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# Soil organic matter management across the EU best practices, constraints and trade-offs

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# Soil organic matter management across the EU – best practices, constraints and trade-offs

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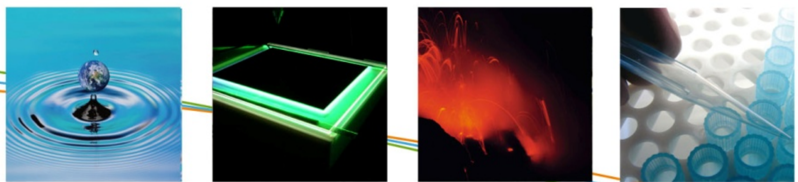

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# EXECUTIVE SUMMARY

The aims and objectives of the report "Soil organic matter management across the EU – best practices, constraints and trade-offs" are to assess the relative contributions of the different inputs and outputs of organic carbon and organic matter to and from the soil. From this assessment we evaluate the environmental consequences in view of improving the management of soil and biomass resources at the EU level. In this report we present the following issues:

- A literature review of the importance of soil organic matter (SOM) in ecosystems and its relevance to climate change (Chapter 2);
- An approach to assess soil organic carbon stocks and soil organic matter fluxes for agriculture, forests and peatlands, and to explore selected environmental policy and resource management options using scenario analysis (Chapter 3);
- Assessments of soil organic carbon stocks and fluxes for agriculture, forests and peatlands for the Baseline Period 2000-2005 and on the basis of investigating the impact of selected environmental policy and resource management options through the use of scenario analysis up to 2030 (Chapter 4, 5, 6, 7, 8, 9 and 10);
- A summary of the main results of the modelling and scenario work (Chapter 11);
- Examples of best practices across Europe to improve SOM management, based on case studies addressing specific issues (Chapter 12 and Annex II); and,
- Recommendations to improve policy and EU regulatory actions (Chapter 13).

European soils store around 73 to 79 billion tonnes of carbon, which is more than 50 times the total $CO_2$-equivalent emissions of the 27 Member States of the European Union in 2009 (4.6 billion tonnes). Particularly important are peatland soils, as they store 17 billion tonnes of carbon (around 20-25% of the total), whilst covering only 31 Mha or 7% of the EU-27 surface area. Peatlands are mainly located in Scandinavia, Ireland, northern Britain and Germany. **Soils are an important carbon stock: more than twice as much carbon is held in soils as compared to vegetation or the atmosphere**. Soil organic carbon (SOC) stocks are dynamic and changes in land use, land management and climate all have significant impacts. Both the European Commission and the United Nation's IPCC identify the decline of SOC worldwide as an environmental risk that undermines not only soil fertility and productivity, and hence food security, but also the progressive stabilisation and subsequent reduction of atmospheric $CO_2$ concentration levels.

Agricultural area in the EU-27 covers 166 Mha (38% of the total land area) and forest and other woodland covers 177 Mha (41%) in 2005 (Eurostat, 2010). There is a large spatial variability of soil under agriculture and forestry, so soil organic matter content is highly variable. In general terms the more sandy soils (coarse texture) retain lower amounts of soil organic matter, because organic matter is more quickly decomposed, due to greater soil pores and so higher decay rates. Soil organic matter monitoring programmess, long term experiments and modelling studies all indicate that changes in land use significantly affect soil organic matter levels. Soil organic matter losses occur when grasslands, forests and natural vegetation are converted to cropland. The reverse is true if croplands are converted to grasslands, forests and natural vegetation. **Land use changes can result in rapid carbon losses (i.e. instant), whereas gains accumulate more slowly (i.e. decadal)**.

**The soil organic matter or carbon cycle is based on continually supplying carbon in the form of organic matter as a food source for microorganisms, the loss of some carbon as carbon dioxide, and the build up of stable carbon in the soil (a process called assimilation) that contributes to soil aggregation and formation.** Carbon assimilation is a dynamic process necessary for nutrient availability and cycling. Different sources of organic matter have different assimilation and decomposition characteristics, and result in different soil organic matter fractions. **If the rate of assimilation is less than the rate of decomposition, soil organic matter will decline and, conversely if the rate of assimilation is greater than the rate of decomposition, soil organic matter will increase**. Both the assimilation and decomposition processes occur concurrently, but are of a different order of magnitude. Like for land use changes, organic matter can be lost instantaneously (e.g. by fire), whereas its build up is spread over several decades.

**Soil organic matter influences several critical soil functions and is affected by land management practices.** Because organic matter enhances water and nutrient holding capacity and improves soil structure, appropriate soil carbon management can enhance productivity and environmental quality, and can reduce the severity and costs of natural phenomena, such as droughts and floods. In addition, the practice of increasing soil organic matter levels may help in reducing atmospheric $CO_2$ that contribute to climate change. Decreases in soil organic matter content, through cultivation or tillage intensification, are often related to the deterioration of soil structure. Effects include the loss of aggregate stability, increased crust formation, increased runoff and soil erosion, increased compaction, slower water infiltration and a slower exchange of water/gasses.

A scenario approach is adopted in this study to explore the potential effects of selected environmental policy and resource management issues on land use and soil organic matter levels. For each environmental policy and resource management type we vary one or more parameters so as to define a set of scenarios. We use the regional organic matter balance model (REGSOM) to estimate regional carbon stocks and fluxes, and a dynamic land use change model for CAP impact assessment on the rural landscape (LUMOCAP) to analyse the effect of selected policies on land use area. We make the assumption that the average soil organic carbon (SOC) stock of the surface horizon reflects the equilibrium state; therefore, the differences between SOC stocks under different land uses reflect the change from one equilibrium state to another. Soil organic carbon fluxes on the other hand are like snapshots in time of the impact of resource management on the soil. **Carbon fluxes are therefore snapshots of carbon input and cannot be directly compared or added on to carbon stocks**. For all the scenarios the baseline period is 2000 – 2005, and the end year is 2030. The Hadley Climate model output with 1% compound increase of GHG is used in the LUMOCAP model. The scenarios allow for plausible quantified projections **but are by no means intended to predict the future**: their purpose is to illustrate "what-would-happen-if" type of situations.

The scenarios are summarised in the Table below. The starting point is the Business as Usual (BAU) scenario or central column. Scenarios are compared to the BAU in terms of environmental policy and resource management options that aim to maintain, increase (C-Rich and C-Medium), or decrease SOM (C-Low and C-Poor). For the agriculture and forest land use change scenarios, the difference between the topsoil soil organic stock of one land use compared to another land use is based on spatial analysis of the organic carbon content in topsoils in Europe database (Jones et. al. 2004), hosted by the JRC. The assumption is that the average SOC stock of the surface horizon under different land uses for a given NUTS region reflects the equilibrium state. The parameter(s) that are modified in each scenario in comparison to the BAU are indicated in bold.

*Table. Scenarios to assess the effect of selected environmental policy and resource management issues and options on soil organic matter levels in the EU to the 2030 horizon*

| *Environmental policy / resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Agriculture and forests - land use changes*** | | | | | |
| *Maintenance of grassland (Chapter 4)* | Grassland maintained as per current rules | Grassland maintained as per current rules | Grassland maintained as per current rules | Grassland maintained as per current rules | **Grassland restric-tions abolished** |
| *Use of set-aside (for EU-15 only) (Chapter 5)* | **25% former set aside to afforest-ation** | **10% former set aside to afforest-ation** | Former set aside to arable | Former set aside to arable | Former set aside to arable |
| *Change from Utilised Agricultural Area (UAA) to forest (Chapter 6)* | **Faster decrease of the UAA in favour of forests** | Current change of UAA in favour of forests | Current change of UAA in favour of forests | Current change of UAA in favour of forests | Current change of UAA in favour of forests |
| ***Agriculture – resource management options*** | | | | | |
| *Use of crop residues and straw (Chapter 7)* | 10% crop residues and straw to bio-energy | 10% crop residues and straw for bio-energy | 10% crop residues and straw for bio-energy | **30% crop residues and straw for bio-energy** | **50% crop residues and straw for bio-energy** |
| *Use of manure and compost (Chapter 8)* | Current manure and **50% more compost** available for application | Current manure and **25% more compost** available for application | Current manure and compost available for application | **20% manure used for bio-energy** | **40% manure used for bio-energy** |
| ***Forests – resource management options*** | | | | | |
| *Use of forest residues (Chapter 9)* | **No forest residues removed for bio-energy** | 10% forest residues removed for bio-energy | 10% forest residues removed for bio-energy | **20% forest residues removed for bio-energy** | **25% forest residues and 10% area stumps removed for bio-energy** |
| ***Peatlands – conservation*** | | | | | |
| *Conservation of peatlands (Chapter 10)* | **No further drainage of peatlands allowed** | **50% reduction of historical rates (1980-2000) for peat drainage** | Continuation of historical rates (1980-2000) of peatland drainage | Continuation of historical rates (1980-2000) of peatland drainage | Continuatio n of historical rates (1980-2000) of peatland drainage |

**Agriculture and forests - land use changes**

The **maintenance of grassland scenario** examines the effect of grassland area changes on levels of soil organic carbon stock. The maintenance of grassland areas is related to the Good Agricultural and Environmental Condition (GAEC) standards for permanent pastures that are in place for farmers to adhere to if they want to receive benefits from the Single Farm Payment Scheme. In this scenario we compare the impact of abolishing restrictions on maintaining permanent pasture areas (C-Poor 2030) with maintaining the current rules (BAU 2030). At the EU-27 level there is on average 31 tonnes/ha of SOC stock loss due to conversions of grassland to arable land. The distribution of these losses at the Member State level shows that the difference between SOC stock in arable and grassland soils is much larger in Central European Member States as compared to Southern European Member States. The different scenario options for maintaining permanent grasslands on SOC stock shows that the conversion from grass to arable will have a negative effect on soil carbon stocks. The highest SOC stock losses for the C-Poor scenario are Ireland, Austria and UK, whereas the lowest SOC stock losses are for Mediterranean countries. The average change for EU-27 is -17.2 tonnes/ha for the C-Poor scenario (permanent pastures GAEC rescinded) and -13.2 tonnes/ha for the BAU 2030 (the change in carbon stocks under BAU results from the LUMOCAP model and reflects land use changes and climatic effects to 2030). **Abolishing permanent grassland restrictions would have a negative effect on soil organic carbon stocks, which at EU level can be quantified in a carbon stock loss 30% higher than in the case of maintaining the current permanent grassland restrictions**.

The **use of set-aside scenario** examines the implications of putting set-aside under arable or under different degrees of afforestation. This scenario applies only to EU-15 Member States because set-aside only became compulsory, under the guaranteed price system of the Common Agricultural Policy, in 1992 and was not introduced at all to the new EU-12 Member States. There are no data on SOC stocks for set-aside areas, therefore we assume that the carbon stocks of set-aside areas are equal to the average values for grassland carbon stocks (natural vegetation of set-aside has characteristics similar to permanent grassland habitats with grasses covering around 75% of the fields) – even though this assumption is less realistic for the Base year (2000 – 2005) than for 2030. The conversion options are mostly to arable land and the majority of SOC stock changes are negative, with the BAU option being the most negative (the change in carbon stocks under BAU results from the LUMOCAP model and reflects land use changes and climatic effects to 2030). The SOC stock losses are much higher for Denmark (BAU 2030 loss is -36 tonnes/ha), Germany (-20 tonnes/ha) and Austria (-12 tonnes/ha), than for Member States such as the Netherlands (-0.6 tonnes/ha), Portugal (-1.2 tonnes/ha), Greece (-1.5 tonnes/ha) and Belgium (-2.5 tonnes/ha). The differences can be traced back to the relative importance of set-aside for the different Member States – for instance Denmark had more than 225 000 ha of set-aside, whereas Belgium only had 29 000 ha – but also to the soil organic matter content of the soils in the region. The average soil organic carbon stock loss for EU-15 is -5.2 tonnes/ha for BAU 2030, -4.2 tonnes/ha for C-Medium and -1.8 tonnes/ha for C-Rich. **Promoting the afforestation of 10% and 25% former set-aside land in the EU-15 would therefore reduce the loss of soil organic carbon by 2030 by 19% and 65% respectively compared to a business as usual (BAU) scenario**.

The **change from utilised agricultural area (UAA) to forest scenario** examines the effect on the soil organic carbon stock of converting agricultural land to forest at a higher rate (2% higher) (C-Rich) than the current conversion rates (BAU). The scenario is related to agri-environmental measures that encourage farmers to convert agricultural land to forest. At the EU-27 level there is on average 47 tonnes/ha of SOC stock gain due to conversions of UAA to forest land. The distribution of these gains at the Member State level shows that the difference between SOC stock in arable and forest soils is much larger in Central European Member States compared to Southern European Member States. The highest relative SOC stock gains for the C-Rich scenario are in Slovakia and the Czech Republic, +54.8 tonnes/ha and +43 tonnes/ha respectively. Finland and Sweden on the other hand have very minor relative gains of less than 6 tonnes/ha, because in these Member States the share of UAA is very small compared to forest area. The average SOC stock change for EU-27 is +18.2 tonnes/ha for the BAU 2030 scenario, compared to +20 tonnes/ha for the C-Rich scenario. **At the EU level an increase of the afforestation rate by 2% compared to business as usual would result in a 10% increase in carbon stock levels by 2030.**

The overall assessment of soil carbon stocks under agriculture and forests (Figure A) indicates that the combined effect of land use changes to and from agricultural land use in the different scenarios and for different Member States demonstrates an EU-27 average -9.7 tonnes/ha SOC stock loss for the C-Poor option and a +5.0 tonnes/ha SOC stock gain for C-Rich option. In general, the analysis confirms that forests sequester more carbon than agricultural land; and that grass sequesters more carbon than arable. When considering all the addressed land use changes and respective scenario options we have shown that there are much greater differences between Member States than between scenario options.

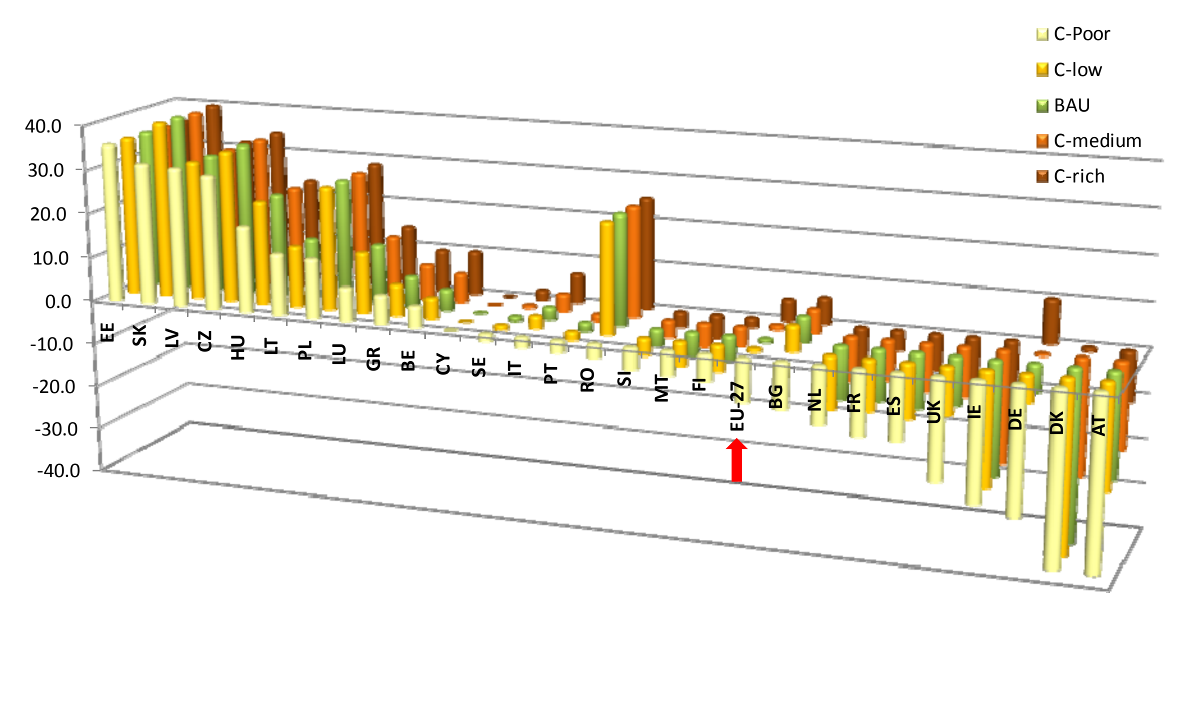



*Figure A. SOC stock changes (in tonnes/ha) due to land use conversions to and from agricultural land, i.e. UAA to forest, arable-grassland conversions and set-aside.*

### Agriculture – resource management options

The **use of crop residues and straw scenario** examines the impact of the use and management of crop residues and straw on soil organic carbon fluxes. The agricultural potential for organic matter sources depends on residue production such as crop residues from annual and perennial crops and manure application. In this scenario we compare using 30% (C-Low) and 50% (C-Poor) of crop residues and straw for bio-energy, respectively, with 10% crop residues and straw (BAU). In addition to the scenarios indicated in the above Table we introduce a Worst Case scenario to examine the impact of extreme practices, whereby 100% of residues are harvested. Data on harvest indices, root/shoot ratios and effective organic carbon content are combined with cropping areas and crop production to calculate the amount of agricultural residues produced across Europe, from which humified organic carbon content is estimated. The amount of humified organic carbon assimilated to the soil depends, firstly, on the yields, as these directly relate to potential residue production, and secondly, on the prevailing climate with cold temperatures and dry moisture regimes being less favourable.

For *cereal* production two management options are presented: straw left as residue on the field and straw harvested. The practice of leaving cereal straw in the field has the potential of doubling the effective organic matter input. Member States with a high production such as Belgium, the Netherlands, Ireland, Denmark, the United Kingdom, Germany, Luxembourg and France have therefore a higher potential for sequestering carbon into the soil than the average for EU-27 at 0.86 tonnes/ha for all straw incorporated and at 0.44 tonnes carbon/ha for all straw harvested. The regional distribution further confirms this with Southeastern England, Northern France, Northern Belgium, the Netherlands and Northern Germany displaying the largest humified organic carbon of Europe. At the EU-27 level the scenario analysis shows that C-Low 2030 and C-Poor 2030 humified organic carbon (HOC) levels are 7% and 21%, respectively, lower than BAU 2030 levels, and that this reduction is 38% below for the Worst Case scenario, where all residues are used for bio-energy.

*Sugar beet* only has half of the capacity for assimilating humified organic carbon to the soil as compared to cereal. For sugar beet production the option of shoots incorporated into the soil is compared to combined root and shoot harvesting. Although the potential to introduce organic matter into the soil is lower as compared to cereals, residue management has a large impact. Root and shoot harvesting leaves little organic matter after cultivation: with an EU-27 average of 0.046 tonnes HOC/ha a factor 10 less as compared to residue incorporation into the soil. At the EU-27 level the scenario analysis shows that C-Low 2030 and C-Poor 2030 humified organic carbon levels are 20% and 40%, respectively, lower than BAU 2030 levels, and that this reduction is 90% below for the Worst Case scenario, where all residues are used for bio-energy.

In the case of *oilseed* the option of straw harvesting is compared to straw incorporation into the soil. The ratio of grain to straw on a weight basis is lower than cereal and therefore results in an average 30% (EU-27) higher flux of humified organic carbon to the soil as compared to cereal. When oilseed straw is incorporated in to the soil it results in an average flux of 1.12 tonnes HOC/ha (EU-27) which is five times higher as compared to harvesting residues. The lowest fluxes are found in Bulgaria and Romania, and the highest in Belgium and IrelandE. The highest fluxes are found in mid-Germany, Northern France, Northern Belgium, the Netherlands and Southern United Kingdom. At the EU-27 level the scenario analysis shows that C-Low 2030 and C-Poor 2030 humified organic carbon levels are 9% and 22%, respectively, lower than BAU 2030 levels, and that this reduction is 47% below for the Worst Case scenario, where all residues are used for bio-energy.

Although fluxes under grassland relate to permanent grassland for which regular harvesting is assumed, the case of grass ploughing is considered as it relates to a common farming practice for *temporary grasslands* and provides for a comparison with regular harvesting under permanent grassland. Grass ploughing provides for an instant large flux of organic matter to the soil that results in a European average of 1.74 tonnes HOC/ha. Regular harvesting of grass biomass ensures an average of 0.43 tonnes HOC/ha realised. Grass ploughing may realise up to 4.5 times the amount of tonnes HOC/ha as compared to regular biomass harvesting. Since the addition provides for an instant fairly substantial flux, temporary grass is common in arable rotations of livestock farms. The carbon sequestration potential of this practice (long-term), however, should be further evaluated against the fluxes of subsequent arable crop growth in common rotation schemes.

The distribution of input of stable or humified organic carbon to the soil under different crops and for different scenarios of crop residues harvested shows large differences across the different regions of Europe. High cereal production in Western European regions are responsible for higher input of stable or humified organic carbon to the soil. At the same time favourable weather conditions (warm and moist) explain increased ability to assimilate organic material in the form of humified organic carbon into the soil. For grassland a comparison is made between grass harvesting and grass ploughing. Although grass ploughing provides for an instant addition of large quantities of organic material into the soil and hence humified organic carbon, the soil reserve is more easily exposed to organic matter decline. This practices explains the benefit of incorporating grass into rotations as it provides for a large instantaneous flux. Under high productive conditions, the effect is more pronounced.

The projected areas for cereals, oilseed and sugarbeet in 2030, according to the LUMOCAP BAU scenarios, are 65 Mha, 10 Mha and 2 Mha, respectively. **This means that residue management of cereals has a much larger impact on carbon fluxes than oil seed and sugar beet.**

From the overall assessment of the soil organic carbon fluxes under agriculture (Figure B) we see that the BAU 2030 scenario option for grass residues (1.58 tonnes/ha) can provide more than three times the levels of humified organic carbon than sugar beet (0.5 tonnes/ha). The Worst Case scenario option of removing all residues from the field, reduces the humified organic carbon from grass to 0.58 tonnes/ha, but this is still higher than the BAU 2030 scenario option for sugar beet.

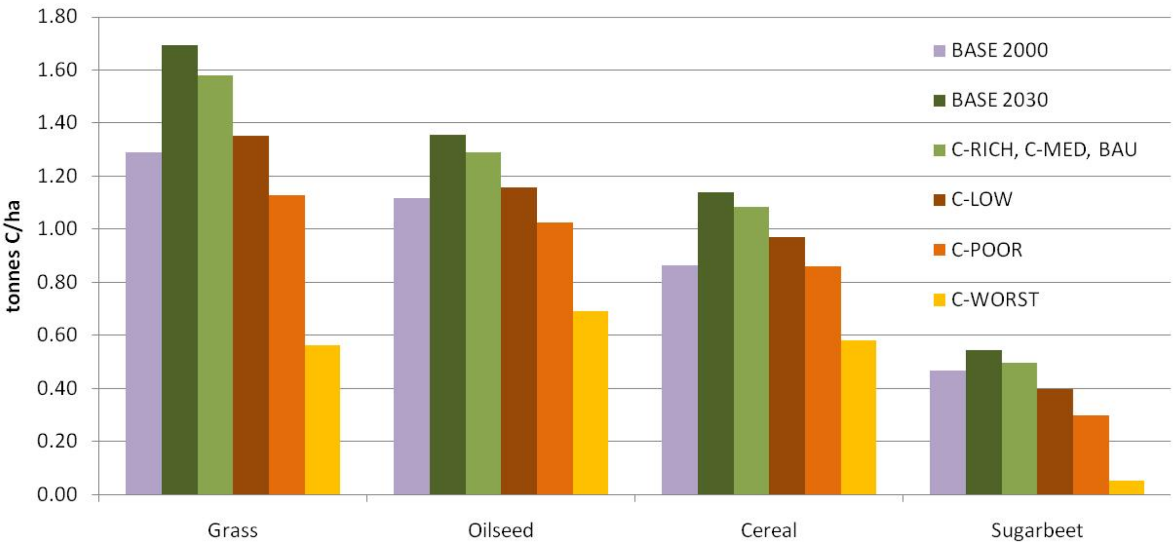



*Figure B. Flux of Humidified Organic Carbon (tonnes per ha) into the soil from grass, oilseed, cereal and sugar beet residues based on BAU 2030, C-Low, C-Poor and C-Worst Case scenarios at the EU-27 level. In Base all residues remain on the field, whereas in C-worst all residues are removed from the field.*

The **use of manure and compost scenario** assesses the availability of organic carbon to agricultural land from manure produced by farms and compost produced from the urban waste. For the C-Rich and C-Medium scenarios manure application rates are at the current potential level, whereas 50% more compost and 25% more compost are applied, respectively. The C-Low scenario assumes that 20% of manure produced is used for energy production and C-Poor scenario assumes that 40% of manure produced is used for energy production. Results for compost application show a clear difference between the extreme scenarios of the potential application rate and the reduced application rate when 40% of the biowaste production is used for bio-energy purposes rather than compost. In the latter case this results in a reduced potential to influence soil organic matter levels.

