) based on the Map of OC Topsoils in cases when other information is not available.

The distribution of peatlands in Europe was also assessed by Byrne *et al.* (2004), who reported a total actual peat area in current EU Member States and Candidate Countries of ca. 339,000 km², including about 58,000 km² drained peatlands and 234,000 km² undrained mires. Their estimate of total C store in peat was 17 Pg, around 20-25%. Their total peatland area is somewhat larger than the 318,000 km² calculated from Table 5.

The difference between 339,000 and 318,000 km² is explained by correcting the drained peatland area for an overestimation of 35,000 km² in Finland. Other differences with the current data in both directions are smaller and harder to track.

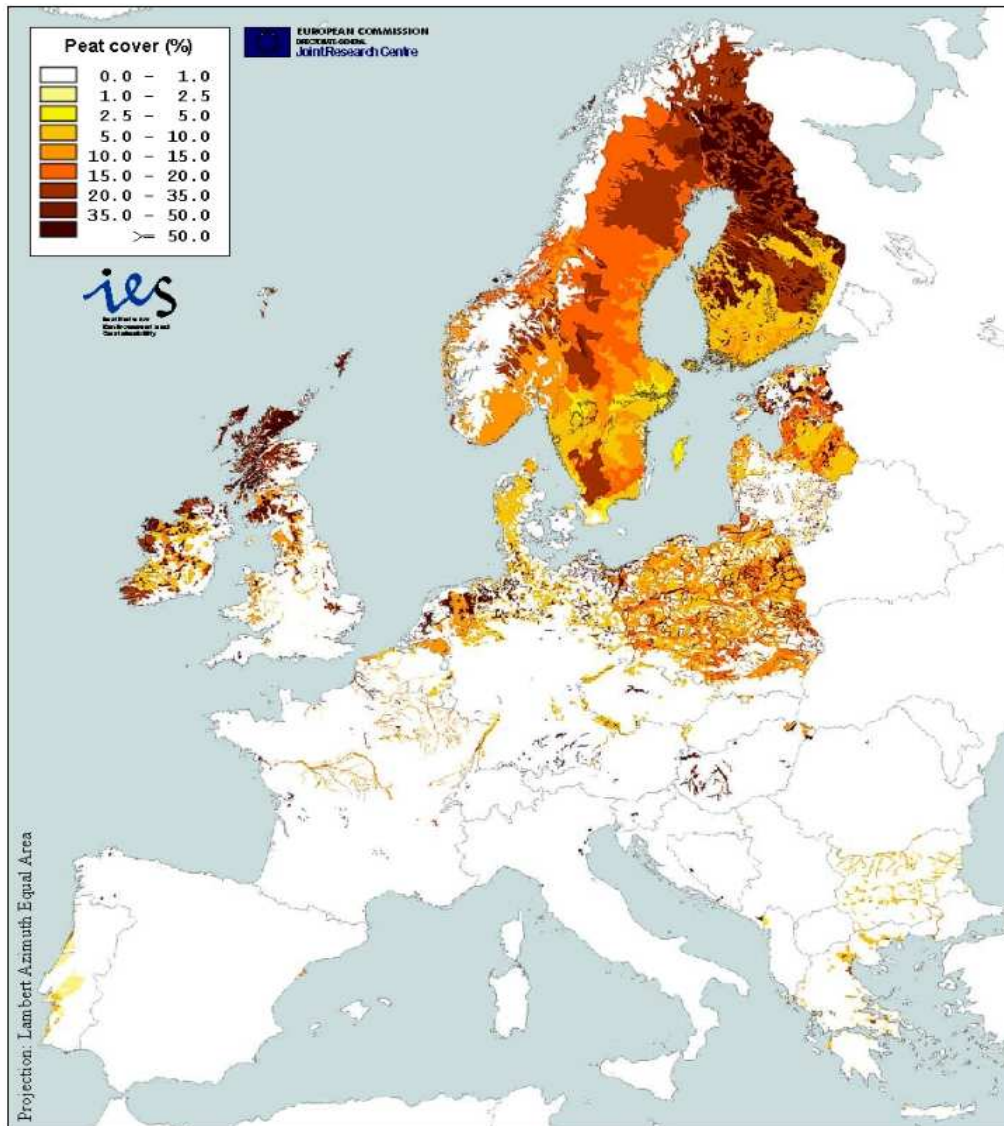


Figure 8 Map of peat cover in Europe (JRC).

Table 5 Occurrence of peat covered land area (km²) in the European Union Member States and Candidate Countries.

State	In forestry	In agriculture or pasture	In peat extraction	Drained total	Undrained	Peat layer lost	Under reservoir	Current area of peatlands	Original peatland area	Area of peat topped soils (European Soil Database, OC>25%)	References
	km ²	km ²	km ²	km ²	km ²	km ²	km ²	km ²	km ²	km ²	
<i>Member states of the EU</i>											
Austria		0						220		134	1
Belgium		252		252				180		95	1
Bulgaria		1863						30		0	
Cyprus				0				0		0	1
Czech Republic				123	147			270		251	1
Denmark		184	12					1420		66	1, 2
Estonia	2580	840	180	7091	3250	150		10091		6889	3
Finland	57000	2930	570	62150	31370	12070	60	91930	104000	98535	4, 15
France		0		645				555	1200	775	1
Germany	200	14133	325	14658				14205		6279	1
Greece		67						101		0	1
Hungary		16750	2	16752				1000		401	5
Ireland	3018	1190	1000	10648	336	5440		11757	11776	12725	1
Italy		90						1200		1	2
Latvia		14571		3555	3167			6722		3382	1, 2
Lithuania	5900		25	7192	750			7942		1489	1
Luxembourg				0				0		0	1
Malta				0				0		0	
Netherlands		2310		2310	290			2790		2022	2
Poland	1270	7600	25	10529	2019			12050	12548	4677	1
Portugal		0						200		0	1
Romania								70		39	2
Slovakia				234	26			260		1	6
Slovenia								101		0	
Spain		0						385		184	7, 8
Sweden	15000	2400	80	17483	86000	3		104000	104000	90785	10, 11, 15
United Kingdom	2200	392	54	2646				50424		44411	9
<i>Candidate countries</i>											
Croatia			?					26		0	
Former Yugoslavian Republic of Macedonia			?					?		0	
Turkey			?		?			240			13
Total	87168	65572	2273	172676	127355	17663	60	318169	233524	273141	

Key to references: 1 Lappalainen (1996), 2 Bragg (2003), 3 Turunen (2008), 4 Irish Peatland Conservation Council, 5 BOGFOR Research Project, Ireland, 6 Repe (2004), 7 NIR/Sweden (2007), 8 Vasander et al. 2003, 9 Burton (1996), 10 Shrier (1996), 11 Milne & Brown (1997), 12 Bašić, F.; European Soil Bureau-Research Report 9, 13 Denzig et al. (2008), 14 NIR/EU (2005), 15 Paappanen & Leinonen (2005)

Figure 9 includes a comparison of estimates for the area of peatland derived by Montanarella *et al.* (2006) from the map of organic carbon in topsoils of Europe with the available estimates of peatland in different countries. Although the basis for both estimates is quite different, the estimates correlate well and seem to be going in the right direction. Especially small areas of peat are generally underestimated in national estimates and are absent in the European Databases. In time, this may improve as accuracy and spatial coverage in databases will improve.

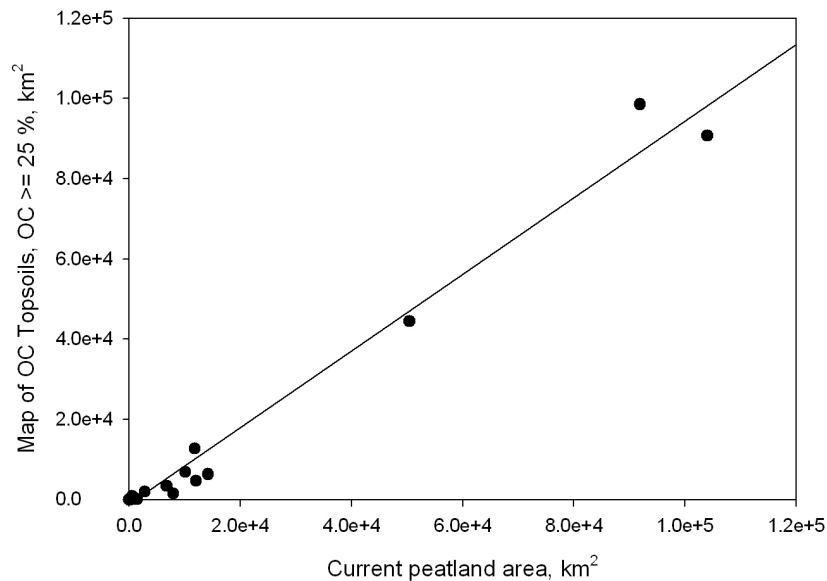


Figure 9 Comparison of areas collected from literature (Current peatland area) and areas derived from the Map of OC in Topsoils of Europe (Montanarella *et al.*, 2006). The points represent those countries for which both estimates could be derived.

A serious problem is in the lack of availability or the ambiguity of information on the various land uses for peat soils across Europe. In some countries such as the Netherlands, the area of peat in agriculture is well known, but less documented in other countries. Commonly the estimate of total drained peatland area cannot be summed from estimates of different land use types in the same country e.g. because the estimates do not always originate from a single common assessment. Similarly, the sum of a single land use type area may not be reliable for the EU for the same reason.

5.3.1 Peat extraction

Peat harvesting for energy production and for substrate in horticulture affects only a relatively small part of the total European peatland area, yet it represents the most severe potential land-use impact on the C balance of peat soils. Extraction for energy has declined since the mid 19th century, but remains significant in Ireland, Finland, Sweden, the Baltic States and Russia (Byrne *et al.*, 2004). Extraction for horticulture has led to the loss of a large part of the lowland bog area in the United Kingdom (Moore, 2002).

In the period 1990 - 2005, the total peat extraction in all European countries has been rather stable and amounts to 13.5 Mt per year (Figure 10). The large annual variance from 7 to 18 Mt likely follows annual differences in weather conditions for peat extraction in those countries with the biggest production volumes, i.e. Finland and Ireland. A clear example is the wet summer of 2004 with low volume of extraction of peat. There is no clear increasing or decreasing trend in the total amount of peat extracted.

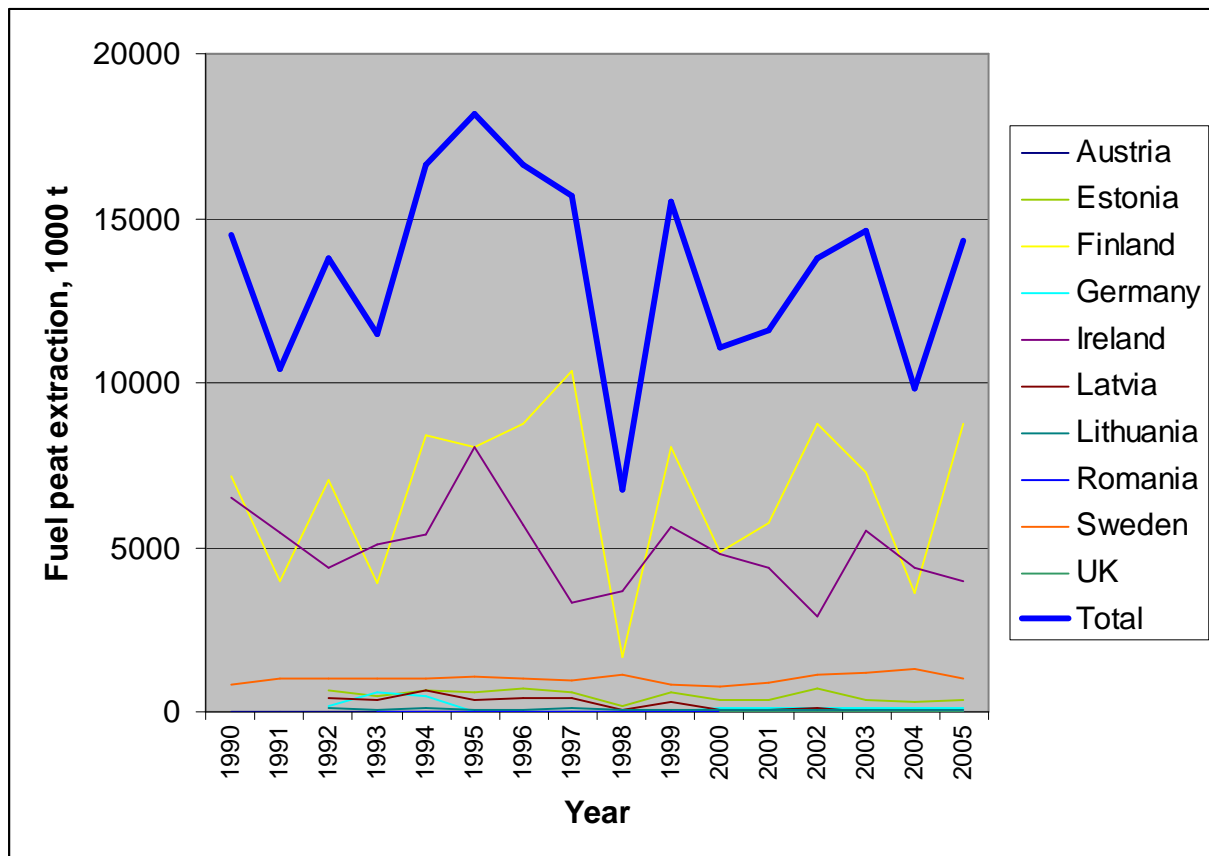


Figure 10 Fuel peat extraction (1000 ton) in 1990-2005 in selected EU countries according to statistics collected by the UN.

Paappanen & Leinonen (2005) published a report on the socio-economic and energy impact of peat uses for energy purposes in the European Union. That report outlines the prospects of fuel peat production in the most significant countries currently active in peat extraction, Finland, Ireland, Sweden, Estonia, Latvia, and Lithuania. In these countries upto 5% of the energy production comes from peat combustion (Annex 8).

5.3.2 Peat soils used in agriculture

Around 16% of the overall European peatland area is used in agriculture (cropland and grassland), including the vast majority of peats in continental Western Europe (Byrne *et al.*, 2004) (Annex 8). In the Netherlands, Germany and Poland this is even 70 – 85 % of the peatland area. The lack of availability of reliable data on the area of peat soils in agricultural use and whether it is grassland or arable land is to primarily caused by land use changes and degradation of peat soils that have turned into mineral soils following oxidation of the peat.

The subsidence of peatsoils in agricultural use due to shrinkage, consolidation and oxidation is estimated at 1 – 3 cm per year (Kasimir-Klemedtsson *et al.*, 1997). About 70% of the subsidence is caused by oxidation (Eggelsman, 1976; Schothorst, 1982) and subsidence continues until all peat is oxidized. In the Netherlands the loss of area of peat soils in agricultural use was about 20% of the total peatland area in the last 30 – 40 years. This is a loss as soil unit. As soon as the peat layer becomes thinner than 40 cm the soil is by definition not any longer a peat soil. At that moment the peatlayer is still almost 40 cm thick, so there will be still a considerable emission of CO₂ caused by oxidation of the remaining organic layer. After about 20 – 30 years of agricultural use this remaining layer will be so thin that it will be ploughed through the mineral soil underneath the organic layer. The lost peatland areas were covered with rather shallow peat layers. Moreover a large part of these shallow peat soils were in arabale use with deep ditchwaterlevels causing high oxidation rates and so speeding up the vanishing of the peat soil.

The major part of the remaining peatsoils in the Netherlands now are deep peatsoils with depths of 3 meters up to 14 meters, so the rate of decrease of peatland area will slow down in the near future. Also in other countries a large amount of the original peatland area has evidently vanished during agricultural use. Burton and Hodgson (1987) estimate that two thirds of the peat areas of East Anglia Fenland of approximately 240 km² in 1985, will be lost by the year 2050 as 56% of the peat deposit is less than 1 m thick. Also land use change is a major cause of a decreasing area of peat soils in agricultural use. Berglund and Berglund (2009) reported that the peatland area in agricultural use in Sweden was 6,300 km² in 1946, 3,400 km² in 1961 and 2,500 km² in 2003, of which 630 km² in crop production.

In the UK, the Netherlands, Germany and Poland ever more peatlands in agricultural use are reconverted into wet ‘natural’ peatland. Since the 1990s peatland restoration measures covering over 20,000 ha have been implemented in the German federal state of Mecklenburg-Vorpommern (Theuerkauf *et al.*, 2006) and rewetting is planned of another 70,000 ha of degraded peatland. In Belarus rewetting of 42,000 ha degraded peatland is financed and rewetting of an additional 260,000 ha is planned (Joosten, 2007).

5.4 Emissions of greenhouse gases from drained peatland

Utilization of peatlands for forestry, agriculture or peat extraction involves drainage by ditching. It is clear that peatland drainage causes an increase of oxidation in the drained peat layer, resulting in increased emissions of CO₂. In nutrient rich peatlands also the emissions of N₂O may rise, while as compared with undrained mires, the emissions of CH₄ are lowered.

The total areas of peatlands drained for forestry, cultivation, or peat extraction are used to estimate the emissions of CO₂, CH₄, and N₂O due to peatland drainage in the EU and the Candidate

Countries. It is worth to note that the (anthropogenic) emissions from managed peatlands are currently reported to the UNFCCC (United Nations Framework Convention on Climate Change) by those countries where peatland management has importance. According to the guidance given by the IPCC for the Land Use, Land Use Change and Forestry (LULUCF) Sector, no emissions from undrained mires are reported. Thereby the binding of atmospheric CO₂ in the formation of peat, or the emissions of CH₄ from the anoxic peat layers are not accounted for in the reporting.

Table 6 Emissions of CO₂, CH₄, and N₂O (in ton km⁻² a⁻¹) estimated according to the drained peatland area. Typical annual emissions for each land use type are derived from the IPCC Emission Factor Database (www.ipcc-nggip.iges.or.jp/EFDB) for boreal and temperate peatlands, denoted by “*”, and from Alm *et al.* (2007); all unmarked emission factors. CO₂-equivalents are calculated using GWP (100 yr) conversion factors 21 for CH₄, and 296 for N₂O, respectively.

Drained for	Area km ²	CO ₂ , t km ⁻² a ⁻¹		CH ₄ , t km ⁻² a ⁻¹		N ₂ O, t km ⁻² a ⁻¹		CO ₂ -eq., Mt a ⁻¹		
		Min	Max	Min	Max	Min	Max	Min	Max	Aver
Forestry	87,168	719	2035	-0.6	3.5	0*	0.283*	62	191	127
Cultivation	65,572	290	4033	0*	0*	0.283*	0.283*	25	270	147
Peat extraction	2,273	73*	4033*	0.3	9	0	0.06	0	1	1

The largest central estimate (average of min and max estimates) of CO₂-equivalent emissions are due to peatland cultivation (Table 6). The uncertainty limits illustrated by the Min-Max ranges are huge. One important reason for the large ranges in the estimates follow from the unknown distribution of nutrient poor and nutrient rich peatlands in the various land use categories. In peatlands used for agriculture, the variance in emissions follows from differences in management, ploughing increases the oxidation and CO₂ emissions, respectively. It is clear that peatlands reclaimed for agriculture are the nutrient rich ones. On the other hand, forest drainage may have occurred also on nutrient poor peatlands. The overall emissions from peat extraction areas are low due to their relatively small total area.

The emission listed in Table 6 originate from the decomposition of the peat layer, and do not describe the complete balance of the gas exchange. In forested peatlands the tree stand binds atmospheric CO₂ producing an increasing standing crop and an annual litterfall. Such carbon inputs to the soil may exceed the soil losses in CO₂ emission from the decomposition of the peat layer. For example, the soil in peatland forests in Finland loses 6 Mt C, while the annual gain by the tree stand currently corresponds to a removal of approximately 14 Mt C from the atmosphere (NIR 2007, Finland). A similar ratio may be applicable also elsewhere in the boreal zone. However, after harvesting the tree stand, the ratio at least temporarily changes in favor of net carbon loss, stressing the importance of sustainable forest management practices in peatland forests. Furthermore, it should be underlined that CO₂ emission due to the decomposition of the peat layer and CO₂ removal because of tree growth cannot be compared in climate change terms. This originates from the fact that the carbon emitted because of peat decomposition is a *net* addition to the atmosphere, while the carbon absorbed by the trees will eventually go back to the atmosphere when the forest ceases to exist.

5.4.1 Emissions from peat soils used in agriculture

The oxidation of peat soils in agricultural use results in large CO₂ emissions (Figure 11). The data in Figure 11 is derived from a literature study by Couwenberg *et al.*, (2008). Good direct measurements of CO₂ emissions are difficult, because not only the CO₂ emission of the peat is measured, but also the oxidation of fresh organic material and respiration and sequestration of CO₂ of the crop is measured. Further on during the day, season and depending on meteorological input CO₂ emissions change. Therefore data collection was restricted to peatlands from temperate Europe and only data on yearly emissions were used – based either on year-round measurements or on sound model-extrapolations. Only net CO₂ balances (with net ecosystem exchange of CO₂ or net ecosystem productivity) from reliable models using both daylight (uptake of CO₂) and night fluxes (respiration of CO₂) as input were used. Also net-emission estimates based on observations of peat subsidence were included by Couwenberg *et al.* (2008). The data of Van den Akker (unpublished data) is calculated from subsidence.

The calculation of CO₂ emissions on the basis of the annual long-term subsidence rate of peat is very robust, because a mass balance based on the subsidence is in a long-term perspective and primarily caused by the net loss of organic matter by oxidation as CO₂. Moreover subsidence is usually measured over many years and sometimes decades; as a result, seasonal and yearly variations in CO₂ emissions are averaged over a long period. The main problem here is to determine which part of subsidence of drained peat soils is caused by oxidation of the organic matter of the peat soil and which part is caused by consolidation of the peat layer and permanent shrinkage of the upper part of the peat soil above groundwater level.

Armentano and Menges (1986) estimate the fraction Fr of subsidence due to oxidation of organic matter compared to the total subsidence to vary between 0.33 to 0.67. This fraction will vary strongly over time as subsidence rates directly after drainage are very high due to shrinkage and consolidation. Eggelsman (1976) gives a value of 0.7 for the fraction Fr and this value was also used by Kasimir-Klemedtsson *et al.* (1997) in their calculation of CO₂ emissions from subsidence. They used in their calculation the bulk density and the carbon content of the surface peat layers (upper 20 cm). A further problem might be that bulk density and carbon content of the upper 20 cm can vary considerably. Van den Akker *et al.* (2008) takes another approach to avoid problems with estimating Fr and bulk densities and suggested to assume that over time there will be an equilibrium in carbon content in the layer just above the deepest groundwater level. The CO₂ emission is then calculated based on the amount of carbon in a layer at a depth just below the deepest groundwater level and a thickness of the yearly subsidence. The results on the basis of the approach taken by Van den Akker *et al.* (2008) are presented in Figure 11.

These calculated emissions are in good agreement with Höper (2007) for German peat soils in agricultural use. Höper (2007) also found that bog peatsoils have more or less the same GHG emissions as fen peatsoils at high ditchwater levels (40 – 60 cm below soil surface) and relatively much lower GHG emissions at low ditchwater levels (around 90 cm and deeper below soil surface).

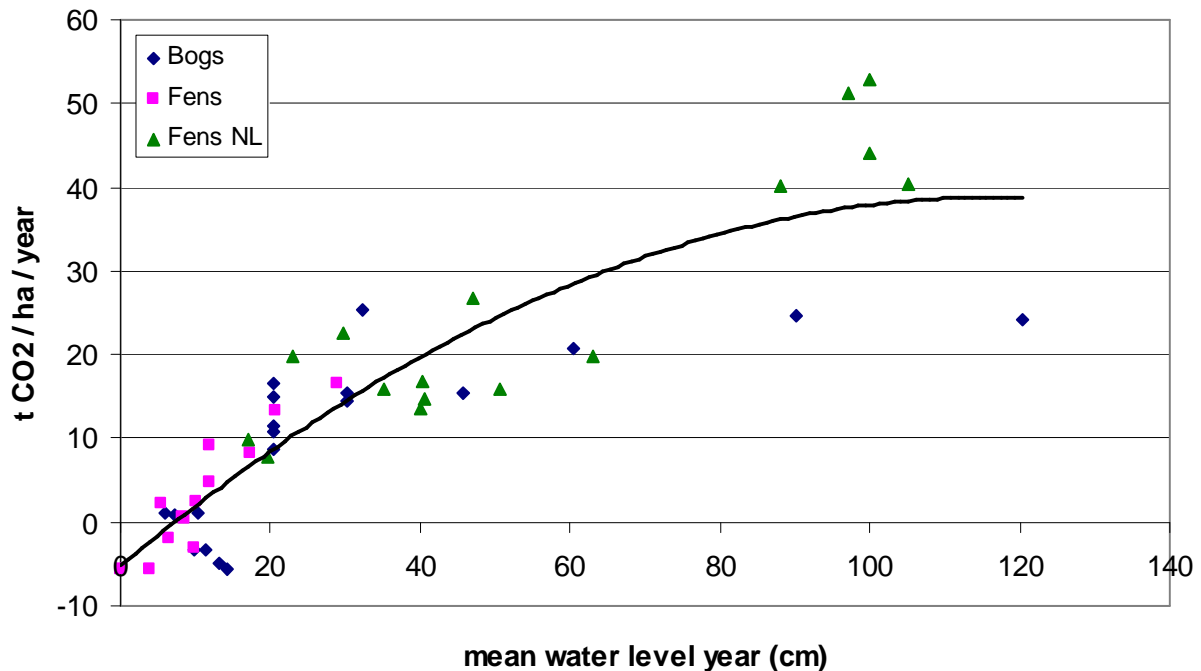


Figure 11 CO₂ emission of peat soils. Agricultural peat soils have at least a mean ditchwater level of 20 cm minus soil surface. Data collected by Couwenberg *et al.* (2008) are based on direct measurements of CO₂ emissions and data by Van den Akker (Fens NL, unpublished data) are based on CO₂ emissions calculated from measured mean annual subsidence.

The results in Figure 11 and findings in a very recent review by Oleszczuk *et al.* (2008) are used to calculate the CO₂ emission of European peat soils in agricultural use (Table 7). In Table 7 the emission of CH₄ is not taken into account, because these are very low in peatsoils in agricultural use (Höper, 2007; Couwenberg *et al.*, 2008). Emissions of N₂O are difficult to measure and have a very high temporal variability and results found in literature are very diverse (Couwenberg *et al.*, 2008). Therefore we simplified the determination of N₂O emission to an estimation that 2 % of the mineralized N will be converted into N₂O (Mosier *et al.*, 1998).

In Table 7, not all EU countries are represented, however, only countries with a very low area of peatsoil in agricultural use were skipped. So we can estimate the total emissions of GHG from peatsoils in agricultural use in the EU as 98.51 Mt CO₂ equivalent per year. This is about 34 Mt CO₂ equivalent per year higher than estimated by Byrne *et al.* (2004). The main reason for the difference is that Byrne *et al.* (2004) use lower GHG emissions per ha, especially for grassland on bog peatland and cropland on fen peatland. Looking at the data used by Byrne *et al.* (2004) we conclude that unrealistic low values were included in their calculation of average emissions. Therefore we think that the emissions presented by Byrne *et al.* (2004) are too conservative and consider the emissions in Table 7 to be more realistic.

Table 7 Emissions of GHG of peatsoils in agricultural use. Calculation are based on: grassland emissions 20 tonne CO₂ ha⁻¹ a⁻¹; cropland emissions 40 tonne CO₂ ha⁻¹ a⁻¹ (see Fig. 5 and Oleszczuk *et al.*, 2008); C/N ratio = 20 (assuming that the major part of agricultural peat soils are fen peats); 1.25 % of mineralized N converted into N₂O (Mosier *et al.*, 1998). Cropland area and grassland area are based on Byrne *et al.*, 2004.

Country	Agricultural area km ²	Crop area km ²	Grass area km ²	CO ₂ - C Mt / a	CO ₂ Mt / a	N ₂ O CO ₂ eq Mt / a	Total CO ₂ eq Mt / a
<i>Member states of the EU</i>							
Belgium	252	25	227	0.15	0.55	0.05	0.60
Denmark	184	0	184	0.10	0.37	0.03	0.40
Estonia	840	0	840	0.46	1.68	0.14	1.82
Finland	2930	0	2930	1.60	5.86	0.49	6.35
Germany	14133	4947	9186	10.41	38.16	3.18	41.33
Ireland	2136 ^a	896	1240	1.65	6.06	0.50	6.57
Italy	90	90	0	0.10	0.36	0.03	0.39
Latvia	1000 ^a	1000	0	1.09	4.00	0.33	4.33
Lithuania	1900 ^b	1357	543	1.78	6.51	0.54	7.06
Netherlands	2050 ^c	75	1975	1.16	4.25	0.35	4.60
Poland	7600	55	7545	4.18	15.31	1.27	16.58
Sweden	2500 ^d	630	1870	1.71	6.26	0.52	6.78
UK	392	392	0	0.43	1.57	0.13	1.70
Total EU	36007	9467	26540	24.80	90.95	7.57	98.51
<i>Other European countries</i>							
Iceland	1300 ^a	0	1300	0.71	2.60	0.22	2.82
Norway	6100 ^a	4200	1900	5.62	20.60	1.71	22.31
Russia (European)	26400 ^a	2640	23760	15.84	58.08	4.83	62.91
Belarus	9630 ^a	963	8667	5.78	21.19	1.76	22.95
Ukraine	5000 ^a	5000	0	5.45	20.00	1.66	21.66

^a based on Byrne *et al.*, 2004; ^b based on Oleszczuk *et al.*, 2008; ^c based on Kuikman *et al.*, 2005;

^d based on Berglund and Berglund, 2009.

5.5 Effect of land use and soil management on carbon stocks of peat soils

The vast quantity of C stored within European peatlands clearly necessitates their effective protection. Intact European mires are generally still functioning as sinks for C; Byrne *et al.* (2004) collated literature estimates of long-term C accumulation in Finland, Russia and Sweden, most of which were in the range 15-25 g C m⁻² a⁻¹. UK peatland accumulation rates have been estimated at 20-50 g C m⁻² a⁻¹ (Cannell *et al.*, 1999).

The largest per-area losses of SOC from peatlands occur where the C stocks are largest and are caused by either drainage, cultivation or liming. The potential for loss of SOC following land use

change on high organic peat soils is very large too. Emissions from drained organic soils can be reduced to some extent by practices such as avoiding row crops and tubers, avoiding deep ploughing, and maintaining a shallower water table. But the most important mitigation practice is avoiding the drainage of these soils in the first place or re-establishing a high water table (Freibauer *et al.*, 2004). We will discuss the impact of soil management on carbon stocks in peat soils in more detail below.

Draining peat

Peats accumulate SOC due to the suppression of decomposition processes, primarily due to a lack of oxygen in the waterlogged conditions. However, up to half of all European peatlands have been subject to artificial drainage. A major part of this has been for forestry and specific issues related to peat afforestation are discussed below. Drainage is also undertaken for peat extraction, to convert lowland peats to intensive agriculture, and with the intention of improving grazing quality in upland blanket bogs. In the United Kingdom, upland drainage took place extensively from the 1960s and 70s (Holden *et al.*, 2004), often generating very dense artificial drainage networks. In general, while increased peat aeration resulting from drainage may reduce CH₄ emissions (e.g. Waddington and Price, 2000), most studies show that rates of CO₂ production are greatly increased (Moore and Dalva, 1993; Silvola *et al.*, 1996; Waddington and Price, 2000). However, the consistency of this response, particularly over the longer term, has been questioned (Laiho, 2006). Increased CH₄ emissions from drainage ditches may offset reductions within the drained peat mass (e.g. Sundh *et al.*, 2000). Increased CO₂ losses are likely to be most pronounced in naturally wetter peats, where large pools of labile C have accumulated near the surface (Laiho, 2006). For a range of drained ombrotrophic bogs across Europe, Byrne *et al.* (2004) estimate that the average net GHG flux due to drainage is 125 g CO₂-C eq m⁻² a⁻¹, versus 19 g CO₂-C eq m⁻² a⁻¹ in natural bogs.

In some peatland areas, such as the blanket bogs of Northern England, artificial drainage has coincided with extensive problems of gully development, attributed to a combination of overgrazing, and the loss of *sphagnum* with decay-resistant litter (Lee *et al.*, 1993). Drainage itself may contribute to gully development, and has also been shown to cause soil structural change, notably development of soil pipes (Holden, 2005). Rapid water movement through drains, gullies and pipes, together with the exposure of bare peat surfaces, greatly increase rates of particulate organic carbon (POC) loss. Holden (2006) estimates that POC losses from a drained peat slope could halve the long-term net C sink of an undrained blanket peat. Some studies have also shown large increases in dissolved organic carbon (DOC) leaching from drained catchments (e.g. Mitchell and McDonald, 1995; Wallage *et al.*, 2006) further reducing the sink function of peat soils.

Restoration of peat

Peatland restoration, though blocking of drainage channels, is becoming increasingly widespread, with aims including restoration of ecological quality, improved drinking water quality, and restoration of the function of peatlands as a C sink. Drain (and gully) blocking will clearly reduce POC loss, and may also reduce DOC loss (Wallage *et al.*, 2006). This will also increase C sequestration rates, although Waddington and Price (2000) noted that restored peatlands did not, at least in the short term, sequester C at the same rate as pristine systems. Based on a range of studies, Byrne *et al.* (2004) provided an estimate of net GHG emission from restored bogs of 74 g CO₂-C eq m⁻² a⁻¹, intermediate between values for pristine and drained systems. Their data suggest that restoration of a drained bog will decrease CO₂ emissions by 48 g CO₂-C m⁻² a⁻¹, and also decrease the CH₄ efflux by 10.5 g CO₂-C eq m⁻² a⁻¹. The latter value, based two studies, must be considered

doubtful, because raising water tables following restoration can be expected to increase CH₄ emissions. Waddington and Day (2007) found that restoration raised CH₄ effluxes by 88 g CO₂-C eq m⁻² a⁻¹ (a factor of 4.6) relative to unrestored cutover peatland. Most of these increased emissions derived from flooded ponds and the ditches themselves. In general, the net effect of peatland restoration on GHG balance will be highly dependent on the nature of the restoration, in particular the location of the post-restoration water table, and extent of flooded areas such as old drainage ditches.

Previously cultivated organic soils that have been restored emit less CO₂ from oxidation of peat and this emission reduction is an order of magnitude higher than the increased emissions of CH₄ (in CO₂ equivalents) from restored peat soils. However, there is a significant range and variability in these estimates. Many factors impact the balance between CO₂ emission reduction and CH₄ increase after restoration and include the type of organic soil, duration of maintaining a high water table during the year, e.g. year-round *versus* seasonal (Kasimir-Klemedtsson *et al.*, 1997; Freibauer *et al.*, 2004; Smith *et al.*, 2008), and the level of CO₂ emissions from the cultivated organic soil which may vary greatly (Nykänen *et al.*, 1995; Maljanen *et al.*, 2001, 2004; Lohila *et al.*, 2004).

Restoration of a functioning peatland ecosystem, with a high water table and establishment of peat-forming vegetation, provides greater potential for C sequestration. In two restored peatlands in the Jura Mountains, Bertoluzzi *et al.* (2006) measured net C accumulation of 67-183 g C m⁻² a⁻¹, although other studies (Tuittila *et al.*, 1999; Yli-Petäys *et al.*, 2007) reported smaller net CO₂ sinks, and Yli Petäys *et al.* (2007) suggest that C accumulation may peak during initial regeneration, and subsequently decline. Yli Petäys *et al.* (2007) and Waddington and Day (2007) both report significant elevated CH₄ effluxes from restored peatlands.

Peat extraction

Carbon losses from peat extraction are associated directly with the removal (and ultimate burning or mineralisation) of peat material; in Finland, peat combustion alone is estimated to generate 15% of the country's net GHG emissions (Lapveteläinen *et al.*, 2007). Additional C losses are associated with persistently elevated decomposition of the residual, unvegetated peat (Waddington *et al.*, 2002). Rates of C loss from cutaway peatlands have been estimated at around 250 g C m⁻² a⁻¹ for Finland (Alm *et al.*, 2007), 60-280 g C m⁻² a⁻¹ for Sweden (Sundh *et al.*, 2000), and 19-32 g C m⁻² a⁻¹ at a mountain bog in the Jura Mountains, France (Bortoluzzi *et al.*, 2006). Conversion of cutaway peatlands to forestry may reduce rates of CO₂ loss, but accumulation of forest carbon may not offset continued high rates of peat decomposition (Alm *et al.*, 2007).

Grazing of peatland

Although peatlands are in general less intensively grazed, they may be highly sensitive to grazing impacts (Haigh, 2006). C accumulation in peatlands is dependent on the presence of plant species that generate decay-resistant litter, such as *sphagnum* mosses (Belyea, 1996), which may be affected by grazing intensity. Ward *et al.* (2007) found moderate reductions in dwarf shrub and moss biomass with grazing, versus long-term ungrazed controls. Smith *et al.* (2003) found that complete cessation of grazing on ombrotrophic mires resulted in growth of dwarf shrubs and hypnoid mosses at the expense of peat-forming *sphagnum*. This result suggests that grazing, at a low intensity, may be beneficial to maintaining the C sink, at least in areas of dryer/drained peatlands. However, where effects on C cycling have been measured, results are unclear. Garnett *et al.* (2000) found no significant difference in long-term C accumulation on grazed and ungrazed peatland in the 50 year Hard Hills exclusion experiment in Northern England. For the same experiment, Ward *et al.* (2007)

measured a 22% reduction in above-ground biomass with grazing, a stimulation of both photosynthesis and respiration, a large increase in CH₄ efflux and small increase in DOC loss, non-significantly lower carbon stocks in the litter layer and upper mineral soil layers, but no measurable difference in soil C stocks to a depth of 1m. Worrall *et al.* (2007) found no effect of grazing on DOC at the same site.

Effects of trampling associated with grazing are more severe on peats, due to the low bulk density and depth of the organic layer; Overgrazing has been a major cause of blanket peat erosion in Northern England (Haigh, 2006; Holden *et al.*, 2007), with erosion triggered by relatively low stocking densities (0.55 sheep ha⁻¹, Rawes and Hobbs, 1979). Erosive effects may be concentrated in areas of livestock movement or shelter, with compaction causing an increase in overland flow and potentially triggering or accelerating gully development.

Burning of peatland

Wildfires, in which the peat itself is burnt, have a major detrimental impact on peat carbon stocks; analyses of peat cores suggest that long-term mire accumulation rates may be halved by frequent natural burns (Kuhry 1994; Pitkänen *et al.*, 1999). Increased droughts due to climate change could increase burn frequency, exacerbated in populated areas by accidental or malicious fire-starting. Dikici and Yilmaz (2005) reported very large C losses from a Turkish peatland, drained in the 1950s, which subsequently experienced repeated catastrophic burns. Management burning, on the other hand, is intended to burn only above-ground vegetation. It has historically been less extensive on peats than on dryer heathlands, due to their relative inaccessibility, and because *sphagnum* accumulation in an aggrading bog encourages continuous heather growth without the need for management (Adamson and Kahl, 2003). However, Yallop *et al.* (2006) report significant increases in the area of English blanket bog subjected to rotational burning since 1995. At the Hard Hills experiment, 30 years of managed burning on a 10 year cycle significantly reduced peat C stocks (Garnett *et al.*, 2000). Ward *et al.* (2007) estimated C loss rates due to burning at this site at 17 g m⁻² a⁻¹ from the peat surface, and 9 g m⁻² a⁻¹ from above-ground biomass loss. Burning led to large reductions in moss and shrub biomass, with an increase in grasses, increased gross photosynthesis and respiration rates, and slightly decreased CH₄ efflux. Ward *et al.* (2007) found no significant change in DOC, whilst Worrall *et al.* (2007) observed a significant decrease. Holden *et al.* (2007) note that there has been pressure in the United Kingdom for a ban on burning on blanket bogs, and Flynn and Smith (2006) suggest that heather cover in cool, wet European heathlands may be maintained through low-intensity grazing rather than burning, with likely benefits for C stocks.

5.5.1 Peat soils converted to forests

Around 15 million hectares of boreal and temperate peatlands have been drained; 90% of this is in Fennoscandia and Russia (Moore, 2002). 20% of the European peatland stock has been drained for forestry, including over half of all Finnish peatlands (Byrne *et al.*, 2004). In oceanic bogs without natural tree cover, particularly in the British Isles, large areas have been planted with exotic conifer species during the last century.

Compared to afforestation of organo-mineral soils, levels of soil disturbance associated with afforestation of peats are high. This is due to both the fragility of the peat and the need for intensive drainage to provide aerobic conditions for tree growth. Fertiliser may also be applied too and transpiration by the growing forest further lowers water tables and both will accelerate

decomposition of peat. As noted above, most studies show an increase in peat decomposition rates following drainage, which leads to a loss of existing peat C due to forest planting (e.g. Martikainen *et al.*, 1995; Brake *et al.*, 1999; Hargreaves *et al.*, 2003; Reynolds, 2007; Minkkinen *et al.*, 2007).

In naturally forested peatlands, lowering of the water table through drainage increases timber production. CH₄ emissions are likely to decrease from the drained peat due to increasing soil aeration but may increase within the ditches themselves. Losses of organic matter as POC and DOC may also increase during periods of forestry-related disturbance, particularly following felling (Niemenen, 2004; Reynolds, 2007). As for afforestation of organo-mineral soils, however, some studies suggest that growth of biomass and litter accumulation will outweigh soil C losses over a forest rotation (e.g. Harrison *et al.*, 1997; Hargreaves *et al.*, 2003).

Byrne *et al.* (2004) estimate that, at the European scale, forested peats are net GHG sinks, although the authors caution that this conclusion assumes a mild degree of drainage in which CH₄ emissions are strongly reduced and peat formation still takes place. Hargreaves *et al.* (2003) report initial C losses from newly drained peatland in the range 200-400 g C m⁻² a⁻¹. They further report that the system became a net sink after 4-8 years of around 300 g C m⁻² a⁻¹ due to tree growth. This offsets a continuing yet lower peat decomposition loss of ~100 g C m⁻² a⁻¹ after 4-8 years. Minkkinen *et al.* (2007) measured higher peat CO₂ respiration rates of 250-500 g C m⁻² a⁻¹ in Finnish and Estonian forested peatlands drained for forestry prior to 1960; this indicates that losses of C from peats are sustained in the long term. Furthermore, the C sink associated with tree growth will decrease as forests matures. Cannell *et al.* (1999) suggest that most peat C under plantation forest will eventually be lost.

We conclude that, given the importance of peatlands for global C stocks and the major uncertainties associated with replacing large, old and stable peat C stores with new, potentially more unstable storage in tree biomass, peatland afforestation cannot be considered an effective means of sequestering C.

6 Effect of land use and soil management on soil carbon

6.1 Introduction

The previous chapters dealt with the amounts of carbon in the soil pool and how that pool is affected by climate change. This chapter discusses the effect of land use and soil management on the soil carbon pool.

The first section gives an overview of our current understanding of how land use and soil management affect changes in soil carbon. In order to assess the carbon sequestration potential of land use and soil management, in the second section we then compare, in a wider context, these soil carbon based strategies with mitigation efforts in other sectors. The final section of this chapter analyses the current status of reporting systems for carbon stock changes due to changes in land use and soil management.

6.2 Effect of land use on carbon sequestration

Trends in soil carbon stocks were discussed earlier in Chapter 3. It was shown that grasslands and forests generally sequester carbon, while croplands generally lose soil carbon, however with a large variation (Table 4). It is obvious that land use changes between these categories will likely affect the carbon balance of the soil (Table 8). SOC tends to be lost when converting grasslands, forest or other native ecosystems to croplands. SOC tends to increase when restoring grasslands, forests or native vegetation on former croplands, or by restoring organic soils to their native condition. Where the land is managed, best management practices that increase C inputs to the soil or reduce losses help to maintain or increase SOC levels. Management practices to increase SOC storage are discussed in the next section.

Table 8 Summary of effect of land use change on soil carbon.

From :	To:	Grassland	Forest	Cropland
Grassland			No effect	C loss
Forest		No effect		C loss
Cropland		C gain	C gain	

Most long term experiments on land use change show significant changes in SOC (e.g. Smith *et al.*, 1997; 2000; 2001a; 2002, 2008). Land use change significantly affects soil C stock (Guo and Gifford, 2002). This is likely to continue into the future; a recent modeling study examining the potential impacts of climate and land use change on SOC stocks in Europe, land use change was found to have a larger net effect on SOC storage than projected climate change (Smith *et al.*, 2005a).

Conversion from forest land or grassland to croplands caused significant loss of SOC, as was shown by Guo and Gifford (2002) in a meta-analysis of long term experiments. .

Conversion from forest to grassland did not result in SOC loss in all cases. Total ecosystem C (including above ground biomass), does however, decrease due to loss of the tree

biomass C. Similar results have been reported in Brazil, where total ecosystem C losses are large, but where soil C does not decrease (Veldkamp, 1994; Moraes *et al.*, 1995; Neill *et al.*, 1997; Smith *et al.*, 1999), though other studies have shown a loss of SOC (e.g. Allen, 1985; Mann, 1986; Detwiller and Hall, 1988). Even in the most favorable case, only about 10% of the total ecosystem C lost after deforestation (due to tree removal, burning etc.) can be recovered since much of the carbon lost is from tree biomass (Fearnside, 1997; Neill *et al.*, 1997; Smith *et al.*, 1999).

Conversion from cropland to forest generally increases soil carbon stocks. The afforestation of former agricultural land increases the C pool in the aboveground biomass and replenishes the soil C pool. Accumulation occurs until the soil reaches a new equilibrium between C input (litterfall, rhizodeposition) and C output (respiration, leaching). Although afforestation increases soil carbon, carbon loss may occur in a brief period following afforestation, when C loss by soil microbial respiration and C gain by litterfall are imbalanced. Tree planting leads to soil disturbance and can stimulate the mineralization of soil organic matter (SOM). These losses are not necessarily offset by the low C input by litterfall in a young plantation.

The previous land-use affects the C sequestration potential of afforested sites. Pasture soils already have high C stocks and high root densities in the upper part of the mineral soil, so **conversion from grassland to forest** has a small effect (Guo and Gifford, 2002; Römken *et al.*, 1999; Murty *et al.*, 2002). Chronosequence studies from New Zealand on former pastures, northern Spain on arable land, and northern England on peatland found that soils initially lost, but later gained C (Halliday *et al.*, 2003; Romanyá *et al.*, 2000; Zerva *et al.*, 2005).

Soil C responses at specific sites can vary from the generally expected trends as shown in Table 8, likely due to site-specific characteristics and the long time-lag of litter inputs effecting soil C pools. Additionally, whilst topsoils generally gain C when afforestation occurs, underlying mineral soils may lose C.

The most effective mechanism for reducing SOC loss globally would be to halt land conversion to agriculture, but with the population growing and diets changing in developing countries (Smith *et al.* 2007b; Smith and Trines 2007), more land is likely to be required for agriculture. To meet growing and changing food demands without encouraging land conversion to agriculture will require productivity on current agricultural land to be increased (Vlek *et al.* 2004). In addition to increasing agricultural productivity (without increasing soil disturbance), there are a number of other management practices that can be used to prevent SOC loss. These are discussed further below.

6.3 Effect of soil management on carbon sequestration

6.3.1 Agricultural systems

Smith *et al.* (Smith *et al.*, 2007a&b, 2008). have reviewed the greenhouse gas (GHG) mitigation potential of agricultural management practices and concluded that about 90% of total GHG mitigation potential in agriculture stems from soil C sequestration.

They also examined practices under the broad activities of cropland management, grazing land management, restoration of cultivated organic soils and restoration of degraded lands. The effects of these management practices on SOC are discussed in detail in the sections below. Table 9 presents a quantitative assessment of the potential effect on carbon sequestration of a selection of measures used in the PICCMAT project (<http://climatechangeintelligence.baastel.be/piccmat>).

Previous studies have reported the technical potential for a range of mitigation measures in Europe but more recent studies have shown that little of such potential has been realized (Smith *et al.*, 2005). New methods are now available to access economic potential for different sequestration measures which can be used to achieve better estimates of achievable mitigation potential. In Europe, the economic potentials for C sequestration are around 53-58, 80-87 and 93-102 Mt C (or 0.053-0.058, 0.080-0.087 and 0.093-0.102 Pg C) at C prices of 0-20, 0-50 and 0-100 USD t CO₂-equivalent⁻¹, respectively (from data in Smith *et al.*, 2008).

Table 9 Effect of a selection of mitigation measures on carbon sequestration in agriculture

	Potential implementation cost	Probability of implementation*	Global mitigation potential (Smith <i>et al.</i> , 2008) (tCO ₂ eq./ha/yr)
Catch crops	Low	High	0.29 - 0.88
Reduced tillage	Low	Medium (low in some areas)	0.15 - 0.70
Residue management	Low	High	0.15 - 0.70
Extensification	Medium	Low	1.69 - 3.04
Fertiliser application	No	Medium (already done in some areas)	0.26 - 0.55
Fertiliser type	Low	Medium (already done in some areas)	0.26 - 0.55
Rotation species	No	Medium	0.29 - 0.88
Adding legumes	Low	High	0.26 - 0.55
Permanent crops	Variable	Low (reduces flexibility)	1.69 - 3.04
Agroforestry	Medium	Low (reduces flexibility)	0.15 - 0.70
Grass in orchards & vineyards	Medium/high	Low	1.69 - 3.04
Optimising grazing intensity	Low / medium	Medium (already done in some areas)	0.11 - 0.81
Length and timing of grazing	Medium	Medium	0.11 - 0.81
Grassland renovation	Low	High	0.11 - 0.81
Optimising manure storage	Medium / high	Medium	
Manure application techniques	Medium	Medium	1.54 - 2.79
Application of manure to cropland versus grassland	Low	Medium	1.54 - 2.79
Organic soil restoration	Medium / high	Medium	36.67 – 73.33

* Based on potential uptake by farmers

6.3.1.1 Cropland

Mitigation practices in cropland management include the following, partly-overlapping, categories:

Agronomy: Improved agronomic practices that increase yields and generate higher inputs of carbon residue can lead to increased soil carbon storage (Follett, 2001). Examples of such practices include: using improved crop varieties; extending crop rotations, notably those with perennial crops that allocate more carbon below ground; and avoiding or reducing use of bare (unplanted) fallow (West and Post, 2002; Smith, 2004a, b; Lal, 2003, 2004a; Freibauer *et al.*, 2004). Another group of agronomic practices are those that provide temporary vegetative cover between successive agricultural crops, or between rows of tree or vine crops. These ‘catch’ or ‘cover’ crops add carbon to soils (Barthès *et al.*, 2004; Freibauer *et al.*, 2004). Adding more nutrients, when deficient, can also promote soil carbon gains (Alvarez, 2005). The view has commonly been held that the use of fertilizer-N on agricultural land, by increasing crop yields, and hence crop residue returns, can lead to an increase in soil C, or at least moderate the decline that takes place as a result of tillage. However, a recent review (Kahn *et al.*, 2007) has concluded that fertilizer-N stimulates microbial breakdown of soil organic matter (SOM). The views of Kahn *et al.* (2007) have been challenged by Reid (2008) who suggested that observed decreases in soil C were caused by factors other than addition of fertilizer-N. Reay *et al.* (2008) recently reviewed evidence for the impact of N on soil organic carbon (SOC) stocks. Evidence is contradictory, with some studies suggesting that soil C may decrease under N enrichment, others suggesting no change, and others suggesting that soil C sinks may increase (Annex 10).

Potential side-effects: The benefits from N fertilizer can be offset by higher N₂O emissions from soils and CO₂ from fertilizer manufacture (Schlesinger, 1999; Pérez-Ramírez *et al.*, 2003; Robertson, 2004; Gregorich *et al.*, 2005).

Tillage: Advances in weed control methods and farm machinery now allow many crops to be grown with minimal tillage (reduced tillage) or without tillage (no-till). There are many different ways in which tillage intensity can be reduced, ranging from complete cessation of tillage (zero tillage), through reduced / minimum tillage where deep ploughing stops and surface tillage (scarification, disk harrowing) is used, and ridge tillage (where row crops are grown and only the ridges are tilled). These practices are now increasingly used throughout the world (e.g., Cerri *et al.*, 2004). There are a number of meta-analyses of reduced tillage experiments in the literature (Smith *et al.*, 1998; West and Post, 2002; Ogle *et al.*, 2003; Ogle *et al.*, 2005) Since soil disturbance tends to stimulate soil carbon losses through enhanced decomposition and erosion (Madari *et al.*, 2005), reduced- or no-till agriculture often results in soil carbon gain, but not always (West and Post, 2002; Ogle *et al.*, 2005; Gregorich *et al.*, 2005; Alvarez 2005).

Potential side-effects: Adopting reduced- or no-till may also affect N₂O emissions but the net effects are inconsistent and not well-quantified globally (Smith and Conen, 2004; Helgason *et al.*, 2005; Li *et al.*, 2005; Cassman *et al.*, 2003; Six *et al.*, 2004). The effect of reduced tillage on N₂O emissions may depend on soil and climatic conditions. In some areas, reduced tillage promotes N₂O emissions (MacKenzie *et al.*, 1998), while elsewhere it may reduce emissions or have no measurable influence (Marland *et al.*, 2001). Further, no-tillage systems can reduce CO₂ emissions from energy use (Marland *et al.*, 2003b; Koga *et al.*, 2006).

Residue management: Accompanying change in tillage practice, there is often necessarily a change in residue management, in that residues can no longer be ploughed in to the soil, and tend to be left on the soil surface. For this reason, it is difficult to separate the impacts of reduced tillage intensity and changed residue management. Systems that retain crop residues also tend to increase soil carbon, because these residues are the precursors for soil organic matter, the main carbon store in soil. Improved return of crop residues to the soil would increase soil carbon content, with or without changes in tillage. Many residue incorporation experiments in Europe have shown increases in soil carbon (e.g. Smith *et al.*, 1997).

Water management: About 18% of the world's croplands now receive supplementary water through irrigation (Millennium Ecosystem Assessment, 2005). Expanding this area (where water reserves allow) or using more effective irrigation measures can enhance carbon storage in soils through enhanced yields and residue returns (Follett, 2001; Lal, 2004a).

Potential side-effects: Some of the carbon gains brought about by irrigation may be offset by CO₂ from energy used to deliver the water (Schlesinger 1999; Mosier *et al.*, 2005) or from N₂O emissions from higher moisture and fertilizer-N inputs (Liebig *et al.*, 2005). The latter effect has not been widely measured. Drainage of wet croplands lands can promote productivity (and hence soil carbon) and perhaps also suppress N₂O emissions by improving aeration (Monteny *et al.*, 2006), but in highly organic soils, this could lead to a C loss (see section on restoration of organic soils).

Agro-forestry: Agro-forestry is the production of livestock or food crops on land that also grows trees for timber, firewood, or other tree products. It includes shelter belts and riparian zones/buffer strips with woody species. The standing stock of carbon above ground is usually higher than the equivalent land use without trees, and planting trees may also increase soil carbon sequestration (Oelbermann *et al.*, 2004; Guo and Gifford, 2002; Mutuo *et al.*, 2005; Paul *et al.*, 2003).

Potential side-effects: The effects on N₂O and CH₄ emissions are not well known (Albrecht and Kandji, 2003).

6.3.1.2 Grazing land

Grazers significantly impact on the carbon balance indirectly through their effect on vegetation type, on organic matter inputs to the soil microbial community, and on soil structure through trampling. Grazing lands occupy much larger areas than croplands (FAOSTAT, 2006) and are usually managed less intensively. The following are examples of practices to reduce GHG emissions and to enhance removals:

Grazing intensity: The effects of grazing intensity are inconsistent, owing to the many types of grazing practices employed and the diversity of plant species, soils, and climates involved (Schuman *et al.*, 2001; Derner *et al.*, 2006). Nevertheless, the intensity and timing of grazing (and livestock species) can influence the removal, growth, carbon allocation, and flora of grasslands, thereby affecting the amount of carbon accrual in soils (Conant *et al.*, 2001; 2005; Freibauer *et al.*, 2004; Conant and Paustian, 2002; Reeder *et al.*, 2004).

N management and fertilization: N fertilisation increases productivity in N-limited grassland systems, and if greater than any accompanying increase in decomposition rates, will lead to an overall increase in net ecosystem production (NEP). In mineral soils, fertilisation is generally considered to enhance carbon storage due to enhanced productivity. Irrigating grasslands, similarly, can promote soil carbon gains (Conant *et al.*, 2001) as well.

Potential side-effects: Adding nitrogen, often stimulates N₂O emissions (Conant *et al.*, 2005) thereby offsetting some of the benefits. The net effect of irrigating depends also on emissions from energy use and other activities on the irrigated land (Schlesinger, 1999).

Species introduction: Introducing grass species with higher productivity, or carbon allocation to deeper roots, has been shown to increase soil carbon. For example, establishing deep-rooted grasses in savannahs has been reported to yield very high rates of carbon accrual (Fisher *et al.*, 1994), although the applicability of these results has not been widely confirmed (Conant *et al.*, 2001; Davidson *et al.*, 1995). In the Brazilian Savannah (Cerrado Biome), integrated crop-livestock systems using Brachiaria grasses and zero tillage are being adopted (Machado and Freitas, 2004). Introducing legumes into grazing lands can promote soil carbon storage (Soussana *et al.*, 2004), through enhanced productivity from the associated N inputs.