Households and the service sector (schools, hospitals, offices and shops) are potential providers of compost in urban areas. Two types of compost are considered: kitchen compost made from vegetables, fruit and gardening waste (kitchen-compost) and green compost made from made from prunings, branches, grass and leaf litter (green-compost). **Humified organic carbon is calculated with the (quite conservative) assumption that all compost produced is spread on the entire utilised agricultural area.** Potential kitchen-compost is assumed to be half of the potential green-compost and the yearly production is assumed to be 150 kg/inhabitant. In addition, the stable organic carbon flux is higher from green-compost compared to kitchen-compost. This explains the factor two difference between humified organic carbon (kg/ha) between potential green-compost and potential kitchen-compost spread over the entire utilised agricultural area. Member States such as Belgium and the Netherlands have potential green composting rates of more than 600 kg/ha, since they are heavily populated and have a relatively small utilised agricultural area when expressed per person. In comparison the EU-27 average is 120 kg/ha. **At the EU-27 level the compost scenario options indicate an increase in humified organic carbon from 0.05 tonnes/ha (BASE 2005) to 0.07 tonnes/ha (BAU 2030), representing an increase of 40%. The levels of humified organic carbon are increased to 0.08 tonnes/ha and to 0.095 tonnes/ha for 25% (C-Medium 2030) and 50% (C-Rich 2030) increases in compost generation, respectively. This represents an increase of 14% and 36%, respectively, compared to BAU 2030.** Highly populated Member States, such as the Netherlands and Belgium could reach up to 0.25 and 0.23 humified organic carbon (tonnes/ha), respectively, if potential compost generation is increased by 50% (C-Rich 2030), whereas for lowly populated Member States, such as Estonia, Lithuania, Ireland, Latvia and Slovenia increasing the potential compost generated by 50% would still result in humified organic carbon levels of less than 0.05 tonnes/ha. The distribution of potential green-compost production across Europe shows a similar pattern to kitchen-compost, but the potential for stable carbon assimilation is more or less double. This means that if the organic matter resources are added together, the C-Rich scenario will provide nearly 0.3 humified organic carbon (tonnes/ha) at the EU-27 level. For the Netherlands and Belgium the addition of kitchen and green composts for the C-Rich scenario option results in 0.8 and 0.75 humified organic carbon (tonnes/ha), respectively.

The application of livestock manure to agricultural land is a source of carbon for increasing soil organic matter in the soil profile. Currently there are no statistical data available on livestock manure applications, in terms of manure type, storage practices, C:N ratios, other uses of livestock manure, field application rates and field application methods. In addition the manure application rates reported under the Nitrates Directive are not complete and have not been verified. The approach to calculate the stable or humified organic carbon resulting from the the application of livestock manure on farm areas was based on statistical information on livestock populations and coefficients either reported or found in the literature. The distribution of livestock manure production in terms of N kg/ha and therefore also carbon for the soil indicates that the major producers and users of livestock manure are the regions of Flanders, Britanny, Southern Netherlands and West Denmark. In general Southern Europe, Eastern Europe and Northern Europe are the lowest producers of livestock manure (less than 60 N kg/ha) and Western Central Europe are the highest producers of livestock manure (more than 150 N kg/ha). **At the EU-27 level stable organic carbon input levels drop from 0.19 tonnes/ha to 0.15 tonnes/ha between BASE 2000 and BAU 2030. This 21% reduction is due to the expected reduction in livestock levels between 2000 and 2030. The input levels in 2030 are further reduced to 0.12 and 0.09 tonnes/ha, by using 20% (C-Low) and 40% (C-Poor) of the available manure for bio-energy, respectively. This means that at the EU-27 level C-Low 2030 and C-Poor 2030 scenarios result in 40% and 60% reductions in humified organic carbon levels compared with BAU 2030.** For Member States such as the Netherlands, Belgium and Ireland, where livestock production is important, the differences between the scenario options are greater, but the C-Poor 2030 humified organic carbon levels are still double the Base 2000 levels of many Scandinavian and Eastern European Member States. This indicates that manure as a source to provide more organic matter to soils is highly variable across Europe, as well as being highly variable between regions of some of the larger Member States of EU-27 (e.g. France and Spain).

From the overall assessment of the soil organic carbon fluxes under agriculture for the use of composts and manure (Figure C) it is clear that at the EU-27 level manure and compost production is not as important as crop residues for providing stable organic carbon input. However, at the regional level – where in some regions manure and compost production is high (e.g. Northern Belgium, Southern Netherlands) – these sources might be important supplements to the soil, especially if crops and crop residues are being harvested for bio-energy. Even more so, as one as to remember that it has been assumed that compost is spread on the entire utilised agricultural area. This is an oversimplification made necessary by the lack of more specific information on compost use possibilities at the regional level. In any event, care has to be taken that nutrient applications do not exceed specific application rates set by legislation.

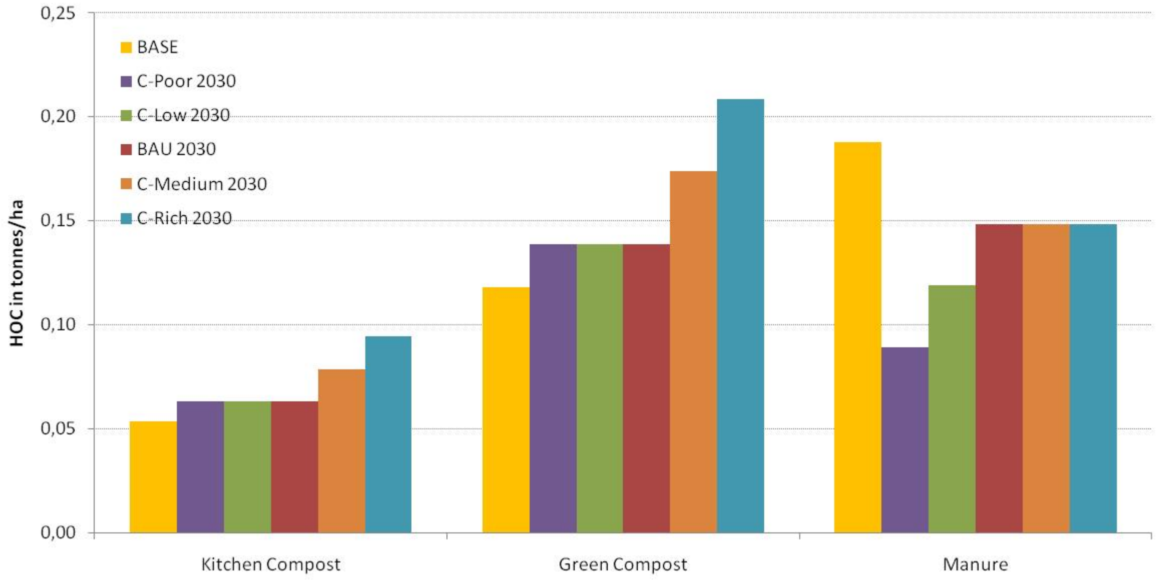



Figure C. Humidified Organic Content in tonnes per ha from the application of kitchen compost, green compost and livestock manure based on Base 2000, C-Poor 2030, C-Low 2030, BAU 2030, C-Medium and C-Rich scenarios at the EU-27 level.

**Forests - resource management options**

The **use of forestry residues scenario** examines the impact of different forest residue management options, with a particular focus on whether residues from branches and roundwoods that are removed from forests for bio-energy production have a detrimental effect on soil organic matter levels. In this scenario we compare using 20% forest residues removed for bio-energy (C-Low) and 25% forest residues and 10% area stumps removed for bioenergy (C-Poor) with 10% forest residues removed for bio-energy (BAU). In addition, we introduce a Worst Case scenario to examine the impact of extreme practices, whereby 70% of the wood residues and 25% of the stumps are removed.

The general carbon balance under forests depends primarily on forest biomass production. The organic matter that contributes to soil organic carbon depends on the interaction between the two major stocks: the forest biomass and the forest soil reserve. Fluxes to and from the soil reserve are influenced by litterfall, natural fellings, deadwood, logging residues and disturbances.

The difference between the baseline (2000) and the C-rich scenario (2030) represents the climatic effect on carbon turnover to humified organic carbon in the soil. For coniferous forests, the climate effects result in up to 7.4% *decrease* in humified organic carbon added to the soil as compared to baseline for Southern European Member States and up to 9.7% *increase* for Northern European Member States. For broadleaved forests, the climate effects are up to 6.3% decline (S-Europe) and 9.2% increase (N-Europe). The decrease links to a drier moisture regime in Southern Europe, whereas the increase in Northern Europe relates to warmer temperatures. Climatic change influences organic matter decay factors and has the largest impact on easily decomposable forest residues, i.e. mainly foliage and fine roots.

The composition of forest residues determines the flux into the soil. The contribution of woody residues represents a slow flux into the soil, but provides for an important carbon reserve that adds to the overall forest carbon stock. Woody residues contribute on average for EU-27 and depending on the scenario less than one third of the humified organic carbon into the soil. Consequently, **the influence of forest residue management across the different scenarios is most noticeable for C-Worst Case scenario with an EU-27 decrease of 35.6% for coniferous forests and 33.6% for broadleaved forests.** The scenarios C-Poor and C-Low are not significantly different for the contribution of woody residue to humified organic carbon due to a double effect of increased wood removal and increased foliage removal with stump harvesting. The increased foliage removal, however, results in a significant difference in carbon flux decline into the soil. Soil carbon assimilation rates for broad leaved forests are on average 1.6 times higher than for coniferous forests.

From the overall assessment of the soil organic carbon fluxes under forests (Figure D) we see that the input of stable or humified organic carbon to the soil for broad leaved forests is 1.75 times that of coniferous forest because of higher decay and assimilation rates. Broad leaved forest can assimilate up to 1.7 tonnes C/ha yearly; coniferous forests up to 1.1 tonnes C/ha. The differences can be attributed predominantly to the share of needles or leaves and fine roots. The composition of forest residues determines the flux into the soil. The contribution of woody residues represents a slow flux into the soil, but provides for an important stable carbon reserve that adds to the overall forest carbon stock. Woody residues contribute, depending on the scenario, less than one third of the humified organic carbon flux into the soil.

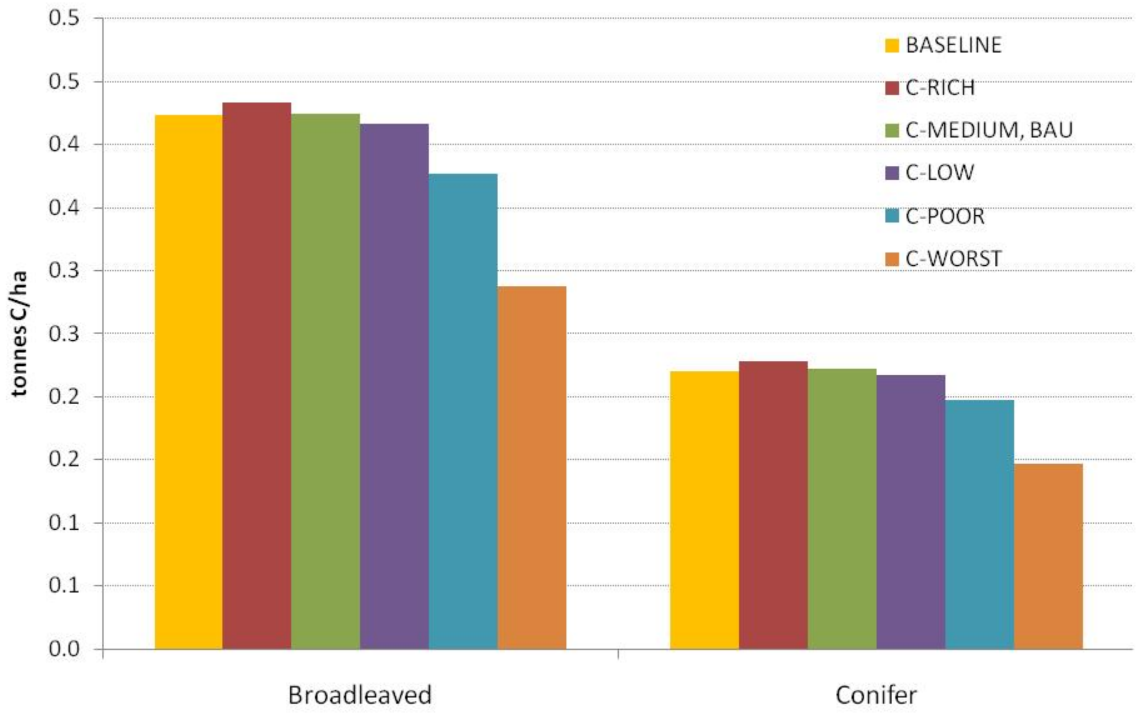



Figure D. Flux of Humidified Organic Carbon in tonnes per ha from broadleaved and conifer forest residues to the soil based on Base 2000, C-Rich 2030, BAU 2030, C-Low, C-Poor and C-Worst Case 2030 scenarios at the EU-27 level.

**Peatlands – conservation**

The **conservation of peatlands scenario** assesses the factors that determine soil organic carbon stock and fluxes under peatlands and examines the impact of different options to conserve peatlands. We compare the following scenarios: BAU 2030 assumes that the historical rates of peatland drainage are continued, C-Medium 2030 assumes a 50% reduction of historical rates, and C-Rich 2030 assumes that no further drainage of peatlands is allowed. In addition, we include a Best Case scenario that assumes that 25% to 100% of existing peatlands are restored by 2030.

There is a great deal of uncertainty in assessing the surface area of peatlands and their carbon stocks, due to their definition, depth and density. The main processes affecting the carbon balance of peatlands are carbon accumulation due to peat formation and in-situ losses due to different types of land management (unmanaged or natural peatlands, forests, grassland, and arable land). Land management affecting the water table will inevitably affect the carbon and greenhouse gas balance. In case peat and/or vegetation are harvested, off-site carbon losses may become important as well (e.g. peat combustion for energy). Exploitation of peatlands for forestry, agriculture or peat extraction involves drainage of the area. As a result, the drained peat layer undergoes oxidation resulting in emissions of $CO_2$. Although there are gaps in the available data on land use in peatlands, it is estimated that 20% of the European peatland area has been drained for agriculture, 28% has been drained for forestry and less than 1% is used for peat extraction. **The distribution of different land uses on peat soils in EU-27 for those Member States having considerable areas of peatland demonstrates that the majority of peatlands are no longer pristine.**

Yearly percentages of peatland carbon stock gains and losses are estimated for EU-27, with and without extraction. The estimates are based on carbon balances (only $CO_2$ and

$CH_4$) using peatland area, historic land use conversion data and emission factors. Peat extraction emissions are estimated from peat production volumes reported in the Industrial Commodity Statistics Database of the United Nations assuming the entire carbon content to be released. The estimated peatland carbon stock loss rates in EU-27 range between 0.13 and 0.36% per year, which means that **13-36% of the current soil carbon stock in European peatlands might be lost by the end of this century**. Large regional differences exist. In some Member States, all peatland carbon reserves may already be gone within a couple of decades. **Obviously curbing current land use conversion rates will be necessary to safeguard the large carbon reserve of peatland soils.**

Peatland carbon and greenhouse gas emission balances are estimated for EU-27 using peatland area and land use data. Multiple estimations are made applying different sets of emission factors for peat soils collected from literature. Over the period 1990-2007, the average total peat extraction in the EU amounted to 21 Mt per year. Off-site emissions are estimated to account for 6-16% of the summed greenhouse gas emissions from peatlands and peat use in EU-27. Compared to the total greenhouse gas emissions, the contribution of off-site emissions from extracted peat is less than 1%, but in some countries it can be much higher. In Finland, for example, peat combustion is estimated to generate about 15% of the country's net greenhouse gas emissions.

Following the assessment of the soil organic carbon fluxes under peatlands (Figure E) - expressed in carbon loss tonnes per hectare per year, current carbon losses are around 1.6 tonnes of carbon per hectare and include peat extraction. The scenarios assume no further peat extraction but assume a continued conversion to forest and agricultural land at both current and lower rates until the year 2030. No further conversion (C-Rich) results in a loss of 1.4 tonnes of carbon per hectare. If 50% of the original peatlands were rewetted, then yearly losses would be 0.58 tonnes of carbon per hectare for EU-27. In the case of 100% peatlands restoration 0.23 tonnes per hectare of carbon can be sequestered annually.

**The BAU 2030 scenario indicates that on the basis of the same historical trends in peatland drainage there will be a 4% decrease in greenhouse gas emissions by 2030, compared to BASE 2000 for EU-27. This compares to a 8% decrease in greenhouse gas emissions by 2030 if the current trends are reduced by 50% (C-Medium 2030) and a 12% decrease in greenhouse gas emissions by 2030, if no further peatlands drainage is allowed (C-Rich 2030). For the Best Case scenarios, whereby up to 100% of peatlands are restored, greenhouse gas emissions are further reduced until peatlands become a sink for GHG emissions rather than a source.**

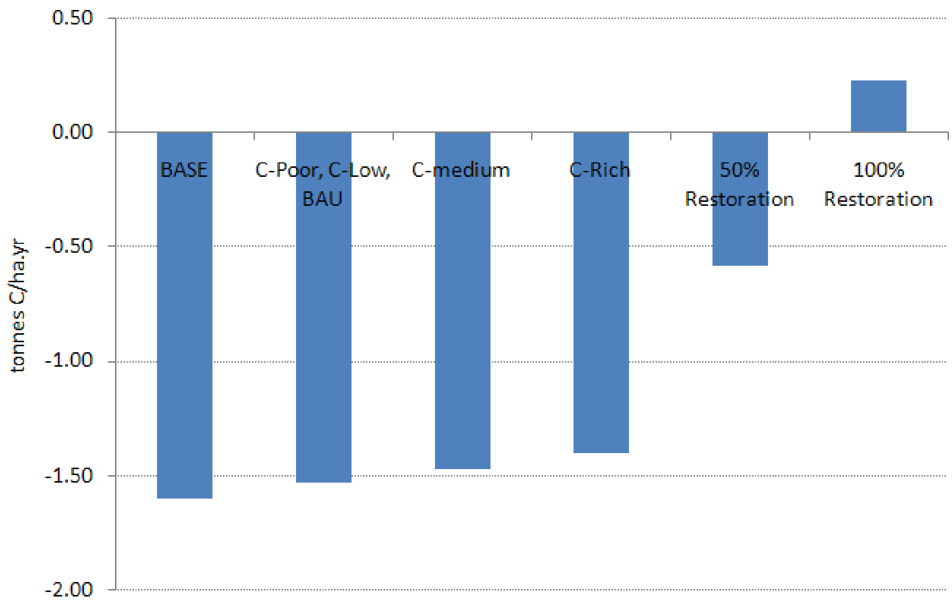



*Figure E. Carbon fluxes from the soil in tonnes per ha per year from peatlands during Base 2000, C-Poor 2030, C-Low 2030, BAU 2030, C-Rich 2030, and 50% and 100% restoration scenarios at the EU-27 level (positive values are gains, negative values are losses).*

The study (including the eight case studies presented in Chapter 12 and in Annex II) enables us to make the following conclusions and recommendations:

1. Bio-geography and pedology are important factors in determining the levels of soil organic matter across Europe, showing that **practices need to be adapted to regional conditions to be most effective.** Policy decisions at the regional level have to take this into account.

2. We have shown that crops or forests grown for bio-energy production, whereby all residues are removed, is detrimental to the soil, resulting in a reduction of soil organic carbon stocks and an increase of carbon dioxide concentrations in the atmosphere. Therefore, we recommend that **a (significant) minimum percentage of residues should be retained in soils for crops and forests grown for bio-energy.** Further work needs to be done to set such minimum percentage values, which could vary between bio-geographic regions as well as crop and forest types. These standards could be introduced through cross-compliance measures or standards for good agricultural and environmental conditions (GAECs) for crops under the Common Agricultural Policy and the use of standards or labels for crop and forestry products used for bio-energy production.

3. The policy implications for compost and livestock manure are also highly regional. Densely populated regions have the potential to provide compost for improving the soil organic status of the surrounding farm areas, however the cost implications of transporting urban produced compost need to be taken into account. Livestock manure can only be used for bio-energy production in highly intensive livestock rearing regions. In these regions, **bioenergy production can be seen as an added environmental benefit for manure** that has to otherwise be kept in storage facilities that are built to reduce N emissions. Indeed farmers should be encouraged to use liquid manure for producing bio-energy and then **transforming this bi-product into a compost rather than spreading or injecting liquid raw manure into the soil.** The case study work indicates that farmers are not keen to add composts to fields when they are not

confident of the quality – so improved standardization or quality labels need to be introduced. However, for both manure and composts, care still has to be taken that nutrient applications do not exceed specific application rates set by legislation.

4. Concerning peatlands we see that the current land use conversion and peat extraction rates enhance drainage and decomposition, thus increasing greenhouse gas emissions, and that the restoration of peatlands turns them from a carbon source into a carbon sink. This means that **the conservation, restoration and management of peatlands should be an important environmental policy concern in terms of both retaining peatlands as a key land use to reduce or even reverse carbon dioxide and also methane emissions**. It is clear, therefore, that peatland drainage, for example for agriculture and forestry, needs to be stopped and reversed, to prevent further emissions. This has implications for Climate Change policy and negotiations, but also for policy measures in the Common Agricultural Policy and NATURA 2000.

5. There is a need to increase the understanding of complex relationships in the soil carbon cycle. There are significant challenges in coming up with cost effective techniques to measure soil organic carbon changes efficiently. Climate change but especially – as this report shows – land use practices and land use changes are likely to have a significant influence on soil carbon stocks and will make it more difficult to predict the sequestration potential of soils and its permanence. **Soil monitoring is therefore vital to provide evidence on the state of, and change, in our soils**, underpinning policy development and allowing to evaluate its effectiveness. This means developing a set of soil quality indicators and new biological indicators of soil quality.