Fire management: Burning can affect the proportion of woody versus grass cover, notably in savannahs, which occupy about an eighth of the global land surface. Reducing the frequency or intensity of fires typically leads to increased tree and shrub cover, resulting in a CO₂ sink in soil and biomass (Scholes and van der Merwe, 1996).

6.3.1.3 Restoration of degraded lands

A large proportion of agricultural lands has been degraded by excessive disturbance, erosion, organic matter loss, salinization, acidification, or other processes that reduce productivity (Batjes, 1999; Foley *et al.*, 2005; Lal, 2001a, 2003, 2004b). Often, carbon storage in these soils can be partly restored by practices that reclaim productivity including: re-vegetation (e.g., planting grasses); improving fertility by nutrient amendments; applying organic substrates such as manures, biosolids??, and composts; reducing tillage and retaining crop residues; and conserving water (Lal, 2001b; 2004b; Bruce *et al.*, 1999; Olsson and Ardö, 2002; Paustian *et al.*, 2004).

Potential side-effects: Where these practices involve higher nitrogen amendments, the benefits of carbon sequestration may be partly offset by higher N₂O emissions.

6.3.2 (Semi-) natural systems

6.3.2.1 Upland semi-natural grasslands

Temperate grasslands comprise around 20% of the land area of Europe, and store significant quantities of C, mostly below-ground (Soussana *et al.*, 2004). In upland areas of Europe 'semi-natural' grasslands, which have not previously been subject to intensive management practices such

as re-seeding or liming, generally occur on relatively organic-rich soils. These ecosystems are therefore sensitive to changes in the intensity of land-management.

Grazing: In relatively low-productivity semi-natural grasslands, carbon removals associated with animals are relatively small (e.g., Allard *et al.*, 2007). The EU GREENGRASS project included assessments of the GHG budget at two cattle-grazed upland semi-natural grassland sites, at Laqueille in the Massif Central, France, and Malga Arpaco in the Italian Alps (Soussanna *et al.*, 2007). Surprisingly, compared to six lower-elevation grasslands with more intensive management, the Italian upland site was the strongest net GHG sink (and one of the strongest C sinks) whereas the French site was one of the weakest. Based on modelling of the Laqueille site, Soussanna *et al.* (2004) concluded that the CO₂ sink would be greatest, and CH₄ source associated with the grazing cattle smallest, at low stocking densities. At a grazed acid grassland on organic soils in Wales, experimental grazing intensification caused a loss of organic horizon C (B. Emmett, unpublished data), whereas at a nearby grassland on mineral soils, 12 years of experimental grazing removal did not change soil C stocks (E. Rowe, unpublished data).

Fertilisation: Experimental results for the Laqueille site (Allard *et al.*, 2007), comparing an intensively grazed and N-fertilised paddock with an extensively grazed unfertilised paddock, showed a strong increase in the GHG sink in the year following extensification, but pronounced weakening of this sink (relative to the intensive management) in subsequent years, as the production declined due to lower N supply and a higher proportion of dead vegetation cover. Allard *et al.* (2007) conclude that reductions in grazing, if accompanied by reduced fertilisation, may not be an effective GHG mitigation option. Soussanna *et al.* (2004) state that N fertilisation may increase net ecosystem production in moderately fertile systems, as the increase in production outweighs any concurrent increase in decomposition. In more organic-rich mountain pastures, due to the relatively large pool of organic matter available for decomposition, N fertilisation may trigger large carbon losses.

Liming: Liming has been widely used to increase productivity in acidic grasslands, with the greatest use (supported in some countries by agricultural subsidies) in the middle of the 20th century. It has also been used in some areas, such as Germany and Scandinavia, to ameliorate the effects of acidification on natural ecosystems. Because decomposition rates in many upland soils are constrained by acidity, increases in pH due to liming consistently lead to accelerated C turnover, CO₂ production and DOC export (e.g. Andersson and Nilsson, 2001; Rangel-Castro *et al.*, 2004). Increased pH may also increase N₂O losses (Yamulki *et al.*, 1997). Although liming has become less prevalent since the mid 20th century, past liming is likely to have residual effects on soil acidity. Additionally, reductions in atmospheric sulphur deposition across Europe since the 1980s may be having a similar impact on soil decomposition rates as liming at larger spatial scales, removing the constraints on soil decomposition caused by anthropogenic acidification (Evans *et al.*, 2007).

Mowing: In many mountain regions, grasslands have been managed historically for hay production through annual mowing, although this practice is declining.

Uhlířová *et al.* (2005) suggest, for a Czech mountain grassland, that annual mowing is the most appropriate management for maintaining SOM status due to its positive influence on soil microbial biomass. However, annual mowing removes a substantial proportion of produced grass from the system (Soussanna *et al.*, 2007; Franzluebbers and Stueddeman, 2008).

6.3.2.2 Heathlands

Heathlands, or moorlands, are important ecosystems within the British Isles, parts of Scandinavia, Alpine regions, and in other temperate upland regions globally. They are characterised by low-growing vegetation (dwarf shrubs, mosses, sedges and acid grassland species) and organic-rich soils, which develop because environmental factors such as waterlogging and acidity constrain decomposition rates. Information regarding management impacts on soil carbon in heathlands is largely restricted to the British Isles, where extensive heathland areas developed following forest clearance in the mid-Holocene. Under conditions of extreme waterlogging, severely restricted decomposition has led to peat development. Since peats are critically important as carbon stores, and subject to specific management pressures, these are considered separately below. However, heathlands also occur widely on organo-mineral soils (soils with a peaty O horizon of up to 30-40 cm depth), and management pressures and mitigation options relating to these ecosystems are considered here.

Reducing grazing pressure: In a review of pressures on UK moorlands, Holden *et al.* (2007) state that heather will only grow at grazing levels below two sheep ha⁻¹, and that the overall area of UK moorland in which stocking densities exceeded this level rose to 29% in 1977, and 71% in 1987, as a result of CAP subsidies. In severely degraded ecosystems, where characteristic species or surface soil have been lost, recovery is likely to be very slow, and may require complete cessation of grazing (Britton *et al.*, 2005). However, exclosure studies in areas of heavily grazed Welsh moorland have shown the restoration of dwarf shrub dominance within 10 years of sheep exclusion (Hill *et al.*, 1992), and Hope *et al.* (1998) recorded a general shift in dominance from grasses to dwarf shrubs (and at one site, birch invasion) following long-term grazing removal. Even in areas that continue to be grazed, impacts may be minimised by removing animals during the wetter winter months, when soils are most susceptible to physical erosion (Grant *et al.*, 1982) and animals are most likely to graze on vulnerable heathland plants.

Potential side-effects: Reductions in domesticated grazers, such as sheep, may be offset by increases in 'wild' grazers such as deer, with consequences for habitat condition, and hence C sequestration, that are difficult to predict (Albon *et al.*, 2007).

Fertilisation: In general, N fertilisation is not practiced on heathlands, but they are widely subject to elevated atmospheric N deposition, which may have similar impacts on C cycling. The response of organo-mineral soils to elevated N inputs is hard to predict, as it depends strongly on the effect of increased N supply on decomposition rates within the large soil organic matter pool. In general, increased N inputs may accelerate decomposition of more reactive organic matter, but constrain decomposition of more recalcitrant material (e.g. Berg *et al.*, 1998; Hagedorn *et al.*, 2003). Some studies have shown some evidence of carbon accumulation in heathlands exposed to long term experimental N additions, at rates in the order of 15-30 g C g⁻¹ N (Evans *et al.*, 2006; de Vries *et al.*, in review), but it is doubtful whether these results can be generalised. In particular, elevated N deposition may, in the long-term, lead to species changes towards plant species with more degradable litter (e.g. dwarf shrubs to grasses), thereby reducing rates of soil organic matter formation.

Potential side-effects: N additions have also been shown to increase N₂O emissions from moorland soils (Skiba *et al.*, 1999; Pilkington *et al.*, 2005). Both liming and reduced sulphur deposition can, as in acidic grasslands, be expected to increase decomposition rates, leading to elevated CO₂ and DOC loss (e.g. Hornung *et al.*, 1986; Reynolds *et al.*, 1994).

Burning: Heathlands are frequently managed through **controlled burning**, initially to hold back succession to woodland, and more recently and intensively in the UK to maintain a supply of young heather for game birds. The implications of management burning for carbon stocks are not well understood (Holden *et al.*, 2007). Controlled burns remove most of the above-ground biomass, converting much of the C contained within this material to CO₂. However they have little impact on litter or O horizon organic matter. Post-burn, nutrient and pH levels may be raised by ash, and soil temperatures may increase, all of which may increase decomposition rates (Kim and Tanaka, 2003; Stevenson *et al.*, 1996). However, burning also removes nutrients in smoke and through volatilisation, and managed burns have been estimated to remove 60% of biomass N (Terry *et al.*, 2004), as well as significant amounts of phosphorous (Kinako and Gimingham, 1980). These nutrient losses may decrease productivity in the long-term, but conversely the maintenance of low-nutrient conditions through burning may be effective in offsetting ecosystem eutrophication due to elevated N deposition (Pilkington *et al.*, 2007), which might otherwise cause vegetation change from heathland to grassland species, increase decomposition rates, and hence decrease C accumulation. Holden *et al.* (2007) conclude that some degree of management (burning or cutting) is required to maintain current heathland vegetation on dryer (organo-mineral) soils, but the overall impact of burning for C and GHG budgets are clearly uncertain.

Wildfires have a more severe impact on ecosystem C stocks as they generally occur during drought periods (managed burning generally takes place when soils are wet) and burn at higher temperatures, leading to combustion of litter and soil organic matter. Exposure of bare soil following severe burns also increases susceptibility to physical erosion; a post-burn study of a heathland on organo-mineral soil in Northeast England showed a tenfold increase in erosion rates in areas subject to severe burning (Ineson, 1971). Because managed burning reduces the stock of above ground biomass, particularly the build up of combustible woody debris, it may provide some protection for soil organic matter stocks against the risk of wildfires

Forestry: Most heathlands on organo-mineral soils would revert to woodland without a certain level of management through grazing or burning. While afforestation leads to soil C losses, tree growth itself leads to accumulation of above-ground biomass, and increased litter inputs. Forest soils may therefore gain C in surface horizons, while simultaneously losing C at depth (Emmett *et al.*, 1997; Post and Kwon, 2000). The net effect of afforestation on ecosystem C balance thus depends on whether the increase in NPP exceeds the increase in soil decomposition; on the long-term stability of organic matter added to the soil; and on the fate of forest biomass removed from the site. Overall, Cannell *et al.* (1999) estimated that the 2 Mha of UK forest plantation were accumulating C at a rate of 113 g C m⁻² a⁻¹. Various site-based studies synthesised by Reynolds (2007) provide estimates of soil C accumulation rates at afforested sites in the range 21 to 74 g C m⁻² a⁻¹.

Potential side-effects: By far the dominant form of afforestation in moorland areas, particularly within the British Isles, has been plantation with non-native conifers. The levels of disturbance, associated with this type of moorland afforestation in particular, are high, including road building, ploughing, drainage and subsequent harvesting, generally leading to a loss of existing soil C

(Reynolds, 2007). This loss occurs through accelerated decomposition of disturbed soils, and increased POC loss through physical erosion; Soutar (1989) estimated that, in the long-term, afforestation increased sediment losses by a factor of 3-4. Losses may be much higher during periods of maximum disturbance associated with road building, ploughing, drainage and harvesting (Reynolds, 2007), although this may be greatly reduced by improved forestry practices. DOC losses may also increase after planting and harvesting, although the evidence for this is equivocal (Reynolds, 2007).

Although forest planting on organo-mineral moorland soils does appear to lead to a net ecosystem C gain, then, it is worth noting that this represents a shift in the type of C stored, with new C being added while older stored C is lost (e.g. Karlton *et al.*, 2005; Reynolds, 2007). The long-term benefit of moorland afforestation as a means of C sequestration is thus critically dependent on the long-term stability and fate of the newly accumulated C.

6.3.3 Forests

Jandl *et al.* (2007) have reviewed the effects of forest management on soil carbon sequestration. Other more general reviews focused on the potential contribution of forest management on GHG mitigation (Schelhaas *et al.*, 2007) and the forest GHG budget including nitrous oxide and methane (Lindner *et al.*, 2004).

Because of the large spatial variability of soils, changes in soil carbon are not always easy to detect (Conen *et al.*, 2005). Moreover, due to the long management cycles and the irregular natural disturbance effects there are large temporal fluctuations in the GHG budget of forests at the stand scale. This is particularly true for the soil organic layer, which responds relatively fast to changes in management (Jandl *et al.*, 2007). Even at the continental scale, unequal age class distributions can cause a shift from a GHG sink to a GHG source (Kurz *et al.*, 1995; Karjalainen *et al.*, 2003). Consequently, management induced changes in GHG budgets can often be overlain by natural variability. This may explain that there are quite often conflicting results reported for similar management changes. However, when the underlying processes affected by management can be identified, it is easier to assign clear effects to the management changes.

Species influence: Despite long research on the role of the vegetation on soil formation, a generalized understanding on the extent of the effect of tree species across site types has not yet been reached (Augusto *et al.*, 2002; Binkley and Menyailo, 2005). Tree species affect the C storage of the ecosystem in several ways. Shallow rooting coniferous species tend to accumulate soil organic matter in the forest floor but less in the mineral soil, compared to deciduous trees. Species-specific site productivity affects biomass production and litter fall differently along site quality gradients. The influence of tree species was studied in common garden experiments. In Denmark, a study of seven species replicated at seven different sites along a soil fertility gradient showed that Lodgepole pine (*Pinus contorta*), Sitka spruce (*Picea sitchensensis*) and Norway spruce had much higher forest floor C stocks than European beech (*Fagus sylvatica*) and oak (*Quercus sp.*) (Vesterdal and Raulund-Rasmussen, 1998). A German experiment showed more C in the forest floor under pine than under beech, but in pine stands that have been underplanted with beech, the depth gradient of soil C was changed (Fischer *et al.*, 2002). While the effect of tree species on forest floor C stocks is faster, for the permanence of C sequestration it is more relevant to select tree species that increase the pool of stabilized C in the mineral soil. The driving process is the production of belowground

biomass (Jobbágy and Jackson, 2000, Vesterdal *et al.*, 2002a). However, little evidence for the size of the effect is available.

Stand management: Under stand management we consider differences in the thinning and harvesting regimes, the rotation length in even-aged forest management systems, and the overall silvicultural system.

The effects of **thinning** are unclear. Thinning affects the distribution of biomass in a forest stand and changes the microclimate. Decomposition of forest floor C is temporarily stimulated (Aussenac, 1987, Piene and van Cleve, 1978). The stand microclimate returns to previous conditions unless the thinning intervals are short and intensities are high. Litterfall is temporarily lowered in strongly thinned stands. This reduces forest floor accumulation, but the input of thinning residues into the soil may compensate for that (de Wit and Kvindesland, 1999). Forest floor C stocks decreased with increasing thinning intensity in field studies in New Zealand, Denmark and USA (Vesterdal *et al.*, 1995; Carey *et al.*, 1982; Wollum and Schubert, 1975). A thinning intervention in an experimental site with flux measurements in Finland did not result in a net release of C from the ecosystem, because the enhanced growth of the ground vegetation compensated for the reduced C sequestration of the tree layer and the increase of heterotrophic soil respiration was balanced by a decrease in autotrophic respiration of similar magnitude (Suni *et al.*, 2003). In a Korean study, neither soil CO₂ efflux nor litter decomposition were increased with increasing thinning intensity (Son *et al.*, 2004). Any effects on soil respiration rates were apparently overruled by root respiration as indicated by a positive relationship between stand density and soil CO₂ efflux.

Harvesting removes biomass, disturbs the soil and changes the microclimate more than a thinning operation. In the years following harvesting and replanting, soil C losses may exceed C gains in the aboveground biomass. The long-term balance depends on the extent of soil disturbance. Harvest residues left on the soil surface increase the C stock of the forest floor, disturbance of the soil structure leads to soil C loss. A review on harvesting techniques suggested that their effect on soil C is rather small (Johnson and Curtis, 2001). Whole-tree harvest caused a small decrease in A-horizon C stocks, whereas conventional harvesting, leaving the harvest residues on the soil, resulted in a small increase. Although soil C changes were noted after harvesting, they diminished over time without a lasting effect.

Longer rotation periods have been proposed to foster C sequestration in forests. The effect of increased rotation lengths is mainly determined by the current management practice. Longer rotation lengths with more old forests lead to higher C pools than short rotations with only young plantations. Old-growth forests have the highest C density, whereas younger stands have a larger C sink capacity. After harvest operations, soil C pools in managed forests recover to the previous level. Short rotation lengths where the time of harvest is close to the age of maximum mean annual increment will maximize aboveground biomass production, but not C storage. Longer rotation periods imply that the disturbance frequency due to forest operations is reduced and soils can accumulate C (Schulze *et al.*, 1999). Growth and yield tables suggest that stand productivity declines significantly in mature forest stands. However, a mature Siberian Scots pine forest and old-growth forests in USA transferred a higher proportion of its C into the soil than in the early stages of the stand development and continuously increased the soil C stock (Harmon *et al.*, 1990; Schulze *et al.*, 2000). The accumulation of C continues until the C gain from photosynthesis is larger than respiration losses. Late-successional species (*e.g.* beech, Norway spruce) are able to maintain high C sequestration rates for longer than pioneer tree species. Over-

mature forest stands are not able to close canopy gaps created by natural mortality or thinning. Consequently the decomposition of SOM is enhanced and decreases the soil C pool. Several modelling studies suggest that very long rotation lengths not necessarily maximize the total C balance of managed forests (Cannell, 1999; Liski *et al.*, 2001; Harmon and Marks, 2002).

Silvicultural changes are often associated with changes in species composition and mixture, which are difficult to assess as reported above. Continuous cover forestry with selective harvesting is linked with reduced soil disturbance compared with clear-cut harvesting which may decrease soil C losses (ECCP-Working group on forest sinks, 2003).

Minimising site preparation: Site preparation techniques include manual, mechanical, chemical methods and prescribed burning, most of which include the exposure of the mineral soil by removal or mixing of the organic layer. The soil disturbance changes the microclimate and stimulates the decomposition of SOM, thereby releasing nutrients (Palmgren, 1984; Johansson, 1994). A review on the effects of site preparation showed a net loss of soil C and an increase in stand productivity (Johnson, 1992). The effects varied with site and treatment. Several studies that compared different site preparation methods found that the loss of soil C increased with the intensity of the soil disturbance (Johansson, 1994; Örlander *et al.*, 1996; Schmidt *et al.*, 1996; Mallik and Hu, 1997). At scarified sites, organic matter in logging residues and humus, mixed with- or buried beneath the mineral soil, is exposed to different conditions for decomposition and mineralization compared to conditions existing on the soil surface of clear-cut areas. The soil moisture status of a site has great importance for the response to soil scarification. The increase in decomposition was more pronounced at poor, coarsely textured dry sites than on richer, fresh to wet sites (Johansson, 1994). Sandy soils are particularly sensitive to management practices, which result in significant losses of C and N (Carlyle, 1993). Intensive site preparation methods might result in increased nutrient losses and decreased long-term productivity (Lundmark, 1988).

Tending and weed control: Tending includes all activities in forest plantations after planting up to the moment of the first (commercial) thinning. Trees and weed cut in tending operations usually are not removed from the site. The decomposition of their foliage, stems and roots increases soil C content (Paul *et al.*, 2002). However, weed control by e.g. soil scarification could result in the loss of soil carbon due to accelerated decomposition of organic matter and wind and water erosion (Johnson, 1992; Paul *et al.*, 2002). Tending in combination with thinning can have a beneficial effect of up to 10% on carbon sequestration, because the remaining trees will grow better (Kairiukstis and Juodvarkis, 2005).

Increased productivity (including fertilization and liming): As for agricultural lands, forests can be improved by a variety of measures that promote productivity. A meta-analysis by Johnson and Curtis (2001) showed that fertilization had an overall increasing effect on soil C storage due to increased litter production and reduced soil respiration. Nitrogen fertilization stimulates biomass production, but the effect on the soil C pool is more complex. It stimulates the microbial decomposition of SOM, which can lead to a net C loss from the soil and can lead to the formation of nitrogen oxides (Jandl *et al.*, 2007).

Potential side-effects: The effect of C sequestration in the aboveground biomass is partly offset by the production of N₂O. This has been shown in agricultural as well as in forest ecosystems (Brumme and Beese, 1992; Mosier *et al.*, 1998). In Central and Northern Europe many forest soils have been limed in the past in order to regulate soil and surface water chemistry, to prevent the ecosystem from

irreversible acidification and to mobilize recalcitrant forest floor material (Fiedler *et al.*, 1973; von Wilpert & Schäffer, 2000). However, the target of mobilizing the forest floor is in conflict with the objective of C sequestration. Liming causes a net loss of C in temperate and boreal forest soils due to increased microbial activity and DOC leaching (Brumme and Beese, 1992; Jandl *et al.*, 2003; Lundström *et al.*, 2003).

Protecting against disturbances: The role of **fire** in ecosystem C changes is not straightforward. Several experiments showed that wildfire had caused increases in soil C, which may be driven by the incorporation of charcoal into soils and new C inputs via post-fire N₂ fixation (Hirsch *et al.*, 2001; Johnson and Curtis, 2001; Johnson *et al.*, 2004; Schulze *et al.*, 1999). Nevertheless, N-fixing plants are not common to all fire-prone ecosystems. Forest management can temporarily decrease forest fire risk by manipulating the fuel characteristics. A very important measure is to disrupt the continuity of the fuel, both within stands (open forest) and between stands (fire breaks, variation in stand characteristics). Planning at the landscape level is also very important (Hirsch *et al.*, 2001). The amount of fuel can be reduced by prescribed burning, or by active removal (Fernandes and Botelho, 2003). Other management options are to manage the forest to create an open structure (combined with removal of felling debris) or to change tree species to less flammable species.

Storm damage may result in strongly increased amounts of coarse woody debris on the forest floor. The C dynamics after the disturbance are also affected by subsequent management decisions. In case of a severe reduction of the value, the stand will be harvested and damaged timber will be salvaged. When only parts of the canopy are broken and the stand is already mature, it may be wise to continue the originally planned production cycle (Thürig *et al.*, 2005). Uprooting of trees by windthrow destroys soil structure, which in turn makes protected C accessible for decomposers. Important options for increased stability are well designed thinning regimes (including no-thinning regimes in stands at high risks) and carefully planned fellings in order to minimize the length of exposed edges (Gardiner and Quine, 2000). Tree species choice also plays a decisive role in stand stability. Especially Norway spruce and Sitka spruce are known to be sensitive to wind throw. In conclusion, disturbances consistently lead to the mobilization of C and present a potentially large C source. There are many interdependencies with management activities such as choice of tree species, regulation of stand structure, thinning intensity, and rotation length. Without forest management interventions, the importance of disturbances for C dynamics increases.

Potential side-effects: The policy of fire suppression can delay but cannot prevent wildfires over the long term. It leads to C accumulation with the side effect of higher fuel loads, which increases the risk of large C release during catastrophic fires.

Removing harvest residues: The effects of harvest residue extraction on soil carbon and long term site productivity are still not fully clear. With increasing demand for renewable bio-energy, biomass removals from forests after harvest operations have increased a lot recently in Sweden, Finland and other European countries. Harvest residues left on site are decomposing rather quickly in temperate forest conditions and therefore removing them for fossil fuel substitution may have a positive effect on the GHG balance (Johnson and Curtis, 2001). Findings from the North American long-term soil productivity experiment suggest that under moderate and warmer climates, carbon from harvest residues is mainly respired as CO₂, and very little carbon is incorporated into the soil (Powers *et al.*, 2005).

Potential side-effects: While many short-term studies showed no negative effect of harvest residue removal on growth (Roberts *et al.*, 2005), it is possible that negative growth impacts occur in the long term. This has been shown in Northern Sweden for whole-tree harvesting in Scots pine stands on nutrient-poor sites, where growth declines were revealed only 12 – 24 years after harvesting (Egnell and Valinger, 2003). Therefore, utilising forest harvest residues on poor sites could be detrimental to site productivity and long-term soil carbon storage without compensatory fertilisation (Sverdrup and Rosen, 1998; Richardson *et al.*, 2002; Raulund-Rasmussen *et al.*, 2008). With a doubling of biomass removals in intensive biomass harvesting, the nutrient removal may increase up to 6-7 times (Raulund-Rasmussen *et al.*, 2008). Even on more fertile soil types it is beneficial to retain foliage on the site (Samuelsson, 2002). Thus it is beneficial to exclude small branches and foliage from the biomass removals by extracting dry residues in the case of coniferous species (to allow needles to drop before chipping), or in the case of broadleaved species to harvest in the winter months (Richardson *et al.*, 2002).

When foliage and roots are removed as well, e.g. in whole tree harvesting and stump extraction, there is a risk for detrimental impacts, especially on nutrient-poor sites. More research is needed to reveal whether wood ash recycling or conventional fertilization will be sufficient to sustain long term site productivity under such conditions by replenishing the exported nutrients (Raulund-Rasmussen *et al.*, 2008).

Contrasting evidence has been found regarding the effects of stump extraction on soil carbon dynamics and site productivity. Stump removal may improve growth of the regenerated stand on sites infected with root rot fungi (Thies and Westlind, 2005; Vasaitis *et al.*, 2008). However, Zabowski *et al.* (2008) found an extended decrease in mineral soil total N and C and forest floor depth in five stands in the Pacific Northwest of America 22-29 years after stump removal. The stumped areas showed 20% lower mineral soil nitrogen concentration, 24% lower mineral soil carbon concentrations and 24% lower forest floor depth. A non-significant trend of lower foliar N was also observed with stump removal. The results were consistent in all five soil types, suggesting that the reduction in the organic component of the soil may be a concern for nutrient cycling and long-term productivity on poor sites (Zabowski *et al.*, 2008). These results are in line with the most drastic management scenario of the North American long-term soil productivity experiment, where the removal of the forest floor (in addition to harvest residue removal) also led to reduced nitrogen availability and significant reductions in soil carbon concentrations down to a depth of 20 cm (Powers *et al.*, 2005).

6.4 Comparison of the potential of soil management and land use measures to mitigate climate change with mitigation efforts in other sectors

6.4.1 Potential of soil carbon sequestration

Soil C sequestration can be achieved by increasing the net flux of C from the atmosphere to the terrestrial biosphere by increasing global C inputs to the soil (via increasing NPP), by storing a larger proportion of the C from NPP in the longer-term C pools in the soil, or by reducing C losses from the soils by slowing decomposition. For soil C sinks, the best options are to increase C stocks in soils that have been depleted in C, i.e. agricultural soils and degraded soils, or to halt the loss of C from cultivated peatlands (Smith *et al.*, 2007a).

Early estimates of the potential for additional soil C sequestration varied widely. Based on studies in European cropland (Smith *et al.*, 2000), U.S. cropland (Lal *et al.*, 1998), global degraded lands (Lal 2001) and global estimates (Cole *et al.*, 1996; IPCC 2000a), an estimate of global soil C sequestration potential of $0.9 \pm 0.3 \text{ Pg C y}^{-1}$ was made by Lal (2004a, b), between a 1/3 and 1/4 of the annual increase in atmospheric C levels. Over 50 years, the level of C sequestration suggested by Lal (2004a) would restore a large part of the C lost from soils historically.

The most recent estimate (Smith *et al.*, 2007a) is that the technical potential for SOC sequestration globally is around 1.3 Pg C y^{-1} , similar to the estimate of Lal (2004). The estimates made in Smith *et al.* (2008) for both technical and economic potential (where other estimates existed) were compared. Almost all (global and regional) were found to be close (Smith *et al.*, 2007a, 2008). The technical mitigation potential is very unlikely to be realized. Economic potentials for SOC sequestration estimated by Smith *et al.* (2007a) were 0.4, 0.6 and 0.7 Pg C y^{-1} at carbon prices of 0-20, 0-50 and 0-100 USD t CO₂-equivalent⁻¹, respectively. At reasonable C prices, then, global soil C sequestration seems to be limited to around $0.4\text{-}0.7 \text{ Pg C y}^{-1}$. In Europe, the economic potentials for C sequestration are around 53-58, 80-87 and 93-102 Mt C (or 0.053-0.058, 0.080-0.087 and 0.093-0.102 Pg C) at C prices of 0-20, 0-50 and 0-100 USD t CO₂-equivalent⁻¹, respectively (from data in Smith *et al.*, 2008). Even then, there are barriers (e.g. economic, institutional, educational, social) that mean the economic potential may not be realized (Trines *et al.*, 2006; Smith and Trines, 2007). These are discussed further in section 5.3.3. The estimates for C sequestration potential in soils are of the same order as for forest trees, which have a technical potential to sequester about 1 to 2 Pg C y⁻¹ (IPCC, 1997; Trexler, 1988 [cited in Metting *et al.*, 1999]), but economic potential for C sequestration in forestry is similar to that for soil C sequestration in agriculture (IPCC WGIII, 2007).

Many reviews have been published recently discussing options available for soil C sequestration and mitigation potentials (e.g. IPCC, 2000a; Cannell, 2003; Metting *et al.*, 1999; Smith *et al.*, 2000; Lal 2004a; Lal *et al.*, 1998; Nabuurs *et al.*, 1999; Follett *et al.*, 2000; Freibauer *et al.*, 2004; Smith *et al.*, 2007a).

Compared to abiotic carbon sequestration, soil carbon sequestration potential is small. Abiotic carbon and capture and storage (CCS) sequestration can take the form of oceanic sequestration through deep ocean injection of CO₂ (5000-10000 Pg C potential), geological sequestration through the capture, liquefaction, transport and injection of CO₂ into coal seams, old oil wells, stable rock strata or saline aquifers, or scrubbing of CO₂ and mineral carbonation at point of CO₂ emission (Lal, 2008). However, Lal (2008) points out that abiotic technologies are expensive, have leakage risks and may not be available for routine use until 2025 and beyond, whereas soil carbon sequestration is natural, cost-effective, with ancillary benefits and is immediately applicable (Lal, 2008).

6.4.2 Barriers to implementation of soil carbon sequestration measures

Despite significant economic potential for GHG mitigation through agricultural carbon sequestration, there are many barriers that could prevent the implementation of these measures. These have recently been reviewed by Smith *et al.* (2007b), Trines *et al.* (2006) and Trines and Smith (2007):.

- **Economic barriers** include the cost of land, competition for land, continued poverty, lack of existing capacity, low price of carbon, population growth, transaction costs and monitoring costs.

- **Risk related barriers** include the delay on returns due to slow system responses, issues of permanence (particularly of carbon sinks) and issues concerning leakage and natural variation in carbon sink strength.
- **Political and bureaucratic barriers** include the slow land planning bureaucracy and the complexity and lack of clarity in carbon / greenhouse gas accounting rules, resulting in a lack of political will.
- Among **logistical barriers** considered by Trines *et al.* (2006) were the fact that land owners are often scattered and have very different interests, that large areas are unmanaged, the managed areas can be inaccessible and some areas are not biologically suitable.
- The **education / societal barriers** relate to the sector and legislation governing it being very new, stakeholder perceptions and the persistence of traditional practices.

Competition with other land uses is a barrier that necessitates a comprehensive consideration of mitigation potential for the land-use sector. It is important that forestry and agricultural land management options are considered within the same framework to optimise mitigation solutions. Costs of verification and monitoring could be reduced by clear guidelines on how to measure, report and verify GHG emissions from agriculture.

Transaction costs, on the other hand, will be more difficult to address. The process of passing the money and obligations back and forth between those who realise the carbon sequestration and the investors or those who wish to acquire the carbon benefits, involves substantial transaction costs, which increases with the number of landholders involved. Given the large number of small-holder farmers in many developing countries, the transaction costs are likely to be even higher than in developed countries, where costs can amount to 25% of the market price (Smith *et al.*, 2007b). Organisations such as farmers' collectives may help to reduce this significant barrier by drawing on the value of social capital. Farmers are in touch with each other, through local organisations, magazines or community meetings, providing forums for these groups to set up consortia of interested forefront players. In order for these collectives to work, regimes need to be in place already, and it is essential that the credits are actually paid to the local owner.

For a number of practices, especially those involving carbon sequestration, risk related barriers such as delay on returns and potential for leakage and sink reversal, can be significant barriers. Education, emphasising the long term nature of the sink, could help to overcome this barrier, but fiscal policies (guaranteed markets, risk insurance) might also be required.

Education / societal barriers affect many practices in many regions. There is often a societal preference for traditional farming practices and, where mitigation measures alter traditional practice radically (not all practices do), education and extension would help to reduce some of the barriers to implementation.

A significant barrier to implementation of mitigation measures in poorer parts of Europe is economics. Given the challenges many farmers in these regions already face, climate change mitigation may be a low priority. Capacity building and education in the use of innovative technologies and best management practices would also serve to reduce barriers.

Maximizing the productivity of existing agricultural land and applying best management practices would help to reduce greenhouse gas emissions (Smith *et al.*, 2007b). Ideally agricultural mitigation measures need to be considered within a broader framework of sustainable development. Policies to encourage sustainable development will make agricultural mitigation more achievable.

The UK's Stern Review (www.sternreview.org.uk) warns that unless we take action in the next 10-20 years, the environmental damage caused by climate change later in the century could cost

between 5 and 20% of global GDP every year. The barriers to implementation of mitigation actions need to be overcome if we are to realise even a proportion of the global agricultural climate mitigation potential. In both environmental and economic terms, we cannot afford not to act strongly and quickly.

6.4.3 Soil carbon sequestration in comparison to the GHG mitigation potential in other sectors

Soil carbon sequestration potential in agricultural soils is of a similar size as that available through forest carbon sequestration and prices up to 100 USD t CO₂⁻¹. Figure 12 shows the findings from the IPCC Fourth Assessment Report on global economic mitigation potential (IPCC WGIII, 2007).

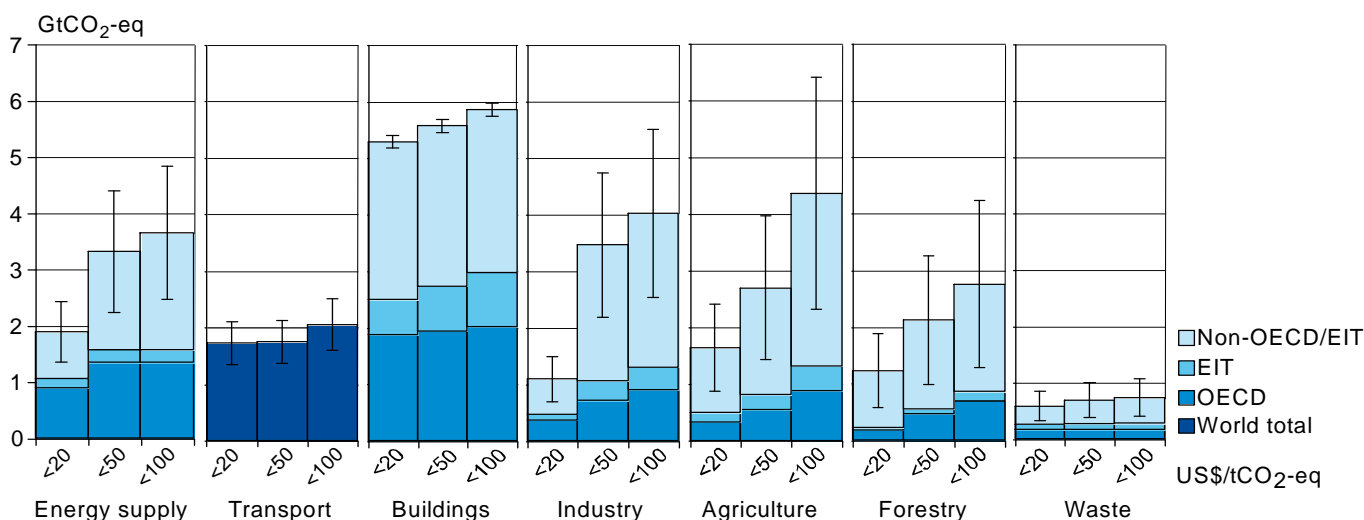


Figure 12 Global economic mitigation potential

At all carbon prices, the greatest mitigation potential is in the buildings sector. At low carbon prices (20 USD t CO₂-eq.⁻¹), agricultural mitigation potential (of which 90% is due to carbon sequestration; Smith *et al.*, 2007a, 2008) is similar to the potential in the energy and transport sectors and is higher than that in the industry, forestry and waste sectors. At medium carbon prices (50 USD t CO₂-eq.⁻¹), the mitigation potential for soil carbon sequestration is lower than the potential in the buildings, energy and industry sectors, but is higher than the potential in the transport, forestry and waste sectors. At high carbon prices (100 USD t CO₂-eq.⁻¹), the mitigation potential from agricultural soil carbon sequestration is similar to the industry and energy supply sectors, lower than the buildings sector, but higher than the transport forestry and waste sectors. It should be noted that there is considerable uncertainty (denoted by error bars in Figure 12) associated with the mitigation potential in all sectors, but especially in the energy supply, industry, agriculture forestry and waste sectors (IPCC WGIII, 2007). In another analysis of cross-sectoral mitigation potentials, Enqvist (2007) reported similar potentials from the agricultural sector, but that analysis considered only non-CO₂ GHG emission reduction. The same organization (McKinsey) has released an updated cross-sectoral assessment of GHG mitigation potentials in October 2008 that also considers agricultural soil carbon sequestration, and contains regional breakdown of mitigation potentials.

To put the figures for soil carbon sequestration potential in the context of global annual C emissions and the annual rise in atmospheric CO₂ concentration, at 100 USD t CO₂-eq.⁻¹, 0.7 Pg C yr⁻¹ can be sequestered each year in agricultural soils (Smith, 2007). The current annual emission of CO₂-carbon to the atmosphere is 6.3 ± 1.3 Pg C y⁻¹. Carbon emission gaps by 2100 could be as high as 25 Pg C y⁻¹ meaning that the C emission problem could be up to 4 times greater than at present. The maximum annual global C sequestration potential is about 0.7 Pg C y⁻¹ (Smith *et al.*, 2007a) meaning that even if these rates could be maintained until 2100, soil C sequestration would contribute a maximum of about 1-3% towards reducing the C emission gap under the highest emission scenarios. When we also consider the limited duration of C sequestration options in removing C from the atmosphere, we see that C sequestration could play only a minor role in closing the emission gap by 2100. It is clear from these figures that if we wish to stabilize atmospheric CO₂ concentrations by 2100, the increased global population and its increased energy demand can only be supported if there is a large-scale switch to non-C emitting technologies in the energy, transport, building, industry, agriculture, forestry and waste sectors (IPCC WGIII, 2007).

This demonstrates that soil C sequestration alone can play only a minor role in closing the C emission gap by 2100. Nevertheless, if atmospheric CO₂ levels are to be stabilized at reasonable concentrations by 2100 (e.g. 450-550 ppm), drastic reductions in emissions are required over the next 20-30 years (IPCC, 2000b; IPCC WGIII, 2007). During this critical period, all measures to reduce net C emissions to the atmosphere would play an important role – there will be no single solution (IPCC WGIII, 2007). IPCC WGIII (2007) show that there is significant potential for greenhouse gas mitigation at low cost across a range of sectors, but for stabilization at low atmospheric CO₂ / GHG concentrations, strong action needs to be taken in the very near future, echoing the findings of the Stern Review (Stern, 2006). Given that C sequestration is likely to be most effective in its first 20 years of implementation, it should form a central role in any portfolio of measures to reduce atmospheric CO₂ concentrations over the next 20-30 years whilst new technologies, particularly in the energy sector, are developed and implemented (Smith, 2004; 2007).

6.5 Inventory and reporting systems for measuring the carbon stock changes due to land use and land use changes

An analysis has been carried out to assess the extent to which the IPCC/UNFCCC inventory and reporting system reflects the findings published in recent peer-reviewed literature and the confidence limits of the estimates of carbon sequestration. Recommendations for updating the current IPCC/UNFCCC inventory and reporting systems are given. Furthermore the status of development of reporting schemes outside the EU will be considered, and evaluated to assess the potential value for Europe.

6.5.1 Current status of the inventory and reporting systems for measuring the carbon stock changes in soils in the land use, land use change and forestry sector

According to the United Nations Framework Convention on Climate Change (UNFCCC) Articles 4 and 12, Parties are required to develop and submit to the UNFCCC national inventories of anthropogenic emissions by sources and removals by sinks of all greenhouse gases not controlled by

the Montreal Protocol on an annual basis. The report (so called national inventory reports, or NIRs) and the associated Common Reporting Format (CRF) tables, where estimates should be reported, have to be prepared in accordance with the guidelines “Updated UNFCCC reporting guidelines on annual inventories following incorporation of the provisions of decision 14/CP.11” (FCCC/SBSTA/2006/9) adopted by the Conference of the Parties (COP) in 2006 (this supersedes the previous reporting guidelines, the UNFCCC Reporting Guidelines on Annual Inventories as adopted by Decision 18/CP. 8., FCCC/SBSTA/2004/8).

There are several principles of reporting that are relevant for soils. These include the five main principles, i. e. accuracy, completeness, consistency, comparability and transparency. In addition, it is important to note that the GHG inventory must be done for the entire country. This means that emissions and removals must be estimated for areas that are unusually large compared to the scale of most scientific or monitoring programs for most countries. This also involves the application of yet another principle of the GHG inventories, which can be abbreviated as the “*practicability principle*”. This principle states that the inventories should be accurate as far as *practicable*. The interpretation of what is practicable is of course up to the countries, and, indeed, one consequence of this is that countries usually invest as little resource in the inventory as possible, and often rely on IPCC default values.

Additionally, the estimation must be done annually, which requires that at least annual changes of land use (i.e., how large areas under the various land use categories have been converted to other land use) are registered. Also, it is emissions and removals that must be estimated, not stocks, which may have been in the focus of soil inventories so far, whether scientific or for certain monitoring purposes. Finally, taking soil carbon and other soil related measurements is regarded as being rather expensive. All this makes it very difficult for countries to conduct an inventory based on measurements so that reporting could satisfy all requirements that are usually set for statistical sample based inventories.

In contrast, emissions from and removals by soil, if measured, are often a key category⁴ in the national greenhouse gas inventories. This indicates two issues. One simply is that, in order to be accurate in the sense of “completeness”, it would be important to make efforts to estimate these emissions and removals. The other issue is that the estimation methodology would require higher Tier (i.e., Tier 2 or 3 in the sense of IPCC terminology, i.e. IPCC, 2003, IPCC; 2006) to ensure the accuracy that is usually required for such key categories.

Concerning the methodology, it has two basic elements: one is how land is identified (this has three so called Approaches), and the other is how carbon stocks or their changes are estimated (so called Tiers). These approaches and tiers are in practice combined in a number of ways. Below is a summary of the more common combinations that occur in practice:

- Approach 1, Tier 1: area statistics of land use and land management categories for each year are available, but no country specific soil carbon information is available, and the IPCC default soil organic carbon (SOC) and so called F values, which depend on the management practice, are used.
- Approach 1, Tier 2: in addition to area statistics, country specific SOC and F values are available.

⁴ A category is deemed key if it is prioritised within the national inventory system because its estimate has a significant influence on a country's total inventory of greenhouse gases in terms of the absolute level of emissions and removals, the trend in emissions and removals, or uncertainty in emissions or removals. Whenever the term key category is used, it includes both source and sink categories. For more details, see Chapter 4 of Volume 1 of IPCC (2006).

- Approach 2, Tier 1: annual land use change matrices are available, but there are no country specific soil carbon data, therefore, the IPCC default SOC and F values are used.
- Approach 2, Tier 2: same as above, but country specific SOC and F values are available.
- Approach 3, Tier 1: time series of geographically identified locations of land use categories are available, however, there are no country specific soil carbon data, therefore, the IPCC default SOC and F values are used. This is just a theoretical possibility.
- Approach 3, Tier 2: same as above, but with country specific SOC and F values.
- Approach 3, Tier 3: time series of geographically identified locations of land use categories are available, and country specific database exists, which either comes from statistical soil inventory and is the basis for the emission and removal estimation, or which is used to calibrate a model. This model can either be a country specific one, or taken from another country and adapted to local conditions.

Most countries use Approach 1 and Tier 1 for most land use and land use change categories where estimation has been attempted. In many instances, however, a high degree of uncertainty is associated with the land area activity data in general, and the consistent identification of the various land use or land use change categories is not possible. This of course makes it difficult, or impossible, to report on certain emissions or removals. Thus, land identification itself may represent obstacles for the estimation, and resolving this issue should be the number one priority for soil C monitoring, as well as for reporting on other sources of emissions and removals by sinks. Furthermore, the area of land converted to other land, which can be an important source or sink in many countries, is usually relatively small compared to the land remaining in the same land categories, which makes it difficult to identify them. For example, area of land converted to forest land is not easily estimated with sample-based forest inventories. Therefore, the uncertainty associated with the emissions/removals of these subcategories is significantly higher than for land remaining in the same category.

However, as this report mainly concerns itself with methodological issues for estimating C stock changes, we do not touch this issue in any more detail except that it is noted that stratifying land within any land use or land use change category should be done with respect to possible variations in carbon stock *changes*.

In Table 10 and Annex 9, the most important pieces of information are summarized concerning what is reported and how by the various EU countries with respect to soils. The tables are detailed according to how countries report information in the CRF tables, as well as in the methodological sections of the NIRs. Estimates of removals or emissions are either reported in the CRF tables numerically, or a so called notation key is applied that they are reported in another category (“IE”, i.e. included elsewhere), or that they are not reported (“NO” i.e. not occurring, “NE” i.e. not estimated, or “NA” i.e. not applicable, respectively). The tables here are only meant to demonstrate how often countries are able to report, and in which categories, and not to analyse the emission or removal estimates. The numbers reported in the tables are mean values per unit area, and depend on country specific soil and forest characteristics, but also on artifacts like which categories are included and which are not, and other methodological details. Thus, these numbers are not to be compared between countries or categories.

Table 10 Emissions or removals per unit area for mineral and organic soils for the main land use and land use change categories for the EU countries that submitted CRF tables based on the most recently submitted national inventory reports to the UNFCCC (usually 15 April 2008 submissions). Categories are denoted by abbreviations of the category in the previous year followed by the category in the current year, e.g. FL-FL for forest land remaining forest land, and L-FL for (any) land converted to forest land. L means (any) land, CL is for cropland, GL is for grassland, WL is for wetland, SE is for settlements, and OL is for other land (ie. the land use categories by IPCC). IE means 'included elsewhere', NO means 'not occurring', NE means 'not estimated', NA means 'not applicable'.

Country	Soil type	FL			CL			GL			WL			SE			OL		
		Total	FL-FL	L-FL	Total	CL-CL	L-CL	Total	GL-GL	L-GL	Total	WL-WL	L-WL	Total	SE-SE	L-SE	Total	OL-OL	L-OL
Austria	Mineral	0.134	NO	2.106	-0.320	0.070	-1.009	0.241	-0.011	0.798	-0.557	NE	-3.808	-0.295	NE	-1.128	-4.001		-4.001
	Organic	NO	NO	NO	NO	NO	NO	IE,NO	IE	NO									
Belgium	Mineral	0.043			-0.183									NE				NE	
	Organic	NE			NE			NE											
Bulgaria	Mineral	NE			NE	NE	NE	NE,NO			NE,NO	NE	NE,NO	NE,NO				NE,NO	
	Organic	NE			NE	NE	NE	NE,NO											
Czech Rep	Mineral	0.002		0.157	-0.004	0.001	-0.356	0.099	NE	0.481								NA	
	Organic	0.000		NA,NO	NA,NO	NO	NA,NO	NA,NO	NO	NA,NO									
Germany	Mineral	NE	NE	NE	-0.509	-0.440	-30.780	-0.684	-0.731	11.090	NE	NE	NE	NE	NE	NE	NE	IE,NE	IE,NE
	Organic	NE	NE	NE	NE	NE	NE	NE	NE	NE									
Denmark	Mineral	0.011	NE	0.149	0.029	0.029	NA	IE,NA	IE	NA	0.412	-0.500	0.513	NE	NE	NE	NE	NE	NE
	Organic	NE,NO	NE	NE,NO	-4.451	-4.451	NA	-1.250	-1.250	NA									
Estonia	Mineral	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
	Organic	-1.090	-1.090	NE	NE	NE	NE	NE	NE	NE									
Spain	Mineral	NE,NO	NE	NE,NO	NE,NO	NE	NO	NE,NO	NE	NE,NO	NE,NO	NE	NO	NE,NO	NE	NE,NO	NO	NO	NO
	Organic	NO	NO	NO	NO	NO	NO	NO	NO	NO									
Finland	Mineral	0.061	0.061	IE	0.286	0.286	NE	-3.124	-3.124	NE	-0.066	NE	-2.354	NE	NE	NE	NA,NE	NA,NE	
	Organic	-0.328	-0.328	IE	-4.900	-4.900	NE	-0.250	-0.250	NE									
France	Mineral	0.046	0.012	0.290	-0.147		-1.293	0.220		12.078	NO	NO	NO	-0.591	NO	-0.591	NO	NO	
	Organic	0.000																	
Greece	Mineral	0.000			0.016	0.016	NO				NO	NO	NO	NE,NO	NE	NE,NO	NE,NO	NE,NO	
	Organic	0.000									NE	NE	NE	NE,NO	NE	NE,NO	NE,NO	NE,NO	
Hungary	Mineral	IE,NE,NO	NE	IE,NO	0.000	0.000	IE,NE,NO	IE,NO	IE	IE,NO	IE,NE,NO	NE	IE,NE,NO	IE,NE	NE	IE,NE	IE,NE	IE,NE	
	Organic	NE,NO	NE	NO	NO	NO	NO	NO	NO	NO									
Ireland	Mineral	-0.144	NE	-0.275	-0.038	0.074	-0.677	0.013	NO	0.321	-0.163	-0.163	NE,NO	NO	NO	NO	0.003	0.166	
	Organic	-3.953	NE	-3.953	NO	NO	NO	-0.250	-0.250										
Italy	Mineral	1.019	0.677	33.409	-0.031		-5.785			NO			NO	-0.235	NE	-42.275	NO	NO	
	Organic	NO	NO	NO	-10.00	-10.00	NO	NO	NO	NO									
Lithuania	Mineral	NE	NE	NE	NE	NE	NE	NA,NE	NE	NA,NE	-0.041	-0.041	NA,NE	NA,NE	NE	NA,NE	NE	NE	
	Organic	NA,NE	NE	NE	NE	NE	NE	NA,NE	NE	NA,NE									
Luxembourg	Mineral	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	
	Organic	NE	NE	NE	NE	NE	NE	NE	NE	NE									
Latvia	Mineral	IE,NE	NE	IE,NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	
	Organic	IE,NE	NE	IE,NE	IE,NE	IE	NE	IE,NE	IE	NE									
Malta	Mineral	NA,NE	NE	NA	NE,NO	NO	NE,NO	NE,NO	NE	NE,NO	NO	NO	NO	NE,NO	NO	NE,NO	NE,NO	NE,NO	
	Organic	NA,NE	NE	NA	NE,NO	NO	NE,NO	NE,NO	NE	NE,NO									
Netherlands	Mineral	IE,NE	NE	IE	0.010	NE	0.500	-0.810	-0.894	4.646	NE	NE	NE	0.609	NE	3.272	-0.249	-137.2	
	Organic	IE,NE	NE	IE	IE,NE	NE	IE,NE	IE,NE	IE	IE,NE									
Poland	Mineral	0.426	0.359	1.975	-0.142	-0.142	NE,NO	NE,NO	NO	NE,NO	NA,NE,NO	NA	NE,NO	NA,NE	NA	NE	NE	NE	
	Organic	NE,NO	NO	NE,NO	NE,NO	NE	NE,NO	NE,NO	NE	NE,NO									
Portugal	Mineral	0.013	0.000	0.314	-0.348	-0.030	-1.144	1.154	NO	1.154	-3.015	NO	-3.095	-2.809	-0.268	-2.955	0.228	0.321	
	Organic	NO	NO	NO	NO	NO	NO	NO	NO	NO									
Romania	Mineral	NE	NE	NE	NE	NE	NE	NE	NE	NE	NA,NE	NE	NA,NE	NA,NE	NE	NA,NE	NA,NE	NA,NE	
	Organic	NE	NE	NE	NE	NE	NE	NE	NE	NE									
Sweden	Mineral	0.044	0.044	NA	-0.019	-0.020	NA,NO	0.025	0.032	NA	-0.002	-0.003	NO	NA	NA	NA	NA	NA	
	Organic	-0.022	-0.022	NA	-3.004	-3.004	NA,NO	NA	NA	NA									
Slovenia	Mineral	IE,NO,NE	NE	IE,NE,NO	NE,NO	NE	NE,NO	NE,NO	NE	NE,NO	NE,NO	NE	NO	NE,NO	NE	NE,NO	NO	NO	
	Organic	IE,NO,NE	NE	IE,NE,NO	NE,NO	NE	NE,NO	NE,NO	NE	NE,NO									
Slovakia	Mineral	IE,NO,NE	NE	IE,NO	NE	NE	NE	NE,NO	NE	NO	IE	IE	IE	IE	IE	IE	-0.21	-1.787	
	Organic	IE,NO,NE	NE	IE,NO	NE	NE	NE	NE,NO	NE	NO									
UK	Mineral	0.273	NO	0.433	-0.340	NO	-0.704	0.507	NO	0.507	IE	IE	IE	-1.356	NO	-1.356	NO	NO	
	Organic	0.544	NO	0.544	-2.093	-2.093	IE,NO	-12.950	-12.95	IE,NO									

The most important general conclusions that can be drawn from the tables, from relevant sections of the NIR of the various countries, and also by comparing the estimates in NIR with other estimates, are the following:

- Many countries are still not able to report, or fully report, on soils in most land use and land use change categories.
- In many cases, one reason for this is that land identification in the required IPCC categories is not or not yet possible in the respective countries. This is partly due to the underdeveloped (in terms of sampling density and frequency) land statistical data collection schemes, but also partly that these schemes are not able to differentiate enough the various land uses and land use changes. (It must be noted here that remote sensing methods are only able to detect land cover, which may not directly translate to land use.)
- The reporting on soils is often not transparent enough to evaluate the methods, assumptions and data applied. This often means missing information or inadequate description.
- Most countries apply Tier 1 method in various steps of the estimation.
- The methods applied by the various countries are usually rather different, which is due to the fact that very different databases and monitoring schemes can be found in the EU countries. This makes it difficult not only to compare estimates, but also to harmonize methods at the European scale.
- Many countries rely on IPCC default values.
- There are a few countries that apply soil models for estimating country level emissions and removals. These models include Yasso (European countries), Roth-C (Australia), Century (USA and Japan, also Canada for croplands), and CBM-CFS3 (Canada, for forests).
- In some other cases, further modelling was used to report on the effect of afforestations (Denmark, Hungary). This modelling was done based on case studies, and this modelling was also used in reasoning to conclude that the soil pool is not a source.
- Finally, special models were also used by some countries to develop inventories from available information on land use and carbon stock changes from case studies. However, the description of these specific models may only be found in the respective NIR, and thus some of these models have not been subject to scientific scrutiny.

When evaluating models in country-scale inventories, Peltoniemi et al. (2007b) concluded that:

- model selection is strongly guided by availability of representative input data;
- simple models may be the only reasonable option to estimate soil C changes
- process-based models are needed when soil responses to e.g. management practices are assessed.

A recent meeting of the IPCC (Helsinki, 13-15 May 2008), however, concluded that further guidance is needed to supplement the guidance in IPCC (2006) as to how a model should be described and verified in order that it is scientifically acceptable and transparent. This especially relates to uncertainties associated with the model.

Concerning the application of models for country level reporting in comparison with soil monitoring, Mäkipää *et al.* (2007) found that:

- currently available models can be used in national GHG inventory for estimation of soil C changes;
- soil monitoring with repeated measurement is laborious;

- the minimum number of sample plots for repeated soil measurements is >80 in a cohort of high rate of change, and >20 soil samples per plot are needed for reliable mean estimate of the C stock of organic layer; and
- sampling efficiency can be improved and monitoring costs reduced by using existing networks of measures plots, by increasing sampling interval, or by stratification according to predicted changes of soil C.

It must also be noted that estimates in the NIRs of recent consecutive years have shown an accelerating development. In fact, considerable improvements may have occurred in several countries since the latest reports which served as the basis for this analysis. This is because the first “Kyoto reports” are only due in 2010. A new assessment of the situation will be available at a workshop to be organized by the JRC in November 2008. However, this also means that reported estimates changed from year to year, and may change in the near future, too, and also that reported estimates still considerably differ, at least in case of several countries, from estimates using different methodologies (Janssens *et al.*, 2005).

Uncertainty estimation is rather rare in the NIRs. Even in cases when reporting is based on soil monitoring, the uncertainty estimation was done based on IPCC default values and expert judgement (Austria). However, examples of Tier 2 based uncertainty assessments can also be found in the literature. For example, uncertainty of the change in soil carbon due to afforestation was analysed by Paul *et al.* (2003); uncertainty in a forest carbon budget model by Smith and Heat (2001); and uncertainty of the sinks and stocks of forest soil and vegetation by Peltoniemi *et al.* (2006) and Monni *et al.* (2007). In agricultural soils, uncertainty was estimated by Ogle *et al.* (2003) and Vandengygaart *et al.* (2004). Whereas the uncertainty of carbon stocks are reasonable (i.e. at the order of 30%, e.g. Ogle *et al.*, 2003), that of carbon stock changes can still be as high as 100% or more (e.g. Monni *et al.*, 2007) even for countries with more developed methodology and relatively large resources for monitoring.