# Table of Contents

**Executive Summary** ____ III

**Table of Contents** ____ XVII

**List of Figures** ____ XXI

**List of Tables** ____ XXV

**List of Boxes** ____ XXVI

**Abbreviations** ____ XXVII

**Chapter 1 Introduction** ____ 1

*1.1 Aims and objectives of the project* ____ 1

*1.2 Scope of the Report* ____ 1

**Chapter 2 The role of soil organic matter in ecosystems and society** ____ 3

*2.1 Current state of soil organic matter across Europe* ____ 3

*2.2 Soil organic matter dynamics* ____ 5

*2.3 Soil organic matter and its functions* ____ 10

*2.4 Soil organic matter quantity and quality* ____ 13

2.4.1 Organic matter supplements to the soil ____ 13

2.4.2 Land management to maintain or increase soil organic matter ____ 14

*2.5 Optimal and Maximum Input Potential* ____ 16

*2.6 The organic matter cycle* ____ 17

2.6.1 The global carbon cycle ____ 17

2.6.2 Fluxes in ecosystems ____ 19

2.6.3 Fluxes in society ____ 21

*2.7 Economic value of sequestering carbon in soils* ____ 21

**Chapter 3 Approach to assess regional soil organic matter balances and scenario analyses** ____ 25

*3.1 Introduction* ____ 25

*3.2 The regional organic matter balance* ____ 25

3.2.1 Concepts ____ 25

3.2.2 Components ____ 27

*3.3 Scenario analysis to assess the effect of selected environmental policy and resource management issues on soil organic matter levels* ____ 29

**Chapter 4 Maintenance of grassland** ____ 31

*4.1 Introduction* ____ 31

*4.2 Scenario Approach and Method* ____ 31

*4.3 Soil organic carbon stock under agriculture* ____ 33

4.3.1 Surface area of agricultural land ____ 33

4.3.2 Status of soil organic matter under agriculture ____ 33

*4.4 Results ____ 34*

4.4.1 Distribution of soil organic carbon content in the surface horizon of grasslands ____ 34

4.4.2 Distribution of soil organic carbon content in the surface horizon of arable land ____ 34

4.4.3 Impact on soil organic carbon stocks of converting grasslands to arable ____ 35

4.4.4 Change in grassland areas and the change in the grassland share of agriculture due to the scenario option maintaining the current rules for the GAEC permanent pastures (BAU 2030), and for the scenario option abandoning the current rules for the GAEC permanent pastures (C-Poor 2030) ____ 36

4.4.5 Change in soil organic carbon stock (tonnes/ha) due to the scenario option maintaining the current rules for the GAEC permanent pastures (BAU 2030), and for the scenario option abandoning the current rules for the GAEC permanent pastures (C-Poor 2030) ____ 38

**Chapter 5 Use of set-aside (for EU-15 only) ____ 41**

*5.1 Introduction ____ 41*

*5.2 Scenario approach and method ____ 41*

*5.3 Soil organic carbon stock under agriculture ____ 42*

*5.4 Results ____ 42*

**Chapter 6 Change from utilised agricultural area (UAA) to forest ____ 45**

*6.1 Introduction ____ 45*

*6.2 Scenario approach and method ____ 45*

*6.3 Soil organic carbon stock under forests ____ 46*

6.3.1 Surface area of forest and other wooded land ____ 46

6.3.2 Status of soil organic matter under forests ____ 49

*6.4 Results ____ 51*

6.4.1 Trends in forest areas at Member State level ____ 51

6.4.2 Distribution of soil organic carbon content in the surface horizon of forests ____ 52

6.4.3 Impact on soil organic carbon stocks of converting UAA to forest ____ 53

6.4.4 Change in forest areas and the change in the forest share due to the scenario adopting the current change of UAA in favour of forests (BAU 2030) and adopting a faster decrease of UAA in favour of forests (C-Rich 2030) 54

6.4.5 Change in soil organic carbon stock loss (tonnes/ha) due to the scenario adopting the current change of UAA in favour of forests (BAU 2030) and adopting a faster decrease of UAA in favour of forests (C-Rich 2030) 56

**Chapter 7 Use of crop residues and straw ____ 59**

*7.1 Introduction ____ 59*

*7.2 Scenario Approach and Method ____ 59*

*7.3 Regional organic matter balance for crop residues ____ 60*

*7.4 Results ____ 69*

**Chapter 8 Use of manure and compost** 73
*8.1 Introduction* 73
*8.2 Scenario approach and method* 73
8.2.1 Production of organic matter from urban areas 74
8.2.2 Reported compost production 74
8.2.3 Potential compost production 75
8.2.4 Humified Organic Carbon of compost as spread on UAA 77
8.2.5 Organic carbon flux of livestock manure 78
*8.3 Results* 82
8.3.1 Projected trends in livestock manure production 82
8.3.2 Projected trends in potential compost production 83
8.3.3 Impact of different resource management options on regional soil organic carbon fluxes from livestock manure and potential compost production 84
8.3.4 Lifestock manure 86
8.3.5 Compost 86
**Chapter 9 Use of forest residues** 89
*9.1 Introduction* 89
*9.2 Scenario approach and method* 89
*9.3 Production of organic matter from forests* 90
9.3.1 Forest biomass production 90
9.3.2 Forest biomass compartments 93
9.3.3 Fluxes from living forest biomass to the soil 96
9.3.4 Fluxes from felled biomass to the soil 97
9.3.5 Forest organic carbon balance 97
*9.4 Loss of organic matter from forest products* 100
*9.5 Results* 101
**Chapter 10 Conservation of peatlands** 105
*10.1 Introduction* 105
*10.2 Scenario approach and method* 105
*10.3 Soil organic carbon stock under peatlands* 106
10.3.1 Surface area under peatland 106
10.3.2 Current state of soil organic carbon under peatlands 108
10.3.3 SOC stock loss under peatlands 110
*10.4 Soil organic carbon fluxes under peatland* 112
10.4.1 Positive SOC fluxes under peatland 112
10.4.2 Negative carbon fluxes for peatland 112
*10.5 Contribution of peatland to GHG balance* 114

10.5.1 Contribution of peatland to GHG balance ______ 114

10.5.2 Contribution of peatlands to GHG emissions ______ 115

*10.6 Results ______ 117*

10.6.1 BASE 2000: Impact of peat extraction rate on carbon stock gains and losses ______ 117

10.6.2 Scenarios ______ 117

**Chapter 11 Summary and conclusions of soil organic carbon stock and fluxes analysis ______ 121**

*11.1 Soil organic carbon stocks under agriculture and forests ______ 121*

*11.2 Soil organic carbon fluxes under agriculture ______ 124*

11.2.1 Crop residues ______ 126

11.2.2 Compost/manure ______ 126

*11.3 Soil organic carbon fluxes under forests ______ 127*

*11.4 Soil organic carbon stocks and fluxes under peatlands ______ 128*

**Chapter 12 Identification of best practices in relation to soil organic matter management on the basis of selected case studies ______ 131**

*12.1 Selection of case studies ______ 131*

*12.2 The effect of long term crop rotations on soil organic matter status in North Eastern Italy ______ 134*

*12.3 Long term effect of reduced tillage systems on soil organic matter in Northern France ______ 134*

*12.4 Evaluation of crop residue management options on soil organic matter levels in Jutland (Denmark) ______ 135*

*12.5 Production and management of compost in Northern Belgium ______ 135*

*12.6 Production and management of sugar-beet composts (vinasse) in South Western Spain ______ 136*

*12.7 Effects of afforestation on arable land in Northern Europe ______ 136*

*12.8 Conservation of mires in Latvia ______ 137*

*12.9 Restoration of bogs in Ireland ______ 137*

**Chapter 13 Recommendations ______ 139**

**References ______ 141**

**Annex I Description of LUMOCAP ______ 145**

**Annex II Case Studies ______ 149**

## List of Figures


Figure 1 Status in 1990 of organic carbon content in topsoils (0–30cm) in Europe 3
Figure 2 The breakdown of organic material such as natural plant residues, forest litter, compost, manure or bio-waste into soil organic matter pools 7
Figure 3 Composition of soil organic matter 8
Figure 4 The global carbon cycle and carbon reservoirs 18
Figure 5 The organic matter balance 26
Figure 6 Share of arable land, permanent grassland, permanent crops and set aside to utilised agricultural area ranked by share of arable land according to 2006 Eurostat farm statistics, where PGrass = permanent grassland, PCrops= Permanent crops. 33
Figure 7 Mean topsoil organic carbon content (%) for permanent grasslands in the EU 34
Figure 8 Mean topsoil organic carbon content (%) for arable land in the EU 35
Figure 9 Potential SOC stock loss in tonnes/ha as a result of converting grassland to arable land on the basis of the topsoil organic carbon maps and assuming a surface horizon thickness of 20 cm 36
Figure 10 Grassland share of total utilised agricultural area in the baseline year (BASE 2000) in the background (shade of brown), with the percentage change in grassland area for the scenario maintaining the current rules for the GAEC permanent pastures (BAU 2030 – blue bars), and for the scenario abandoning the current rules for the GAEC permanent pastures (C-Poor 2030 – red bars) 37
Figure 11 Potential SOC stock loss (in tonnes/ha) resulting from maintaining the current rules for the GAEC permanent pastures (BAU 2030) and abandoning the current rules for the GAEC permanent pastures (C-Poor 2030) 39
Figure 12 Arable share of agricultural area in the baseline year (BASE 2000) in the background (shade of brown), with the percentage change in arable area for the scenario all set-aside changing to arable (BAU 2030 – blue bars), for the scenario 10% of set-aside changing to forest (C-Medium 2030- green bars), and for the scenario 25% of set-aside changing to forest (C-Rich 2030 – purple bars) 42
Figure 13 SOC stock loss (in tonnes/ha) due to conversion from set-aside area 44
Figure 14 The 2000 forest cover map 49
Figure 15 The distinguishing characteristics of mor, moder and mull humus forms in forest soils 50
Figure 16 Organic carbon content (left) and organic carbon stock (right) as related to humus type in forest plots 51
Figure 17 Topsoil organic carbon content (%) for forest soils per NUTS2 region. 53
Figure 18 Stock gain in tonnes C/ha as a result of afforestation of arable land on the basis of the topsoil organic carbon map 54
Figure 19 Forest share of MS in the baseline year (BASE 2000) in the background (shade of brown), with the percentage change in forest area for the scenario all set-aside changing to arable (BAU 2030 – blue bars),and for the scenario adopting a faster decrease of UAA in favour of forests (C-Rich 2030 – purple bars) 55
Figure 20 SOC stock changes (in tonnes/ha) due to conversion from arable land to forest, weighted for the total area of land use change under BAU, C-Poor, C-medium and C-rich scenarios. 57
Figure 21 SOC stock changes (in tonnes/ha) due to conversion from arable land to forest, weighted for the entire forest area under C-poor, C-medium, C-rich scenarios. 58

Figure 22 Comparison of average humified organic carbon production (tonnes/ha) under cereal with straw incorporated into the soil (green) and straw harvested (yellow). 62
Figure 23 Comparison of average humified organic carbon production (tonnes/ha) under sugar beet with shoot & head incorporated into the soil (green) and shoot & head harvested (brown). 63
Figure 24 Comparison of average humified organic carbon production (tonnes/ha) under oilseed with straw incorporated into the soil (green) and straw harvested (brown). 63
Figure 25 Comparison of average humified organic carbon production (tonnes/ha) under grass with grass incorporated into the soil (green) and grass harvested (yellow). 64
Figure 26 Humified organic carbon production (tonnes/ha) under cereal with straw incorporated into the soil (top) and straw harvested (bottom) across Europe 65
Figure 27 Humified organic carbon production (tonnes/ha) under sugar beet with shoots & heads incorporated into the soil (top) and shoots & heads harvested (bottom) across Europe 66
Figure 28 Humified organic carbon production (tonnes/ha) under oilseed with straw incorporated into the soil (top) and straw harvested (bottom) across Europe 67
Figure 29 Humified organic carbon production (tonnes/ha) with grass ploughing (top) and grass harvesting (bottom) across Europe. 68
Figure 30 Humified organic carbon (tonnes/ha) under cereal production for different scenarios of residue management (0%, 10%, 30%, 50% and 100% of straw removed; 0% removed equals all residues incorporated into the soil) 70
Figure 31 Humified organic carbon (tonnes/ha) under sugar beet production for different scenarios of residue management (0%, 10%, 30%, 50% and 100% of shoots & heads removed; 0% removed equals all residues incorporated into the soil) 70
Figure 32 Humified organic carbon (tonnes/ha) under oilseed production for different scenarios of residue management (0%, 10%, 30%, 50% and 100% of straw removed; 0% removed equals all residues incorporated into the soil) 70
Figure 33 Distribution of humified organic carbon (HOC tonnes/ha) across EU-27 under cereal production with different levels of residue management 71
Figure 34 Distribution of humified organic carbon (tonnes/ha)across EU-27 under grass with harvest in 2000 (left) and 2030 (right) 72
Figure 35 Distribution of humified organic carbon (tonnes/ha)across EU-27 under grass ploughing in 2000 (left) and in 2030 (right) 72
Figure 36 Regional map of reported compost production from urban areas (2005) 75
Figure 37 Population map of Europe (2005) at NUTS2 level 76
Figure 38 Regional distribution of potential compost production in 2005 for EU-27 based on assumptions by Barth et al. (2008) 76
Figure 39 Humified organic carbon (in kg/ha UAA) from actual and potential Kitchen (K-) and Green (G-) compost in 2005 78
Figure 40 Regional distribution of cattle, sheep and pig livestock units (LU) per ha of UAA 79
Figure 41 Livestock manure applied to agricultural land (N kg/ha) 81
Figure 42 Distribution of humified organic carbon from livestock manure applied to agricultural areas (C tonnes/ha) 82
Figure 43 Regional map of potential compost production (tonnes/ha) in 2005 (left) and 2030 (right) across EU-27. 83

Figure 44 Evolution of potential kitchen compost until 2030 as compared to current actual compost as spread over the Utilised Agricultural Area per Member State 84
Figure 45 Evolution of potential green compost until 2030 as compared to current actual compost as spread over the Utilised Agricultural Area per Member State 84
Figure 46 Humified organic carbon (tonnes C/ha) from projected manure production applied to the UAA per NUTS 2 region (BAU 2030) 85
Figure 47 Humified organic carbon (tonnes C/ha) from 60% of projected manure production applied to the UAA per NUTS 2 region (C-Poor 2030) 85
Figure 48 Comparison of mean humified organic carbon (HOC in tonnes/ha) for different manure management options (Base 2000, BAU 2030, C-Low 2030 and C-Poor 2030) for MS 86
Figure 49 Comparison of mean humified organic carbon (HOC in tonnes/ha) for different kitchen compost management options (Actual 2005, Potential 2005, BAU 2030, C-Medium 2030 and C-Rich 2030) for MS 87
Figure 50 Comparison of mean humified organic carbon (HOC in tonnes/ha) for different green compost management options (Actual 2005, Potential 2005, BAU 2030, C-Medium 2030 and C-Rich 2030) for MS 87
Figure 51 Different available databases for linking forest surface area and biomass production 91
Figure 52 Growing Stock (in m³/ha) of broadleaved forests based on data from UNECE (2000) 92
Figure 53 Growing Stock (in m³/ha) of coniferous forests based on data from UNECE (2000) 93
Figure 54 Felling and removal (in m³/ha) in broadleaved forests based on data from UNECE (2000) 95
Figure 55 Felling and removal (in m³/ha) in coniferous forests based on data from UNECE (2000) 95
Figure 56 Structure of a tree and relation to biomass sources (A Foliage, B Branches, C Top, D Stem, E Trunk & roots, F Fine roots, G Small trees, H Litterfall) 96
Figure 57 Humified Organic Carbon from broadleaved forest (tonnes HOC/ha) per Member State 98
Figure 58 Distribution of Humified Organic Carbon from broadleaved forest (tonnes HOC/ha) 99
Figure 59 Humified Organic Carbon from coniferous forest (tonnes HOC/ha) per Member State 99
Figure 60 Distribution of Humified Organic Carbon from coniferous forest (tonnes HOC/ha) 100
Figure 61 Contribution of forest residue to Humified Organic Carbon (tonnes/ha) in coniferous forest across Europe according to different scenarios 101
Figure 62 Contribution of forest residue to Humified Organic Carbon (tonnes/ha) in broadleaved forest across Europe according to different scenarios 102
Figure 63 Distribution of humified organic carbon (tonnes/ha) across EU-27 under coniferous forest with different levels of forest residue management 103
Figure 64 Distribution of humified organic carbon (tonnes/ha) across EU-27 under broad leaved forest with different levels of forest residue management 104
Figure 65 Relative cover (%) of peat and peat-topped soils in the Soil Mapping Units (SMUs) of the European Soil Database 107
Figure 66 Relative contribution of peatland areas in the EU Member States to the total EU-27 peatland area 108
Figure 67 Mean topsoil organic carbon content (%) for inland wetland areas 109
Figure 68 Land use on peat soils in EU-27 for those countries having more than 1400 ha peatland, inset 110

Figure 69 Relative contribution of different land uses on peat soils to the peatland GHG emission budget for EU-27 and for 10 selected countries with considerable peatland areas. 115
Figure 70 Annual carbon emission (i.e. $CO_2$ and $CH_4$) as % of estimated peatland C stock with and without current rates of peat extraction (unit is %/yr) – BASE 2000 (estimated C stock is the total soil carbon stock in peatland areas). 117
Figure 71 Relative increase of carbon emissions due resulting from the BAU 2030 peatland conservation scenario (continued trend in historical conversion rates) 118
Figure 72 Relative increase of carbon emissions due resulting from the C-Medium 2030 peatland conservation scenario (50% reduction in historical conversion rates) 118
Figure 73 Relative decrease of carbon emissions due to different rates of peatland restoration. 119
Figure 74 SOC stock changes (in tonnes/ha) due to land use conversions in a C-poor scenario 122
Figure 75 SOC stock changes (in tonnes/ha) due to land use conversions in a C-rich scenario 122
Figure 76 SOC stock changes (in tonnes/ha) due to land use conversions to and from agricultural land, i.e. UAA to forest, arable-grassland conversions and set-aside. 123
Figure 77 Flux of Humidified Organic Carbon (tonnes per ha) into the soil from grass, oilseed, cereal and sugar beet residues based on BAU 2030, C-Low, C-Poor and C-Worst Case scenarios at the EU-27 level (In Base 2000 all residues remain on the field, whereas in C-worst all residues are removed from the field) 124
Figure 78 Humidified Organic Content in tonnes per ha from the application of kitchen compost, green compost and livestock manure based on Base 2000, C-Poor 2030, C-Low 2030, BAU 2030, C-Medium and C-Rich scenarios at the EU-27 level 125
Figure 79 Flux of Humidified Organic Content in tonnes per ha from broadleaved and conifer forest residues to the soil based on Base 2000, C-Rich 2030, BAU 2030, C-Low, C-Poor and C-Worst Case 2030 scenarios at the EU-27 level. 127
Figure 80 Carbon fluxes from the soil in tonnes per ha per year from peatlands during Base 2000, C-Poor 2030, C-Low 2030, BAU 2030, C -Rich 2030, and 50% and 100% restoration scenarios at the EU-27 level (Positive values are gains, negative values are losses). 129
Figure 81 Screenshot of the LUMOCAP PSS 145
Figure 82 The LUMOCAP system diagram. 146

## List of Tables


Table 1 C:N of common soil amendments and manure types 14
Table 2 SOC content in soils as related to soil function 17
Table 3 Global estimates of land area, net primary productivity (NPP), and carbon stocks in living plants and soil organic matter for ecosystems of the world (Amthor et al. 1998) 20
Table 4 Data sets used to calculate fluxes of organic matter to and from the soil 26
Table 5 Scenarios to assess the effect of selected environmental policy and resource management issues and options on soil organic matter levels in the EU to the 2030 horizon 30
Table 6 Grassland area (ha) in the Member States for the baseline year (BASE 2000), for the scenario maintaining the current rules for the GAEC permanent pastures (BAU 2030), and for the scenario abandoning the current rules for the GAEC permanent pastures (C-Poor 2030) 38
Table 7 Set-aside area (ha) per Member State (EU-15) in 2000 43
Table 8 Surface area in 1000 ha of forest and other wood land (FOWL), of which forest and protected forest, and percentages of broadleaved, coniferous and mixed forest (source data: UNECE, 2000; CLC; COSTE4). 48
Table 9 Change in forest area (based on UNFCCC reporting) for the period 1990 to 2007 52
Table 10 Forest areas and forest area changes per Member State for BASE 2000, BAU 2030 and C-Rich 2030 56
Table 11 Average crop parameters for organic matter production on agricultural land 61
Table 12 Reported and calculated potential compost production (tonnes/year) in 2005 across EU-27 77
Table 13 Mean nitrogen content coefficients, carbon to nitrogen ratios and humification coefficients different livestock categories for EU-27 based on OECD data (2004) 80
Table 14 Percentage change in livestock population between 2000 and 2030 based on LUMOCAP projections 83
Table 15 Turn-over rate, mass fractions, decomposition and humification rates for each compartment and for coniferous (CON) and broadleaved (BL) forest as used in REGSOM. 97
Table 16 Proportions of tree components to standing volume (adapted from Eggers, 2002; Marklund, 1988) 97
Table 17 Contribution of woody residue (stemwood, branches and stump) to humified organic carbon into the soil and decline due to residue harvesting. Figures in italic show ranges based on Member State values. Baseline is in 2000, all scenarios are in 2030. 102
Table 18 Estimates of European carbon storage in peatlands. Rough carbon storage estimates for the entire Russian and Canadian peatlands included for comparison 108
Table 19 Carbon loss (in million tonnes per year and in tonnes per ha per year) in peat soils under agricultural land use; surface areas are based on Byrne et al. (2004) 111
Table 20 Average emission factors (kg C or N.ha$^{-1}$.yr$^{-1}$) based on measured fluxes from European bogs and fens under different land uses 113
Table 21 Peatland carbon and GHG balance and relative contribution of the national peatland GHG budget to the total GHG emissions per Member State. 116
Table 22 Summary information concerning the selected case studies 133

## List of Boxes


| | | |
|---|---|---|
| Box 1 | Organic carbon stock in a soil layer | 5 |
| Box 2 | Important terminology related to soil organic matter | 9 |
| Box 3 | Nitrogen in organic matter | 12 |
| Box 4 | Effective organic matter | 14 |
| Box 5 | Carbon in peat soils | 16 |
| Box 6 | Key messages | 23 |
| Box 7 | Important biomass production definitions from forest inventories (as reported in UNFCCC, FAO, UNECE, 2000) | 94 |

# ABBREVIATIONS

| | |
|---|---|
| BAU | Business as usual |
| C | Carbon |
| $CO_2$ | Carbon Dioxide |
| CFI | Carbon Financial Instruments |
| CEC | Cation exchange capacity |
| CCX | Chicago Climate Exchange |
| CRU | Climate Research Unit |
| CAP | Common Agricultural Policy |
| CLC | Corine Land Cover |
| DG ENV | Directorate General Environment |
| DG JRC | Directorate General Joint Research Centre |
| DOM | Dissolved organic matter |
| EEA | European Environment Agency |
| EFISCEN | European Forest Information Scenario Database |
| EFSOS | European Forest Sector Outlook Study |
| EOC | Effective Organic Carbon |
| EU | European Union |
| FSS | Farm Structure Survey |
| FC | Field Capacity |
| FAO | Food and Agricultural Organisation |
| FAO-FRA | Food and Agricultural Organisation |
| FOWL | Forest and other woodland areas |
| FAWS | Forest areas available for wood supply |
| FSCC | Forest Soil Co-ordinating Centre |
| GAEC | Good Agricultural and Environmental Conditions |
| GHG | Green house gas |
| Ha | Hectare |
| HOM | Humus organic matter |
| IOM | Inert organic matter |
| Kg | Kilogrammes |
| MIP | Maximum input potential |
| Mha | Million hectares |
| Mt | Million tons |
| NVZ | Nitrates vulnerable zones |
| N | Nitrogen |
| NAI | Net annual increment |
| NUTS | Nomenclature of Units for Territorial Statistics |
| OM | Organic matter |
| OECD | Organisation for Economic Cooperation and Development |
| ppm | parts per million |
| POM | Particulate organic matter |
| PWP | Permanent wilting point |
| SOC | Soil organic carbon |
| SOM | Soil Organic Matter |
| S | Sulphur |
| T | Tons |
| UAA | Utilised agricultural area |
| UNECE | United Nations Economic Commission for Europe |
| VFG | Vegetable, fruit and garden |
| WHC | Water holding capacity |

# CHAPTER 1 INTRODUCTION

## 1.1 Aims and objectives of the project

The aims and objectives of the report "Soil organic matter management across the EU – best practices, constraints and trade-offs" are to assess the relative contributions of the different inputs and outputs of organic carbon and organic matter to and from the soil. From this assessment we evaluate the environmental consequences in view of improving the management of soil and biomass resources at the EU level.

## 1.2 Scope of the Report

**Chapter 2** provides a concise overview of the role of soil organic matter (SOM) in ecosystems and its relevance to climate change. It is based on a literature review of peer reviewed literature and reports.

**Chapter 3** outlines the approach to assess soil organic carbon stocks and soil organic matter fluxes for agriculture, forests and peatlands and explore selected environmental policy and resource management options using scenario analysis. Regional soil organic matter balances (REGSOM) are based on existing statistical information, simplified models and geographic overlay analysis using GIS. Scenarios compare selected environmental policy and resource management issues with options that are classed as falling under C-Rich, C-Medium, Business as Usual, C-Low and C-Poor. The LUMOCAP model is used to analyse the effect of selected policies on land use area – it is underpinned by the Hadley climate model with a 1% increase of greenhouse gases.

**Chapters 4, 5, 6, 7, 8, 9 and 10** provide the results related to the assessment of soil organic carbon stocks and fluxes for agriculture, forests and peatlands for the Baseline Period 2000-2005 and on the basis of investigating the impact of selected environmental policy and resource management options through the use of scenario analysis up to 2030.

**Chapter 11** provides a summary of the main results of the modelling and scenario work and the main conclusions related to the assessment of soil organic carbon stocks and changes under agriculture and forests, the soil organic carbon fluxes from agriculture and forests, and the carbon fluxes under peatlands.

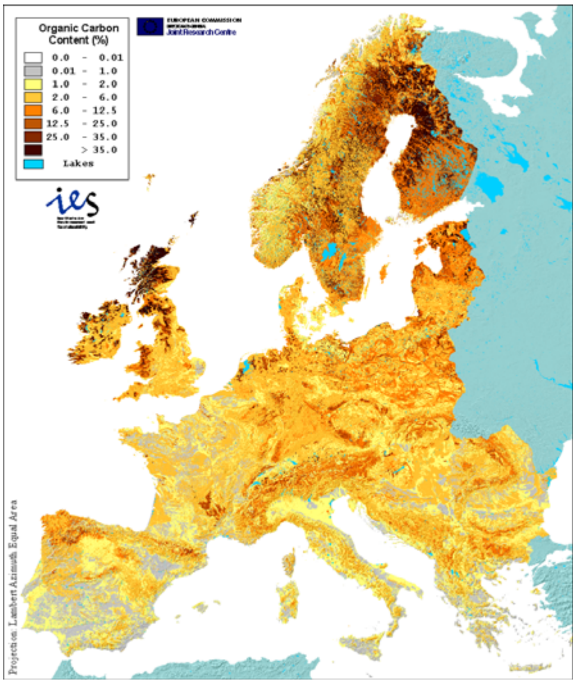
**Chapter 12** assesses the contribution of work done to identify best practices from eight case studies from around Europe (Annex II) to understand the results and implications for farmers and policy makers for the different issues related to soil organic matter management. The case studies address the following issues: crop rotation, reduced tillage practices, the use of crop residues, the use of different composts, the management of peatlands and afforestation.