Some key elements of uncertainty at the country level, as reported in the NIRs, include:

- The uncertainty associated with applying IPCC defaults to a specific country is unknown (however, applying the same defaults across countries may yield more consistent estimates).
- The uncertainty associated with the application of Tier 1 assumption that there is no change in the carbon stocks (e.g. in forests) can result in underestimating emissions: if change is reported (“estimated”) to be zero, can it happen that there are still emissions due to e.g. forest operations like soil preparation?
- Several countries use various country specific values in their reporting when applying Tier 2 methodology. These country specific values are very rarely compared to IPCC default values. Examples of such comparisons include those from the US where stock change factors associated with management impacts on mineral soils (Figure 13) and carbon loss rates for organic soils under agricultural management (Figure 14) are compared with IPCC default values. Overall, there is a generally good agreement, however, differences up to some 20% do occur.

Figure 13 One-to-one copy of Table A-209 of NIR USA (2008) where US factors are compared to IPCC default values.

	IPCC default	Warm Moist Climate	U.S. Factor		
			Warm Dry Climate	Cool Moist Climate	Cool Dry Climate
Land-Use Change Factors					
Cultivated ^a	1	1	1	1	1
General Uncult. ^{a,b} (n=251)	1.4	1.42±0.06	1.37±0.05	1.24±0.06	1.20±0.06
Set-Aside ^a (n=142)	1.25	1.31±0.06	1.26±0.04	1.14±0.06	1.10±0.05
Improved Grassland Factors ^c					
Medium Input	1.1	1.14±0.06	1.14±0.06	1.14±0.06	1.14±0.06
High Input	Na	1.11±0.04	1.11±0.04	1.11±0.04	1.11±0.04
Wetland Rice Production Factor ^c	1.1	1.1	1.1	1.1	1.1
Tillage Factors					
Conv. Till	1	1	1	1	1
Red. Till (n=93)	1.05	1.08±0.03	1.01±0.03	1.08±0.03	1.01±0.03
No-till (n=212)	1.1	1.13±0.02	1.05±0.03	1.13±0.02	1.05±0.03
Cropland Input Factors					
Low (n=85)	0.9	0.94±0.01	0.94±0.01	0.94±0.01	0.94±0.01
Medium	1	1	1	1	1
High (n=22)	1.1	1.07±0.02	1.07±0.02	1.07±0.02	1.07±0.02
High with amendment ^c	1.2	1.38±0.06	1.34±0.08	1.38±0.06	1.34±0.08

Note: The "n" values refer to sample size.
^a Factors in the IPCC documentation (IPCC 2006) were converted to represent changes in SOC storage from a cultivated condition rather than a native condition.
^b Default factor was higher for aquatic soils at 1.7. The U.S. analysis showed no significant differences between aquatic and non-aquatic soils, so a single U.S. factor was estimated for all soil types.
^c U.S.-specific factors were not estimated for land improvements, rice production, or high input with amendment because of few studies addressing the impact of legume mixtures, irrigation, or manure applications for crop and grassland in the United States, or the impact of wetland rice production in the US. Factors provided in IPCC (2003) were used as the best estimates of these impacts.

Figure 14 One-to-one copy of Table A-211 of NIR USA (2008) where US factors are compared to IPCC default values.

Region	Cropland		Grassland	
	IPCC	U.S. Revised	IPCC	U.S. Revised
Cold Temperate, Dry & Cold Temperate, Moist	1	11.2±2.5	0.25	2.8±0.5 ^a
Warm Temperate, Dry & Warm Temperate, Moist	10	14.0±2.5	2.5	3.5±0.8 ^a
Sub-Tropical, Dry & Sub-Tropical, Moist	20	14.0±3.3	5	3.5±0.8 ^a

^a There were not enough data available to estimate a U.S. value for C losses from grassland. Consequently, estimates are 25 percent of the values for cropland, which was an assumption used for the IPCC default organic soil C losses on grassland.

Conclusions

As a conclusion, reporting on soil carbon stock changes under the UNFCCC and its Kyoto Protocol is quickly changing, but considerable difficulties are still expected in a number of countries. A better and country-specific focus on land use and land use change identification, on identifying relevant sources of emissions and removals by sinks, on extending monitoring programs, on efforts to collect more country specific data from case studies, and on modelling (including both calibration, as well as verification) will be needed in the near future to meet challenges. The main gap in the reporting is data availability, but some methodological problems also remain. Many of these problems are country-specific, and require efforts by countries to put the applied methodologies to scientific scrutiny by the scientific community. However, some of the problems can only be resolved under the auspices of international organizations like the IPCC due to the nature of the methodology that it is to be approved by the international community that has a stake under the UNFCCC process.

7 Analysis of selected EU policies affecting soil carbon stocks

7.1 Introduction

In the countries of the EU, soil management and land use are affected by many different policies, with a wide scope of objectives. Unintended, these policies may therefore affect soil carbon sequestration. It is especially important to identify those policies that may increase soil carbon losses or negatively affect soil carbon sequestration. This section presents an overview of relevant EU policies and their potential effects on soil carbon.

7.2 Common Agricultural Policy

7.2.1 The policy

Council Regulation 1782/2003, covering decoupling of farm payments and cross compliance, and the introduction of the single payment scheme, and Council Regulation 1698/2005, covering rural development, came into force in October 2003, with implementation generally beginning in 2005 and continuing in the following years as different sectors became partially and fully decoupled in the different Member States (MS). Further developments will come about as a result of the CAP 'Health Check' which was finalised in November 2008. The two main elements of CAP reform are decoupled payments (farm payments separated from production) and cross compliance (compliance with legislation being linked to receipt of payments). Only those farm enterprises receiving direct payments are subject to cross-compliance. This would exclude e.g. vineyards, fruit production and sugar production (Hudec *et al.*, 2007).

Historically, the principal aim of the CAP had been to maintain food supply and farm incomes by manipulating producer prices and output through the use of measures such as intervention, import duties, production quotas and set aside. During the 1990s, however, the structure of the CAP was radically changed, away from market intervention price support to payments based on farmed area and livestock numbers, together with the introduction of various rural development and agri-environmental measures. Whilst these policies are likely to influence land use and management in different ways, it is also difficult to disaggregate direct policy effects from the influence of other socio-economic trends. These include, for example, technological change, the effects of world markets and international agreements, changing consumer preferences as well as soil and water quality (Rounsevell *et al.*, 2002).

The Single Payment System (SPS), intended to replace the plethora of agricultural support payments, was to have been introduced in all Member States by 2007. The main aim of the SPS, introduced under the 2003 CAP reform, is to end the link between farm payments and agricultural production. Farmers can decide what to produce in the knowledge that they will receive the same amount of aid, allowing them to adjust production to suit demand and become more market focussed. To receive direct payments, farmers must comply with legislation covering public, animal and plant health, the environment and animal welfare. Under the SPS these pieces of legislation are known as Statutory Management Requirements (SMRs) and must be complied with, where relevant. Farmers must also keep their land in good agricultural and environmental condition (GAEC)

irrespective of whether or not they farm. This requirement is intended to avoid the abandonment of agricultural land and its environmental consequences.

Where farmers fail to comply with GAEC and the appropriate SMRs, the direct payments they can claim are reduced or even withdrawn completely for the year concerned (cross-compliance). Only those farm enterprises receiving direct payments are subject to cross-compliance.

Another aspect of the 2003 CAP reform is the requirement for maintenance of permanent pasture. With some exceptions, Member States must ensure that levels of permanent pasture which existed in 2003 are retained.

In addition to mandatory requirements associated with payment under the SPS, farmers may also sign up to voluntary agri-environment schemes under which payments are made for improved environmental management, such as measures to improve biodiversity conservation, that go beyond legal requirements.

Impacts of the CAP reforms have included a reduction in livestock numbers (reduced intensity because payments are no longer linked to production), reduction in inputs including fertilizers (maximising profit and efficiency of inputs rather than volume), improved environmental practices (because of cross-compliance: SMRs and GAEC), and maintenance of grassland and semi-natural areas and, until recently, the use of set-aside.

7.2.2 Potential effects on soil carbon

There has been a clear decline in the area of grassland in Europe since the 1960s. This is largely a result of the increased production of maize at a time when livestock numbers have reduced due to the implementation of milk quotas in 1984. Since the early 1990s, however, grassland areas have remained fairly stable. Two explanations seem plausible: the 1992 CAP price support reforms and the introduction of agri-environmental and rural development measures. The 1992 MacSharry reforms effectively prevented grassland to arable conversions by fixing the area of land that was eligible for arable area payments. Thus, only land that was in arable production on 31 December 1991 could claim the aid payment. The Less Favoured Areas (LFA) policies have probably contributed to the maintenance of permanent pastures in arid and upland grazing areas. Thus, the policy has effectively maintained the status quo in many grassland areas and one could question what land use would have existed if marginal areas were abandoned or converted to other uses. It is possible that the return of land to natural vegetation types would have led to an increase in C stocks in the biomass whilst soil carbon could have rather decreased (Guo and Gifford, 2002; Jackson *et al.*, 2002; Joaris, 2002; Rounsevell *et al.*, 2002).

The EU report *A long-term perspective for sustainable agriculture* (European Commission, 2003), states that the Commission's proposal to sever the link between production and subsidy ("de-coupling") would favour the extensification of production and would secure significant income gains for EU farmers.

Nevertheless, as indicated above, further extensification of grassland, by reducing grazing pressure and management control, might lead to a decrease in soil carbon. This is because reduced stocking density would lead to a reduced return of organic matter to the soil both in the form of excreta directly voided to the land during grazing and due to the reduction in the amounts of manures available for spreading. Moreover, the point needs to be borne in mind that de-coupling has led to producers' decisions being driven by market considerations rather than by the maximisation of farm subsidies. This makes the impacts of CAP reform on soil management difficult to predict.

With respect to soil as a CO₂ sink, a number of SMRs, the requirement to maintain land in GAEC and the requirement to maintain levels of permanent pasture will all improve soil structure and maintain organic matter in soil, which will in turn lock-up atmospheric carbon. Five directives are included as environmental SMRs: the Birds Directive (79/409/EEC), Habitats Directive (92/43/EEC), Groundwater Directive (80/68/EEC), Nitrates Directive (91/676/EEC), and Sewage Sludge Directive (86/278/EEC). From a soil carbon perspective, the two most likely to have an impact on soil organic matter are the Sewage Sludge Directive and the Nitrates Directive, both of which contribute to the maintenance of soil organic matter through regulation and control of spreading of sewage sludge (where applied) and organic fertilizers (in NVZs) (Hudec *et al.*, 2007).

The EU PICCMAT project (<http://climatechangeintelligence.baastel.be/piccmat/index.php> ; Leipprand *et al.*, 2007) assessed the likely impact on soil C from range of EU measures. The aim of that project was to review different EU policies that may affect emissions of greenhouse gases (GHG) from agriculture. The authors report that de-coupling is also expected to have a beneficial effect on agricultural GHG emissions, since it removes or reduces incentives for intensive production. This recent CAP reforms made the agricultural sector more responsive to the market, so farmers are likely to react more strongly to non-policy signals in the future. If market signals were to change, this might also lead to increases in emissions. Hence should market signals favour arable crops over livestock products there could be conversion from grassland to arable'.

The potential impact of CAP reform on agricultural GHG emissions is summarized below in Table 11 from Leipprand *et al.* (2007).

Table 11 CAP reform measures and assumed impact climate-related characteristics of farm systems in Europe

Measure	Expected impact
Decoupling	Reduction of incentives towards intensive production (e.g., extensification, livestock, reduced fertilizer use) Farmers more responsive to non-market signals
Modulation: Reduction in direct payments. Amounts transferred to rural development	Increased budget for rural development --> Stimulate the adoption of environmentally friendly production techniques
Cross-compliance: Direct payments conditional to the respect of Statutory requirements from 19 Community Acts, including 5 environmental Directives Maintenance of agricultural land in GAEC Maintenance of permanent pastures.	Incentive to comply with statutory environmental requirements, e.g. Nitrates Directive (reduced fertilizer use + improved practices) GAEC --> soil conservation, e.g. improved management of soil organic matter (crop rotation, reduced tillage) reduced soil erosion
Set aside: Maintenance of individual historical set-aside obligation (10 %), Maintenance in GAEC	Less fertilizer use Potentially increased carbon sequestration, in particular long term non-rotational set aside
Non-food (energy) crops can be grown on set aside land Energy crops support	Carbon substitution potential: promotion of biofuels (however, GHG may be released when converting long-term grassy set-aside back into crop land)

Thus the aspect of CAP reform that could have led to a decrease in soil carbon was the provision that set-aside could be used for the cultivation of energy crops. However, this is no longer an option as set aside was abolished as part of the 2008 Health Check

Smith *et al.* (2007a) estimated that set-aside in dry climates could mitigate GHG's by 3.93 (-0.07 - +7.9) t CO₂-eq. ha⁻¹ yr⁻¹, comprising an increase in SOC of 1.61 (-0.07 - +3.3) t CO₂-eq. ha⁻¹ yr⁻¹, and decreased N₂O emissions of 2.3 (0.0 - 4.6) t CO₂-eq. ha⁻¹ yr⁻¹. In moist climates the estimates were 5.36 (1.17 - 9.51) t CO₂-eq. ha⁻¹ yr⁻¹, comprising an increase in SOC of 3.04 (1.17 - 4.91) t CO₂-eq. ha⁻¹ yr⁻¹, and decreased N₂O emissions of 2.3 (0.0 - 4.6) t CO₂-eq. ha⁻¹ yr⁻¹. King *et al.* (2004) argued that partial set-aside by expanding field margins could play an important role in mitigating GHG emissions from English agriculture, sequestering the equivalent of 0.95 - 1.22 t C ha⁻¹ yr⁻¹ via increases in SOC, and reductions in energy usage and other GHG emissions. Land that is set aside should, however, be vegetated, as leaving it fallow may reduce C stocks by 0.2 t ha⁻¹ yr⁻¹ (Arrouays *et al.*, 2002).

Armstrong-Brown *et al.* (1996), cited in Storey (1997), studied the impacts of set-aside policies on carbon fluxes in the UK. They estimated that for land set aside from cereal, oilseed and protein crop production, 469,000 tonnes C per year could be being sequestered (although limitations to these estimates were noted). It should be noted, however, that the EU set-aside policies placed constraints on how set aside land could be used, restricting their potential benefits. The rotational nature of the set-aside policy meant the afforestation of set-aside land was impossible if this land was to be eligible for arable area payments. The abolition of permanent set-aside, if the land is put back into production under tillage crops, will lead to a reduction in soil C and hence emissions of CO₂.

Leipprand *et al.* (2007) quote the Scenar 2020 study (Nowicki *et al.*, 2007), which develops projections for the development of European agriculture based on different socio-economic scenarios, also expects that livestock numbers will continue to decrease, due to a decline in beef production on the one hand, but also an increase in productivity on the other hand. The study also predicts fertilizer use to decrease in the EU-15, although it seems to be unclear whether increasing demand for biofuels might change this trend. While there are considerable uncertainties attached to the above conclusions, two possibilities arise which could lead to reductions in soil C. First, a reduction in beef production may lead to surplus grassland being converted to tillage land with consequent reductions in soil C. Second, an increase in the use of fertilizer-N might also lead to a decrease in soil C.

This latter conclusion is at variance with earlier views that the use of fertilizer-N, by increasing crop yields, and hence crop residue returns, can lead to an increase in soil C, or at least moderate the decline that takes place as a result of tillage. The potential impact of changing N inputs on soil C is discussed in section 2.2.3.3 above. It is difficult to draw firm conclusions but it seems sensible to suggest that for agricultural soils:

- previous assumptions that fertilizer-N can increase C sequestration may not be valid;
- increases in crop yields from fertilizer-N do increase the active fraction of SOM, the fraction that is involved in nutrient turnover, aggregate formation and is associated with soil fertility;
- but increases in active SOM may take place while total soil C is decreasing.

Hence measures to reduce N fertilization in order to reduce pollution of watercourses might lead to some increases in SOC, rather than small decreases as had previously been feared.

The removal or reduction of production-related agricultural subsidies may have beneficial impacts in terms of increasing the potential for carbon sequestration through allowing for the reversion of some agricultural land into more natural eco-systems and also the conversion to other

land uses such as forestry. This was, at least initially, the experience in New Zealand, where the removal of subsidies removed an important incentive to farm marginal land, much of which has reverted to natural ecosystems or was being planted to forestry (Shepherd, 1996). On the other hand, the removal or reduction of agricultural subsidies could have negative impacts if it were to lead to agricultural farmland being converted to urban development uses and therefore reducing its function as a sink of CO₂ (Storey, 1997).

There is a strong interrelationship between the CAP and the Renewable Energy Directive in that the latter lead to the establishment of markets for sustainable fuel and energy, but it is the CAP itself which will drive the cultivation of crops to be used for bioenergy production, as an alternative to imported palm oil and other materials. Even though the vast majority of CAP payments have now been decoupled, some measures under the rural development strand of the CAP are specifically aimed at stimulating the cultivation of energy crops, as well as the establishment of small-scale renewable energy micro-generation. Nevertheless, it may be argued that it is the RES Directive, with its targets, that will drive biofuel production, and not the CAP, which merely sets the framework conditions for crop (and biofuel) cultivation

Under Council Regulation 1782/2003 an energy crops scheme was introduced to encourage the cultivation of energy crops; previously farmers had been allowed to grow them on set aside land but there was no specific support. The regulation set an area payment of €45/ha for land on which crops were grown for the purposes of biofuel production and biomass production for heat and electricity generation. This has now been abolished as part of the 2008 CAP Health Check.

Measures to address climate change and renewable energy priorities are to be incorporated within Member States rural development programmes from 2010. Such measures may include schemes to encourage biogas production, the cultivation of perennial energy crops (such as short rotation coppice and *miscanthus*) and the development of processing installations and infrastructure for the production and distribution of energy produced from biomass.

7.2.2.1 Cross Compliance - GAEC

The PICCMAT report (Leipprand et al., 2007) states that 'Annex IV of Regulation 1782/2003 however specifies as a basic framework that standards must cover soil erosion, soil organic matter content, soil structure and a minimum level of maintenance. A further requirement provided by the regulation is that land under permanent pasture must be maintained as such or at least the total area of permanent pasture within a member state must not decrease.'

Thus it appears that under cross compliance there is provision to ensure that in response to CAP reform etc., there should not be an overall conversion from permanent pasture to tillage land. However, while in the case of non-compliance farmers will be sanctioned through reductions of their direct support (and hence the cross compliance instrument is expected to lead to a greater level of compliance with existing regulations), if market prices are sufficiently favourable farmers may decide to forego income support if there appears to be sufficient returns from tillage crops.

An assessment of the potential impacts of CAP reform is likely to become somewhat more complicated in view of forthcoming, and as yet unconfirmed, changes to the CAP under the Health Check. While proposals for this were submitted in 2007 final outcomes to the subsequent consultation are still unknown and may have a further impact on GAEC, SMRs and set aside, thus affecting soil C levels. Furthermore, socio-economic changes under the Health Check (such as phasing out milk quotas) may have an indirect impact on land use and management.

Maintaining land in GAEC includes requirements to reduce soil erosion, maintain soil organic matter and soil structure, and adopt a minimum level of maintenance. All of these elements of soil conservation will contribute to maintaining soil C level but, of course, the most influential will be the requirement to maintain soil organic matter. The appropriate practices for doing this are the establishment of standards for crop rotations and managing arable stubbles. Measures to achieve the latter include, for example, the banning of stubble burning (adopted in 13 MS) and the management of crop remains or incorporation of organic manure adopted as a measure by a number of other MS. Alongside appropriate crop rotations, measures such as soil organic matter/humus analysis, the use of cover crops and incorporation of leguminous plant or other organic materials into the soil have also been adopted in Member States (Hudec *et al.*, 2007).

Other measures associated with achieving other GAEC soil protection objectives will also contribute positively to soil C levels; for instance, the use of cover crops is used in various Member States to help maintain soil structure and to prevent erosion and this can increase soil C sequestration, at least in the short term. Twenty Member States have adopted measures to protect permanent pasture as a way of fulfilling the objective of setting a minimum level of maintenance, and, again, this will benefit soil C levels.

7.2.3 Conclusions

It is difficult to accurately summarize the impacts of such a far-reaching policy, but it appears to have effectively maintained the status quo in many grassland areas albeit one could question what land use would have existed if marginal areas were abandoned or converted to other uses. The requirement to maintain land in GAEC and to maintain levels of permanent pasture will all improve soil structure and maintain organic matter in soil, which will in turn lock-up atmospheric carbon. In contrast, the abolition of permanent set-aside, if the land is put back into production under tillage crops, will lead to a reduction in soil C and hence emissions of CO₂.

However, further extensification of grassland, by reducing grazing pressure and management control, might lead to a decrease in soil carbon, while de-coupling has led to producers' decisions being driven by market considerations rather than by the maximisation of farm subsidies. A reduction in beef production may lead to surplus grassland being converted to tillage land with consequent reductions in soil C. The removal or reduction of production-related agricultural subsidies may have beneficial impacts in terms of increasing the potential for carbon sequestration through allowing for the reversion of some agricultural land into more natural eco-systems and also the conversion to other land uses such as forestry.

7.3 Nitrates Directive

7.3.1 The policy

The Directive came into force in 1991. Implementation by Member States was due by 20 December 1993 although the degree and date of implementation vary between MS. The purpose of the Nitrates Directive is to protect human health and aquatic ecosystems and to safeguard other legitimate uses of

water, by reducing current and preventing future water pollution caused or induced by nitrates from agricultural sources. The Directive aims to mitigate the negative effects of nitrogen (N) fertilization on drinking water sources and ecosystems by limiting the input of inorganic N fertilizers and manure on farmland. Other aspects include requirements for manure and slurry storage and rules covering certain land management practices.

The Nitrates Directive requires Member States to identify and designate Nitrate Vulnerable Zones (NVZs) and to draw up Action Programmes to reduce nitrate pollution in surface and ground waters in these areas. Member States also have the option of declaring their whole territory as a NVZ. Farmers located in NVZs are required to comply with Action Programme measures to reduce nitrate leaching. The main impacts are on application of N fertilizers and manure management practices. There have also been initiatives to increase the area of unfertilized grassland, but these have not always been continued.

The Directive requires M S to establish standards and codes regulating the following issues:

- Periods during which the application of N fertilizer is limited/prohibited
- Crop requirement limits must be respected by not applying more N than a crop requires, taking account of crop uptake, soil N supply, excess winter rainfall, and plant or crop available N from organic manures.
- On top of this, specific limits for N applications (kg/ha) from manures are also set, on a field or farm basis, or both.
- N fertilizer and organic manures should be spread as evenly and accurately as possible.
- Application of manures or N fertilizers on waterlogged, flooded, frozen or snow covered ground is prohibited.
- Application of manures or N fertilizers to steeply sloping fields and in the vicinity of watercourses is prohibited.
- Sufficient manure storage facilities (or alternative arrangements) – storage capacity must exceed that required for storage throughout the longest period during which land application in the vulnerable zone is prohibited.
- Farmers must keep farm and field records on cropping, livestock numbers, N fertilizer usage and manure usage, for a minimum of five years after the relevant activity takes place.

At EU 15 level, the reduction recorded in the period 2000-2003 compared with the previous period 1996-1999 was 6% for N and 15% for phosphate fertilizers respectively, with downwards trends continuing also in 2004 and 2005.

7.3.2 Potential effects on soil carbon

The most likely concern with respect to these measures is that any reduction in fertilizer-N inputs, by reducing crop yields and hence returns via crop residues, might lead to a long-term decline in SOM. However, a modelling study by Webb *et al.* (2003), which assessed the potential impacts of reductions in fertilizer-N applications on soil C in the UK, estimated only very small differences in soil C over the next 100 years from differences in fertilizer-N strategies. More recent work (Kahn *et al.*, 2007) reports that N fertilizer may also lead to increased mineralization of SOM and hence reduce the potential for C sequestration. The results of this paper are discussed under CAP reform. In summary none of these measures are likely to have a deleterious impact on soil carbon and the direction of any impact is also uncertain.

7.3.3 Conclusions

Conventional wisdom may suggest that limiting the addition of N fertilizer may play a role in reducing soil C contents, although this effect may be negligible at the European scale since many soils are N-saturated. However, more recent studies suggest that reducing fertilizer-N inputs, to avoid excess N fertilization, may also preserve SOC.

7.4 *Renewable Energy Sources and Biofuels Directives*

7.4.1 The policy

Directive 2001/77/EC on the promotion of electricity from renewable energy sources in the internal electricity market (Renewable Energy Sources or RES Directive) requires Member States to commit to specific targets for the use of energy from renewable sources for electricity production. These targets must be consistent with the global indicative target of 12 % of gross national energy consumption by 2010 and in particular with the 22.1 % indicative share of electricity produced from renewable energy sources in total Community electricity consumption by 2010. Allowing discretion for Member States to set their own targets means that renewable energy will play a larger role in some energy markets than others; in the UK, for example, the target set is 10% (BERR, 2008). This compares with a target of 3.6% by 2010 for Hungary and, at the other end of the scale, 78% for Austria (European Commission, 2008a).

In addition, it is important to underline that the RES Directive defines 'biomass' as including 'the biodegradable fraction of municipal and industrial waste' (Article 2(b)). This has an important bearing on the way in which biodegradable waste is handled, in the sense that the production of energy from it through incineration gets a premium over other forms of management, for instance composting. The production of compost and its use on land could contribute to the maintenance or increase of SOC in EU soils, especially in those regions that do not have an easy access to other forms of organic soil improvers, e.g. manure.

Liquid biofuels were not a part of the RES Directive, as they were subject to Directive 2003/30/EC on the promotion of the use of biofuels or other renewable fuels for transport (Biofuels Directive), which set a target of 5.75% of all petrol and diesel for transport placed on the market by 31 December 2010 as biofuels. Member States were required to set indicative targets for 2005, taking a reference value of 2% into account. This interim indicative target has not been achieved. Biofuels accounted for 1% of transport fuel in 2005. The Commission's conclusion according to the assessment of the progress is that the target for 2010 is not likely to be achieved – expectations are for a share of about 4.2%.

Recognising that the Biofuels Directive in its current form was unlikely to provide the necessary impetus for the EU to reach the 2010 target of 5.75% market share, the European Commission recently published proposals to reinforce the existing legislative framework. The proposal for a Directive on the promotion of the use of energy from renewable sources⁵ (Renewables Directive) incorporates elements of the Biofuels Directive and aims to establish an overall binding

⁵ COM(2008)19, 23.1.2008.

target of a 20% share of renewable energy sources in energy consumption and a 10% binding minimum target for biofuels in transport to be achieved by each MS, as well as binding national targets by 2020 in line with the overall EU target of 20% (European Commission, 2008a). The Commission also recognizes that some practices in biofuels production can lead to less-than-expected reductions of carbon emissions and to environmental problems. The Commission proposes the introduction of an incentive/ support system to avoid this and to encourage the development of second generation biofuels.

7.4.2 Potential effects on soil carbon

The current RES Directive has set targets for the proportion of electricity produced in the EU from renewable sources, which include the use of biomass as well as wind and water to produce energy. Electricity produced from solid biomass in the EU has seen considerable growth in recent years, with a 16.2% increase between 2004 and 2005. However, biomass used in electricity production will come from a variety of sources and will include by-products (such as straw) as well as crops grown specifically for electricity production, such as short rotation coppice.

The new Renewables Directive will strengthen existing targets to the extent that by 2020, 20% of Europe's energy is provided by renewable energy sources, and for transport fuels that figure is 10%. The Directive will set up a system to ensure the environmental sustainability of biofuels production; however, like the current RES Directive, there is no explicit requirement for increases in cultivation of crops for the production of biofuels and biomass for renewable energy production, since the raw materials could be produced as a result of increased productivity. Such a scenario has been suggested in central European countries in which productivity has declined following the abolition of subsidies for inputs such as fertilizers (Smeets *et al.*, 2004). However, Searchinger *et al.* (2008) argued that in practice farmers will replace most of the crops diverted from food and feed to biofuels because the demand for overall food and feed, as opposed to any particular crop, is inelastic. Searchinger *et al.* (2008) also pointed out that if there is surplus agricultural land such land could be better used to sequester carbon by reversion either to woodland or grassland. Use of such surplus land for biofuels rules out the opportunity for sequestration, which could exceed the carbon saved by using the same land for biofuels. However, Wang and Haq (2008) considered the conclusions of Searchinger *et al.* (2008) may have under-estimated the potential for increased yields of feedstocks as well as under-estimated the extent to which residues from biofuel production could substitute for existing production of livestock feed. Wang and Haq (2008) concluded that that indirect land use changes are much more difficult to model than direct land use changes. To do so adequately, researchers must use general equilibrium models that take into account the supply and demand of agricultural commodities, land use patterns, and land availability (all at the global scale), among many other factors. Efforts have only recently begun to address both direct and indirect land use changes, and it is not clear what land use changes may occur globally as a result of increased biofuel production, although the Renewables Directive may provide further impetus to an increase in crop cultivation.

By raising the value of agricultural land, and increasing the returns to agricultural production in relation to alternative land uses, agricultural support policies result in a cost in terms of carbon sequestration potential foregone. Lippert and Rittershofer (1996), for example, identified European Union agricultural support policies as being the principal factor inhibiting afforestation in the Saxony region of Germany. The EEA make the point that, in line with the cross-compliance

objectives agreed in the last reform of the Common Agricultural Policy, grassland should not to be transformed into arable land. This avoids a release of CO₂ from grassland soils that would occur when such land is ploughed.

Banse and Grethe (2008) used the ESIM model to forecast the impact of a 10% target for biofuels on production and demand for biofuels within the EU. Two policy scenarios were simulated up to 2020: a baseline under which the share of biofuels in total transport fuels increases to 6.9% by 2020, and a scenario with more demanding legislation resulting in a 10% share. Results indicate that a substantial part of the policy-induced demand for biofuels is likely to be met by imports of biofuels and biofuel substrates, especially following the implementation of a potential Doha agreement. In particular, imports of plant oils were forecast to increase. EU production of bioethanol was forecast to decrease substantially, while almost all bioethanol demand was forecast to be met by imports. The authors acknowledge that technological developments could alter their conclusions. Their analysis was based on first-generation technologies for biodiesel and bioethanol production. Second-generation technologies, such as biomass-to-liquids or cellulose conversion into sugars, could result in greater yields per ha and provide the option to use land which is not suited, or is only poorly suited, for the production of food crops. Other studies have indicated that increased crop production on degraded land may lead to increased C sequestration. No mention is made of the implications for land use in the exporting countries.

With respect to the production of plant oils for biofuels, globally there are particular concerns about biofuel production from palm oil. Palm oil expansion, and in particular the increase in concessions granted for palm oil, is reported as one of the leading cause of deforestation in Indonesia and Malaysia (Nellerman *et al.*, 2007). After logging, palm oil and timber plantation are the most important drivers behind the destruction of peatland forests. In addition to the carbon stored in the trees, large amounts of carbon are stored in the peat. Plantation establishment requires intensive drainage which, together with timber burning, lead to the largest CO₂ emissions of any land use change (Hooijer *et al.*, 2006). Palm oil production has also been linked to large-scale deforestation in Colombia, Ecuador, Brazil, Central America, Uganda, Cameroon and elsewhere (Boswell *et al.*, 2007).

The authors conclude that 'In the long run, the political perspective for biofuels in the EU is questionable. In light of the increasing evidence of the arbitrary environmental effects of first-generation biofuel production in the EU and the inefficiently high cost of GHG mitigation through biofuel production, political support may cease'.

Legislation to encourage the production of arable crops to provide feedstocks for biofuels is perhaps the legislation most likely to lead to decreases in the overall carbon content of European soils if they are grown on land which was previously uncultivated. However, if food and livestock feed crops are replaced with those grown for biofuels there would not be any change in soil C in European soils, although food and feed crops so displaced would need to be imported from outside the EU. Alternatively, some studies indicate that much of the demand for biofuels may be met by imports from outside the EU, rather than by domestic production. In either eventuality, there may be serious implications for soil C stocks in those countries which either supply a greater proportion of Europe's food or the biofuels or their substrates.

7.4.3 Conclusions

The proposed Renewables Directive is the one most likely to have adverse effects on SOC if it leads to the conversion of grassland to arable cropping in order to produce feedstocks for biofuels. Given the low elasticity of demand for food and the increasing global population, any significant reduction of food production is unlikely (it would only be possible if more people went undernourished), hence the pressure to convert grassland within the EU to arable land (experience so far indicates that agricultural expansion takes place first on fertile, productive lands rather than on degraded land).

More worrying, if the EU biofuel requirement is, as forecast, met by increased imports, there are serious implications for soil carbon stocks in the exporting countries. For example, production of soybeans in Brazil and palm oil in Southeast Asia have expanded largely at the expense of tropical forest, taking advantage of the fertility arising from mineralization of soil organic matter following land use change. If biofuels policy results in reduction of SOM, it will take many years or decades of biofuel production for the overall carbon balance to become positive. In other words, the increased production of biofuels will result in a significant surge of GHG emissions in the near future, which may then be compensated by the “savings” from the eventual production and use over the coming decades. The GHG balance would be negative for decades. This is very important for two reasons: First, it is widely recognised that early mitigation action contributes more to stabilisation than actions in the future. A policy that promises modest future savings at the expense of a significant increase short-term emissions is questionable. Second, future GHG savings depend on the assumptions that the use of first-generation biofuel crops would indeed make sense even decades from now. Hence, if more efficient alternatives become available (e.g., second-generation fuels or alternatives to the internal combustion engine), then savings may never materialise, but the initial emissions will have arisen.

In addition, it should be recalled that the definition of 'biomass' in the current RES Directive presents the risk of supporting the production of energy through the incineration of biowaste rather than a return of such waste, under controlled conditions, for maintaining or enhancing SOC levels.

7.5 Waste Policy

7.5.1 The policy

European legislation to manage waste was first introduced in 1975, with the Waste Framework Directive 75/441/EEC and the Hazardous Waste Directive 91/689/EEC which put in place the basis of the regulatory structure on waste. However, these pieces of legislation did not touch on the issue of emissions from waste management facilities and processes and led to problems associated with pollution from landfill and incinerators. These problems began to be addressed by the Landfill Directive 199/31/EC and the Waste Incineration Directive 2000/76/EC, which both set standards for pollution into the air or groundwater.

In November 2008 the revised Waste Framework Directive 2008/98/EC was published in the Official Journal of the European Union, replacing the previous version introduced in 1975. It will also streamline other EU waste legislation by replacing two additional directives: the Hazardous Waste Directive and the Waste Oils Directive. The new directive will stand as the central pillar of EU waste management policy and it also represents a shift in thinking from waste as a burden to a

potentially valuable resource. The directive sets new recycling targets for households and the building industry as well as strengthening the emphasis on waste prevention. The legislation also reinforces the five-step 'hierarchy' of waste management options of which prevention is the preferred, followed by reuse, recycling, other forms of recovery and with safe disposal as the last recourse (European Commission, 2008b). The Sewage Sludge Directive 86/278/EEC, which aims to protect the environment when sewage sludge is used in agriculture as a fertilizer and to improve soils, is referred to under the CAP section (5.1.1).

The aim of the Waste Incineration Directive is to prevent or to reduce as far as possible negative effects on the environment caused by the incineration and co-incineration of waste. In particular, it should reduce pollution caused by emissions into the air, soil, surface water and groundwater, and thus lessen the risks which these pose to human health. This is to be achieved through the application of operational conditions, technical requirements, and emission limit values for waste incineration and co-incineration plants within the Community.

The aims of the Landfill Directive were 'to prevent or reduce as far as possible negative effects on the environment, in particular the pollution of surface water, groundwater, soil and air, and on the global environment, including the greenhouse effect, as well as any resulting risk to human health, from the landfilling of waste, during the whole life-cycle of the landfill' (European Commission, 1999). The directive covers the location of landfill sites as well as management of leachate into soil and water. It also sets targets to reduce the amount of municipal biodegradable material that is landfilled. These targets are:

- By 2010 to reduce biodegradable municipal waste landfilled to 75% of that produced in 1995;
- By 2013 to reduce landfill to 50%;
- By 2020 to reduce landfill to 35%.

The directive also required Member States to set up national strategies for reducing the quantity of biodegradable waste going to landfills, such as recycling, composting, biogas production or materials/energy recovery.

The new Waste Framework Directive includes targets for re-use and recycling of materials of 50% for paper, metal and glass from households and 70% for construction and demolition waste. Waste prevention is strengthened with Member States being obliged to establish waste management and prevention programmes. There will also be a target to reduce incineration and landfill even though incineration will be classified as 'recovery' rather than 'disposal' where incineration is used in an efficient way to generate usable energy.

7.5.2 Potential effects on soil carbon

The waste hierarchy, reinforced under the new Waste Framework Directive, includes composting as a method of recycling organic material as an alternative to disposal. The end product, compost, then becomes a useful soil conditioning and fertilizing material which has the potential to replace lost carbon from the soil. Provisions are also made in the directive to ensure the protection of soils, as well as water air and wildlife.

Prior to the proposed new Waste Framework Directive, the increased emphasis on alternative methods of disposal, or recovery, of organic wastes required under the Landfill Directive lead to an increase in the use of compost as a disposal mechanism for biodegradable products, and hence lead to increased production of a valuable soil-improving material. The Commission envisaged for a

while to present a directive on the management of biological waste to encourage composting, but it has so far not emerged. However, Article 22 of the Waste Framework Directive calls on the Commission to 'carry out an assessment on the management of bio-waste with a view to submitting a proposal if appropriate. The assessment shall examine the opportunity of setting minimum requirements for bio-waste management and quality criteria for compost and digestate from bio-waste, in order to guarantee a high level of protection for human health and the environment'. It is therefore to be expected that in future the Commission will consider possible further actions to support composting and digestion of biowaste across the EU.

The Waste Incineration Directive applies to incineration and co-incineration plants. Co-incineration plants include facilities where waste is used as a fuel or is disposed of at a plant where energy generation or production is the main purpose. The directive has little *immediate* influence on soil C, as it is concerned with emission limit values rather than influencing the amounts of organic waste to be incinerated. However, changes in those amounts could have an impact on SOC by diverting organic waste either away from or toward land application. Indeed, the support given to energy production by the Renewable Energy Sources (RES) Directive 2000/77/EC (see section 6.4) could result in waste incineration being promoted over waste recycling, thus limiting the amount of biowaste composted and its use as soil improver for maintaining or enhancing SOC levels.

The Waste Incineration Directive has a possible indirect effect in improvements in air quality, as it will reduce the impact of ozone (O₃) on plant growth by reducing emissions of O₃ precursors and hence remove a constraint to plant growth, carbohydrate assimilation, and return of organic matter to soil.

7.5.3 Conclusions

By encouraging the use of composting as a valid waste recovery option, less organic waste will be sent to landfill; rather, more will be composted with the resultant compost material becoming available for land spreading. However, the potential impacts of this policy for soil organic carbon across the EU are likely to be limited, especially because there is no obligation for the production of high quality compost across the EU. There appear to be little or no likely direct consequences of the implementation of the Waste Incineration Directive for SOC, although the RES Directive tends to promote energy production over material recycling with potential negative effects in terms of the return of SOM to the soil.

7.6 *EU Thematic Strategy for soil protection*

7.6.1 The policy

Current provisions in favour of soil protection are spread across many areas, and are designed in many cases to safeguard other environmental media or to promote other objectives. They do not therefore constitute a coherent soil protection policy. Even if exploited to the full, existing policies fall a long way short of covering all soils and all the threats to soil identified. Hence the need for a coherent strategy to assess and revert soil degradation.

The aim of the Thematic Strategy for soil protection, announced in 2006, is 'to ensure that Europe's soils remain healthy and capable of supporting human activities and ecosystems' (European

Commission, 2006a). The legislative proposal for a Soil Framework Directive (European Commission, 2006b), accompanying the strategy, will oblige Member States to tackle threats such as landslides, contamination, soil erosion, the loss of soil organic matter, compaction, salinisation and sealing wherever they occur, or threaten to occur, on their national territories. However, the strategy allows for flexibility within Member States to set objectives and targets nationally due to the varying nature of soil degradation across all the EU.

7.6.2 Potential effects on soil carbon

An impact assessment was carried out for the proposed Soil Framework Directive (SEC(2006)620). The assessment lists several beneficial effects for climate to be expected from anti-erosion practices, practices to avoid loss of organic matter, and practices to avoid compaction: a reduction in CO₂ and other GHG emissions due to less machinery use (reduced tillage) and reduced stocking rates, and contributions to carbon sequestration. The PICCMAT study (Leipprand et al., 2007) concluded that most, if not all, the measures proposed have the potential to increase C sequestration. However, it should be noted that the Directive has not yet been adopted, let alone implemented.

7.7 Other policies and legislation

A number of other pieces of legislation that could have an impact on soil carbon levels were considered for this study. The Water Framework Directive (Dir 2000/60/EC) and the Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EC) were both looked at but it was agreed that the impacts of both on soil carbon levels would be negligible. However, as with the Nitrates Directive and CAP reform, the encouragement of extensification (under both policies) may lead to reductions in SOC.

Other areas of legislation which may impact land management and soil C levels indirectly relate to livestock and, specifically, animal health policy. How governments react to disease, bovine TB, Foot and Mouth (F&M) and bluetongue, is going to have an impact on that particular livestock sector, and hence numbers of livestock. At one extreme, the UK F&M outbreak of 2001 led to a large decrease in livestock numbers. While in some areas this was only relatively short-term, in others less so. However, disease pressures, together with rising financial stress facing some sectors of the agricultural economy could result in an increase in ungrazed pasture. Combined with the recent doubling of grain prices, an increase in arable area is likely to arise. These factors, however, are likely to be localised and very much depend on Member States attitudes to disease control.

7.8 Assessment

Table 12 presents a qualitative assessment of the potential impact of EU policies on soil carbon. Legislation to encourage the development of markets for renewable energy as well as rural development measures which stimulate the cultivation of energy crops to provide feedstocks for biofuels are the policies most likely to lead to decreases in the overall carbon content of European soils. Taken with other elements of CAP reform, which is expected to lead to further decreases in the numbers of grazing animals, may lead to a decrease in grassland if such land is converted to arable cropping, especially in the absence of set aside.

Table 12 Overview of potential impacts of EU policies on carbon sequestration

Policy	Measure	Impacts	Qualitative assessment of effect on soil C	Uncertainty	Level of Agreement	Remarks
CAP reform	Decoupled payments	Fewer livestock	Neutral to potentially negative. Negative impacts could be large if large areas of grassland converted to arable	Significant uncertainty on the impacts of greater exposure to market forces on livestock industry	Limited	Greater exposure of farmers to the market could lead to increased arable production and decreases in soil C
		Cross-compliance	Reduced inputs	Neutral to potentially negative. But while the direction of change is uncertain, the impacts are likely to be small	Significant uncertainty on the impacts of increases or decreases in use of n fertilizer on soil carbon.	Limited
	GAEC		Maintain or increase	Some	Good	
	maintenance of grassland abolition of set-aside		Maintain or increase Negative	Little Moderate	Good Good	
	Increased biofuel cultivation	Increased arable cropping and SRC	Negative	Little	Good	
Nitrates Directive	N application	Less N applied	May increase or decrease soil C	Significant	Limited	See CAP reform
Renewables Directive	Increased biofuel demand	Increased arable cropping and cultivation of perennial energy crops (in conjunction with CAP)	Potentially large decreases in soil C	Moderate	Reasonable	Outcome depends on balance between arable and permanent crops
	Support to energy production from biowaste	Incentive to incineration rather than composting	Negative	Significant, as it depends on decisions to be taken by each Member State	Limited, as it is argued that energy production is beneficial in GHG terms	Energy production from anaerobic digestion followed by composting would be a win-win option
Waste policy	Emission limits Increase in composting	None Moderate increase in compost additions to soil	None Potential small increase in soil C	Little	Good	The potential amounts of compost are small in comparison with the amounts of livestock manures already applied to land
Framework EU strategy for soil protection	Protect soil organic matter	Should maintain if not enhance SOM	Maintain or increase soil C	Some		Subject to adoption of the Soil Framework Directive by the EU

Glossary

Abatement

Refers to reducing the degree or intensity of greenhouse-gas emissions.

Adaptation

Adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities.

Afforestation

Planting of new forests on lands that historically have not contained forests.

Annex I Parties

The industrialized countries listed in this annex to the Convention which were committed return their greenhouse-gas emissions to 1990 levels by the year 2000 as per Article 4.2 (a) and (b). They have also accepted emissions targets for the period 2008-12 as per Article 3 and Annex B of the Kyoto Protocol. They include the 24 original OECD members, the European Union, and 14 countries with economies in transition. (Croatia, Liechtenstein, Monaco, and Slovenia joined Annex 1 at COP-3, and the Czech Republic and Slovakia replaced Czechoslovakia.)

Annex II Parties

The countries listed in Annex II to the Convention which have a special obligation to provide financial resources and facilitate technology transfer to developing countries. Annex II Parties include the 24 original OECD members plus the European Union.

Anthropogenic greenhouse emissions

Greenhouse-gas emissions resulting from human activities.

Biomass

The total mass of living organisms in a given area or volume; recently dead plant material is often included as dead biomass. The quantity of biomass is expressed as a dry weight or as the energy, carbon or nitrogen content.

Biomass fuels or biofuels

A fuel produced from dry organic matter or combustible oils produced by plants. These fuels are considered renewable as long as the vegetation producing them is maintained or replanted, such as firewood, alcohol fermented from sugar, and combustible oils extracted from soy beans. Their use in place of fossil fuels cuts greenhouse gas emissions because the plants that are the fuel sources capture carbon dioxide from the atmosphere.

Biome

Major and distinct regional element of the biosphere, typically consisting of several ecosystems (e.g., forests, rivers, ponds, swamps) within a region of similar climate. Biomes are characterised by typical communities of plants and animals.

Bottom-up models

Models represent reality by aggregating characteristics of specific activities and processes, considering technological, engineering and cost details.

Carbon Cycle

The term used to describe the flow of carbon (in various forms, e.g., carbon dioxide) through the atmosphere, ocean, terrestrial biosphere and lithosphere.

Carbon market

A popular but misleading term for a trading system through which countries may buy or sell units of greenhouse-gas emissions in an effort to meet their national limits on emissions, either under the Kyoto Protocol or under other agreements, such as that among member states of the European Union. The term comes from the fact that carbon dioxide is the predominant greenhouse gas and other gases are measured in units called "carbon-dioxide equivalents."

Carbon pool

Carbon pools are: above-ground biomass, belowground biomass, litter, dead wood and soil organic carbon. CDM project participants may choose not to account one or more carbon pools if they provide transparent and verifiable information showing that the choice will not increase the expected net anthropogenic GHG removals by sinks.

Carbon sequestration

The process of removing carbon from the atmosphere and depositing it in a reservoir.

Clean Development Mechanism (CDM)

A mechanism under the Kyoto Protocol through which developed countries may finance greenhouse-gas emission reduction or removal projects in developing countries, and receive credits for doing so which they may apply towards meeting mandatory limits on their own emissions.

Common Reporting Format (CRF)

Standardized format for reporting estimates of greenhouse-gas emissions and removals and other relevant information by Annex I Parties.

Compliance

Fulfilment by countries/businesses/individuals of emission and reporting commitments under the UNFCCC and the Kyoto Protocol.

Conference of the Parties (COP)

The supreme body of the Convention. It currently meets once a year to review the Convention's progress. The word "conference" is not used here in the sense of "meeting" but rather of "association," which explains the seemingly redundant expression "fourth session of the Conference of the Parties."

Deforestation

Conversion of forest to non-forest.

Drought

In general terms, drought is a 'prolonged absence or marked deficiency of precipitation', a 'deficiency that results in water shortage for some activity or for some group', or a 'period of abnormally dry weather sufficiently prolonged for the lack of precipitation to cause a serious hydrological imbalance' (Heim, 2002). Drought has been defined in a number of ways. Agricultural drought relates to moisture deficits in the topmost 1 metre or so of soil (the root zone) that affect crops, meteorological drought is mainly a prolonged deficit of precipitation, and hydrologic drought is related to below-normal streamflow, lake and groundwater levels. A megadrought is a longdrawn out and pervasive drought, lasting much longer than normal, usually a decade or more.

Emission reduction unit (ERU)

A Kyoto Protocol unit equal to 1 metric tonne of CO₂ equivalent. ERUs are generated for emission reductions or emission removals from joint implementation project.

Evapotranspiration

The combined process of evaporation from the Earth's surface and transpiration from vegetation.

Global warming potential (GWP)

An index representing the combined effect of the differing times greenhouse gases remain in the atmosphere and their relative effectiveness in absorbing outgoing infrared radiation.

Greenhouse effect

Greenhouse gases effectively absorb thermal infrared radiation, emitted by the Earth's surface, by the atmosphere itself due to the same gases, and by clouds. Atmospheric radiation is emitted to all sides, including downward to the Earth's surface. Thus, greenhouse gases trap heat within the surface-troposphere system. This is called the greenhouse effect. Thermal infrared radiation in the troposphere is strongly coupled to the temperature of the atmosphere at the altitude at which it is emitted. In the troposphere, the temperature generally decreases with height. Effectively, infrared radiation emitted to space originates from an altitude with a temperature of, on average, -19°C, in balance with the net incoming solar radiation, whereas the Earth's surface is kept at a much higher temperature of, on average, +14°C. An increase in the concentration of greenhouse gases leads to an increased infrared opacity of the atmosphere, and therefore to an effective radiation into space from a higher altitude at a lower temperature. This causes a radiative forcing that leads to an enhancement of the greenhouse effect, the so-called enhanced greenhouse effect.

Greenhouse gases (GHGs)

The atmospheric gases responsible for causing global warming and climate change. The major GHGs are carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O). Less prevalent --but very powerful -- greenhouse gases are hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride (SF₆).

Gross Primary Production (GPP)

The amount of energy fixed from the atmosphere through photosynthesis.

Implementation

Actions (legislation or regulations, judicial decrees, or other actions) that governments take to translate international accords into domestic law and policy.

Intergovernmental Panel on Climate Change (IPCC)

Established in 1988 by the World Meteorological Organization and the UN Environment Programme, the IPCC surveys world-wide scientific and technical literature and publishes assessment reports that are widely recognized as the most credible existing sources of information on climate change. The IPCC also works on methodologies and responds to specific requests from the Convention's subsidiary bodies. The IPCC is independent of the Convention.

Joint implementation (JI)

A mechanism under the Kyoto Protocol through which a developed country can receive "emissions reduction units" when it helps to finance projects that reduce net greenhouse-gas emissions in another developed country (in practice, the recipient state is likely to be a country with an "economy in transition"). An Annex I Party must meet specific eligibility requirements to participate in joint implementation.

Kyoto Protocol

An international agreement standing on its own, and requiring separate ratification by governments, but linked to the UNFCCC. The Kyoto Protocol, among other things, sets binding targets for the reduction of greenhouse-gas emissions by industrialized countries.

Land use and Land use change

Land use refers to the total of arrangements, activities and inputs undertaken in a certain land cover type (a set of human actions). The term land use is also used in the sense of the social and economic purposes for which land is managed (e.g., grazing, timber extraction and conservation). Land use change refers to a change in the use or management of land by humans, which may lead to a change in land cover. Land cover and land use change may have an impact on the surface albedo, evapotranspiration, sources and sinks of greenhouse gases, or other properties of the climate system and may thus have a radiative forcing and/or other impacts on climate, locally or globally. See also the IPCC Report on Land Use, Land-Use Change, and Forestry

Land use, land-use change, and forestry (LULUCF)

A greenhouse gas inventory sector that covers emissions and removals of greenhouse gases resulting from direct human-induced land use, land-use change and forestry activities.

Mires

Peat-accumulating wetlands.

Mitigation

In the context of climate change, a human intervention to reduce the sources or enhance the sinks of greenhouse gases. Examples include using fossil fuels more efficiently for industrial processes or electricity generation, switching to solar energy or wind power, improving the insulation of buildings, and expanding forests and other "sinks" to remove greater amounts of carbon dioxide from the atmosphere.

National communication

A document submitted in accordance with the Convention (and the Protocol) by which a Party informs other Parties of activities undertaken to address climate change. Most developed countries have now submitted their fourth national communications; most developing countries have completed their first national communication and are in the process of preparing their second.

Net biome production (NBP)

Net biome production is the net ecosystem production (NEP) minus carbon losses resulting from disturbances such as fire or insect defoliation.

Net ecosystem production (NEP)

Net ecosystem production is the difference between net primary production (NPP) and heterotrophic respiration (mostly decomposition of dead organic matter) of that ecosystem over the same area (see also net biome production (NBP)).

Net primary production (NPP)

Net primary production is the gross primary production minus autotrophic respiration, i.e., the sum of metabolic processes for plant growth and maintenance, over the same area.

Non-Annex I Parties

Refers to countries that have ratified or acceded to the United Nations Framework Convention on Climate Change that are not included in Annex I of the Convention.

Non-linearity

A process is called 'non-linear' when there is no simple proportional relation between cause and effect.

Peat

Peat is formed from dead plants, typically Sphagnum mosses, which are only partially decomposed due to the permanent submergence in water and the presence of conserving substances such as humic acids.

Photosynthesis

The process by which plants take carbon dioxide from the air (or bicarbonate in water) to build carbohydrates, releasing oxygen in the process. There are several pathways of photosynthesis with different responses to atmospheric carbon dioxide concentrations.

Permafrost

Ground (soil or rock and included ice and organic material) that remains at or below 0°C for at least two consecutive years.

Rangeland

Unmanaged grasslands, shrublands, savannas and tundra.

Reforestation

Replanting of forests on lands that have previously contained forests but that have been converted to some other use.

Reservoir (Stock)

A component of the climate system, other than the atmosphere, which has the capacity to store, accumulate or release a substance of concern, for example, carbon, a greenhouse gas or a precursor. Oceans, soils and forests are examples of reservoirs of carbon. Pool is an equivalent term (note that the definition of pool often includes the atmosphere). The absolute quantity of the substance of concern held within a reservoir at a specified time is called the stock.

Respiration

The process whereby living organisms convert organic matter to carbon dioxide, releasing energy and consuming molecular oxygen.

Sink

Any process, activity or mechanism which removes a greenhouse gas, an aerosol or a precursor of a greenhouse gas from the atmosphere. Forests and other vegetation are considered sinks because they remove carbon dioxide through photosynthesis.

Source

Any process, activity or mechanism that releases a greenhouse gas, an aerosol or a precursor of a greenhouse gas or aerosol into the atmosphere.

Sustainable development

Development that meets the cultural, social, political and economic needs of the present generation without compromising the ability of future generations to meet their own needs.

Top-down models

Models applying macroeconomic theory, econometric and optimization techniques to aggregate economic variables. Using historical data on consumption, prices, incomes, and factor costs, top-down models assess final demand for goods and services, and supply from main sectors, such as the energy sector, transportation, agriculture, and industry. Some top-down models incorporate technology data, narrowing the gap to bottom-up models.

Uncertainty

An expression of the degree to which a value is unknown (e.g. the future state of the climate system). Uncertainty can result from lack of information or from disagreement about what is known or even knowable. It may have many types of sources, from quantifiable errors in the data to ambiguously defined concepts or terminology, or uncertain projections of human behavior. Uncertainty can therefore be represented by quantitative measures (e.g., a range of values calculated by various models) or by qualitative statements (e.g., reflecting the judgment of a team of experts).

Uptake

The addition of a substance of concern to a reservoir. The uptake of carbon containing substances, in particular carbon dioxide, is often called (carbon) sequestration.

Vulnerability

The degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate variation to which a system is exposed, its sensitivity, and its adaptive capacity.

Water-use efficiency

Carbon gain in photosynthesis per unit water lost in evapotranspiration. It can be expressed on a short-term basis as the ratio of photosynthetic carbon gain per unit transpirational water loss, or on a seasonal basis as the ratio of net primary production or agricultural yield to the amount of available water.

Wetland

A transitional, regularly waterlogged area of poorly drained soils, often between an aquatic and a terrestrial ecosystem, fed from rain, surface water or groundwater. Wetlands are characterized by a prevalence of vegetation adapted for life in saturated soil conditions.

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Annex 1 Methodologies to estimate changes in soil carbon.

Category 1: Statistical analyses of spatially distributed soil samples

Statistical analyses of spatially distributed soil samples provide information on changes in soil carbon pools when the measurements are taken at two points in time (e.g. Bellamy et al., 2005) or are from a chronosequence (simultaneous measurement at sites with different histories of change behind them, e.g. Covington, 1981). This report concentrates on the former approach of estimating soil carbon changes as a difference between repeated measurements because this approach is more useful to estimate the contribution of climate change to soil carbon changes. Chronosequences cannot provide this information although they may be useful to obtain information on the effects of other factors, such as land use or land management.

When analyzing soil carbon changes based on repeated measurements, relevant issues to be considered are:

- 1) sample design, i.e. selection of study sites and selection of sample points at the sites,
- 2) selection of soil layers to be studied,
- 3) repeatability of sampling and laboratory measurements and
- 4) data analysis.