**Chapter 13** uses the information provided in Chapters 2 to 12 to provide recommendations to improve policy and EU regulatory actions.

# CHAPTER 2 THE ROLE OF SOIL ORGANIC MATTER IN ECOSYSTEMS AND SOCIETY

## 2.1 Current state of soil organic matter across Europe

European soils store around 73 to 79 billion tonnes of carbon (Figure 1; Hiederer et al., 2004; Schils et al., 2008), which is more than 50 times the total $CO_2$-equivalent emissions of the 27 Member States of the European Union (EU-27) in 2009 (4.6 billion tonnes) (EEA, 2010). The smaller estimate (73 billion tonnes) is derived from the SOC map of the USDA Natural Resources Conservation Service (USDA, 2000) based on JRC data and the SOC stocks were estimated to a depth of 100 cm. The larger estimate (79 billion tonnes) is based on the pan-European Spatial Layer developed by the Joint Research Centre (JRC) of the European Commission (Jones et al., 2005). They applied a sophisticated pedo-transfer rule to the Soil Geographic Database of Eurasia which is the most detailed and harmonized spatial data set that currently exists for Europe. The SOC stocks were estimated in the top 30 cm according to the available data from the JRC pan-European Spatial Layer. According to Schils et al. (2008) both methodologies provide similar estimates for most European countries and were of the same order of magnitude as estimates from national or regional studies. In spite of the variation in estimates, both methods can be considered reliable. The variation in estimated SOC stocks is hardly surprising given the uncertainties in the data used (e.g. bulk density, volume of stones, depth of the soil layer).




Based on: soil survey data from the 1980s and climate data from 1960-1989, Jones et al. (2004)

Figure 1 Status in 1990 of organic carbon content in topsoils (0–30cm) in Europe

Worldwide Soil Organic Carbon (SOC) is one of the major pools of carbon. The SOC pool is about double the size of the atmospheric carbon pool and about 3 times the size of the biotic carbon pool. The SOC pool to a depth of 1m is estimated at 1,500 billiion tonnes (Jobbagy & Jackson, 2000; Batjes, 1996). The SOC pool to 1 meter depth ranges from 30 t/ha in arid climates to 800 t/ha in cold regions (Lal, 2004). Due to their very high carbon content organic soils are estimated to account for 20-30% of the global carbon stock (Moore, 2002; Turunen et al., 2002) while covering only 3% of the global surface (Strack, 2008). In Europe nearly half of the total soil organic carbon stock is located in Sweden, Finland and the United Kingdom as these countries have large areas of peatlands.

The global area of northern peatlands is estimated around 346 Mha (Gorham, 1991); over 85% of the peatlands occur in European Russia, Fenno-scandinavia and the British Isles (Byrne *et al*., 2004). The total peat carbon storage of Europe is estimated at 42 billion tonnes, accounting for 10-15% of the carbon stock in northern peatlands. The European peatlands are estimated to cover about 52 Mha (Joosten and Clarke, 2002) out of which about 31 Mha occurs in EU Member States (Schils et al., 2008). About 20% of the European soil organic carbon stock is located in peatlands (Schils *et al*., 2008).

Agricultural area in the EU-27 covers 166 Mha (38% of the total land area) and forest and other woodland covers 177 Mha (41%) in 2005 (Eurostat, 2010). There is a large spatial variability of soil under agriculture and forestry, so soil organic matter content is highly variable. In general terms the more sandy soils (coarse texture) retain lower amounts of soil organic matter, because organic matter is more quickly decomposed, due to greater soil pores and so higher decay rates. Soil organic matter monitoring programmess, long term experiments and modelling studies all indicate that changes in land use significantly affect soil organic matter levels. Soil organic matter losses occur when grasslands, forests and natural vegetation are converted to cropland. The reverse is true if croplands are converted to grasslands, forests and natural vegetation. **Land use changes can result in rapid carbon losses (i.e. instant), whereas gains accumulate more slowly (i.e. decadal)**.

Rates of carbon accumulation in EU soils are very difficult to estimate and the range of the estimated net yearly accumulation of carbon is from 1 to 100 million tonnes. The factors that influence soil organic matter accumulation include: climate, soil texture, hydrology, land use and vegetation. With the wide variety of soil types, land use and climatic conditions across Europe the soil organic carbon content of EU soils varies from less than 35 tonnes C/ha to more than 1250 tonnes C/ha[1] (Figure 1). According to the SoCo Project (EC, 2009), soil organic carbon content in agricultural soils varies from less than 20 to more than 400 tonnes C/ha. It is clear that there is a north south gradient of soil organic matter content in soils across Europe – with southern regions having considerably less soil organic matter compared to the north (Figure 1Figure 1). The meteorological conditions (i.e. the higher temperatures in combination with lower rainfall) in combination with land cover and land management differ considerably along the north south gradient.

Since 1850, soils have lost an estimated 40–90 billion tonnes[2] (Gt) carbon (Lal, 2009) globally through cultivation and disturbance with current rates of C loss due to land use change of about 1.6 ± 0.8 billion tones (Gt) C/y (Lal, 2009). Halting unfavourable land-

[1] Soil organic carbon percentages are converted to specific weight in tonnes C/ha assuming a bulk density of 1.2 and a depth of 30 cm. This enables comparison with agricultural soils.
[2] 1 billion tonnes = 1 gigatonne ($10^9$ tonne) = 1 Petagram ($10^{15}$ gramme)

use conversions and encouraging C-sequestration would be an effective mechanism to reduce soil C losses, but with a growing population and higher bio-energy demands, more land is likely to be required for settlement and for bio-energy production. Maximising the potential of existing agricultural land and applying best management practices to that land would slow the loss of, or in some cases restore, soil organic matter. Management practices that at the same time improve profitability are most likely to be adopted.

## 2.2 Soil organic matter dynamics

The term soil organic matter (SOM) has been used in different ways to describe the organic constituents of soil. SOM is a myriad of organic compounds formed from organic material and microbial decomposition products (Rice in Lal, 2002). The Soil Science Society of America (SSSA) (2008) defines SOM as the total organic fraction of the soil exclusive of undecayed plant and animal residues. Other definitions define SOM as all organic materials found in soils irrespective of their origin or state of decomposition (e.g. Baldock and Skjemstad, 1999). Surface litter is generally not included as part of soil organic matter, nor are living animals. In the Commission for a Soil Framework Directive (COM(2006) 232 final) – soil organic matter is explicitly defined as "the organic fraction of the soil, excluding undecayed plant and animal residues, their partial decomposition products, and the soil biomass".

Since SOM consists of C, H, O, N, P and S, it is difficult to directly measure the SOM content. Most analytical methods of organic matter generally measure only organic compounds or carbon, and estimate SOM through a conversion factor. They are therefore only an approximation of the level of once-living or decomposed matter. SOM is assumed to consist of between 50 to 58% carbon such that SOM is simply the multiplication of measured soil organic carbon (SOC) with a factor 1.724 to 2. Based on the organic carbon content and the soil specific weight, the organic carbon stock can be derived (Box 1).

> The organic carbon stock, expressed in t/ha, is the mass of organic carbon in the soil (layer) and is derived from the organic carbon content and the weight of the soil. The soil organic carbon content is the fraction of organic carbon in the soil expressed as weight percentage (weight organic carbon/weight soil). The weight of the soil (t/ha) is derived from the depth of the soil (cm) and the bulk density or specific weight ($g/cm^3$) of the soil. For a soil with different layers the carbon stock in the soil is the sum of the carbon stock in the soil layers.
>
> *Example* If the organic carbon content of a topsoil layer of 30 cm is 2% and the bulk density is 1.2, then the carbon stock is 72 ton carbon per ha.

Box 1 Organic carbon stock in a soil layer

Total SOC may not be a good indicator for assessing how well a particular soil function is likely to perform; mainly because the different pools, that make up the bulk SOC, vary considerably in their physical and chemical properties, and rates of turnover. The total soil organic matter can be divided into several pools (for definitions see Box 2), the most commonly considered pools for modelling purposes (Jenkinson and Coleman, 1994; Coleman et al., 1997) are:

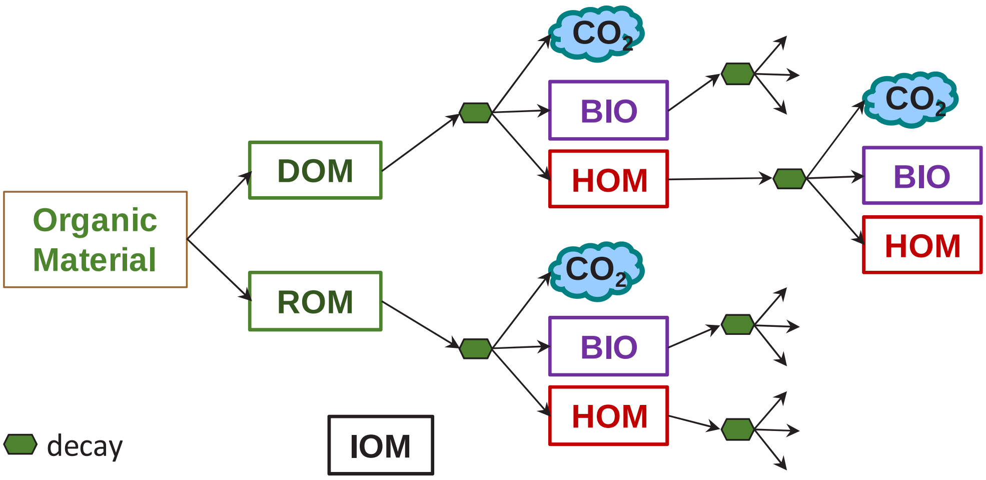
- microbial biomass of microorganisms;
- fresh organic material;
- active organic matter (i.e. partially decomposed residues);

- humified organic matter (i.e. the well-decomposed and highly stable organic material also known as humus); and,
- inert organic matter.

One of the most universal soil forming processes is the conversion of organic material to humus (humification) and its accumulation in topsoil. The process of humification can occur naturally in soil, or in the production of compost. Humus cannot be decomposed readily because of its intimate interactions with mineral soil phases. Some humic fractions are chemically too complex to be used by most organisms. The fraction of the organic material that is still present after one year of decomposition is the effective organic matter and determines the humification coefficient. Decomposition processes of organic matter [and to a lesser extent of humus] (mineralisation) result in the release of $CO_2$ and valuable plant nutrients such as N and P. The decomposition rate is influenced by climate variables such as temperature and rainfall, the soil water balance and the carbon-nitrogen-lignin composition of the organic material. Soil living organisms play a vital role in both humification and mineralisation processes. The processes, however, are difficult to control.

The above classification of SOM pools is useful to describe different organic matter processes. Modelling soil organic matter (SOM) turnover can be approached in different ways (Smith, 2002): 1) process-based multicompartment models; 2) models that consider each fresh addition of plant debris as a separate cohort which decays in a continuous way; and 3) models that account for C and N transfers through various trophic levels in a soil food web. One of the most well-known and widely used models in Europe is the Rothamsted carbon model (Coleman and Jenkinson, 1999; Coleman et al., 1999), which is of type 1) (Figure 2).

**The soil organic matter or carbon cycle is based on continually supplying carbon in the form of organic matter as a food source for microorganisms, the loss of some carbon as carbon dioxide, and the build up of stable carbon in the soil (a process called assimilation) that contributes to soil aggregation and formation.** Carbon assimilation is a dynamic process necessary for nutrient availability and cycling. Different sources of organic matter have different assimilation and decomposition characteristics, and result in different soil organic matter fractions. **If the rate of assimilation is less than the rate of decomposition, soil organic matter will decline and, conversely if the rate of assimilation is greater than the rate of decomposition, soil organic matter will increase**. Both the assimilation and decomposition processes occur concurrently, but are of a different order of magnitude. Like for land use changes, organic matter can be lost instantaneously (e.g. by fire), whereas its build up is spread over several decades.




ROM: Resistant OM; DOM: Decomposable OM; BIO: Microbial Biomass;
HOM: Humified OM; IOM: Inert OM

Adapted from: Coleman et al., 1997.

Figure 2 The breakdown of organic material such as natural plant residues, forest litter, compost, manure or bio-waste into soil organic matter pools

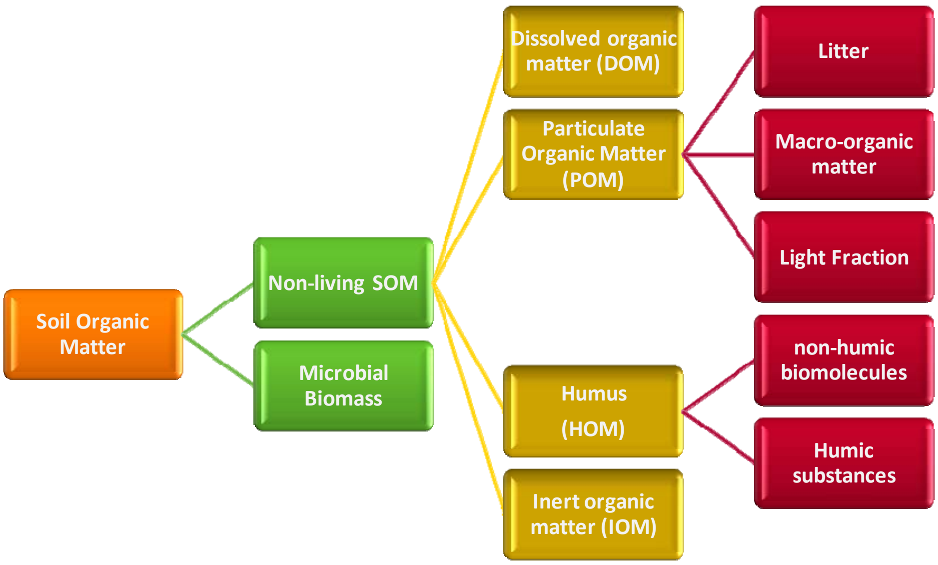
Humus consists of different humic substances that behave chemically similar to weak acids:

- Fulvic acids: the fraction of humus that is soluble in water under all pH conditions. Their colour is commonly light yellow to yellow-brown.
- Humic acids: the fraction of humus that is soluble in water, except for conditions more acid than pH 2. Common colours are dark brown to black.
- Hymatomelanic acids: the fraction of humus that is soluble in alcohol, after having been extracted with alkali and precipitated with acid.
- Humin: the fraction of humus that is not soluble in water at any pH and that can only be extracted with a strong base, such as sodium hydroxide (NaOH). Commonly black in colour.

The different components of soil organic matter decompose at different rates. The rate at which breakdown occurs depends on several factors: oxygenation, temperature, water content, particulate surface area, the recalcitrance of organic matter due to its chemical structure and the carbon to nitrogen ratio. The active pool has a high turnover rate of one to two years and consists of decomposable organic matter, accounting for one third of total soil organic matter. Active organic matter is the portion available to soil organisms. Bacteria tend to use simpler organic compounds, such as root exudates or fresh plant residues. Fungi, the only lignin composing organisms, tend to use more complex compounds, such as fibrous plant residues, wood and soil humus. The intermediate pool or stabilised organic matter, also called humus, turns over in two to five years. Humus accounts for another third of total soil organic matter. The recalcitrant or stable pool, called humus, is well-decomposed organic matter that is chemically or physically resistant to breakdown, taking more than 10 years to turnover. The latter pool consists of a small fraction of the total soil organic matter. In addition the soil is assumed to contain a small fraction of inert organic matter that is not available to microbial activity. Microbial biomass typically accounts for up to 5% and fresh residue for up to 10% of soil organic matter.

Fractionation methods are needed to identify different soil organic matter pools (Figure 3). These may, however, differ from the pools used for modelling purposes (Figure 2) and the challenge is in linking the different fractions to the different pools in the model. Different methods exist for separating SOM into discrete organic pools. A common method using particle size separation enables the separation of non-living soil organic matter into dissolved organic matter (DOM, <0.45µm diameter organic materials in solution), particulate organic matter (POM, >53µm diameter, recognizable structure), humus or humified organic matter (HOM, amorphous) and inert organic matter (IOM, carbonized material, char). POM is further separated into litter, macro-organic matter and the Light Fraction identified using density separation. Humus consists of humic substances (non identifiable chemical structure such as humic acids and humin) and non-humic biomolecules (identifiable chemical structures such as polysaccharids, proteins, waxes and lignin).

Different fractions have different rates of turnover which is related to their chemical composition (Krull et al., 2003) and can be related to the fraction's availability for microbial breakdown. The particulate organic matter (POM) fraction and light fraction (LF) are often considered the active pool and have a relatively fast turnover time of <10 years. The humified organic matter (HOM) fraction links to the HOM pool and is estimated to have a turnover time of 10s of years. The IOM pool may reside for 100s to 1000s of years. Organic material or residues high in lignin and with high carbon to nitrogen ratios are more resistant to decomposition than low lignin residues. The ratio between decomposable and resistant organic material determines the organic carbon turnover and is high for fresh plant residues and manure but medium for compost and low for woody biomass. The decomposition rate is not the only factor determining soil organic matter accumulation. For example, green manure with only 6% lignin and high decomposition rate can result in a higher C accumulation compared with straw (Paustian et al., 1992) due to higher crop productivity and returned inputs in response to the higher N content. Soil organic matter dynamics depend on a number of factors that require further study.




Modified from: Baldock and Skjemstad, 1999.

Figure 3 Composition of soil organic matter

*Active (fraction, pool) organic matter*: organic compounds that can easily be used as food by microorganisms.

*Cation exchange capacity (CEC):* is the capacity of a soil for ion exchange of cations between the soil and the soil solution, and is as a measure of fertility and nutrient retention capacity.

*Dissolved organic matter (DOM):* is organic materials in solution. The organic matter materials have a diameter less than 0.45µm.

*Effective organic carbon*: amount of carbon present in fresh organic matter that can directly be digested by soil micro-organisms estimated as organic carbon present one year after input into the soil.

*Fresh organic material*: the term refers to plant, animal, or other organic residues (e.g. manure) at moisture content that have recently been added to the soil and have only begun to show signs of decay.

*Humification*: conversion of fresh organic matter to a stable form of organic matter in the soil or conversion of organic matter to humus.

*Humification coefficient*: the fraction of effective organic carbon to total organic carbon in fresh organic matter.

*Humus or humified organic matter*: complex organic compounds that remain after many organisms have used and transformed the original material. Humus is not readily decomposed because it is either physically protected inside of aggregates or chemically too complex to be used by most organisms. Humus is the stable fraction of organic matter in the soil.

*Inert organic matter* represents a soil carbon pool that is of great age that is not subject to biological transformation and is therefore constant.

*Lignin*: a hard-to-degrade compound that is part of the fibers of older plants. Fungi can use the carbon ring structures in lignin as food.

*Mineral soil*: anorganic fraction of the soil, i.e. not containing carbon and therefore not from animal or plant origins.

*Mineralisation*: microbial decay of organic matter in the soil. Micro-organisms decompose organic matter into mineral components, $CO_2$ and $NH_4$.

*Organic matter*: fresh organic material such as plant residues, manure, compost, ... consisting of organic compounds predominantly made up of carbon, oxygen, hydrogen and nitrogen.

*Plant nutrient*: substance that can be taken up by plants to grow. Plant roots take up nutrients that are soluble in soil water.

*Particulate organic matter (POM) and Light fraction (LF) organic matter*: POM and LF are larger and lighter than other types of soil organic matter, POM can be separated from the soil by size (using a sieve), LF can be further separated by weight (using a centrifuge – density separation).

*Soil organic carbon*: carbon present in the organic fraction of the soil. Organic matter consists of 50-58% of organic carbon. Soils with a high pH also contain carbon in the form of $CaCO_3$ and $MgCO_3$. SOM is simply the multiplication of measured soil organic carbon (SOC) with a factor 1.724 to 2.

*Recalcitrant organic matter*: organic matter such as lignin-containing material that few soil organisms can decompose.

Box 2 Important terminology related to soil organic matter

## 2.3 Soil organic matter and its functions

Many soil properties have an impact on soil quality. Soil organic matter, however, influences several critical soil functions and is affected by land management practices. Because organic matter enhances water and nutrient holding capacity and improves soil structure, appropriate soil carbon management can enhance productivity and environmental quality, and can reduce the severity and costs of natural phenomena, such as droughts and floods. In addition, the practice of increasing soil organic matter levels may help in reducing atmospheric $CO_2$ levels that contribute to climate change. Decreases in soil organic matter contents, through cultivation or tillage intensification, are often related to the deterioration of soil structure. Effects include the loss of aggregate stability, increased crust formation, increased runoff and soil erosion, increased compaction, slower water infiltration and a slower exchange of water/gasses.

Organic matter impacts the physical, chemical and biological characteristics of the soil. Different organic matter pools affect different soil functions (Baldock and Skjemstad, 1999; Figure 3). Particulate organic carbon (POM) is most important in providing energy for biological processes. The humic fraction (HOM) is an important source of essential soil nutrients and is the principal pool in contributing to the soil's cation exchange capacity (CEC). Soil structure is maintained by both the HOM and POM fractions with POM playing a greater role in sandy soils as a means of physically binding particles together. For soils with a higher clay content, both HOM and POM are required to develop optimal structural support as both chemical and physical binding play critical roles.

The role of soil organic matter is intrinsically linked to biological functioning: organic matter is the main source of food and energy for soil organisms. A soil regularly supplied with different kinds of soil organic matter supports a varied population of soil organisms and maintains a complex food web. An incredible diversity of organisms make up the soil food web (Jeffery et.al., 2010). They range in size from the tiniest one-celled bacteria, algae, fungi, and protozoa, to the more complex nematodes and micro-arthropods, to the visible earthworms, insects, small vertebrates, and plants. Microbial biomass in temperate grasslands is estimated to be 1-2 t/ha (Nannipieri et al., 2003). The biological functions of SOM are primarily to provide metabolic energy that drives biological processes, to act as a supply of macro-and micro-nutrients and to ensure that both energy and nutrients are stored and released in a cycle that connects above- and belowground energy transformations. Importantly, biological processes in turn influence both soil chemical and soil structural properties as they greatly affect soil structure and soil redox reactions. Microorganisms play an important role in the transformation of organic matter and nutrients as 80-90% of the total soil metabolism is due to microbial processes (Brady, 1990). Soil organisms mediate chemical conversions in the soil. The actions of the organisms comprise mineralisation of organic material; nutrient cycling, nutrient mineralisation and nutrient sequestration; degradation of pollutants; control the populations of soil organisms including crop pests; structure formation of the soil; and, fixing $CO_2$. The relationship between soil biodiversity and productivity is often described as a roughly bell-shaped curve, suggesting that there is an optimum level of biodiversity (Loreau et al., 2001).

Soil thermal properties (i.e. the ability to warm up quickly in cold climates) are related to colour, and the inert carbon pool (IOM), which consists of highly aromatic structures such as charcoal, plays the most important role here. The presence of organic matter in the soil results in a darker soil colour and contributes to a higher energy absorption due to a reduced albedo effect as compared to that of a light coloured mineral soil. Dark-coloured soils also hold more water such that it does not necessarily result in a warmer temperature regime. Generally good soil conditions are associated with dark brown colours near the soil surface, which is associated with relatively high organic matter levels, good soil aggregation and high nutrient levels (Peverill et al. 1999).

Soil organic matter dynamics are intrinsically linked to changes in soil structure. When fresh organic matter is added to the soil, soil microbes release polysaccharides that promote the formation of large or macro-aggregates. As the organic matter decomposes over the longer term, different sizes of aggregates are formed that are resistant to physical disruption. Soil aggregates of different size and stability hold SOM of different nature and dynamics, and provide for the physical protection of SOM against further biodegradation (Oades and Waters, 1991). Stable macro-aggregates (>0.2 mm) are found to be richer in SOM than micro-aggregates, and this younger SOM is easier to decompose. A soil with a good physical structure has pores and channels of many sizes. Larger pores allow rainfall to infiltrate without running off and causing soil erosion. In addition, stable soil aggregates resist movement by wind or water because they are larger than primary particles of silt or clay. Soil organic matter helps form and maintain the air passages and channels, protecting the soil from compaction. Soils with increased organic matter have a desirable structure that tends to crumble and break apart easily and is more suitable for crop growth than hard, cloddy structures. A positive relationship exists between aggregate stability and SOM, but there is no effect on structural stability below a threshold value 2% SOC content (Loveland and Webb, 2003) Aggregate stability can be improved by reduced tillage, rotations of crops with pasture, crop rotations and organic amendments (Six et al., 1998).