Sample design, i.e. selection of study sites and selection of sample points at the sites

Sample design is an optimization problem, where the trade-off is between required resources and reliability of resulting estimates. The basic schemes of sample design to choose from are 1) random sampling, 2) systematic sampling and 3) stratified sampling with either random or systematic sampling per stratum. Reliability of estimates obtained using purely random or systematic sampling can usually be improved by dividing the study area into internally more homogenous groups with respect to soil carbon changes. This stratification can be done on the basis of earlier measurements or model-calculated estimates. Stratification is the more effective the more reliably it is possible to estimate the change rate of soil carbon inside a stratum. In other words, it pays off to stratify if it is possible to distinguish groups with high change rates of soil carbon from those with low change rates. In addition, stratification is effective if the change rates inside the strata can be determined reliably either because the spatial variability of the change is low or because it is possible to take a large number of soil carbon samples and thereby obtain a reliable estimate. Peltoniemi et al. (2007) estimated that it would be possible to reduce the standard error of a mean change estimate of soil carbon in the Finnish forests by 9 to 34 %, depending on uncertainty estimates, by dividing the forests into four strata.

In practice, the process of sample design usually consists of answering two questions, namely 1) which sites to sample and 2) where to take soil samples at the sites. The sites are considered as homogenous units compared to the study area as a whole and thus they are used as one basis of stratification. Such two-phase sample design is also practical from the point of view of the logistics of taking the soil samples. Inside the study sites it is possible to operate on foot but some other means of transportation is needed to move from one site to another.

In selecting study sites or sample locations at the sites, systematic sampling is usually preferred to random sampling. When the sites or sample locations are taken from a systematic grid it is possible to control the degree of spatial dependence, either to avoid it by taking samples from an

adequate distance from one another or to make use of it and apply geostatistical methods when analyzing the data. For example, in boreal forests, the spatial dependence of soil carbon density extends to a few meters (Liski, 1995, Liski et al. manuscript).

When changes in soil carbon pools are estimated based on repeated sampling, it is also necessary to decide how the new sample sites and sampling locations will be placed relative to the original. Work carried out in the east of England (Lark et al 2006) has demonstrated that when resampling an existing baseline survey it is best to sample at the original sites rather than between them and that the best strategy depends on the spatial structure of the change in the soil property. Therefore, when the change in soil carbon content is spatially autocorrelated, taking the new soil samples from the same site (for example from within the same 20 x 20 m square) as the original ones helps to increase the statistical reliability of the change estimate because the covariance can be taken into account when analyzing the data.

Selection of soil layers to be studied

Measurements of soil carbon changes are usually carried out in the topmost soil layers (e.g. Bellamy et al., 2005). In those soils which have an organic soil layer on top of mineral soil, only this layer is often sampled. The rationale behind concentrating on the top soil layers is, first, that these soil layers are rich in labile carbon and for this reason the changes are expected to be the largest there, and, second, that these layers are the easiest to sample. These are usually reasonable reasons considering the costs and benefits. However, sometimes the carbon pools of different soil layers may change in opposite directions. For example, at a Finnish forest site, the carbon pool of the organic layer decreased after harvesting while the pool of the topmost 10 cm deep mineral soil layer increased (Liski et al., manuscript). Looking only at the organic layer would give a biased picture of the soil carbon changes at the site. It may not be possible to give a general rule as to which soil layers to sample when measuring soil carbon changes. It seems to be necessary to consider it case by case and perhaps carry out pilot studies to provide the background information.

Repeatability of sampling and laboratory measurements

Ensuring repeatability of sampling and laboratory measurements is particularly important when estimating soil carbon changes based on repeated measurements. The changes are usually small in proportional terms, commonly only a few percent and maybe as small as one percent. A one percent change in a soil carbon pool is challenging to detect as it is of the same order of magnitude as measurement errors for soil bulk density and soil C concentration.

To make repeated soil sampling and laboratory analysis possible, it is necessary to document these practises carefully. It is also advisable to archive all samples for controlling the repeatability of carbon content measurements.

Data analysis

There are two kinds of methods available to analyse data on soil carbon changes obtained from repeated measurements: (1) traditional statistical methods and (2) geostatistical methods. The geostatistical methods make use of spatial autocorrelation in the data and give more reliable results if such autocorrelation exists in the data (e.g. O'Sullivan & Unwin, 2002). However, analysis of the change in organic carbon in a resampled dataset for England and Wales showed there was no spatial

structure in the change and that the only way to estimate change at the sites not resampled was to use the relationship between change and the original carbon at the site (Bellamy et al, 2005).

In general, it is very challenging to identify the contribution of climate change to measured changes in soil carbon. Measurements of soil carbon changes are characterized by a substantial uncertainty and it is well known that landuse and land management changes have large effects on soil carbon. A project currently being undertaken at Cranfield University is investigating the causes of the loss of carbon identified across England and Wales. Initial results using simple models, fitted using Bayesian analysis and Markov-Chain Monte Carlo methods, indicate that past changes in land use and management were probably the main cause and any climate change signal is masked by these other changes (Kirk and Bellamy, 2008) To enable the effects of climate change on soil carbon to be estimated using repeated measurements it is very important for the land use and land management history of the monitoring sites to be known as well as the management between samplings.

Category 2: Measurements of carbon dioxide fluxes

Carbon dioxide fluxes are measured using various methods. The two main types of measurement that include the contribution from the soil (as opposed to foliar gas exchange equipment) use soil chambers and eddy covariance (EC) towers. Both methods suffer from a number of difficulties: distinguishing between fluxes from vegetation and dead organic matter in soil, distinguishing between autotrophic and heterotrophic respiration from soil, standard sampling-related issues concerning location and replication of instruments and, for EC, determining the typically wind-dependent and therefore variable foot print area. Because of these difficulties, the methods based on measuring carbon dioxide fluxes are not very useful to estimate changes in soil carbon pools or heterotrophic soil respiration over large geographical regions. However, when these methods are applied at also otherwise intensively studied sites, they can be very useful to learn more about processes causing changes in soil carbon and to validate estimates of other methods.

Category 3: Process-based modeling studies

Process-based models are widely used to study changes in soil carbon stocks. They are used from the stand scale up to regional and national scale soil carbon assessment studies in different land-use types (Peltoniemi et al., 2007, Powlson et al., 1996, Smith et al., 1997b, Smith et al., 1998, Tiktak et al., 1995). Models vary from relatively simple models like RothC (Coleman and Jenkinson, 1996) and Yasso (Liski et al., 2005) to models covering the soil processes in more detail like CENTURY (Parton et al., 1987, Parton et al., 1994) and DNDC (Li et al., 1992).

Typical input variables that influence the decomposition processes in models are temperature and soil moisture, soil texture as well as chemical characteristics of the litter input and soil (Peltoniemi et al., 2007). When models are used to study the impacts of climate change on soil carbon, the most important driving variables of the simulations tend to be estimates of the litter input to the model as well as climatic variables like mean annual or monthly air temperatures and precipitation. Very often the environmental variables determine decomposition rate in one or more model compartments. The linearity or non-linearity of the dependencies of the modelled soil carbon stocks and stock changes on these driving variables affect the optimal selection of the spatial calculation unit of the model simulations. In case of linear dependencies, models can be run in

coarse resolution whereas in case of non-linear dependencies one should run the models at small scales and sum up the results to obtain wider scale estimates.

With dynamic models, the model results of each time step depend not only on the model parameters and input, but also on the previous values of the state variables. Model initialisation, i.e. assigning values to the state variables at the beginning of the simulations, is therefore an important step in model applications. This initialisation is typically hampered by the lack of measurable counterparts to the model compartments. A means often used in models is to assume the state variables to be in a steady-state with certain input estimates given to the model. The accuracy of the equilibrium assumption depends on the application, and easily leads to underestimation in such applications where the true soil carbon stock is far from equilibrium. This effect is of particular importance for the first years of the simulations (de Wit et al., 2006, Peltoniemi et al., 2006), but the effect can be avoided rather effectively by running the model for some years. Assuming an equilibrium state in model calibrations with soils that are not in equilibrium may also lead to the overestimation of the decomposition rates of the slowest pools and to the overestimation of the stocks of recently disturbed sites (Wutzler and Reichstein, 2007).

The time step of the models varies from daily (in some detailed models like DNDC (Li et al., 1992) some routines are calculated hourly or even sub-hourly) to annual. Simulation periods to predict the effects of changing climate on soil carbon have varied from decades to centuries.

There are different sources of uncertainties in model simulations. Uncertainty propagation from input data and model parameters can be assessed with Monte Carlo simulations. Peltoniemi et al. (2006) carried out such analysis to assess the uncertainty of the Finnish forest carbon balance for which forest inventory information was combined with the Yasso soil carbon model. The uncertainty of the model structure is more difficult to define. Indirectly it can be evaluated with model comparisons that may highlight the range of possible values. Model comparison concerning the effects of climate on soil carbon stock at the global scale was done for example by Jones et al. (2005). The uncertainty related to future predictions is typically handled by using a set of future scenarios spanning a plausible range.

Repeatability is an important criterion in science and this criterion is of special challenge for the modelling studies where the complexity in model structures and explicit and implicit assumptions of the models and modelling processes are difficult to perceive unless they are explicitly and clearly documented.

As measuring changes in soil carbon stocks is laborious and expensive, estimating the changes using soil carbon models appears as a practicable alternative. A few points require particular attention however to ensure that the estimates are reliable. First, it is important that the models used are built and calibrated in an unequivocal and transparent way. It is also important that the applications of the models meet the same criteria. Second, it is necessary that the results of the models are accompanied with uncertainty estimates. It is equally important to describe transparently how the uncertainty estimates are calculated and which sources of the total uncertainty they cover. Ideally, the requirements of using models in estimating soil carbon changes should be as similar as possible with the requirements of estimating the changes based on measurements, i.e. the results should be presented as real probability distributions rather than single mean estimates.

Category 4: Combination of monitoring and process-based modeling

Combining monitoring and process-based modeling may provide benefits in estimating changes in soil carbon pools compared to using any of the methods alone. Process-based modeling may

be used as a basis for sampling design in monitoring programmes. Monitoring may, in turn, be used to test the validity of model-calculated results. This may reduce the total effort of estimating soil carbon changes if the validity of the model-calculated results can be tested adequately in a sub-set of monitoring sites. Monitoring could also in principle be used to determine the status of soil carbon compartments of process-based models in the beginning of the simulations. This would be very useful because determining this status is a particular problem with using the process-based soil carbon models. However, this is still hard to do in practice because the monitored soil carbon pools do not have counterparts in the soil carbon models.

Annex 2 Inventory of available datasets on soil organic carbon (SOC) or soil organic matter (SOM) in cultivated agricultural land (arable land and grassland) and non- cultivated land for the assessments of changes in SOC or SOM content as a result of land use and management in response to the threat “Decline of soil organic matter”; the information has been collected within the RAMSOIL framework (<http://www.ramsoil.eu/UK/Results/Project+Reports+WP2/>).

Country	Depth (cm)	Method applied ¹	Frequency	Spatial coverage	Reference
Belgium, Flanders	0-24 cm	WB (modified)	1990, 1993, 1996, 1999	190000	Sleutel et al., 2003
Belgium, Flanders	plough layer	WB	1952,1990, 2003	116 locations	Sleutel et al., 2006
Belgium, South	Variable 7 databases	Variable (LOI, DC)	1990, 2000	Variable (16-11977)	Lettenens et al., 2005
Belgium, Wallonia, southern part	variable	WB, 4/3	1955 (1950-1970) resampled in 2005	295	Goidts and Wesemael, 2007
Finland	Plough layer (~0-20 cm)	WB (1974)/DC	1974, 1987, 1998	Farm plots (2000, 1320, 705)	Sippola & Yli Halla, 2005
Germany	40	WB	Irregular, 1983, 1989, 1998	Farm plots	Nieder & Richter, 1999
Germany	0-120 cm (8 soil profile layers)	WB (modified)/DC	1969, 1996	Farm plots	Rinklebe & Makeschin, 2002
Ireland	10 (grassland)	WB	1964 a second sampling 1995-1996	678/220	Zhang et al., 2004
Netherlands	5 (grassland) 20 or 25 (arable land)	SOC≤12,%:KU (≤1994); DC (1994) DC (>1995) SOC>12.5%: LOI	1984-2004 Intervals 4-5 years	2-50 ha	Reijneveld et al., (accepted)
Netherlands	5 (grassland) 20 (maize land)	LOI/DC SOC≤12,%:KU (≤1994); DC (1994) DC (>1995) SOC>12.5%: LOI	1984-2004 Intervals 4-5 years	2-50 ha	Hanegraaf et al., 2008 (accepted)
Netherlands	Variable	SOC≤12,%:KU (≤1994); DC (1994) DC (>1995) SOC>12.5%: LOI	Irregular ²	2 -50 ha	Smit et al., 2007
Norway	Variable topsoil depth (1952)/0-20 cm.	Visual assessment (1952)/LOI	1952, 1976, 1986 and 2002	Farm, 25 ha	Riley & Bakkengard, 2006
Norway	0-25 cm	LOI	1991, 2001	291 Farm plots	Riley & Bakkengard, 2006
Sweden	0-25/25-60 cm(1956,1984); 0-25, 25-35, 35-60 cm (2001)	WC (1956/1984 DC (2001))	1956, 1984, 2001	124 (1956), 65 (1984)124 (2001)	Kätterer et al., 2004
UK, England & Wales	0-15 cm	WB modified (%C<15 LOI (%C≥15))	1978-1983 first sampling; Second sampling 1994-1995 arable land; 1995-1996 grassland; 2003 non agricultural land	5661 (1 st sampling); 853/971/535	Bellamy et al., 2005

¹ DC: dry combustion followed by measuring CO₂, KU: Kumies, WB: Walkley & Black, LOI: Loss of ignition, ² not each year at same place.

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Annex 3 Examples of monitoring schemes in European countries

Below are examples of monitoring schemes for soils that have been implemented in several European countries. This is not a comprehensive list.

Austria (Freudenschuß, 2006)

Forest Soil Monitoring is carried out by the Federal Research and Training Centre for Forests, Natural Hazards and Landscape (BFW) and covers 514 sites. Environmental Soil Inventories are conducted at provincial level and comprise a total of *c.* 5.500 sites. There are also permanent soil monitoring sites, managed both by the Provinces, BFW and the Environment ministry and these comprise *c.* 40 sites. Other specific investigations are carried out by Universities, Federal and Provincial Institutions. Most of them are included in the Soil Information System BORIS of the Umweltbundesamt which comprises more than 10,000 sites.

Czech Republic (Němeček & Kozák)

Two separate systematic monitoring systems have been implemented in the Czech Republic: (1) Monitoring of the agricultural land on 200 observation plots, which started in 1992 (Mazanec, 1996), (2) Monitoring of observation plots in forests (Materna, 1996; Moravčík, 1996). The monitoring of soil characteristics is accompanied by the observation of atmospheric emissions. Two systems of monitoring forest soils also exist. The first system started at the beginning of the 1980s and was aimed at studying the input of S and N into soils and their direct effects (along with ozone) on 500 forests sites in endangered areas

Germany (Miehe et al.)

In Germany permanent soil monitoring was introduced in 1986 and there are currently *c.* 800 sites. There are two different types of monitoring site which differ in intensity of investigation: basic monitoring sites used for trait documentation whereas intensive monitoring sites additionally are dealing with process documentation (substance input and output). Soil monitoring sites have been comprehensively documented with respect to pedology, land condition and use. The sites are sufficiently representative for nationwide soil monitoring.

Norway (Arnoldussen)

Norway contributes to the European Forest Monitoring programme. Information on forest vitality is gathered annually from fixed points, lying in a grid system. The soil at each sampling point is described and samples taken for chemical analysis. This inventory will be repeated after a certain period so as to obtain information about trends. Particular emphasis is given to monitoring changes in the content of nitrogen, sulphur and some heavy metals. There is also an additional monitoring system based on sampling of the plough layer in agricultural areas for the farmers for analysis of crop relevant parameters. All information is stored in a database at the Centre for Soil and Environmental Research. Information about these parameters can be retrieved per municipality. Farmers are given advice on what to do in situations in which deficiencies occur.

Sweden (Olsson)

Systematic soil monitoring in Sweden, at national level, is carried out mainly by: (1) The Swedish National Survey of Forest Soils and Vegetation at the Department of Forest Soils, SLU, (2) Integrated Monitoring (IM) through the Department of Environmental Assessment, SLU, (3)

Intensive Monitoring Plots (ICP Forest, Level 2) through the National Board of Forestry and (4) Monitoring of Arable Land, carried out by the Department of Soil Science, SLU. In addition, soil monitoring is performed on a regional scale under the responsibility of County Boards but with a common protocol.

The survey methods have changed since the first survey in 1963, but continue to be stratified random sampling with greater densities in southern Sweden and lesser densities in northern Sweden. The first inventory, during the 10-year period 1963-1972, comprised random sampling on almost 76,800 plots. The country was re-sampled during the 3-year period 1973-1975, with around 23,100 plots and with several investigational pits per plot. The inventory during the 5-year period 1983 to 1987 comprised a total of 23,100 plots on forestland. A new method was implemented for this survey with defined permanent circular plots of a radius between 7 and 10 m. The intention is that the use of permanent plots will enable more accurate assessment of changes over time. The plots are clustered into "tracts". These are quadratic or rectangular with a side, depending on location in the country, within a range of 300-1,800 m. In general, soil pits and soil and site descriptions are made at one to two circular plots per tract. At each circular plot, general site properties such as vegetation type and occurrences of different species, type of soil parent material and hydrological conditions are described. Specific variables include thickness of humus layer, humus form and thickness of E horizon. The inventory also records soil type according to the Swedish system and the FAO-UNESCO legend. Samples are stored in a soil bank. Parent material (C horizon) from selected plots (c. 3,000) has been analysed for the total elemental composition of major and trace elements. Some plots are included in CCP Forest, level 1 programme. This survey is followed by a new one, covering the 10-year period 1993-2002. The results of the measured parameters from The Swedish National Survey of Forest Soils and Vegetation can be related to natural site conditions such as geology and climate and to human impacts such as pollution. Using the data, critical loads for acidity and N deposition have been developed. It has also been possible to verify that the accumulation of carbon is increasing in humus layers. Most of the material is being presented and free to use as maps or as an interactive database (<http://www-markinfo.slu.se>). The web-material is in Swedish but a translation to English is being undertaken.

United Kingdom (SNIFFER, 2006)

National soil sampling schemes in the U.K. include: (1) the National Soil Inventory (NSI), (2) the Representative Soil Sampling Scheme (RSSS), (3) the Countryside Survey (CS) and (4) the Environmental Change Network (ECN). The UK's department of Defra instigated project SP0515 to compare these sampling schemes (SNIFFER 2006). Recommendations from this comparison have fed into a new project designing a UK soil monitoring Network (EA 2008).

Annex 4 Carbon trends in grassland, cropland and forest soil: methods and their reliability

When calculating the carbon balance of grassland soils, Smith *et al.* (2005) analysed also carbon sequestration efficiency of grasslands and the reliability of their calculation method. The carbon sequestration efficiency of grasslands (= ratio NBP/NPP) is 0.147 when it is defined as NBP_{data}/NPP_{data} from the data oriented best estimate $NPP_{DATA} = 758 \text{ g C m}^{-2} \text{ y}^{-1}$. The range of uncertainty for the sequestration efficiency is 0.04 to 0.23. The carbon sequestration efficiency has a very similar value of 0.13, when it is defined as NBP_{model}/NPP_{PASIM} from the model oriented best estimate $NBP_{model} = 101 \text{ g C m}^{-2} \text{ y}^{-1}$, and the NPP_{PASIM} corrected for extensive management ($755 \text{ g C m}^{-2} \text{ y}^{-1}$). Smith *et al.* (2005), using the RothC soil carbon model and NPP change estimates from the LPJ model, estimated that European grassland soils were a net sink in the 1990s of between 8 and 448 Tg C, a net mean sequestration rate of 0.8 to 44.6 Tg C yr⁻¹ for the whole of Europe, smaller than the estimates of Janssens *et al.* (2003, 2005) (Ciais *et al.*, 2008b).

The carbon balance of cropland soils (NBP) is a highly uncertain flux, because we lack inventory data of agricultural soil carbon change with full EU-25 coverage. Instead, we estimated NBP with two process-oriented models, ORCHIDEE-STICS and Roth-C. ORCHIDEE-STICS (Gervois *et al.*, 2007) was integrated between 1901 and 2000, starting from ancestral practice and crop varieties. In this version, the model calculates NPP, harvest and soil carbon decomposition for wheat and maize varieties only. It was driven by rising CO₂ and by climate at a resolution of 10 km, with changing technology after 1950, leading to a total NBP of 0 to 30 gC m⁻² y⁻¹ (Ciais *et al.*, 2008a).

Roth-C is a soil carbon model (Smith *et al.*, 2005) that was prescribed with changing NPP (minus harvest) from LPJ as soil carbon input (Sitch *et al.*, 2003). Roth-C was initialized in, and run from 1900 at a resolution of 10 km to 2100. The cropland NBP values from ORCHIDEE-STICS and Roth-C models are comparable (respectively a sink of 0 to 30 g C m⁻² y⁻¹ and a source of 7.6 g C m⁻² y⁻¹ over 1990-1999). Over the EU-25 cropland area, this translates into a net carbon balance ranging from a 39 Tg C y⁻¹ sink, to a 10 Tg C y⁻¹ source (Ciais *et al.*, 2008a).

Forest soils are estimated to accumulate carbon in each European country. The carbon sink of the forest soils has been reported earlier for only western European countries, i.e. EU-15 plus Norway and Switzerland (Liski *et al.* 2002). For this report, this analysis was extended to cover also other European countries using assumptions applied earlier by Janssens *et al.* (2003 and 2005). Accordingly, the carbon sink of the European forest soils was estimated to be equal to 44 Tg year⁻¹. Uncertainty in this estimate, resulting from uncertainty about parameter values used in the calculations, ranges from about 30 to 60 Tg year⁻¹ (Liski *et al.* 2002).

Annex 5 Case studies for assessing changes in soil carbon stocks

In this Annex we present 3 case studies where measures on changes in soil carbon stocks have been observed or calculated on the bases of repeated measurements. This information is relevant for the assessment of changes in soil carbon stocks as presented in chapter 3. The cases are:

- England and Wales, National Soil Inventory
- Great Britain, countryside survey
- Belgium

These are the only studies that have presented country wide data for determining changes in soil carbon stocks.

Case 1: England and Wales, National Soil Inventory

The National Soil Inventory (NSI) was designed to obtain an unbiased estimate of the distribution of the soils of England and Wales and of the chemistry of the topsoil. Samples were collected and soil profiles described at the intersections of a 5-km grid over the whole area. Urban areas and water bodies were avoided, but otherwise all soils were sampled, this yielded 5,662 sites sampled for soil. The NSI was originally carried out during the period 1978-83 and each of the samples taken analysed for a range of soil properties including organic carbon content, pH, metal concentrations and particle size. A range of other properties were also recorded at each site such as land use, slope aspect, altitude etc.

Sub-sets of the sites were re-sampled at intervals of 12 to 25 years after the original sampling. This was done in three phases: in 1994/95 for arable and rotational grassland sites, in 1995/96 for managed permanent grassland sites, and in 2003 for non-agricultural sites (bogs, scrub, rough grazing, woodland, etc). Roughly 40% of the original sites were re-sampled.

The data on soil organic carbon from the two samplings of the NSI was used to investigate how soil carbon has changed over the interval 1978-2003 (Bellamy et al 2005). To allow for the varying time interval between samplings, annual rates of change in carbon were calculated for each site by assuming that the process of change was linear over the sampling interval. Some differences in rates of change between soils and land uses were apparent: for example, peat soils lost carbon an order of magnitude faster than brown soils and man-made soils, and bogs and upland grass lost carbon an order of magnitude faster than lowland heath, which appears to have gained carbon on average. But no statistically significant relations between rate of change and land use, rainfall class or soil textural class were found, whether for the data as a whole or for outlying data. However, a significant negative linear correlation between rate of change and original organic carbon content (Corg) was found; that is, the rate of loss increased with Corg. Using this relationship it was estimated that carbon was lost from soils across England and Wales over the survey period at a mean rate of 0.6% yr⁻¹ (relative to the existing soil carbon content). This estimate was based on the soil carbon content of the top 15cm of soil. Converting this to carbon stocks (using a pedotransfer function to estimate bulk density) it was estimated that the soils of England and Wales were losing carbon at the rate of 4.44Tg yr⁻¹.

One criticism of this analysis was that the relationship between the original carbon and rate of change in carbon could be affected by regression to the mean. Any statistical relation of change to the baseline value will reflect, at least in part, the phenomenon of regression to the mean. However an analysis reported in Lark et al (2006) demonstrated that for this dataset the conclusion that the rate of change of organic carbon depended on the baseline level was robust, and that the bias in that

relationship due to regression to the mean was small. Another criticism of this paper was the use of a pedotransfer function to estimate bulk density to allow estimation of carbon stocks. A pedotransfer function was used as no bulk density measurements were made at the NSI sites however the function was based on hundreds of measurements that had been collected across the whole range of soils found in England and Wales. This problem highlights the need to measure bulk density in any future monitoring of soil carbon.

As described above Bellamy *et al.* (2005) observed a mean loss of topsoil soil organic carbon (SOC) of $0.6\% \text{ yr}^{-1}$, between 1978 and 2003 in England and Wales, which contradicts strong evidence that the UK and Europe as a whole are a net CO_2 sink (Janssens *et al.*, 2003). For non-agricultural areas, it also contradicts data from another long term study of topsoil SOC in British woodlands (Kirkby *et al.*, 2005). Kirkby *et al.* (2005) sampled, in 1971 and 2000-2003, 1648 plots randomly located in 103 woods; their findings suggest no significant change in SOC over 30 years (slight increase of $+0.38\%$ over 30 years; $\sim +0.01\% \text{ y}^{-1}$). Other repeated sampling studies in Europe have shown contrasting results, with some showing loss of SOC (e.g. for Flemish cropland soils; Sleutel *et al.*, 2003), attributed to changing manure application practices, and others showing no loss of SOC (in Danish croplands; Heidmann *et al.*, 2002 and in Austrian soils; Dersch & Boehm, 1997). Smith *et al.* (2007), using two soil carbon models, suggested that only 10-20% of the loss of C from soils in England and Wales reported by Bellamy *et al.* (2005) could be due to climate change. Recent work (Kirk and Bellamy, 2008) has shown that it is likely that past changes in land use history and land management were dominant reasons for the loss of C.

Case 2: The Countryside Surveys of Great Britain

The Countryside Surveys of Great Britain (GB) are ongoing ecological assessments of the non-urban land in GB. The surveys use a stratified random sample of all the one-kilometre squares in GB (Firbank et al., 2003) and have taken place in 1978, 1984, 1990, 1998 and 2007. Soil samples (0-15 cm depth) were collected alongside land use and vegetation information in 1978 (1197 samples) and 1998 (1098 samples). 754 locations were sampled for soils in both 1978 and 1998, whilst 443 and 344 locations were sampled in 1978 and 1998 only, respectively. Soil C concentration was measured by loss-on-ignition in all samples. Land use was determined from a statistical analysis of the vegetation composition at the soil sampling location which groups vegetation into eight aggregate classes (Bunce et al., 1999; AVC2 - tall grass and herbs - was excluded from analyses due to insufficient sample numbers). Soils were split into four 'types' based on mean C concentration (<40, 40-150, 150-300 and >300 g C kg⁻¹); locations which only contained one C concentration measurement were therefore excluded from analysis by soil type. Statistical analysis utilised all available data (not just that of locations sampled twice) using a mixed effects model which estimates the C concentration in each category for each year and then assesses whether the difference between the values in each year is significantly different from zero.

Average topsoil C concentrations across GB in 1978 and 1998 were 128.8±17.5 and 138.5±17.6 g C kg⁻¹ (mean±95% CI), respectively. The increase of 9.7±6.0 g kg⁻¹ over the 20 years (0.5±0.3 g kg⁻¹ yr⁻¹) was significantly different to zero (P<0.01). Significant increases in soil C concentration were observed in fertile and infertile grasslands, upland woodlands, and heath and bog habitats, and were in the range 0.2-2.1 g kg⁻¹ yr⁻¹. Significant changes in mineral soils were limited to fertile (+0.20±0.03 g kg⁻¹ yr⁻¹) and infertile (+0.33±0.03 g kg⁻¹ yr⁻¹) grasslands, in organo-mineral soils to lowland woodlands (+2.44±0.32 g kg⁻¹ yr⁻¹), and in highly organic soils to moorland grass mosaics (+4.38±0.57 g kg⁻¹ yr⁻¹; Fig. 2). Taken together, these results suggest that GB topsoil C concentration increased slightly in the period 1978-98, although changes differed between soil type and land use. There was no evidence of significant losses of topsoil C.