Soil water holding capacity is controlled primarily by the soil texture, structure and the soil organic matter content. Organic matter can hold up to 20 times its own weight of water As the level of organic matter increases in a soil, the water holding capacity (WHC) also increases, due to the affinity of organic matter for water (Hudson, 1994). The effect of SOM on soil water retention tends to be greater in coarse textured compared to fine textured soils. Low initial organic matter results in decreased effects on water holding capacity compared with higher initial SOC contents, suggesting that a lower threshold value exists for the influence of organic matter on water holding capacity. Fine-textured soils show a greater increase in water holding capacity at field capacity (FC) than at permanent wilting point (PWP), whereas for coarse-textured soils, a larger increase in water holding capacity was observed at permanent wilting point (Rawls et al., 2003). Land and vegetation are better able to withstand the effects of flooding and drought when infiltration and water holding capacity increase. If an increase in SOC causes an increase in moisture content at both field capacity and permanent wilting point, the net result on plant available water (PAW) may not be greatly affected since plant available water is defined as the difference between moisture content at field capacity and permanent wilting point. Soils that hold generous amounts of water are less subject to leaching losses of nutrients or soil applied pesticides. This increases the efficiency of farm management practices and reduces diffuse emissions from agriculture to water bodies.

Soil organic matter contributes to soil fertility and nutrient cycling. Plants obtain essential nutrients from fresh organic residues as they decompose in soil. Soil organic matter has a net negative charge and nutrients such as calcium, magnesium, potassium and ammonium (i.e. cations) have a positive charge. The capacity of a soil to hold plant nutrients so that they are easily released or "exchanged" into the soil solution is measured by the cation exchange capacity (CEC) as the sum of exchangeable cations in the soil. A high CEC is regarded as favourable as it contributes to the capacity of soils to retain plant nutrient cations. There is a linear correlation between CEC and SOC above a threshold of 2%. The active pool of SOM is most closely associated with nutrient supply. The stable soil organic matter pool also improves soil fertility by holding plant nutrients and preventing them from leaching into the subsoil.

Humic and fulvic acids enhance plant growth directly through physiological and nutritional effects. Some SOM fractions function as natural plant hormones and are capable of improving seed germination, root initiation, uptake of plant nutrients and serve as sources of N, P and S. Organic matter releases nutrients in a plant available form through the process of decomposition. The majority of N in the soil is incorporated in SOM and microbial biomass. The bulk of the soil phosphorous is present in organic P, calcium-bound inorganic P, and iron- or aluminum-bound inorganic P; inorganic P is not available to plants. Most of the sulfur in soil is in the SOM. Due to the conversion of energy to heterotrophic organisms, mineralisation of complex organic molecules by primarily microbial processes is possible. Some soil nutrients are used in the synthesis of new biomass, some are immobilised and another portion is mineralised and released as plant-available forms into the soil mineral nutrient pool. In a non-fertilised soil, SOM may provide for 90% of plant available N, 80% of plant available P and 50% of plant available S. In order to maintain this nutrient cycling system, the rate of addition from crop residues and manure must at least equal the rate of decomposition. Excess of N may favour SOM mineralisation which in turn may cause losses of nutrients via leaching or conversion to gaseous forms (N, S) or are the result of immobilization (P). Mineralisation may add to the production of greenhouse gasses ($CO_2$, $SO_2$, $NO_x$).

Compared with simple organic molecules, humic substances are very complex and large, with high molecular weights. The characteristics of the stable part of organic matter, the humus, are very different from those of simple organic molecules. Humus is an important buffer, reducing fluctuations in soil acidity and nutrient availability. Soil acidity is determined by the amount of positively charged hydrogen ($H^+$) ions in the soil solution. Humus buffers the soil by taking up or releasing $H^+$ into the soil solution and stabilising the $H^+$ concentration of the soil solution. Humic acids have the ability to interact with metal ions, oxides, hydroxides, mineral and organic compounds, including toxic pollutants, to form water-soluble and water-insoluble complexes. Through the formation of these complexes, humic acids can dissolve, mobilize and transport metals and organics in soils and waters, or accumulate them in certain soil horizons. This influences nutrient availability, especially those nutrients present at micro-concentrations only. Accumulation of such complexes can contribute to a reduction of toxicity, e.g. aluminium in acid soils, or the capture of pollutants such as herbicides or pesticides.

Soil organic matter plays a key role in nitrogen release through mineralisation. Soil temperature, moisture, pore structure and the proportion of carbon to nitrogen present in organic matter (C:N ratio) are the major controlling factors. The application of organic matter with a high C:N ratio (> 30:1) such as wood and sawdust, may immobilise nitrogen as microbes consume nitrogen in the soil in order to decompose the organic matter. On the contrary, organic matter with a low C:N ratio (< 30:1) such as manure, may lead to excess nitrogen release from the organic matter into the soil. Low C:N organic matter therefore acts as an organic nitrogen fertiliser. Since the majority of crops are harvested before the weather conditions are unfavourable for mineralisation, care has to be taken in avoiding excess nitrogen and nitrate leaching. Catch crops that are cultivated after the main crops are a potential solution to this problem.

*Example*. If a soil contains 72 tonnes C/ha and 2% of the soil organic carbon decays yearly then this amounts to 1.44 tonnes C/ha. If we assume a plausible 10:1 ratio of C:N in the organic matter then this amounts to a release of 144 kg N/ha per year.

Box 3 Nitrogen in organic matter

## 2.4 Soil organic matter quantity and quality

### 2.4.1 Organic matter supplements to the soil

There are many possible sources of fresh organic matter that can be added to the soil for the creation of soil organic matter; examples are crop residues, forest litter, manure and compost. Some organic matter breaks down quickly and some takes longer to degrade. Compost is an example of organic matter that has already been degraded or stabilised. A biologically active soil needs to have a mixture of fresh, partially degraded and previously degraded organic matter. In a steady state, the amount of SOM stored in a soil reflects the balance between C produced (or added) in equilibrium with decomposition and leaching (or C lost).

Crops contribute roots and/or plant parts that remain in the soil as fresh crop residue. Depending on the climatic conditions and soil type, the amount of crop residue produced may vary from place to place and over time. In wet years and on heavy clays, large amounts of crop residue can be difficult to incorporate and results in cold, wet soils in the spring. Some crops contribute more organic matter than others: cover crops or green manure provide food for the soil food web and ultimately soil organic matter. Green manure consists of crops grown for ploughing in, thus increasing fertility through the incorporation of nutrients and organic matter into the soil. Leguminous green manures such as clover and vetch contain nitrogen-fixing symbiotic bacteria in root nodules that fix atmospheric nitrogen in a form that plants can use.

Animal manure has been a primary component added to the soil for thousands of years and contains large amount of organic matter. Common forms of animal manure include farmyard manure and farm slurry or liquid manure. Liquid manure is produced by more intensive livestock rearing systems where concrete or slats are used, instead of straw bedding. Farmyard manure also contains plant material (often straw) which has been used as bedding for animals and has absorbed the faeces and urine. The manure from each type of animal has different characteristics and therefore requires different application rates. Liquid manure, like slurry, is often injected directly into the soil to prevent runoff to surface waters and contributes to SOM. Mineralisation rates are lower for farmyard manure as compared to slurry.

In contrast to fresh plant residues or animal manure, composted organic materials decompose slowly when added to soil because they have already undergone a significant amount of decomposition during the composting process. Composting is the result of a complex feeding pattern where aerobic microbes feed on organic matter and break it down into a nutritious soil amendment. Kitchen composts are made from vegetables, fruit and garden waste, whereas green composts are made solely from prunings, branches, grass and leaf litter.

Effective organic matter (EOM) is defined as the organic matter that is still available one year after incorporation in the soil. For every type of organic matter, standard data are used for estimating the remaining percentage of organic matter (humified organic matter) after one year of incorporation in the soil. Cereals contribute between 1 and 1.21 tonnes C/ha, whereas sugar beet contributes 0.51 tonnes C/ha and potatoes 0.4 tonnes C/ha.

*Example*. If a soil contains 72 tonnes C/ha and 2% of the soil organic carbon decays yearly then this amounts to a loss of 1.44 tonnes C/ha. The amount lost cannot be replenished with residues from cereals, sugar beet or potatoes.

Box 4 Effective organic matter

Table 1 C:N of common soil amendments and manure types

| **Category** | **C:N ratio** | **C*** | **EOM** |
|---|---|---|---|
| Horses - farm yard manure | 20.8 | 1.04 | 0.52 |
| Cattle slurry | 1.9-7.5 | 0.38 | 0.15 |
| Dairy cattle – farm yard manure | 11.2 | 0.93 | 0.46 |
| Other Ruminants – farm yard manure | 10.5 | 1.22 | 0.61 |
| Pigs slurry | 3.8-5.4 | 0.25 | 0.10 |
| Pigs – farm yard manure | 10.2 | 0.31 | 0.15 |
| Poultry slurry | 4.5-8.2 | 0.49 | 0.20 |
| Kitchen Compost | 12.8 | 1.54 | 1.32 |
| Green compost | 16.6 | 1.16 | 1.10 |

* in ton per 10 tonnes fresh material

#### 2.4.2 Land management to maintain or increase soil organic matter

Environmental factors such as soil moisture, temperature and aeration that increase biological activity induce changes in organic matter degradation. Since soil texture can effect soil aeration, it also influences soil organic matter break down. While a slow decomposition is desirable, some land management practices can lead to the undesired rapid loss of organic matter. To effectively maintain or increase SOC, the rate of input must exceed the rate of loss from decomposition and leaching processes. In most agricultural cases, this is achieved by stubble retention, rotating crops with pasture, afforestation, or the addition of organic residues such as animal manure, litter or sewage sludge. Frequent cultivation and crop rotation can stimulate soil microorganisms by providing more oxygen, whereas monoculture will lead to reductions in organic matter. Application of excess nitrogen fertiliser, however, stimulates soil bacteria to degrade more organic matter. Incorporation of large amounts of nitrogen-rich fresh organic matter may equally over stimulate the soil food web and ultimately result in organic matter reduction. The optimum level of mineral and organic nitrogen fertiliser is specific to pedo-climatic regions, which are a combination of climatic zones and soil associations.

The carbon sequestration potential depends on many factors such as land management, soil type or climate. Long-term experiments show that increases in soil carbon are often greatest soon after a land-use or land-management change is implemented until a new equilibrium is established. Soil organic matter must ideally be maintained at a level necessary to support optimal soil structure (or tilth). Whilst clay soils accumulate carbon relatively quickly, sandy soils may accumulate only small amounts of carbon even after a century of high carbon inputs. Soils in colder climates, where decomposition is slower, accumulate organic matter more rapidly than soils in warmer climates.

The largest source of soil organic matter readily available is the residue contributed by current crops. Consequently, crop yield and type, method of handling residues and frequency of fallow are all important factors. The value of forage crops in rotations with cereals and oilseeds has long been recognised. Most modern arable crop rotations including cereals, however, are often not sufficient to help maintain, let alone increase soil organic matter. The planting of genetically similar or uniform crop varieties over large tracts of land, sometimes without rotation to other crops in space or time depletes soil organic matter. Systems without narrow crop rotations or without cover cropping equally deplete soil organic matter in agricultural soils, because soil microbial activity is retarded if there are no crop changes. Crop rotations involving perennial forages tend to stabilise soil organic matter at a higher level than crop rotations involving fallow. A production system that includes cover crops, legumes for nitrogen fixation, crop rotation with cereals or grain maize and temporary grass contributes substantially to the increase of organic matter in the soil. Conversion from arable land to grassland is still the most successful conversion for enhancing soil organic matter levels, but this might not suit an arable farming system – therefore including cover crops and legumes in the arable crop rotation can be adopted.

Reduced tillage systems involve the removal of one or more tillage operations to increase residue cover on the soil and to use standing stubble to encourage infiltration and soil moisture which permits the winter survival of winter wheat. Reduced tillage systems include direct seeding where maximum surface residue is maintained until seeding, at which time high disturbance seed openers are used for seedbed preparation, residue management and weed control. Ridge tillage is a type of reduced tillage where row crops (such as corn) are planted on pre-formed ridges. During the planting operation, crop residues are cleared from the row area and moved to the furrow between rows. Minimum tillage is a type of reduced tillage that employs a reduction in one or more tillage operations from conventional practices (such as no fall tillage) and uses low disturbance seed openers. Zero tillage (or no-till) is a type of cropping system in which crops are planted into previously undisturbed soil by opening a narrow slot of sufficient width and depth to obtain proper seedbed coverage. Regardless of the type of conservation tillage system, all will result in lower seedbed disturbance/fewer passes than in a conventional tillage system. Zero tillage and reduced tillage will result in a higher concentration of soil organic matter particularly in the top 10 cm of the soil but will not help increase organic matter in the deeper soil layers below the plough zone. In addition, reduced or zero tillage will help conserve the soil carbon reserve. Risks of increased weed infestation, however, may result in increased herbicide use.

The amount of crop residue produced varies with climatic conditions and soil type. In times of drought and on soils prone to erosion, maximizing the amount of crop residue produced is beneficial. Burning crop residue or forest litter, may destroy soil organic matter, remove nutrients and cause air pollution problems.

Conversion of arable land to grassland or woodland, and reversion of surplus farmlands to natural ecosystems are land use options that tremendously increase the organic matter input to the soil. The extensification of arable production by the introduction of perennial components (e.g. field margins with trees) and the permanent revegetation of arable land (e.g. permanent grassland) are well-known practices that have a positive effect on soil organic matter stocks.

An important land management practice to avoid soil organic matter losses is the conservation of soils rich in soil organic matter. In particular, the exploitation, drainage and cultivation of peatland leads to serious organic matter decline (Box 5). Peatland restoration through reinstalling a shallow water table helps conserve the organic matter stock and helps stop the release of enormous GHG emissions. When restoring peatland, the availability of fresh plant material is the major factor in methane production; old (recalcitrant) peat plays only a subordinate role. The annual mean water level is a surprisingly good indicator for methane emissions (Drösler, 2005), and care has to be taken in finding the optimal water depth with balance between the GHGs $CO_2$ and $NH_3$. The practice of peatland restoration, however, is often not a favourite option for the farming community for obvious reasons such as increased risks of flooding and other related land management problems.

Peat contains around 50% carbon on a dry weight basis, which at a moisture content of 55% is 23% carbon, and has a density of around 300 kg/m³. If all this carbon is released during mineralisation of the peat, this equates to 247 kg $CO_2$ /m³.

When peat is extracted it becomes exposed to aerobic conditions. Peat used as a growing medium or soil conditioner mineralises rapidly, releasing carbon as $CO_2$. Although a small fraction of the carbon from peat will be sequestered in stable humic compounds in the soil, 99.5% of the carbon would have decayed within 100 years. Carbon decay rates imply a half life of 12 years for carbon in peat.

Box 5 Carbon in peat soils

## 2.5 Optimal and Maximum Input Potential

There are many factors and processes that determine the direction and rate of change in SOM content when vegetation and soil management practices are changed. Factors important for increasing SOC storage include (Post and Kwon, 2000): (1) increasing the input rates of organic matter; (2) changing the decomposability of organic matter with inputs that increase the light fraction or particulate organic matter in particular; (3) placing organic matter deeper in the soil either directly by increasing belowground inputs or indirectly by enhancing surface mixing by soil organisms; and, (4) enhancing physical protection through either intra-aggregate or organo-mineral complexes. Conditions favouring these processes generally occur when soils are converted from cultivated use to permanent perennial vegetation.

The interactions among soil functions, the different requirements for optimal SOC levels for each function and the individual soil mineralogical characteristics precludes a generic number for optimal SOC levels. SOC requirements are likely to differ according to function and soil type. Increasing SOC is of greater importance in sandy compared with clayey soils to obtain a higher Cation Exchange Capacity (CEC). SOC is required in larger amounts in sandy soils because most clayey soils can provide a substantial proportion of CEC through charge derived from clay minerals. For biological processes, provision of nutrients and thermal properties, SOC is required irrespective of clay content. Since different fractions influence different soil functions, the addition of organic material adding to a certain fraction (e.g. fresh material or humus) may be necessary.

The total amount of SOC in the soil profile and the distribution with depth is greatly influenced by the typical soil forming factors (e.g. mineral composition, climate, topography, soil biota, management) and the changes made to the system (e.g.

deforestation, conversion from grassland to arable land, drainage, plantation). Perturbed systems will result in different conditions under which SOC enters and exits the system. The process of attaining a new equilibrium between input and output of C content could take more than 50 years (Baldock and Skjemstad, 1999). Any measurements of SOC have to take into account that the soil may still be in the process of re-estabilishing a new equilibrium. Various climatic and management combinations as well as soil types influence equilibrium SOC content. In addition the soil function envisaged plays an important role: the amount required to ensure an adequate nutrient supply and uptake differs from the amount required to ensure structural stability (Table 2). Lower thresholds for productivity, however, appear to be set at 1% (Kay and Angers, 1999) which is lower than thresholds set for enhancing soil physical properties.

Table 2 SOC content in soils as related to soil function

| Soil function | SOC value | References |
|---|---|---|
| Crop nutrient uptake & yield | > 2% no increase | Janzen (1987); Howard and Howard (1990) |
| Aggregate stability | < 2% unstable<br>2-2.5% stable<br>> 2.5% very stable<br>= 4.5% max stability | Greenland et al. (1975), Carter (1992) |
| Vegetation production (biomass) | < 1% non-productive<br>Clay = 4% => equil SOC = 1-1.5%<br>Clay = 21% then opt SOC = 2%<br>Clay = 38% => equil SOC = 3.5-4.5% | Körschens et al. (1998) |

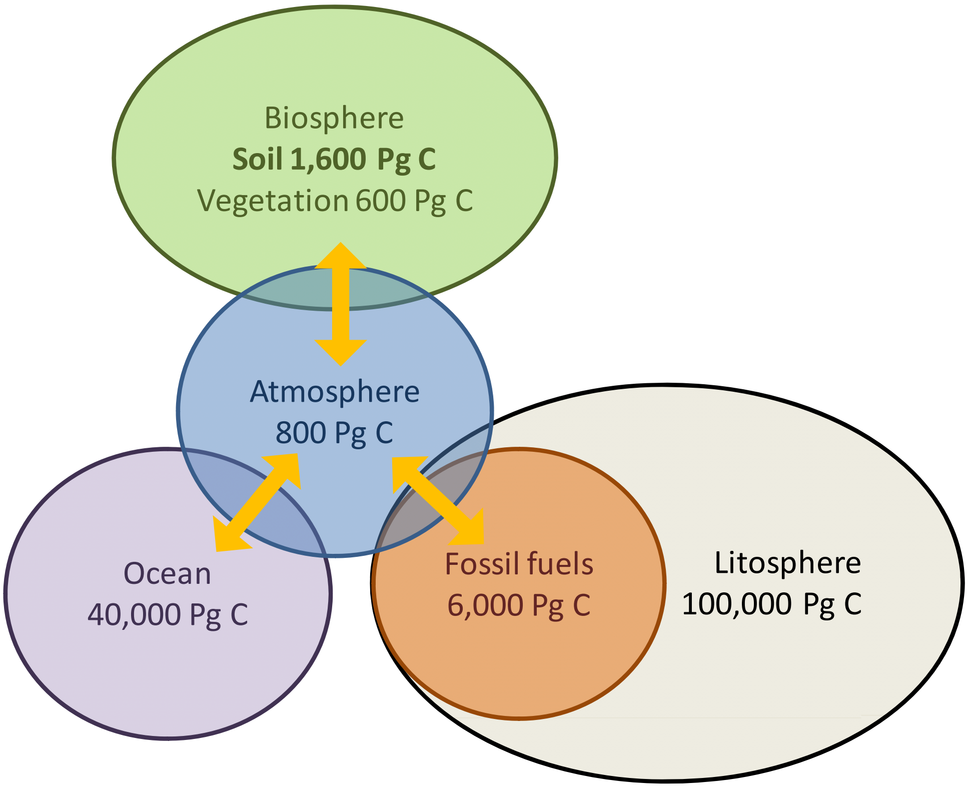
## 2.6 The organic matter cycle

Understanding the different streams of organic matter and carbon fluxes is important to soil organic matter dynamics. In this section we address: the global carbon cycle; carbon fluxes in ecosystems; and biomass fluxes in society.

### 2.6.1 The global carbon cycle

The carbon cycle describes the exchanges between the various carbon reservoirs (Figure 4): land biosphere, ocean, atmosphere, and fossil fuels. Carbon dioxide ($CO_2$) uptake and release at the Earth's surface is expressed as fluxes and the carbon dioxide exchange is described from an 'atmospheric point of view'. Since $CO_2$ concentrations in the atmosphere reflect the sum of all the $CO_2$ exchanges at the surface, they form the ultimate record of the combined human and natural influences on greenhouse gas levels.

The major organic carbon reserve on earth is the lithosphere: an estimated 100,000 billion tonnes carbon is stored in sedimentary rocks, not partaking in the global carbon cycle, and some 150 billion tonnes in marine sediments. Fossil fuels such as coal, oil, and natural gas account for 6,000 billion tonnes C. The oceans contain approximately 40,000 Bt C in the form of dissolved $CO_2$, bicarbonate, carbonate ions, calcium carbonate in shells and organic matter in saltwater vegetation. The land biosphere contains 600 billion tonnes C in terrestrial plants and 1600 billion tonnes C in soils. Soils contain twice as much carbon as the atmosphere (800 billion tonnes), and land cover change may release up to 50% of the soil carbon stocks. Much of this loss in soil organic carbon can be attributed to reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition (Post and Kwon, 2000).




Source: data based on IPPC (2007), where 1 Pg = $10^9$ tonnes = 1 billion tonnes

Figure 4 The global carbon cycle and carbon reservoirs

Since the Industrial Revolution, atmospheric levels of $CO_2$ have increased from about 275 ppm in the early 1700s to around 365 ppm nowadays. Future atmospheric levels of carbon dioxide could reach an amount between 450 to 600 ppm by the year 2100. The major $CO_2$ sources include fossil fuel combustion and the modification of natural plant cover found in grassland, woodland, and forested ecosystems. Emissions from fossil fuel combustion account for about 65% of the additional carbon dioxide currently found in the Earth's atmosphere. The other 35% is derived from deforestation and the conversion of natural ecosystems into agricultural systems. Natural ecosystems can store between 20 to 100 times more carbon dioxide than agricultural land use (Pidwirny, 2006).

Currently, an estimated 15 to 20% of atmospheric carbon dioxide emitted by human activities results from deforestation. The rate of growth in atmospheric $CO_2$ could be reduced substantially by decreasing the current rate at which forest is being converted to other land uses. Key findings from the Food and Agricultural Organisation's (FAO) Global Forest Resources Assessment have shown that deforestation slowed down between 2000 and 2010, with a global loss of about 13 million hectares of forest compared with 16 million during the previous decade. In Europe, forest areas continue to increase, although at a slower pace than previously.

Europe consists of a mosaic of different land uses. An accurate assessment of land-use data as well as information on agricultural and forest management strategies is crucial in estimating reliably the net carbon balance. Based on current land-based measurements, the forest sector in Europe is estimated to be a sink of about 380 Mt C per year (flux from the atmosphere to the biosphere) whereas agriculture is considered to be a source of 200 Mt C per year (flux from the biosphere to the atmosphere) (Smith et al., 2004). The estimate for agriculture refers to changes in the soil carbon reserve but does not include emissions of other GHGs ($CH_4$ and $N_2O$) from animal farming, pasture and cropland. The changes in soil organic carbon reserve in agricultural soils take into account land cover changes such as converting grassland to arable land, land management changes such as tillage operations and climate change with increased decay due to temperature increases.

### 2.6.2 Fluxes in ecosystems

Atmospheric carbon consists mostly of carbon dioxide ($CO_2$) and has two major sinks: terrestrial ecosystems and marine ecosystems. Both ecosystems assimilate carbon during photosynthesis and release carbon during respiration. Terrestrial ecosystems and climatic systems are closely coupled, particularly by carbon cycling between vegetation, soils, and the atmosphere. Global environmental changes (such as climate change, atmospheric $CO_2$ increases and anthropogenic land use changes) affect plant photosynthesis, respiration, and decomposition, thus leading to changes in plant $CO_2$ sequestration and the carbon stocks in vegetation and soils (Schimel et al. 1995; Lal, 1999). Local conditions could modify the frequency and severity of natural risks such as forest fires, strong winds etc., increasing the probability of carbon loss from these systems and hence increased $CO_2$ emissions (Heimann and Reichstein 2008).

Quantification of terrestrial $CO_2$ uptake or release has become one of the most important areas in global change science in the last decade, not least because the Kyoto Protocol has included some biological carbon sinks and sources in a legally binding framework for mitigating the anthropogenic greenhouse effect. Terrestrial ecosystems sequestered about 2.6 x$10^6$ t C per year of all atmospheric emissions during the period 2000-2007 representing net reductions of about 30% (according to 2006 levels) (GCP, 2008). Various terrestrial ecosystems such as forests, grasslands, agricultural systems and degraded land, have different potential of carbon storage. Net Primary Productivity (NPP) is defined as the net flux of carbon from the atmosphere into green plants per unit time. NPP refers to a rate process, i.e., the amount of vegetable matter produced (net primary production) per day, week, or year. Forest ecosystems contain more carbon per unit area in vegetation than any other ecosystem, accounting for 77% of C in all above ground biomass (share of forest to total Plant C in Table 3) and 18% of C in the soil profile (share of forest to total soil C in Table 3). Grass and shrub dominant ecosystems account for 20% of C in all ecosystems above ground biomass and for 44% of total C in the soil profile (Table 3). The highest carbon density is found in peatland ecosystems: 134 kg/m², which is 17 times higher than cultivated and permanent crop ecosystems (8 kg/m²).

The $CO_2$ uptake through photosynthesis[3] and plant growth, and loss of $CO_2$ through respiration[4] and decomposition of organic matter from terrestrial ecosystems are significant fluxes in Europe and therefore play an important part in carbon dynamics and vegetation modelling. C3 plants (95% of plants) produce three-carbon organic acids in carbon fixation. C4 plants (3%), including maize, sorghum, sugarcane, and millet, produce four-carbon organic acids and more sugar compared with C3 plants.

[3] Photosynthesis is the process that converts carbon dioxide into organic compounds, especially sugars, using the energy from sunlight (photons) thereby releasing oxygen to the atmosphere.
[4] Respiration is the metabolic process by which an organism obtains energy by reacting oxygen with glucose to give water, carbon dioxide and ATP (organism energy).

CAM plants or xerophytes, such as cacti and most succulents, fix the $CO_2$ at night, when their stomata are open. In hot and dry conditions, plants close the stomata in their leaves to prevent the loss of water which interferes with gas exchange. As $CO_2$ is used up, the balance of $CO_2$:$O_2$ inside the leaf alters in favour of $O_2$ which slows down photosynthesis, reduces carbon fixation in C3 plants and increases photorespiration. At low $CO_2$:$O_2$ ratios, C4 plants can achieve a relatively high carbon fixation and yield by suppressing photorespiration, but this is at the expense of the plant's energy.

Table 3 Global estimates of land area, net primary productivity (NPP), and carbon stocks in living plants and soil organic matter for ecosystems of the world (Amthor et al. 1998)

| **Ecosystem** | **Area ($10^{12}$ $m^2$)** | **NPP (gC / $m^2$ / y)** | **NPP (BtC / y)** | **Plant C (g / $m^2$ )** | **Plant C (Bt[5]C )** | **Soil C[6] (g / $m^2$ )** | **Soil (BtC)** | **Total (BtC)** |
|---|---|---|---|---|---|---|---|---|
| Tropical forest | 14.8 | 925 | 13.7 | 16500 | 244.2 | 8300 | 123 | 367 |
| Forest Temperate & plantation | 7.5 | 670 | 5 | 12270 | 92 | 12000 | 90 | 182 |
| Boreal forest | 9 | 355 | 3.2 | 2445 | 22 | 15000 | 135 | 157 |
| Temperate Wood land | 2 | 700 | 1.4 | 8000 | 16 | 12000 | 24 | 40 |
| Chaparral | 2.5 | 360 | 0.9 | 3200 | 8 | 12000 | 30 | 38 |
| Tropical Savannas | 22.5 | 790 | 17.8 | 2950 | 65.9 | 11700 | 263 | 329 |
| Temperate grasslands | 12.5 | 350 | 4.4 | 720 | 9 | 23600 | 295 | 304 |
| Tundra, arctic & alpine | 9.5 | 105 | 1 | 630 | 6 | 12750 | 121 | 127 |
| Desert & semi desert scrub | 21 | 67 | 1.4 | 330 | 6.9 | 8000 | 168 | 175 |
| Extreme desert | 9 | 11 | 0.1 | 35 | 0.3 | 2500 | 23 | 23 |
| Perpetual ice | 15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Lake and streams | 2 | 200 | 0.4 | 10 | 0 | 0 | 0 | 0 |
| Wetland | 2.8 | 1180 | 3.3 | 4300 | 12 | 72000 | 202 | 214 |
| Northern Peatland | 3.4 | 0 | 0 | 0 | 0 | 133800 | 455 | 455 |
| Cultivated & permanent crop | 14.8 | 423 | 6.3 | 200 | 3 | 7900 | 117 | 120 |
| Human areas | 2 | 100 | 0.2 | 500 | 1 | 5000 | 10 | 11 |
| **Total** | 151 | | 59 | | 486 | | 2056 | 2542 |

The type of photosynthesis plays an important role not only in sequestering carbon but also in the composition of litter and residues which enter the soil and contribute to soil organic matter. The composition of litter and residues, however, requires the calculation of carbon allocation between different plant organs (leaves, stems, roots), which can be estimated from total carbon uptake using fractional parameters. The newly allocated carbon to plant organs will be accumulated or enter the soil as vegetation litter. The equilibrium state is reached when carbon gain through the allocation equals carbon loss through litter production.

[5] Bt = Billion tonne = Gigatonne = 109 tonnes

[6] Soil C values are for the top 1 m of soil only, except for peatlands, in which case they account for the total depth of peat.

### 2.6.3 Fluxes in society

The use of biomass is increasingly being promoted as a renewable energy source, since the carbon in the displaced fossil fuel remains in the ground rather than being discharged to the atmosphere as $CO_2$. Using current technologies, the most efficient way to convert biomass to useful energy is to burn the biomass for heat or electricity generation, displacing the use of coal.

The biomass cycle reflects carbon fluxes in society. Biomass is a biological material from living, or recently living organisms that assimilates carbon. Harvested biomass, including residues, can be converted into different products such as building materials, paper, fuels, food, animal feed and other products such as plant-derived chemicals. Selected residues from the town (biowaste)may be combined with forestry and crop residues, animal wastes, and biomass crops to provide input for biomass processing. Bio-refineries make a range of products such as fuels, chemicals, new bio-based materials, and electric power. Animal feed is an important byproduct of some processes. Biomass processing facilities aim to use efficient methods to minimise waste streams and to recycle nutrients and organic materials to the land, thus closing the cycle. Throughout the cycle, $CO_2$ and other GHGs are released back into the atmosphere from the processing plants and from the urban and rural communities.

The net impact of biomass fluxes in society on soil organic matter dynamics will primarily depend on land use and land management, how the biomass products are used and recycled, and the time frame of the analysis. In the case of biomass production for bio-energy, the need for food production is involved in decisions on land use and will affect the amount of land available for reforestation or for bio-energy crops. Clearance of natural ecosystems in favour of biomass production may adversely affect the green house gas balance. Poor residue management may adversely affect the flux of fresh biomass to the soil resulting in an increasingly poor soil biodiversity and hence soil carbon dynamics. Cases in peatland, agricultural and forest ecosystems in this report clearly illustrate the potential negative impacts (Chapters 4,5, 6 and 7).

The productivity, or rate of growth becomes an important consideration in the production of biomass. While slow-growing trees can take a very long time before carbon is captured, fast-growing trees can assimilate carbon rapidly. Mature forests, however, achieve a balance between the carbon taken up in photosynthesis and the carbon released back to the atmosphere from respiration, oxidation of dead organic matter, and fires and pests. If production of bio-energy is the goal, a fast-growing herbaceous crop such as switch grass may be the best choice depending on the type of land and bio-energy technology. If the growing of bio-energy crops is optimised to add organic matter to the soil, there may even be some net sequestration or long-term fixation of carbon dioxide into soil organic matter. The cultivation of bio-energy systems, however, consumes energy to grow and to harvest which also needs to be taken into account.

## 2.7 Economic value of sequestering carbon in soils

As part of their global warming mitigation strategies, nations seek to minimise carbon emissions in the production of goods, energy, materials and services. In order to encourage low-carbon economies, a cost can be attributed to greenhouse gases through means such as emission trading scheme and/or carbon tax. Since almost every type of economic activity results in greenhouse gas emissions, different sectors of the economy have different ways to price carbon.

The Chicago Climate Exchange (CCX) is a voluntary but legally binding integrated trading system set up in 2003 to reduce emissions of all six greenhouse gases (GHGs) considered under the UNFCCC/Kyoto Protocol, with offset projects worldwide and based in the USA. CCX employs independent verification and has been trading GHG emission reductions since 2003. CCX Members that cannot reduce their own emissions can purchase credits from those who make extra emission cuts or from verified offset projects. CCX issues tradable Carbon Financial Instrument (CFI) contracts to owners or aggregators of eligible projects on the basis of sequestration, destruction or displacement of GHG emissions. Approved agricultural offset methodologies include (1) rangeland soil carbon management offsets, (2) agricultural soil carbon management, (3) forest carbon emission offsets and (4) agriculture methane emissions. Shortly after being set up in 2003 and European equivalent was started called the European Climate Exchange (ECX).

Agricultural soil management offsets[7] on eligible areas include the following conditions:

- Conservation Tillage: Minimum five year contractual commitment to continuous no-till or strip till (conservation tillage) on enrolled acres. Tillage practice must leave at least two-thirds of the soil surface undisturbed and at least two-thirds of the residue remaining on the field surface.
- Permanent Grassland: Minimum five year contractual commitment to maintain the conversion of cropland to grasslands. Projects initiated on or after January 1, 2003 in CCX eligible counties may qualify.
- CCX CFI contracts are issued for conservation tillage at a rate between 0.2 and 0.6 metric tons $CO_2$ per acre per year.

Sustainably Managed Rangeland Soil Carbon Sequestration Offsets[8] within designated land resource regions include the following conditions:

- Non-degraded rangeland managed to increase carbon sequestration through grazing land management that employs sustainable stocking rates, rotational grazing and seasonal use in eligible locations and a contingency for drought. Minimum 5 year contractual commitment.
- Offsets are issued at standard rates depending on project type and location.
- Rates vary from 0.12 to 0.32 metric tons of $CO_2$ per acre per year.

In October 2010 the CCX ceased trading reportedly because the US Government was not able to set up a federally enacted cap and trade scheme. The sister organisation in Europe the European Climate Exchange (ECX) is however still trading.

[7] http://www.chicagoclimatex.com/docs/offsets/CCX_Conservation_Tillage_and_Grassland_Conversion_Protocol_Final.pdf

[8] http://www.chicagoclimatex.com/docs/offsets/CCX_Sustainably_Managed_Rangeland_Soil_Carbon_Sequestration_Final.pdf

**Role of Soil Organic Matter in the ecosystem - Key messages**

1. Soil organic matter is declining across Europe due to changes in land use, land management and climate.
2. Different fractions of soil organic matter (SOM) support different but often multiple soil functions. They relate to carbon pools, each characterised by age and decomposition rates. If the stable organic matter fraction diminishes, then older carbon is released into the atmosphere.
3. Soil biodiversity plays a key role in both humification (i.e. the creation of stable humus) and mineralisation processes. Different soil biota have different access to soil organic matter. High C/N and high lignin content render OM difficult to decompose.
4. Soil organic matter levels can only be maintained if a steady state is reached. The amount of SOM stored in a soil reflects the balance between C produced (or added) in equilibrium with decomposition and leaching (or C lost). This requires careful management of input and control of loss.
5. Carbon fluxes at ecosystem and global scale, and biomass fluxes in society help understand the influence on soil carbon stocks and fluxes.

Box 6 Key messages

# CHAPTER 3 APPROACH TO ASSESS REGIONAL SOIL ORGANIC MATTER BALANCES AND SCENARIO ANALYSES

## 3.1 Introduction

Regional organic matter balances are established for soils under agriculture, forests and peatlands, on the basis of existing statistical information, models and geographic analysis in a GIS. The estimates of soil organic carbon stocks in the topsoil under agriculture and forests are presented as a % or C tonnes/ha. The estimates of soil organic carbon fluxes from agriculture and forests are presented as humified organic carbon (C tonnes/ha). Peatlands are treated differently, with carbon fluxes presented as a gas balance (tonnes $CO_2$/ha/year).

## 3.2 The regional organic matter balance

### 3.2.1 Concepts

The regional organic matter balance (REGSOM) across Europe consists of the following components:

- the current **input (I) of organic matter** from different sources such as crop residues, livestock manure, compost, and forest residues going to the soil and their relative potential to contribute to SOM formation;
- the current **output (O) of organic matter** lost from the soil because of different causes such as energy purposes, climate change, inadequate agricultural practices;
- the effect on inputs and outputs (E) of different land use changes such that the current net contribution of organic matter to SOM levels can be evaluated; and,
- the maximum amount of organic matter from the different sources that could be going back to the soil represents the **maximum input potential (MIP) of organic matter** to the soil, and its contribution to SOM levels.

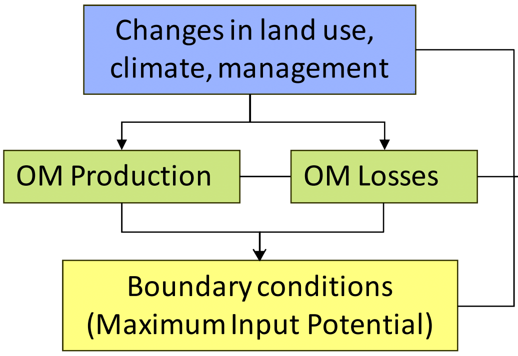



Figure 5 The organic matter balance

In the following sections the different components of the organic matter balance are described and quantified for agriculture, forestry, pristine peatlands and the urban fabric. Each of these land uses have an important influence on the organic matter production and loss per Member State. In addition the different land uses are chosen such that they can be related to major land cover classes as presented in the Corine Land Cover database. The production and loss of organic matter can be estimated using land use in combination with biomass production data at several embedded layers of geographical detail (EU-27, Member States and NUTS2 administrative regions), biogeochemical parameters from literature, climate and soil data (Table 4).

Estimates of the organic matter balance are coupled to the different land uses. Commercial activities (industry) and population densities are used to relate urban land uses to organic matter production (i.e. sewage sludge, biowaste, compost). Utilised agricultural area (UAA) consists of the three major agricultural land use classes: arable land, grassland and permanent cropland. The agriculture sector also comprises non-soil related activities such as animal husbandry producing animal manure and greenhouse agriculture producing green waste.

The amount of organic matter produced differs for each of the land uses and related activities. The databases that were consulted relate to the reporting obligations of environmental directives, to statistical (often voluntary) reporting and other relevant European datasets (Table 4).

Table 4 Data sets used to calculate fluxes of organic matter to and from the soil

| **Date type** | **Land use type** | | | |
|---|---|---|---|---|
| | **Agriculture** | **Forestry** | **Peatlands** | **Urban fabric** |
| ***Land use area*** | FSS, LUMOCAP | CLC, LUMOCAP | ESB, IPS | CLC |
| ***Biomass production*** | FSS, LUMOCAP | UNECE | IPS | ESTAT, ECN |
| ***Bio-geochemical parameters*** | OECD, Rothamstead C, Nitrates Directive | EFI(SCEN) | Jordan and Clarke (2001) | - |
| ***Climate data*** | CRU and Hadley Climate Model (HadCM2GGa1 scenario) | | | |
| ***Soil data*** | Organic Carbon content of topsoil (JRC) | | | |

### 3.2.2 Components

**Organic matter production**

The **inputs** of organic matter in the organic matter balance are equal to the **organic matter production**. Biomass production is the most important source of organic matter. Different sources of organic matter exist according to different land uses and related activities on the land, for example:

- Agriculture: farm related organic matter (crop residues, animal manure, green manure);
- Forestry: forest related organic matter (litter, in–situ organic matter production as a result from felling);
- Peatlands: very slow natural growth; and,
- Urban fabric: household waste (sewage sludge, biowaste, compost), commercial activities (sewage sludge, bio-waste) food and wood industry (green waste, compost).

**Organic matter losses**

The output of organic matter comprises the **organic matter losses or exits**, i.e. organic matter that is considered not available for spreading on land or in-situ losses due to soil processes. A certain percentage of organic matter produced will end up in landfills or will be used in **bio-energy** plants. Potential losses of organic matter are in the form of biomass for the production of bio-energy. In the Directive on Renewable Energy, biomass is defined as the biodegradable fraction of products, waste and residues from biological origin from agriculture (including vegetal and animal substances), forestry and related industries including fisheries and aquaculture, as well as the biodegradable fraction of industrial and municipal waste. Consequently biomass can be derived from different types of organic matter: energy plants (oilseeds, plants containing sugar) and forestry, agricultural crop residues or urban waste including wood and household waste. Biomass can be solid (plants, wood, straw and other crop residues), gaseous (from organic waste such as manure and slurry, landfill waste) or liquid (derived from crops such as wheat, rapeseed, soy or from lignocellulosic material). When employed for bio-energy purposes, all of these forms of organic matter are no longer available for application on the land.

Another major source of potential losses are **in-situ soil losses**. There are two groups of factors that influence soil organic matter content in the soil: natural factors (climate, soil parent material, land cover and/or vegetation and topography), and human-induced factors (land use, management and degradation). The most homogeneous and comprehensive data on the organic carbon/matter content of European soils remain those that can be extracted and/or derived from the European Soil Database in combination with associated databases on land cover, climate and topography. The Soil Portal makes available the maps of organic carbon content (%) in the surface horizon of soils in Europe. This information provides the basis for evaluating the current state of organic matter across the European soils under different land use/cover types. For all different land cover types the topsoil organic carbon content indicates a clear gradient with temperature increasing and moisture regime decreasing from northern to southern Europe. This gradient is taken into account when calculating organic matter turnover.

**Possible effects on production and losses**

The **effects** on the soil organic matter balance are expected from changes in climate, management and land use/cover. The effect of different land use changes on production and losses of organic matter are quantified in order to evaluate the current and future net contribution of organic carbon and organic matter to SOM levels across the EU for different land use and land management scenarios. The effects on SOM production and on SOM losses relate to changes in climate, land use and land management.

Soil has a complex relationship with our warming world. Soil helps take carbon dioxide out of the air and as such it absorbs millions of tonnes each year, but with **climate change** and **the tillage of soils** micro-organisms grow faster, consume more soil organic matter and release carbon dioxide at least under optimum moisture conditions (Smith et al., 2000). The net result is a relative decline in soil organic carbon.

With a growing population and higher bio-energy demands, more land is likely to be required for settlement, for commercial activity and for bio-energy production. **Conversions** between different land uses and from terrestrial ecosystems to urban and commercial activity will alter both the production and losses of organic matter, and have an indirect impact on potential SOM levels. Conversions between different terrestrial ecosystems have a direct impact on SOM levels. Net SOM losses are reported for several land use conversions, e.g. from grassland to arable land, from wetlands to drained agricultural land, and from crop rotations to monoculture.

Maximising the productivity of existing agricultural land and applying best **management** practices to that land slows the loss of, or in some cases restores SOM. Changes in land use/cover such as ploughing up grassland, drainage of peatland and removal of crops or crop residues (e.g. straw for heating purposes) are known to cause major changes in the soil organic carbon stock. Land management effects are linked to practices such as ploughing (depth, frequency), tillage, green manure, harvesting practices and the use of soil amendments & fertiliser (sludge, animal manure and compost application). Land management practices that at the same time improve land profitability are most likely to be adopted.

**Maximum Input Potential**

The **Maximum Input Potential** equals the maximum amount of organic matter that could enter the soil and depends on the different organic matter sources that are available in different regions across Europe. The maximum input potential of organic matter depends on boundary conditions set by environmental factors, soil management practices and legislation (e.g. Nitrates Directive). The environmental factors depend on the soil forming factors, climate, mineral composition (clay content), topography and soil biota. Soil management practices include for example tillage operations, crop rotations, cover crops and application of soil amendments. Very high levels of organic matter may hamper soil management practices such as soil tillage, ridging and irrigation as well as the general workability of the soil. The boundary conditions set by legislation relate to the Nitrates Directive, whereby Member States identify Nitrate Vulnerable Zones, where limits are imposed on the manure application rates. Some Member States have requested derogations to increase limits in certain areas, which are accounted for in the boundary conditions.

In the following chapter the methodology is described for each of the identified major contributing land related activities: agriculture, forestry, peatlands and urban fabric. The production of, loss of, and effect on organic matter can be estimated using land use in combination with statistical data and estimates of crop yields, wood, peat and the use of compost and manure. However, not all of the SOM balance components may be relevant to each land activity.

## 3.3 Scenario analysis to assess the effect of selected environmental policy and resource management issues on soil organic matter levels

A scenario approach is adopted in this study to explore the potential effects of selected environmental policy and resource management issues on land use and soil organic matter levels. For each environmental policy and resource management type we vary one or more parameters so as to define a set of scenarios. We use the regional organic matter balance model (REGSOM) to estimate regional carbon stocks and fluxes, and a dynamic land use change model for CAP impact assessment on the rural landscape (LUMOCAP) to analyse the effect of selected policies on land use area. We make the assumption that the average soil organic carbon stock of the surface horizon reflects the equilibrium state; therefore, the differences between SOC stocks under different land uses reflect the change from one equilibrium state to another. Soil organic carbon fluxes on the other hand are like snapshots in time of the impact of resource management on the soil. **Carbon fluxes are therefore snapshots of carbon input and cannot be directly compared or added on to carbon stocks**. For all the scenarios the baseline period is 2000 – 2005, and the end year is 2030. The Hadley Climate model output with 1% compound increase of GHG is used in the LUMOCAP model. The scenarios allow for plausible quantified projections **but are by no means intended to predict the future**: their purpose is to illustrate "what-would-happen-if" type of situations.

The scenarios are summarised in Table 5 below. The starting point is the Business as Usual (BAU) scenario or central column. Scenarios are compared to the BAU in terms of environmental policy and resource management options that aim to maintain, increase (C-Rich and C-Medium), or decrease SOM (C-Low and C-Poor). For the agriculture and forest land use change scenarios, the difference between the topsoil soil organic stock of one land use compared to another land use is based on spatial analysis of the organic carbon content in topsoils in Europe database (Jones et. al. 2004), hosted by the JRC. The assumption is that the average SOC stock of the surface horizon under different land uses for a given NUTS region reflects the equilibrium state. The parameter(s) that are modified in each scenario in comparison to the BAU are indicated in bold.