Case 3: Belgian soils

In Belgium, a comprehensive national soil survey was carried out between 1950 and 1970. At each location soil pits were dug and over 13,000 soil profiles were recorded. For each horizon, depth and thickness, textural fraction, rock fragment content and organic carbon content were recorded, along with site information such as land use, and location (Van Meirvenne et al., 1996).

Re-sampling of the national survey has taken place piecemeal since 1989, and many assessments of soil C changes are now available in the literature. Two main approaches have been used to examine change: a paired-sample approach, and a landscape unit characterisation approach. In the former, only soils from locations sampled twice are considered. This method detects change at individual locations, and then averages the changes at those locations to examine overall trends in the dataset. In this way, Van Meirvenne et al., 1996, identified an increase in C stocks in permanent arable fields of 930 g C m^{-2} between 1950 and 1990 (a rate of $23 \text{ g C m}^{-2} \text{ yr}^{-1}$). Sleutel et al (2006) then extended this time-series with a further sampling of some of the locations in 2003-4, and observed a decrease in soil C stock of 250 g C m^{-2} ($-19 \text{ g C m}^{-2} \text{ yr}^{-1}$) since 1990.

In the second approach, data from the original and resurvey are used together with spatial data on land-use in Belgium to produce estimates of SOC content for individual landscape units. This method utilises all available data, including data from the original survey even where locations were not re-sampled. Change in soil C is then assessed at the landscape unit level, and reported by land use (e.g. Lettens et al., 2005a). The patterns of soil C change suggest that arable soils have lost C since the original survey (although not before 1990; Van Meirvenne et al., 1996) at a rate of $3\text{-}114 \text{ g C m}^{-2} \text{ yr}^{-1}$. Grasslands were reported either to be sequestering C in soils at rates of 22 or $44 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Lettens et al., 2005a; Goidts and van Wesemael, 2007, respectively), or losing C at $90 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Lettens et al., 2005b). Similar differences in trends in soil C stocks are reported for forests, which are either gaining C at a rate of $73 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Lettens et al., 2005a), or losing C at a rate of $23 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Stevens and van Wesemael, 2008)

Some of these differences result from differing study areas (All of Belgium vs. regions), but more likely result from the datasets used. Although many use the 1950-1970 national survey, resurveys have varied considerably in their methods. Some workers have sought to combine very disparate datasets in an attempt to estimate C stock and change (e.g. Lettens et al., 2005a; Lettens et al., 2005b). In the data they used there were differences in sampling depth and type (6 – 200 cm, by fixed depth or by horizon), analytical methods (Walkley-Black, loss-on-ignition) and additional information gathered (bulk density, texture). To combined the datasets, data was converted to C stock (using estimated bulk densities, where required), extrapolated to standard depths, and assigned to landscape units. In these circumstances, a large number of assumptions must be made to produce comparable data.

None of the resamplings of the 1950-1970 survey have used the same sample collection methods; whilst in the original work a soil pit was dug and soils sampled by horizon, all resampling has used a bulked sample from an auger either to a fixed depth or to the bottom of the plough layer. Similarly, whilst bulk density was measured in the original survey, most resurveys have assumed the bulk density is unchanged, and have estimated bulk density from the literature pedotransfer functions (a rare exception being Sleutel et al., 2006, who measured the BD of resampled soils).

Annex 6 Share of soil organic carbon in 0-30 and 0-100 cm.

The share of soil organic carbon found in the top 30 cm soil layer compared to the amount found in the top 100 cm varies between soil units, from 36 % in Histosols to 77 % in Podzoluvisols. The average across all soil units is 52 %. Values around 50 % are common for wide-spread soil types in Europe

Table. Mean organic carbon contents of two depth intervals by FAO/UNESCO soil unit (kg/m²) (Batjes 1996)

Soil unit	0-30 cm	0-100 cm	30/100
Acrisols	5.1	9.4	54 %
Cambisols	5	9.6	52 %
Chernozems	6	12.5	48 %
Podzoluvisols	5.6	7.3	77 %
Rendizinas	13.3	n.a.	
Ferrasols	5.7	10.7	53 %
Gelysols	7.7	13.1	59 %
Phaeozems	7.7	14.6	53 %
Lithosols	3.6	n.a.	
Fluvisols	3.8	9.3	41 %
Kastanozems	5.4	9.6	56 %
Luvisols	3.1	6.5	48 %
Greyzems	10.8	19.7	55 %
Nitosols	4.1	8.4	49 %
Histosols	28.3	77.6	36 %
Podzols	13.6	24.4	56 %
Arenosols	1.3	3.1	42 %
Regosols	3.1	5	62 %
Solonetz	3.2	6.2	52 %
Andosols	11.4	25.4	45 %
Rankers	15.9	n.a.	
Vertisols	4.5	11.1	41 %
Planosols	3.9	7.7	51 %
Xerosols	2	4.8	42 %
Yermosols	1.3	3	43 %
Solochaks	1.8	4.2	43 %

Annex 7 Definitions of organic soil and Histosols (FAO, 1998. World reference base for soil resources, World Soil Resources Report 84, Food and Agriculture Organization of the United Nations, Rome).

Key to the reference soil groups of the world reference base for soil resources

Soils having a *histic* or *follic* horizon,

1. **either** a. 10 cm or more thick from the soil surface to a lithic or paralithic contact; **or** b. 40 cm or more thick and starting within 30 cm from the soil surface; **and**
2. lacking an *andic* or *vitric* horizon starting within 30 cm from the soil surface.

HISTOSOLS (HS)

FAO - Soil Unit Classification Scheme: HISTOSOLS

HISTOSOLS (HS)

Soils having 40 cm or more organic soil material (60 cm or more if the organic materials consist mainly of sphagnum or moss or have a bulk density of less than 0.1 Mg/m³) either extending down from the surface or taken cumulatively within the upper 80cm of the soil. The thickness of the organic surface horizon may be less if it rests on rock or on fragmental material in which the interstices are filled with organic matter.

Synonym: peat and muck soils; (from Gr. *histos*, tissue)

Parent material: incompletely decomposed plant remains, with or without admixtures of sand, silt or clay.

Environment: the majority of all Histosols have formed in boreal regions. Elsewhere, histosols are confined to poorly drained basins and decompressions, swamp and marshlands with shallow groundwater, and highland areas with a high precipitation/evapotranspiration ratio.

Profile development: mostly H or HCr profiles. Transformation of plant remains through biochemical desintegration and formation of humic substances create a surface layer of mould. Translocated organic material may accumulate in deeper tiers but is more often leached from the soil.

Use: peat lands are used for various forms of extensive forestry and/or grazing or lie idle. If carefully reclaimed and managed, Histosols can be very productive under capital-intensive forms of arable cropping/horticulture. Deep peat formations are best left untouched.

Subclasses:

Gelic Histosols (HSi)

Histosols having permafrost within 200cm of the surface.

Thionic Histosols (HSt)

Other Histosols having a sulfuric horizon or sulfidic materials at less than 125cm from the surface.

Folic Histosols (HSI)

Other Histosols that are well drained and are never saturated with water for more than a few days.

Fibric Histosols (HSf)

Other Histosols having raw or weakly decomposed organic materials, the fibre content of which is dominant to a depth of 35cm or more from the surface; having very poor drainage or being undrained.

Terric Histosols (HSs)

Other Histosols having highly decomposed organic materials with only small amounts of visible plant fibers and a very dark grey to black color to a depth of 35cm or more from the surface, having an imperfect to very poor drainage.

MASTER HORIZONS AND LAYERS

The capital letters **H**, **O**, **A**, **E**, **B**, **C** and **R** represent the master horizons and layers of soils. The capital letters are the base symbols to which other characters are added to complete the designation. Most horizons and layers are given a single capital letter symbol, but some require two. Currently seven master horizons and layers are recognized.

The master horizons and their subdivisions represent layers which show evidence of change and some layers which have not been changed. Most are genetic soil horizons, reflecting a qualitative judgement about the kind of changes which have taken place. Genetic horizons are not equivalent to diagnostic horizons, although they may be identical in soil profiles. Diagnostic horizons are quantitatively defined features used in classification.

H horizons or layers: Layers dominated by organic material, formed from accumulations of undecomposed or partially decomposed organic material at the soil surface which may be underwater. All **H** horizons are saturated with water for prolonged periods or were once saturated but are now artificially drained. An **H** horizon may be on top of mineral soils or at any depth beneath the surface if it is buried.

O horizons or layers: Layers dominated by organic material, consisting of undecomposed or partially decomposed litter, such as leaves, needles, twigs, moss, and lichens, which has accumulated on the surface; they may be on top of either mineral or organic soils. **O** horizons are not saturated with water for prolonged periods. The mineral fraction of such material is only a small percentage of the volume of the material and generally is much less than half of the weight.

An **O** layer may be at the surface of a mineral soil or at any depth beneath the surface if it is buried. An horizon formed by illuviation of organic material into a mineral subsoil is not an **O** horizon, though some horizons formed in this manner contain much organic matter.

Histic horizon

General description. The histic horizon (from Gr. *histos*, tissue) is a surface horizon, or a subsurface horizon occurring at shallow depth, which consists of poorly aerated *organic* soil material.

Diagnostic criteria. A histic horizon must have:

1. *either* - 18 percent (by weight) organic carbon (30 percent organic matter) or more if the mineral fraction comprises 60 percent or more clay;
or - 12 percent (by weight) organic carbon (20 percent organic matter) or more if the mineral fraction has no clay;
or - a proportional lower limit of organic carbon content between 12 and 18 percent if the clay content of the mineral fraction is between 0 and 60 percent. If present in materials characteristic for *andic* horizons, the organic carbon content must be more than 20 percent (35 percent organic matter); *and*
2. saturation with water for at least one month in most years (unless artificially drained); *and*
3. thickness of 10 cm or more. A histic horizon less than 20 cm thick must have 12 percent or more organic carbon when mixed to a depth of 20 cm.

Annex 8 Overview of fuel peat use in selected countries

Finland

Peat is mainly used in combined heat and power production (CHP) plants, serving both public and private sector. Use of peat has produced 5-7 % of total energy in Finland. The basic idea is to provide decentralized energy production from many fuels and sufficient domestic content.

Ireland

Of total energy produced in Ireland, 5 % comes from combustion of peat. Future demand of electricity is predicted to increase by 3 % per annum in 2000-2015, and may require the current peat power plants to be kept running. The future of currently healthy peat industry is felt uncertain.

Sweden

Peat has a 4 % share among biofuels in Sweden. The amount of fuel peat extraction is rather low, and has mostly local importance in district heating. Energy peat import has increased in Sweden over the last ten years. Peat is mostly imported from Lithuania and Estonia.

Estonia

Peat reserves in Estonia are large and about two thirds of peatlands have been drained during the Soviet era . Peat is third important domestic energy source after oil shale and wood. The share of peat in primary energy resources in Estonian energy sector was 2.4 % in 2003. About 65 % (expert estimate) of peat extracted is exported, mainly to Sweden.

Latvia

Fuel peat extraction has virtually ceased over the recent years in Latvia, because there are practically no consumers left. Only 0.05 % of the total primary energy consumption is peat. There is not fuel peat export currently.

Lithuania

Fuel peat has not much socio-economic impact in Lithuania. The Government of Lithuania is not encouraging peat use in energy production because of the relatively high atmospheric warming impact of peat combustion.

Annex 9 Summary of methodological choices of countries on soil categories by relevant land use and land use change categories based on the respective national inventory reports submitted to the UNFCCC in April 2008. Note that information of only those countries is included that provided appropriate methodological information in their report. Note also that much more information may be available in the upcoming new round of the national inventory reports due in April 2009.

Austria							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T2	GPG	country specific by land use/change type, developed based on previous soil inventory	IPCC default values were modified	based on country specific literature value, expert judgement and IPCC default	Depth where SOC was measured is 0-50 cm
	Organic	for grasslands; T2					
Land converted to another land use category	Mineral	T2	GPG	country specific by land use/change type, developed based on previous soil inventory	IPCC default values were modified	based on country specific literature value, expert judgement and IPCC default	
	Organic	-					

Belgium								
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other	
Land remaining in the same land use category	Mineral	T3	YASSOO (forest soils); RothC (agri soils)	The SOC estimations are based on a number of heterogeneous databases and modelling efforts. Significant increases between 1990 and 2000 were reported.	-		Depth where SOC was measured is 0-20 cm	
	Organic	-						
Land converted to another land use category	Mineral							
	Organic	-						

Czech Republic							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T1	no change is assumed		-	-	Depth where SOC was measured is 0-30 cm
	Organic	-					
Land converted to another land use category	Mineral	T2/T3	detailed land use/change matrices (cadastral statistics) combined with soil carbon maps	The SOC estimations are based on over six thousand soil samples.			
	Organic	-					

Denmark							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T1	no change is assumed				
	Organic	-					
Land converted to another land use category	Mineral	-	case studies have not yielded consistent data on soil carbon stock changes due to afforestation		-	-	Data are published at http://www.sil- .kvl.dk/afforest/
	Organic	-					

Estonia							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T1	no change is assumed	page 197 of NIR			Soil thickness in 232 research plots varied between 24 and 92 cm
	Organic	T1					
Land converted to another land use category	Mineral	-	IPCC	IPCC default values were used			
	Organic	T1					

Finland							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T3	YASSO model	N/A	N/A	92% relative standard error for the carbon stock change in	Detailed model description and main data used in YASSO are reported in the
	Organic	T3					
Land converted to another land use category	Mineral	N/A					
	Organic	N/A					

France							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T1	IPCC	N/A	N/A		
	Organic						
Land converted to another land use category	Mineral	T3	IPCC	aggregated country-level average, based on INRA [2003], for FL, CL, GL	N/A		
	Organic						

Hungary							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T1	IPCC	N/A	N/A		
	Organic						
Land converted to another land use category	Mineral	T1 for forests	trend statistics for land use categories are available from country specific databases for forests, international database was used to estimate soils carbon from above ground	N/A	N/A are used	partial uncertainty assessment is done using IPCC default values and expert specific judgement	no data for croplands
		T1 for cropland		T1			
	Organic	T3 for forests, T1-T2 for others					
Land converted to another land use category	Mineral			country specific and IPCC default values are used			right after the conversion, so a transition period of one year is applied
	Organic						

Latvia							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	no activity data for forests	IPCC	IPCC default values are used		no quantitative estimation is done	
	Organic	for cropland and grassland : T1					
Land converted to another land use category	Mineral	no activity data for forests	IPCC	IPCC default values are used		no quantitative estimation is done	
	Organic	for cropland and grassland : T1					

Lithuania							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral						
	Organic	for forests	IPCC	IPCC default values are used		no quantitative estimation is done	
Land converted to another land use category	Mineral						
	Organic	for forests	IPCC	IPCC default values are used		no quantitative estimation is done	

The Netherlands							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T1				T2+T1; *uncertainty is the uncertainty of the change in carbon content in mineral soil, which is calculated at 38%*	
	Organic	T2	IPCC	country specific data			soil carbon stock changes are obtained using measured values as well as soil carbon map
Land converted to another land use	Mineral	T2	IPCC	country specific data			
	Organic	T2	IPCC	country specific data			

UK							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use	Mineral	T1					
	Organic		IPCC	N/A	N/A		
Land converted to another land use category	Mineral	T3	Detailed land use change matrix is used together with transition functions	detailed changes of C content of the soil due to land use change types are available	N/A	uncertainty assessment is done using a Monte Carlo based analysis	Change of carbon stock is assessed to 1 m depth
	Organic						

Norway							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T3	Yasso model				no country specific parameters are used
	Organic	T2 for cropland					
Land converted to another land use category	Mineral	T1 or not estimated for forests; T2 for cropland		C-stock changes due to change in practice are larger than in other countries due to climatic reasons			detailed description of estimating emissions due to erosion
	Organic	T2 for cropland, no change is assumed in some deforestations					

New Zealand								
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other	
Land remaining in the same land use category	Mineral	T1						
	Organic							
Land converted to another land use category	Mineral	T1	IPCC	country specific data			under development	
	Organic	T2	IPCC	country specific data				

Canada							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T3	Century model; CBM-CFS3 (for forests)	country specific values are used to calibrate models	N/A	under development	
	Organic	T3					
Land converted to another land use category	Mineral	T3					
	Organic	T3					

USA							
Land use/change category	Soil category	Tier	Methodology	Reference soil carbon data	F factors applied	Uncertainty assessment	Other
Land remaining in the same land use category	Mineral	T2 for forests, T3 and T2 for cropland and grassland	forests: changes in SOC; cropland: Century model	country specific values are used	N/A	reported based on country specific information	only the uppermost 20 cm layer is modelled
	Organic	T3					
Land converted to another land use category	Mineral	T3					
	Organic	T3					

Annex 10 Effect of nitrogen on SOC

A recent review (Kahn et al., 2007) has concluded that fertilizer-N stimulates microbial breakdown of soil organic matter (SOM). They report that after 40-50 years of fertilizer-N application, that exceeded grain N removal by 60 to 190%, a net decline occurred in soil C despite increasingly large incorporation of C in crop residues. Decreases in SOC were reported to a depth of 46 cm. However, the decrease occurred in rotations including grain maize and there are other data that suggest the cultivation of this crop leads to decreases in SOC. In this study root to shoot ratios were quoted which ranged from 1.0 for alfalfa to 0.5 for grain maize. Hence there will be a tendency in rotations dominated by grain maize for less return of C via the roots than for other crops. Other work is quoted which found additions of N and P fertilizers stimulated mineralization of subsoil SOM. The authors consider that studies which appear to report increases in C sequestration as a result of N fertilization do so erroneously, on the basis of comparisons with soil in the unfertilized control, rather than as a result of proper assessments of changes in SOC over time. The authors recommend that fertilizer-N applications are based on the economic optimum requirement, taking account of available soil N, rather than being based on anticipated yield. Such recommendations have been implemented within the EU in order to reduce the risk of nitrate leaching.

The views of Kahn et al. (2007) have been challenged by Reid (2008) who suggested that observed decreases in soil C were caused by factors other than addition of fertilizer-N. However, Kahn et al. (Reid, 2008) replied that soil C had declined in long-term experiments despite a large increase in C returns from crop residues following increased use of fertilizer-N.

Reay et al. (2008) recently reviewed evidence for the impact of N on SOC stocks. Most of this section draws on that analysis and review. Evidence for changes in soil C sinks under reactive nitrogen (N) enrichment comes from a variety of sources, including changes in soil respiration / carbon mineralization rates in the laboratory or field, changes in litter decomposition rates, and changes in soil organic carbon stocks. Evidence is contradictory, with some studies suggesting that soil C may decrease under N enrichment, others suggesting no change, and others suggesting that soil C sinks may increase (Table).

The response of the soil carbon sink to changing N deposition will depend upon the balance between the N-induced increases in carbon inputs to the soil through increased plant growth, and the influence of increased N on carbon losses via soil organic carbon decomposition, respiration, and mineralization.

N deposition might be expected to increase plant production in systems that are N-limited (Magnani *et al.*, 2007) though, as discussed previously, some authors have questioned the magnitude of this impact (De Vries *et al.*, 2008).

In agricultural soils, N fertilization can enhance soil organic carbon (SOC) mineralization (Lisovoi *et al.*, 2001; Shevtsova *et al.*, 2003) but studies of soil respiration suggest no change (Kowalenko *et al.*, 1978). Mineralization has been shown to be retarded at very high N concentrations (Henrikson & Brelland, 1999) and in long term experiments in agricultural systems, artificial N fertilization at much higher rates than received from natural deposition has reportedly led to some small increases in SOC (Glendining & Powlson, 1995). However, a recent examination of SOC at an experimental site receiving synthetic N fertilisation over a 40-50 year period indicated a net decline in soil C (Khan *et al.*, 2007).

In forest soils too, the evidence is contradictory. Increases in soil respiration (i.e. short term carbon loss) in response to N fertilization have been reported (Gallardo & Schlesinger, 1994; Brume & Beese, 1992). While long-term (13 year) continuous high N addition suppressed soil respiration

by 41% in both hardwood and pine stands (Bowden *et al.*, 2004). It has also been suggested that (relatively low) rates on N addition can suppress soil respiration (Magnani *et al.*, 2007).

Additions of N to forest soils often appear to lower the C/N ratio without causing major changes in the total amount of soil carbon (Nielsen *et al.*, 1992; Harding & Jokela, 1994; Schlesinger & Andrews, 2000). And an examination of soil C after 15 years of N addition to Harvard Forest found no significant change (Magill *et al.*, 2004). More recently, however, consistent increases in soil C in N-fertilized forest plots have been reported with accumulation rates appearing to be strongly dependent on soil N status (Hyvönen *et al.*, 2007a). N fertilisation also increased SOC sequestration at N-rich sites, where the tree-growth response was low, suggesting that reduced decomposition rates after N addition may contribute to soil C accumulation (Hyvönen *et al.*, 2007a).

The contradictory evidence suggests that it may not be possible to make sweeping statements about how soil C sinks will respond to increased N deposition. Laboratory incubations of soils from two long term forest fertilisation experiments showed a 30% reduction of the mineralization rate in the organic layer of plots that had received N additions compared to control plots (Persson *et al.*, 2000), and modelling of bomb-¹⁴C data from one of these sites, showed that the reduced mineralisation rate would significantly increase SOC stocks in the long term (Franklin *et al.*, 2003). About 60% of this increase was estimated to be from a decreased decomposition rate, and the rest from increased litter production. It has been suggested that the decreased decomposition rate was driven by a fertilizer-induced increase in decomposer efficiency (production-to-assimilation ratio), a more rapid rate of decrease in litter quality, and a decrease in decomposer basic growth rate (Persson *et al.*, 2000). Overall then, elevated N deposition may lead to a decrease of the mineralization rate and an accumulation of C in the organic layer (Hyvönen *et al.*, 2007b). Very recently, increased SOC accumulation in surface soil layers under low mineral N addition rates in northern temperate forests, attributable to decreased SOC decomposition rates rather than increased detrital inputs, has also been reported (Pregitzer *et al.*, 2008).

It is not clear whether large increases in soil carbon could be expected in areas that receive excess atmospheric N deposition (Nielsen *et al.*, 1992). The evidence remains mixed (Table 3), but the majority of recent studies in N limited systems do suggest that N enrichment may suppress soil C loss (De Vries *et al.*, 2006; Hyvönen *et al.*, 2007a; Magnani *et al.*, 2007; studies listed in Table 3) and may therefore serve to enhance soil C sinks. SOC responses to N in the studies presented in Table 3 range from 0 g C gN⁻¹ (Harding & Jokela, 1994; Schlesinger & Andrews, 2000; Magill *et al.*, 2004) to 23 g C gN⁻¹ (Pregitzer *et al.*, 2008), with those studies, all of which are for forest soils, showing an increase in SOC ranging from 7 to 23 g C gN⁻¹ (Duyzer *et al.*, 1992; Nadelhoffer *et al.*, 1999; De Vries *et al.*, 2006; Pregitzer *et al.*, 2008). As such, soil C stocks may increase as a consequence of increased N deposition in the future, but the saturation of this response remains unexplored.

The review of Loveland and Webb (2003) suggested that fertilizer-N addition, by increasing crop yields and residue returns, increased the 'active' fraction of soil organic matter and hence nutrient turnover. That review reported the results of long-term studies in which the active fraction of SOM was increased by fertilizer-N use, but total soil C decreased (e.g Grace *et al.*, 2005), albeit such decreases are usually attributed to cultivation rather than to addition of fertilizer-N. Hence part of the apparent inconsistency in results may have arisen because increases in active SOM may take place while total soil C is decreasing.

Whilst N addition may increase SOC stocks in N limited forests, and may increase SOC stocks very slightly in agricultural soils (due to increased crop productivity and thereby increased C inputs to the soil; Glendining & Powlson, 1995; though evidence is mixed, see Table 3), N addition will also increase nitrous oxide emissions. Given that nitrous oxide is 296 times more powerful per

kg than CO₂, the overall greenhouse gas benefit of N addition to increase SOC stocks would likely be negative (Smith et al., 2008).

Fertilizer-N use has made a large contribution to the growth in agricultural crop productivity over the last 50 years, but further increased use will lead to greater emissions of N₂O. Hence, perhaps, future emphasis should be concentrated on the other main driver of productivity, improved crop varieties, albeit the development of cereal varieties which partition a greater proportion of total assimilate to grain have played a crucial role in increasing productivity. Perhaps the next goal for plant breeding, would be perennial cereal crops which could maintain, if not increase, soil carbon.

Table: Summary of studies providing evidence that soil C may increase, decrease or remain unchanged under N enrichment

	Evidence	Ecosystem / soil	Type of N addition	Notes
Studies suggesting soil C loss under N enrichment				
Lisovoi <i>et al.</i> (2001)	Decrease in SOC – long term measurements	Agricultural	Long term mineral fertilizer	
Shevtsova <i>et al.</i> (2003)	Decrease in SOC – long term measurements at many sites	Agricultural	Long term mineral fertilizer	Some sites showed no change or soil C gain but most showed soil C loss
Gallardo and Schlesinger (1994)	Increased soil respiration	Forest	Long term mineral fertilizer	
Brume and Besse (1992)	Increase in litter decomposition	Forest	Long term mineral fertilizer	
Hobbie (2000)	Increase in litter decomposition	Forest	Long term mineral fertilizer	
Vestgarden (2001)	Increase in litter decomposition	Forest	Long term mineral fertilizer	
Knorr <i>et al.</i> (2005)	Increase in litter decomposition	Forest	Long term mineral fertilizer	Only at sites with low ambient N deposition (5 kg ha ⁻¹ yr ⁻¹) and for high quality (low-lignin) litters
Studies suggesting no change in soil C under N enrichment				
Kowalenko <i>et al.</i> (1978)	No change in soil respiration	Abandoned agricultural	Long term mineral fertiliser	
Neilsen <i>et al.</i> (1992)	No change in soil organic carbon	Forest	Long term mineral fertiliser	Change in C/N ratio but no large change in SOC
Harding & Jokela (1994)	No change is soil organic carbon	Forest	Long term mineral fertiliser	Change in C/N ratio but no large change in SOC
Magill <i>et al.</i> (2004)	No change is soil organic carbon (15 years)	Forest	Long term mineral fertiliser	
Prescott (1995)	No change in litter decomposition rate	Forest	Long term mineral fertiliser	
Hobbie & Vitousek (2000)	No change in litter decomposition rate	Forest	Long term mineral fertiliser	

Knops <i>et al.</i> (2007)	No change in litter decomposition rate	Natural grassland	Mineral fertiliser (2 year)	Change in litter chemistry but not decomposition
Studies suggesting soil C gain under N enrichment				
Glendining & Powlson (1995)	Long term SOC meta-analysis – gain in SOC	Agricultural	Long term mineral fertiliser	Very small gain in SOC
Henriksen & Breland (1999)	C mineralisation decrease during incubation	Agricultural	Long term mineral fertiliser	Mineralization may be retarded at very high N concentrations
Hyvönen <i>et al.</i> (2007a)	Soil organic carbon increase - multiple sites	Forest	Long term mineral fertiliser	13 year continuous high N addition suppressed soil respiration by 41% in both hardwood and pine stands
Bowden <i>et al.</i> (2004)	Suppressed soil respiration	Forest	Long term mineral fertiliser	
Magnani <i>et al.</i> (2007)	Suppressed soil respiration	Forest	N deposition (low rates)	
De Vries <i>et al.</i> (2006)	Soil organic carbon increase - multiple sites	Forest	N deposition	
Prescott (1995)	Litter decomposition decrease	Forest	Long term mineral fertiliser	
Magill & Aber (1998)	Litter decomposition decrease	Forest	Long term mineral fertiliser	
Knorr <i>et al.</i> (2005)	Litter decomposition decrease	Forest	Long term mineral fertiliser	Only at sites with moderate ambient N deposition (5-10 kg ha ⁻¹ yr ⁻¹) or for low quality (high-lignin) litters
Persson <i>et al.</i> (2000)	C mineralisation decrease during incubation	Forest	Long term mineral fertiliser	30% reduction in mineralisation rate
Franklin <i>et al.</i> (2003)	Modelled SOC accumulation from ¹⁴ C measurements	Forest	Long term mineral fertiliser	60% of increased SOC estimated to be from decreased decomposition rate and the rest a result of increased litter production.
Craine <i>et al.</i> (2007)	C mineralisation decrease during incubation	Rangeland	Mineral fertiliser	N limitation increased decomposition
Evans <i>et al.</i> (20067)	Measured and modelled increase in SOC	Heathland	Long term mineral fertiliser	Increased growth rates and litter production
Pregitzer <i>et al.</i> (2008)	Surface SOC increased	Forests	Low rates (3 g NO ₃ ⁻ -N m ⁻² yr ⁻¹) of mineral fertilizer	SOC increases due to decreased decomposition rate rather than increased detrital inputs