Table 5 Scenarios to assess the effect of selected environmental policy and resource management issues and options on soil organic matter levels in the EU to the 2030 horizon

| ***Environmental policy / resource management issue*** | **C-Rich** | **C-Medium** | **BAU** | **C-Low** | **C-Poor** |
|---|---|---|---|---|---|
| ***Agriculture and forests - land use changes*** | | | | | |
| *Maintenance of Grassland (Chapter 4)* | Grassland maintained as per current rules | Grassland maintained as per current rules | Grassland maintained as per current rules | Grassland maintained as per current rules | **Grassland restric-tions abolished** |
| *Use of Set-aside (for EU-15 only) (Chapter 5)* | **25% former set aside to afforestation** | **10% former set aside to afforestation** | Former set aside to arable | Former set aside to arable | Former set aside to arable |
| *Change from Utilised Agricultural Area (UAA) to forest (Chapter 6)* | **Faster decrease of the UAA in favour of forests** | Current change of UAA in favour of forests | Current change of UAA in favour of forests | Current change of UAA in favour of forests | Current change of UAA in favour of forests |
| ***Agriculture – resource management issues*** | | | | | |
| *Use of crop residues and straw (Chapter 7)* | 10% crop residues and straw for bio-energy | 10% crop residues and straw for bio-energy | 10% crop residues and straw for bio-energy | **30% crop residues and straw for bio-energy** | **50% crop residues and straw for bio-energy** |
| *Use of manure and compost (Chapter 8)* | Current manure and **50% more compost** available for application | Current manure and **25% more compost** available for application | Current manure and compost available for application | **20% manure used for bio-energy** | **40% manure used for bio-energy** |
| ***Forests – resource management issues*** | | | | | |
| *Use of forest residues (Chapter 9)* | **No forest residues removed for bio-energy** | 10% forest residues removed for bio-energy | 10% forest residues removed for bio-energy | **20% forest residues removed for bio-energy** | **25% forest residues and 10% area stumps removed for bio-energy** |
| ***Peatlands – conservation*** | | | | | |
| *Conservation of Peatlands (Chapter 10)* | **No further drainage of peatlands allowed** | **50% reduction of historical rates (1980-2000) for peat drainage** | Continuation of historical rates (1980-2000) of peatland drainage | Continuation of historical rates (1980-2000) of peatland drainage | Continuation of historical rates (1980-2000) of peatland drainage |

# CHAPTER 4 MAINTENANCE OF GRASSLAND

## 4.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Agriculture and forests - land use changes*** | | | | | |
| *Maintenance of Grassland* | Grassland maintained as per current rules | Grassland maintained as per current rules | Grassland maintained as per current rules | Grassland maintained as per current rules | **Grassland restric-tions abolished** |

The maintenance of grassland scenario examines the effect of grassland area changes on levels of soil organic carbon stock. The maintenance of grassland areas is related to the Good Agricultural and Environmental Condition (GAEC) standards for permanent pastures that are in place for farmers to adhere to if they want to receive benefits from the Single Farm Payment Scheme. The permanent pasture GAEC is a Compulsory Standard, described in Annex III of Council Regulation (EC) No. 73/2009. Each Member State has established a reference ratio, and each year an annual ratio is determined by dividing the permanent pasture area of the year by the utilised agricultural area declared. The GAEC defines "permanent pasture" as land used to grow grasses or other herbaceous forage that has not been included in crop rotation of the holding for five years or longer. Each Member State monitors the maintenance of the annual ratio using the Land Parcel Information System / Integrated Administration and Control System (LPIS/IACS)[9]. The Commission guidelines indicate that an annual ratio decrease of up to 5% from the reference ratio[10] is the threshold whereby farmers applying for aid under any direct payment scheme can convert land under permanent pasture without prior authorisation. If the annual ratio decreases by 10% from the reference ratio then the Member State will oblige farmers to re-convert back to permanent pastures.

## 4.2 Scenario Approach and Method

The LPIS/IACS databases control all cross compliance information (including the permanent pastures GAEC) and are managed by Member States of the EU. These databases provide an accurate record of permanent pasture areas at the farm level – but the information is confidential. Therefore with the permission of DG AGRI the following information from reported LPIS/IACS data at MS level was used to constrain

[9] LPIS/IACS is managing the spending organised in the European Agricultural Guarantee Fund.

[10] The base years reference ratio are 2003 for EU-15, 2004 for EU-10 and 2007 for EU-2.

LUMOCAP land use change simulations for: (1) overall permanent pasture areas, (2) utilised agricultural areas (UAA), and (3) allowed minimum permanent pasture to UAA ratios. The definition of permanent pastures is very precisely defined in the LPIS/IACS databases, but LUMOCAP uses grassland areas based on a combination of CORINE Land Cover and EUROSTAT data. This means, for example, that there is not a distinction between permanent and temporary grassland areas by LUMOCAP. Therefore there will be differences between CORINE Land Cover, EUROSTAT data and LPIS/IACS data with respect to both grassland and utilised agricultural areas. The surface areas for grassland and utilised agricultural area are larger on the basis of LUMOCAP (i.e. CLC and Eurostat) than on LPIS/IACS data.

We adopt two contrasting options for the maintenance of grassland scenario (Table 5):

1. Grassland maintained using current rules (BAU)
2. Grassland restrictions abolished (C-poor)

For "Grassland maintained using current rules" it is assumed that the ratio of permanent pasture over UAA does not decline by more than 5%, because this is the threshold percentage decline that farmers can allow before requesting authorisation. For "Grasslands restrictions abolished" it is assumed that no limit is set to permanent pasture decline.

The following steps are taken in the analysis:

1. The soil organic content in the surface horizon of both grasslands and arable land is determined at regional level (NUTS 2 administrative unit) by overlaying grassland areas from Corine Land Cover (CLC) with the soil organic carbon map from the Joint Research Centre (Jones et al., 2004);
2. The carbon content, expressed as weight percentage in the SOC map, is converted to SOC stock in tonnes per hectare by assuming a surface horizon thickness of 20 cm11 and using a pedotransfer function for deriving the bulk density. The average SOC stock of the surface horizon is assumed to reflect the equilibrium state - the differences between SOC stocks under different land uses reflect the change from one equilibrium state to another;
3. The differences in grassland areas and the grassland share of agriculture by 2030 are assessed under the two scenario options, i.e. maintaining the current rules for the GAEC permanent pastures (BAU 2030) and abandoning the current rules for the GAEC permanent pastures (C-Poor 2030). The grassland area changes are assumed to be conversions to and from arable land; and,
4. The area changes between arable and grassland are subsequently linked to SOC stock changes. Spatial analysis is used to combine land use changes from the scenario analysis and the top soil organic carbon map from the Joint Research Centre (Jones et al., 2004) to ascertain the SOC stock change due to the conversion of grasslands to arable land at regional level (NUTS 2 administrative unit).

[11] We adopt a 20 cm depth, because the surface horizon in Figure 1 ranges from 0 to 30 cm, and so using 30 cm as a depth could lead to overestimates of SOC stock.

## 4.3 Soil organic carbon stock under agriculture

### 4.3.1 Surface area of agricultural land

According to Eurostat farm statistics the utilised agricultural area (UAA) in the EU-27 covered 172.2 Mha in 2006 or 40% of the total land area. The largest share of UAA to total land area is in Denmark (62%), Ireland (60%), Romania (58%) and France (50%); the smallest shares are in Finland and Sweden (7%). In absolute area, France has the largest surface area of UAA (27.5 Mha; 16% of total UAA), followed by Spain (14%) and Germany (10%).

The largest share of utilised agricultural area is arable crops (59% for EU-27; Figure 6). For Ireland and Bulgaria, the share of permanent grassland is larger than the arable area (75% and 60%, respectively). The share of arable area is an important indicator for impacts of farming on soil organic matter dynamics for it is known that arable areas can deplete soil carbon stocks rapidly (Stoate et. al., 2001).

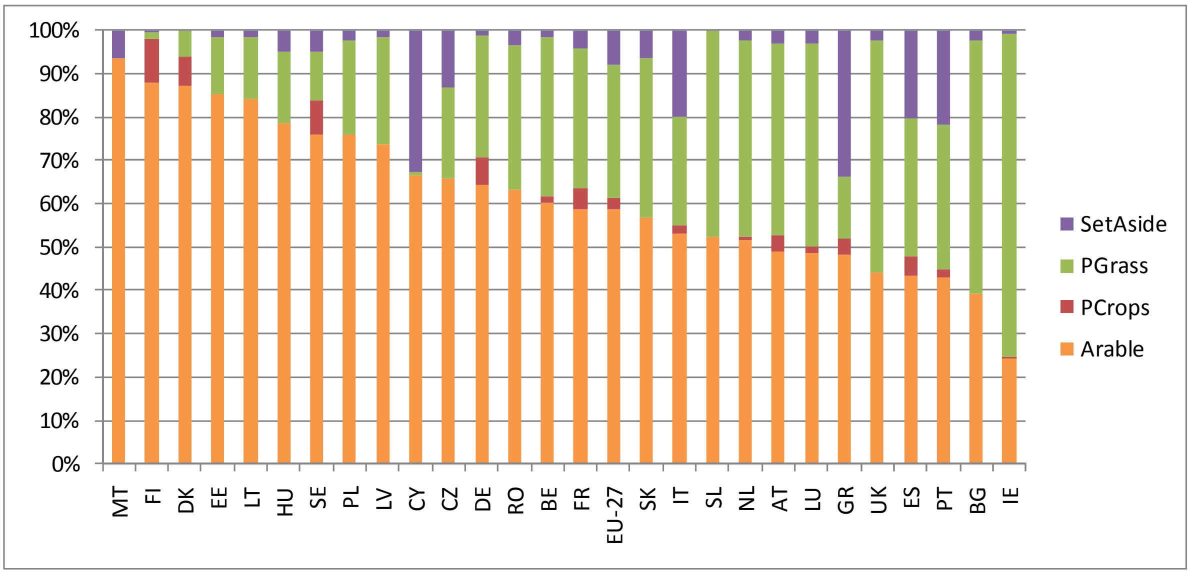



Figure 6 Share of arable land, permanent grassland, permanent crops and set aside to utilised agricultural area ranked by share of arable land according to 2006 Eurostat farm statistics, where PGrass = permanent grassland, PCrops= Permanent crops.

### 4.3.2 Status of soil organic matter under agriculture

Crop production generally results in a decline in soil organic matter levels and a accompanying decline in soil fertility (Smith et al., 2000; Easter et al., 2007). On reasonably fertile soils, with reliable water supply, yields have been maintained at very high levels in long-term arable agricultural systems, despite the decline in soil organic matter, by applying substantial amounts of organic and mineral fertilisers and other soil organic amendments. Changes in land use and land cover, with associated clearing and/or tillage operations, are likely to result in changes in organic residue input and turnover rate. The input and turnover rates affect the total soil organic carbon, which change in time to a new equilibrium value.

For all different land cover types the topsoil organic carbon content indicates a clear gradient with temperature increasing and moisture regime decreasing from northern to southern Europe. On average, topsoil organic carbon contents are highest under wetlands, followed by grassland, and lowest under arable land. Many grassland areas are located on land with waterlogged conditions or on peatland such that topsoil organic matter content is high. Under arable land soil organic matter is often reduced. The influence of climate, land cover and soil type is reflected in the topsoil organic carbon content.

## 4.4 Results

### 4.4.1 Distribution of soil organic carbon content in the surface horizon of grasslands

The soil organic carbon content in the surface horizon of grasslands is much higher in Northern Europe (>10%), because of the prevalence of grasslands on peat soils. In Central Europe the soil organic carbon content is mainly between 4-8%, and in Southern Europe there are many regions where the soil organic carbon content is 1-4% (Figure 7).

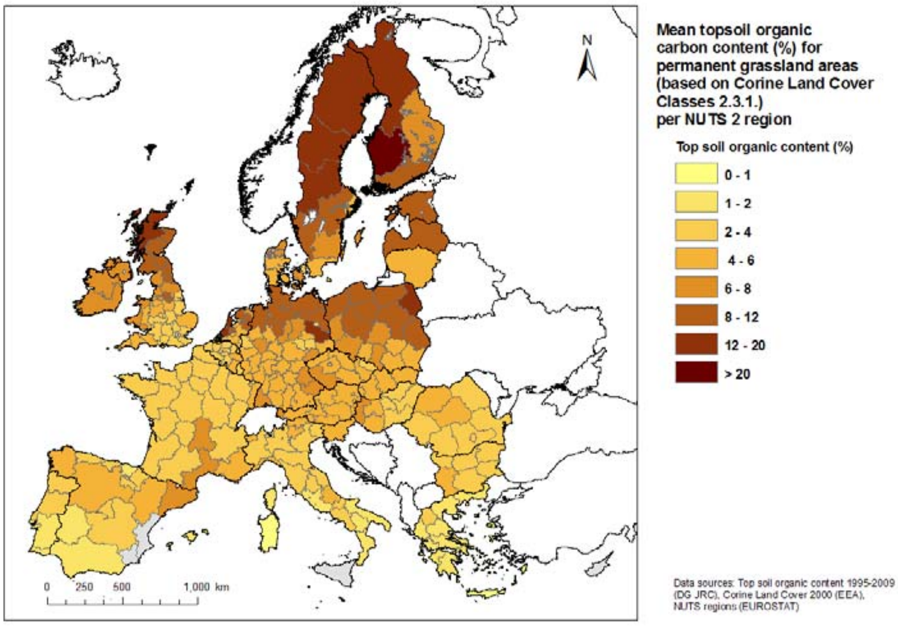



Figure 7 Mean topsoil organic carbon content (%) for permanent grasslands in the EU

### 4.4.2 Distribution of soil organic carbon content in the surface horizon of arable land

The soil organic carbon content (expressed as percentage of total dry matter in the surface horizon of arable land) is much higher in Scandinavia (>8%), because of the prevalence of arable land on peat soils. In Central Europe the soil organic carbon content is mainly between 3-6%, and in Southern Europe there are many regions where the soil organic carbon content is 0-2% ( Figure 8).

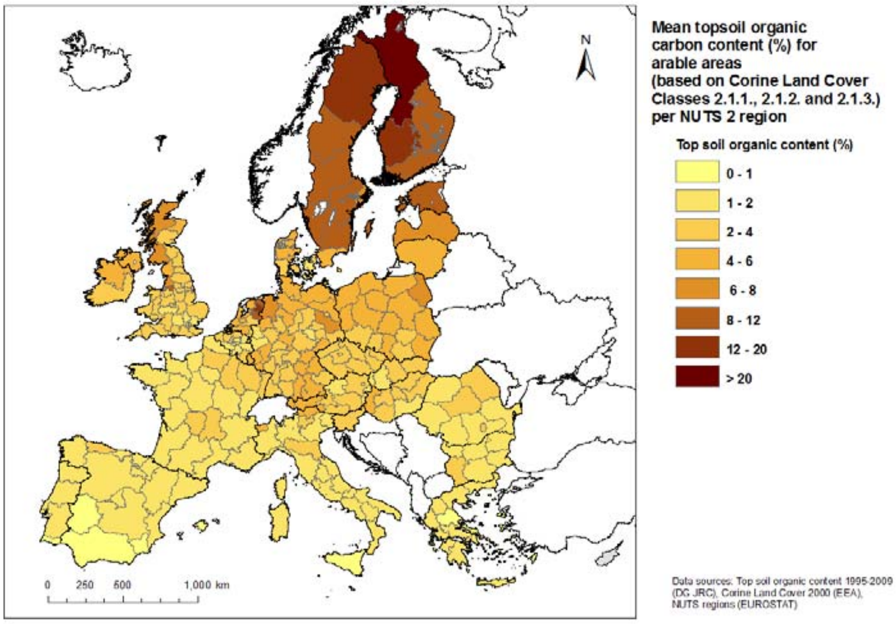



Figure 8 Mean topsoil organic carbon content (%) for arable land in the EU

#### 4.4.3 Impact on soil organic carbon stocks of converting grasslands to arable

On the basis of comparing the average topsoil organic carbon content of grasslands and arable lands at the NUTS 2 level it is possible to assess the change of carbon stock when converting grassland to arable in a given Member State. This assumes that the average SOC stocks of the surface horizon reflects the average equilibrium state. The comparison was carried out at the national level, rather than per NUTS 2 region, because more detailed data cannot be used for confidentiality reasons. At the EU-27 level there is on average 31 tonnes/ha of SOC stock loss due to conversions of grassland to arable land. The distribution of these losses at the Member State level shows that the difference between SOC stock in arable and grassland soils is much larger in Central European Member States as compared to southern European Member States (Figure 9). For example, in Poland converting grasslands to arable would result in a potential soil organic carbon loss of 60 tonnes/ha, whereas in Portugal the same conversion would be 15 tonnes/ha. This means that converting grasslands to arable in Central Europe has a greater impact on the total amount of existing SOC stocks than in southern Europe, where topsoil organic carbon content is already low. Notwithstanding the total losses, the further depletion of soils with already low or very low organic matter content has a siginificant influence on soil fertility too.

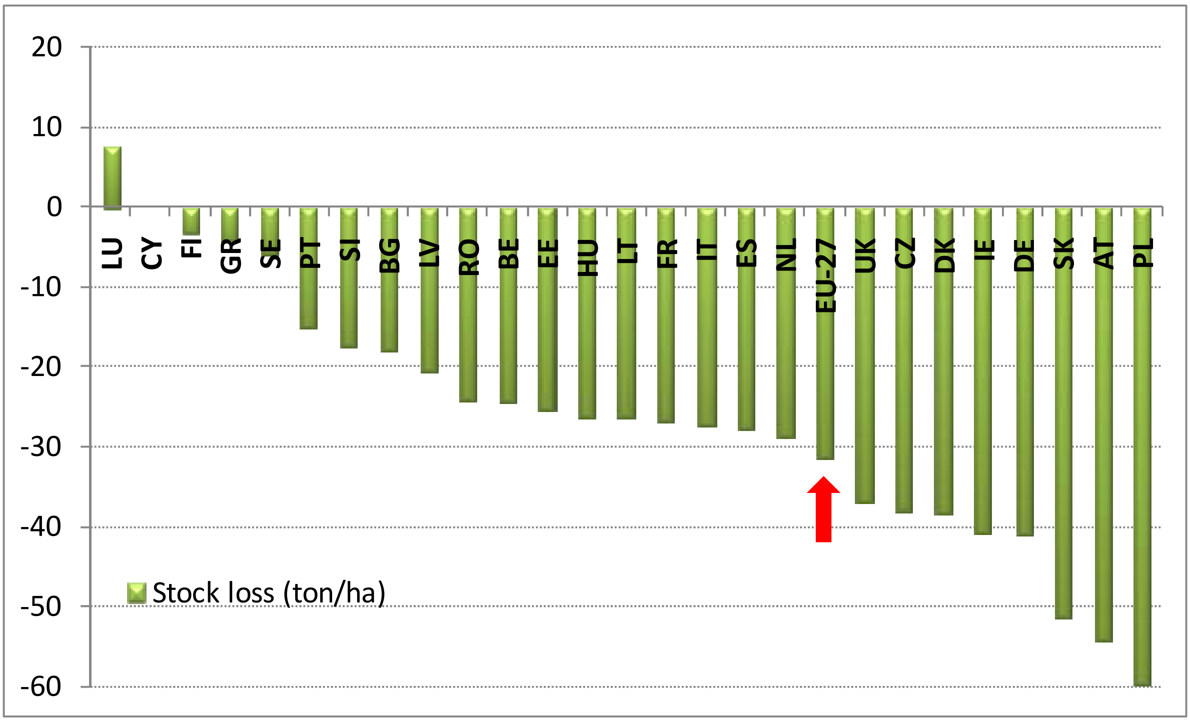



Figure 9 Potential SOC stock loss in tonnes/ha as a result of converting grassland to arable land on the basis of the topsoil organic carbon maps and assuming a surface horizon thickness of 20 cm

#### 4.4.4 Change in grassland areas and the change in the grassland share of agriculture due to the scenario option maintaining the current rules for the GAEC permanent pastures (BAU 2030), and for the scenario option abandoning the current rules for the GAEC permanent pastures (C-Poor 2030)

The utilised agricultural area (UAA) at national level will change because of different factors, regardless of the regulatory environment. The UAA is composed of all agricultural parcels, including arable land, permanent crops and pasture. An absolute decrease of the UAA is the result of a decline in farming area and the abandonment of agricultural parcels, but could also be due to pressures from other land uses such as increases in forest, nature areas, infrastructure or urban sprawl In addition, the current GAEC rules do not prescribe a fixed share of pasture over arable. Rather, the key factor is whether the annual ratio pasture:UAA decreases by more than 5%, or 10%, from the national reference ratio. There is therefore a need to evaluate both the evolution of the national UAA and the yearly evolution of the relative share of grassland out of the national UAA.

The grassland share of agricultural area is highest in Ireland, the UK and Bulgaria (Figure 10). Comparing the BAU 2030 and BASE 2000, there is a decline in grassland area for all Member States, apart from Cyprus. This reflects the general trend of total agricultural area and grassland area decline to either urban, forests or nature land uses between 2000 and 2030. According to LUMOCAP modelling results, the change in grassland area for EU-27 is -19% between 2000 and 2030 under the current rules. The main driving factors determining the LUMOCAP results are the growth of non-agricultural land uses, and competition for land between arable and grasslands. The largest percentage change in grassland areas occurs in Eastern Europe - Bulgaria (-34%), Latvia (-34%), Estonia (-33%) and Lithuania (-33%), whereas the smallest change in grassland areas occurs in Cyrus (+4%), Denmark (-6%) and Germany (-6%).

In terms of the impact of abandoning the current rules for the GAEC permanent pastures (i.e. from BASE 2000 to C-Poor 2030), the largest grassland area change is in Romania (-47%) and Bulgaria (-46%), and the smallest grassland area change is in Cyprus (-8%) and Finland (-11%). At the EU-27 level the change in grassland area between C-Poor 2030 and Base 2000 is -26%.

In Member States with a high grassland share of UAA (for example, Ireland, the UK and Bulgaria), the difference between C-Poor 2030 and the BAU 2030 is small compared to Member States with a low grassland share of UAA (for example, Denmark, Sweden). On the basis of this analysis we can conclude that if the GAEC rule to maintain permanent pastures is rescinded the net result is that Member States with a high grassland share of UAA will see more land use changes away from grassland than Member States with a low grassland share of UAA. In addition, abandoning the current GAEC rules would result, at EU-27 level, in a loss of grassland area 37% higher than if the rules were maintained.

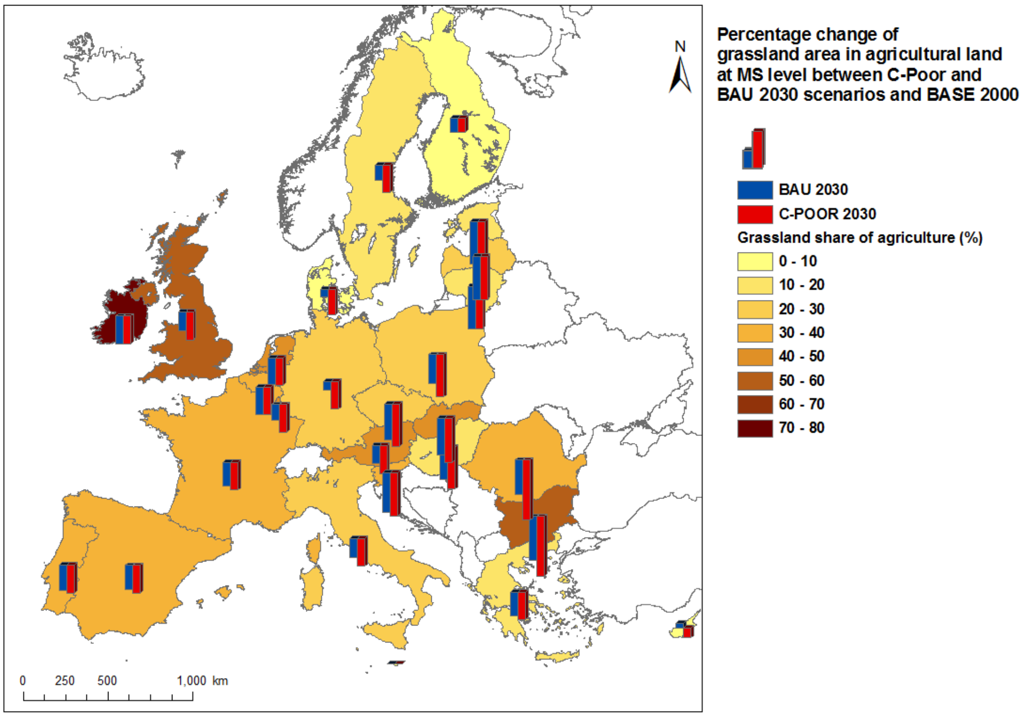



Figure 10 Grassland share of total utilised agricultural area in the baseline year (BASE 2000) in the background (shade of brown), with the percentage change in grassland area for the scenario maintaining the current rules for the GAEC permanent pastures (BAU 2030 – blue bars), and for the scenario abandoning the current rules for the GAEC permanent pastures (C-Poor 2030 – red bars)

The information contained in a visual form in Figure 10 above is presented in a tabular format in Table 6 below.

Table 6 Grassland area (ha) in the Member States for the baseline year (BASE 2000), for the scenario maintaining the current rules for the GAEC permanent pastures (BAU 2030), and for the scenario abandoning the current rules for the GAEC permanent pastures (C-Poor 2030)

| Member State | BASE 2000 (ha) | BAU 2030 (ha) | C-Poor 2030 (ha) | % change BASE to BAU 2030 | % change BASE to C-Poor 2030 |
|---|---|---|---|---|---|
| AT | 1,197,220 | 1,028,188 | 938,628 | -14 | -22 |
| BE | 645,550 | 507,703 | 507,703 | -21 | -21 |
| BG | 3,353,329 | 2,227,704 | 1,804,260 | -34 | -46 |
| CY | 3,692 | 3,823 | 3,399 | +4 | -8 |
| CZ | 962,321 | 696,826 | 647,159 | -28 | -33 |
| DE | 6,049,795 | 5,662,608 | 4,754,102 | -6 | -21 |
| DK | 194,091 | 183,055 | 155,255 | -6 | -20 |
| EE | 197,637 | 132,097 | 132,097 | -33 | -33 |
| ES | 8,122,339 | 6,617,602 | 6,357,921 | -19 | -22 |
| FI | 52,874 | 47,251 | 47,251 | -11 | -11 |
| FR | 10,553,780 | 8,643,030 | 8,285,551 | -18 | -21 |
| GR | 765,303 | 627,279 | 603,374 | -18 | -21 |
| HU | 1,056,471 | 796,750 | 713,102 | -25 | -33 |
| IE | 3,498,667 | 2,748,983 | 2,740,376 | -21 | -22 |
| IT | 3,877,131 | 3,292,361 | 3,039,570 | -15 | -22 |
| LT | 572,727 | 383,926 | 383,926 | -33 | -33 |
| LU | 66,335 | 58,336 | 52,055 | -12 | -22 |
| LV | 701,636 | 465,027 | 465,192 | -34 | -34 |
| NL | 1,141,747 | 909,876 | 898,327 | -20 | -21 |
| PL | 4,341,329 | 3,372,378 | 2,918,684 | -22 | -33 |
| PT | 1,444,998 | 1,158,344 | 1,129,509 | -20 | -22 |
| RO | 4,463,249 | 3,282,875 | 2,386,491 | -26 | -47 |
| SE | 441,279 | 388,852 | 348,479 | -12 | -21 |
| SK | 893,898 | 621,370 | 596,684 | -30 | -33 |
| SL | 340,672 | 243,956 | 223,340 | -28 | -34 |
| UK | 7,637,680 | 6,508,273 | 5,985,882 | -15 | -22 |
| EU-27 | 62,575,750 | 50,608,473 | 46,118,317 | -19 | -26 |

#### 4.4.5 Change in soil organic carbon stock (tonnes/ha) due to the scenario option maintaining the current rules for the GAEC permanent pastures (BAU 2030), and for the scenario option abandoning the current rules for the GAEC permanent pastures (C-Poor 2030)

The impact of different scenario options for maintaining permanent grasslands on SOC stock is assessed. The conversion from grass to arable will have a negative effect on soil carbon stocks, as shown in Figure 7 and Figure 8. In incorporating the impact of the scenarios we have assumed that the change in SOC stock will correspond to the average SOC stock difference in the topsoil between current grassland and arable land. The outcome of this analysis is that the average change of SOC stock for EU-27 is -17.2 tonnes/ha for the C-poor scenario (permanent pastures GAEC rescinded), compared to

-13.2 tonnes/ha for the BAU 2030 scenario (the change in carbon stocks under BAU results from the LUMOCAP model and reflects land use changes and climatic effects to 2030). The highest SOC stock losses for the C-poor scenario are Ireland (-35 tonnes/ha), Austria (-34 tonnes/ha) and the UK (-32 tonnes/ha), whereas the lowest SOC stock losses are for Mediterranean countries such as Greece (-2 tonnes/ha) and Portugal (-4 tonnes/ha). There are also interesting differences between the C-Poor and BAU 2030 scenarios – for example in Germany the difference between C-Poor and BAU 2030 is more than 10 tonnes/ha, whereas in Ireland the difference is less than 1 tonne/ha. The large differences are due to a combination of SOC stock and grassland area. Some regions have grasslands under higher soil organic content than others – meaning that the conversion from grasslands to arable in the C-Poor scenario has a larger impact in terms of SOC stock loss. However, there is also some uncertainty in the data – for example one would not expect such high losses for Austria, because it is not known for having high soil organic carbon soils. Likewise one would not expect arable soils to have a higher soil organic carbon than grassland – which seems to be the case in Luxembourg.

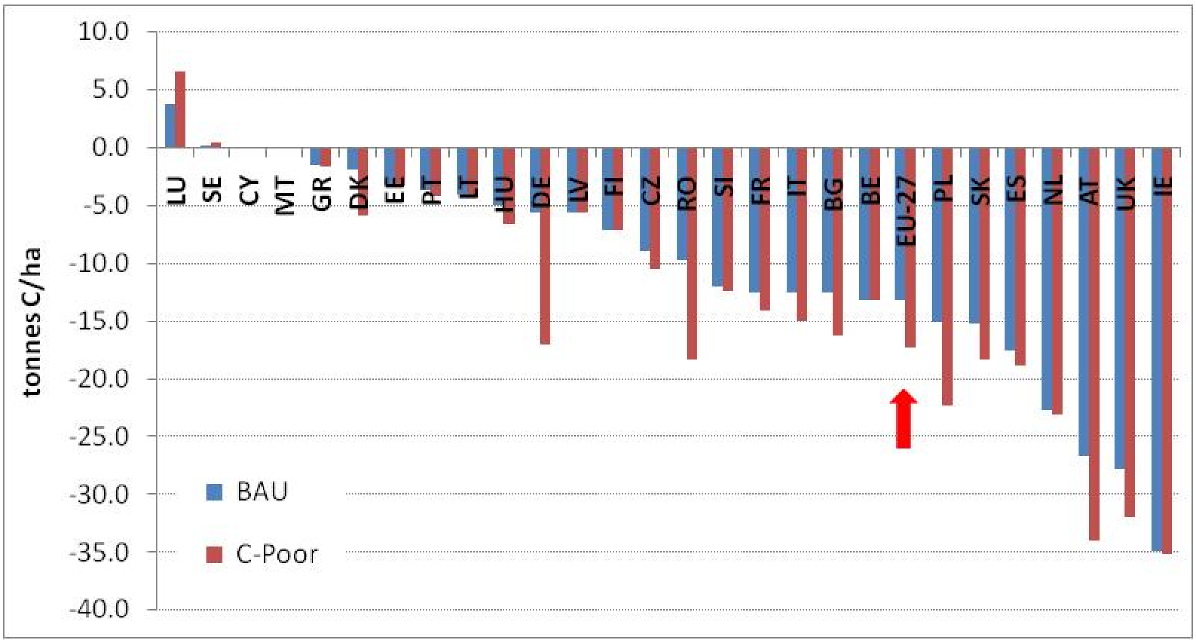



Figure 11 Potential SOC stock loss (in tonnes/ha) resulting from maintaining the current rules for the GAEC permanent pastures (BAU 2030) and abandoning the current rules for the GAEC permanent pastures (C-Poor 2030)

**Abolishing permanent grassland restrictions would have a negative effect on soil organic carbon stocks, which at EU level can be quantified in a carbon stock loss 30 % higher than in the case of maintaining the current permanent grassland restrictions.**

# CHAPTER 5 USE OF SET-ASIDE (FOR EU-15 ONLY)

## 5.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Agriculture and forests land use changes*** | | | | | |
| *Use of Set-aside (for EU-15 only)* | **25% former set aside to afforestation** | **10% former set aside to afforestation** | Former set aside to arable | Former set aside to arable | Former set aside to arable |

Set-aside was introduced by the European Union (EU) in 1988 and became compulsory in 1992 to help reduce the surpluses produced in Europe under the guaranteed price system of the Common Agricultural Policy (CAP) and to deliver some environmental benefits as a result of the intensification of agriculture. The Set-aside rate was originally set at 15% and it was then reduced to 10% in 1996. Following the introduction of decoupled payments in 2005, farmers who had historically claimed set-aside were awarded a number of set-aside 'entitlements' equivalent to the area they had previously set-aside. In order to receive payment on these set-aside entitlements, an equivalent number of hectares had to be removed from agricultural production. The European Union decided in November 2008 to abolish set-aside completely on the basis of the CAP Health Check. This was to help mitigate shortages in the EU cereals market and therefore reduce prices following two consecutive lower EU harvests. In the "Use of set aside" scenario we examine the impact of set aside areas being changed to arable or to forest.

## 5.2 Scenario approach and method

This scenario is only relevant for MS of EU-15, because EU-10 and EU-2 MS were not members of the EU when the set-aside policy in the CAP was established. For BAU the assumption is that all set-aside returns to arable land, for C-Medium the assumption is that 10% of former set-aside is changed to forest, and for C-Rich the assumption is that 25% of former set-aside is changed to forest. The last two options reflect policy to encourage higher SOM in soils and greater carbon sequestration.

The following steps are taken in the analysis:

1. The carbon content, expressed as weight percentage in the SOC map, is converted to SOC stock in tonnes per hectare by assuming a surface horizon thickness of 20 cm and using a pedotransfer function for deriving the bulk density. The average SOC stock of the surface horizon is assumed to reflect the equilibrium state - the differences between SOC stocks under different land uses reflect the

change from one equilibrium state to another. Our assumption is that the set-aside equilibrium is equivalent to grassland in a given region;

2. The differences in arable and forest areas and the arable share of agriculture by 2030 are assessed under the three scenario options, i.e. all set-aside is changed to arable (BAU 2030), 10% of former set-aside is changed to forest (C-High 2030), 25% of former set-aside is changed to forest (C-Rich 2030); and,

3. The area changes between set-aside, arable and forest are subsequently linked to SOC stock changes. Spatial analysis is used to combine land use changes from the scenario analysis and the top soil organic carbon map from the Joint Research Centre (Jones et al., 2004) to ascertain the SOC stock change due to the conversion of set-aside to arable and forest at the regional level (NUTS 2 administrative unit).

## 5.3 Soil organic carbon stock under agriculture

For relevant data and information see Section 4.3.

## 5.4 Results

The **change in arable and forest areas due to all set-aside changing to arable (BAU 2030), 10% of set-aside changing to forest (C-Medium 2030), 25% of set-aside changing to forest (C-Rich 2030)** results in contrasting areal changes across the EU-15 for the different scenarios.

In most EU-15 MS there is an increase in arable area for the BAU 2030 scenario option – but in some countries the percentage change is negative (e.g. UK, Ireland and the Netherlands) because compared to 2000 arable areas there is still a large decrease in arable areas by 2030 even if all set-aside is converted to arable. This is because the importance of set-aside is relatively small for these Member States.

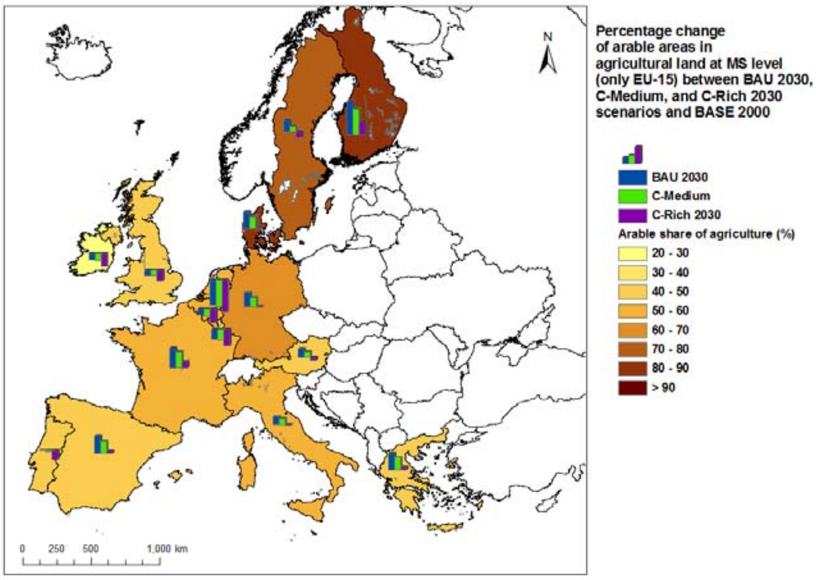



Figure 12 Arable share of agricultural area in the baseline year (BASE 2000) in the background (shade of brown), with the percentage change in arable area for the scenario all set-aside changing to arable (BAU 2030 – blue bars), for the scenario 10% of set-aside changing to forest (C-Medium 2030- green bars), and for the scenario 25% of set-aside changing to forest (C-Rich 2030 – purple bars)

The information contained in a visual form in Figure 12 above is presented in a tabular format in Table 7 below.

Table 7 Set-aside area (ha) per Member State (EU-15) in 2000

| MS | Set-aside (2000) | BAU 2030 | | C-Medium 2030 | | C- Rich 2030 | |
|---|---|---|---|---|---|---|---|
| | Area (ha) | Arable (ha) | Forest (ha) | Arable (ha) | Forest (ha) | Arable (ha) | Forest (ha) |
| AT | 107,039 | 107,039 | 0 | 96,335 | 10,704 | 80,279 | 26,760 |
| BE | 29,489 | 29,489 | 0 | 26,540 | 2,949 | 22,117 | 7,372 |
| DE | 1,346,682 | 1,346,682 | 0 | 1,212,014 | 134,668 | 1,010,012 | 336,671 |
| DK | 225,318 | 225,318 | 0 | 202,786 | 22,532 | 168,989 | 56,330 |
| ES | 1,113,614 | 1,113,614 | 0 | 1,002,253 | 111,361 | 835,211 | 278,404 |
| FI | 290,538 | 290,538 | 0 | 261,484 | 29,054 | 217,904 | 72,635 |
| FR | 1,627,011 | 1,627,011 | 0 | 1,464,310 | 162,701 | 1,220,258 | 406,753 |
| GR | 186,303 | 186,303 | 0 | 167,673 | 18,630 | 139,727 | 46,576 |
| IE | 32,590 | 32,590 | 0 | 29,331 | 3,259 | 24,443 | 8,148 |
| IT | 309,064 | 309,064 | 0 | 278,158 | 30,906 | 231,798 | 77,266 |
| LU | 2,132 | 2,132 | 0 | 1,919 | 213 | 1,599 | 533 |
| NL | 26,209 | 26,209 | 0 | 23,588 | 2,621 | 19,657 | 6,552 |
| PT | 83,557 | 83,557 | 0 | 75,201 | 8,356 | 62,668 | 20,889 |
| SE | 310,397 | 310,397 | 0 | 279,357 | 31,040 | 232,798 | 77,599 |
| EU-15 | 5,689,947 | 5,689,943 | 0 | 5,120,949 | 568,994 | 4,267,457 | 1,422,486 |

There are no data on SOC stocks for set-aside areas, therefore we assume that the carbon stocks of set-aside areas are equal to the average values for grassland carbon stocks (natural vegetation of set-aside has characteristics similar to permanent grassland habitats with grasses covering around 75% of the fields) – even though this assumption is less realistic for the Base year (2000 – 2005) than for 2030. The conversion options are mostly to arable land and the majority of SOC stock changes are negative, with the BAU option being the most negative (the change in carbon stocks under BAU results from the LUMOCAP model and reflects land use changes and climatic effects to 2030). The SOC stock losses are much higher for Denmark (BAU 2030 loss is -36 tonnes/ha), Germany (-20 tonnes/ha) and Austria (-12 tonnes/ha), than for Member States such as the Netherlands (-0.6 tonnes/ha), Portugal (-1.2 tonnes/ha), Greece (-1.5 tonnes/ha) and Belgium (-2.5 tonnes/ha) (Figure 13). The differences can be traced back to the relative importance of set-aside for the different Member States – for instance Denmark had more than 225 000 ha of set-aside, whereas Belgium only had 29 000 ha – but also to the soil organic matter content of the soils in the region. The average soil organic carbon stock loss for EU-15 is -5.2 tonnes/ha for BAU 2030, -4.2 tonnes/ha for C-Medium and -1.8 tonnes/ha for C-Rich. **Promoting the afforestation of 10% and 25% former set-aside land in the EU-15 would therefore reduce the loss of soil organic carbon by 2030 by 19% and 65% respectively compared to a business as usual (BAU) scenario.**

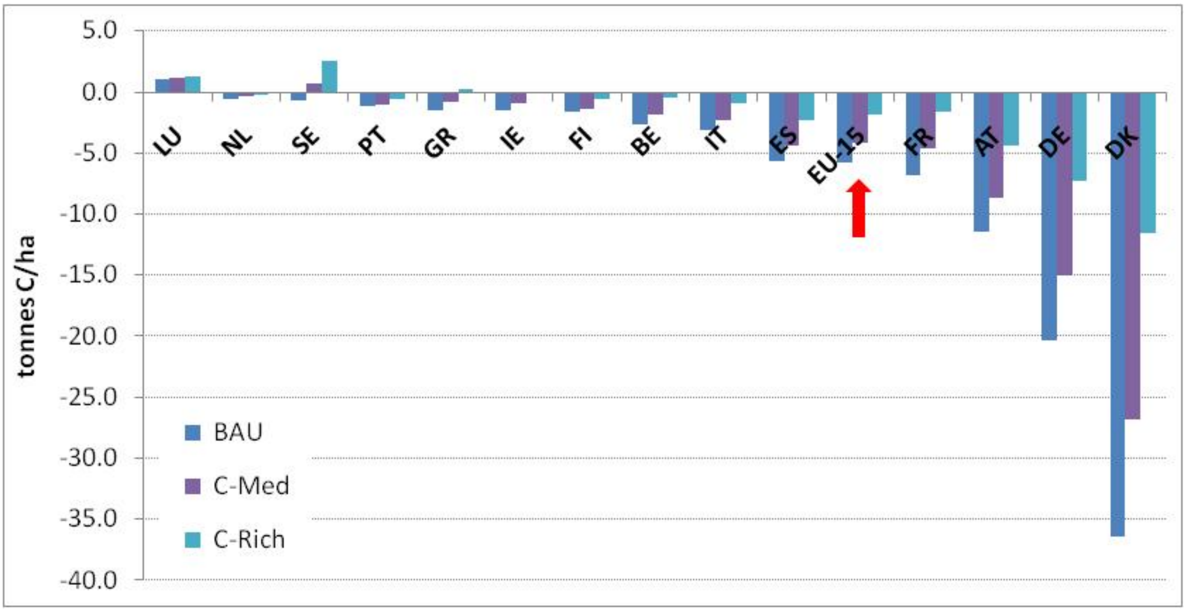



Figure 13 SOC stock loss (in tonnes/ha) due to conversion from set-aside area

# CHAPTER 6 CHANGE FROM UTILISED AGRICULTURAL AREA (UAA) TO FOREST

## 6.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Agriculture and forests land use changes*** | | | | | |
| *Change from Utilised Agricultural Area (UAA) to forest* | **Faster decrease of the UAA in favour of forests** | Current change of UAA in favour of forests | Current change of UAA in favour of forests | Current change of UAA in favour of forests | Current change of UAA in favour of forests |

The change from utilised agricultural area (UAA) to forest scenario examines the effect on the soil organic carbon stock of converting agricultural land to forest at a higher rate (2% higher) (C-Rich) than the current conversion rates (BAU). The scenario is related to agri-environmental measures that encourage farmers to convert agricultural land to forest.

## 6.2 Scenario approach and method

Data obtained from the reporting submissions to the UNFCCC (LULUCF) are used to assess the forest area trends for each MS between 1990 and 2007[12]. The LUMOCAP model indicates historical changes in UAA using data from 1990 to 2008. Therefore we combine changes in forest area reported to the UNFCCC with changes in UAA from the LUMOCAP model to indicate realistic rates for UAA to forest changes.

We adopt two contrasting scenario options for the change from utilised agricultural area (UAA) to forest scenario (Table 5):

1. Current change of UAA in favour of forests (BAU 2030)
2. Faster decrease of UAA in favour of forests (C-Rich 2030)

For "Current change of UAA in favour of forests" it s assumed that the forest trends calculated from the UNFCCC database per Member State is continued until 2030. For "Faster decrease of UAA in favour of forests (C-Rich 2030)" the "faster decrease" is taken to be 2% higher than the trends specified at Member State level.

[12] Reporting submissions to the UNFCCC (LULUCF) can be downloaded from http://unfccc.int/files/kyoto_protocol/application/pdf/awgkplulucfdataeu051109.pdf

The following steps are taken in the analysis:

1. Assess the trends in forest areas at Member State level, and use these trends in the LUMOCAP scenarios. We assume that the arable part of the UAA is converted to forests. The soil organic content in the surface horizon of both arable and forest land is determined at regional level (NUTS 2 administrative unit) by overlaying arable and forest areas from Corine Land Cover (CLC) with the soil organic carbon map from the Joint Research Centre (Jones et al., 2004);
2. The carbon content, expressed as weight percentage in the SOC map, is converted to SOC stock in tonnes per hectare by assuming a surface horizon thickness of 20 cm and using a pedotransfer function for deriving the bulk density. The average SOC stock of the surface horizon is assumed to reflect the equilibrium state; the differences between SOC stocks under different land uses reflect the change from one equilibrium state to another;
3. The differences in forest areas and the forest share of a Member State by 2030 are assessed under the two scenario options, i.e. adopting the current change of UAA in favour of forests (BAU 2030) and adopting a faster decrease of UAA in favour of forests (C-Rich 2030); and,
4. The area changes between UAA and forests are subsequently linked to SOC stock changes. Spatial analysis is used to combine land use changes from the scenario analysis and the top soil organic carbon map from the Joint Research Centre (Jones et al., 2004) to ascertain the SOC stock change due to the conversion of UAA to forests at regional level (NUTS 2 administrative unit).

## 6.3 Soil organic carbon stock under forests

### 6.3.1 Surface area of forest and other wooded land

In order to quantify a regional soil organic matter balance, the forest area must be taken into account. The forest area under broadleaved and coniferous forests is reported in several datasets ranging from land use maps based on remote sensing to national forest inventory data and statistics. Forest land cover derived from Corine Land Cover was compared to the total forest area as reported to the United Nations Framework Convention on Climate Change (UNFCCC) and FAO's Forest Resources Assessment (FRA). Most statistical data sources depend on national inventories but forest areas reported are different between different data sources. The differences can be explained by the resolution of reporting, level of aggregation and the definition of forest or wooded area in terms of minimum area, minimum width, minimum crown cover, the diameter at breast height per area, the presence or absence of newly planted trees.

Comparison of forest protection between regions in Europe is extremely difficult, because there is such a wide variation of strategies, procedures and constraints. In addition, there is a paradigm shift from total protection in segregated areas to 'precision protection' of specific locations and to combining protection and timber production in a holistic, integrated concept of modern management of forests. The latter influences harvested wood figures and national inventories since commercial forests have a higher yield as compared to multi-functional forests. Protected forests and reserves are derived from the European Forest Institute (Parviainen et al., 1999) and are assumed not to contribute to statistics on wood production or national inventories. The data are based on a combination of the results of the research network COST action E4 and the FAO-FRA for countries where no data were available in the first dataset.

Another important factor in organic matter management is the area percentage allocated to different species. National forest inventories usually have records of species data but these do not feature in international statistical databases. We analysed different datasets (UNECE, EFISCEN, CLC) for species composition and could derive an average area percentage of coniferous dominant, broad leaved dominant and entirely mixed forests. The European Forest Information Scenario Database, EFISCEN, is a forest inventory database of European countries, based on input from national inventory experts (Schelhaas et al., 2006). The database contains information of the forest resources of 30 countries including forest area, standing volume and increment, covering 23 European Member States. Spain, Greece, Cyprus and Malta are unfortunately missing in the database. Since spruce represents a very important part of coniferous forest, the share of spruce to the total forested area was calculated based on EFISCEN inventory data.

The forest and other wood land (FOWL) area in the EU 27 is 170 Mha (UNECE), or 40% of the total land area, (Table 8, Figure 14) with 34% forest and 6% other wooded land. The largest area covered with forests and other wood land (FOWL) is in Sweden (30 Mha hectares or 67% of land), Spain (26 Mha or 51% of land), Finland (23 Mha or 67% of land), France (17 Mha or 31% of land), Italy (11 Mha or 36% of land) and Germany (11 Mha or 30% of land). Together these six Member States accounted for more than two thirds of total FOWL area in the EU-27. Finland and Sweden have the highest percentage of their land area covered by FOWL followed by Slovenia (58% of the country), Spain, Greece (49%), Estonia (48%) and Austria (47%). The area of protected FOWL, i.e. forest reserves and protected FOWL (Parviainen et al., 1999), is less than 8% of the total land area. The four largest protected areas, located in Finland (3.9 Mha ha or 17% of its FOWL), Spain (3.0 Mha or 12%), Sweden (1.4 Mha or 5%) and Greece (1.1 Mha or 17%) account for almost 70% of the total protected FOWL in EU-27.

More than half of the forest and other wood land (FOWL) area in 2000 is covered by conifers (UNECE, 2000). In terms of forest type distribution the Member State with the largest share of broad leaved forest out of the total FOWL is Hungary (78% ), followed by Italy (72%), Romania (70%), Bulgaria (67%) and France (64%). The largest share of conifer forest out of the total FOWL is Ireland (83% ), followed by Finland (80%), Sweden (77%), and Austria (68%). The distribution of broad leaved, coniferous and mixed forest is similar for the EFISCEN plot data as compared to the distribution for UNECE forest area. In addition EFISCEN plot data enabled the share of some species to the forest type, e.g. spruce makes up one third of the coniferous forest area in eight Member States (IE, AT, BE, DK, CZ, UK, LU and DE). The latter is important for further biomass calculations.

Table 8 Surface area in 1000 ha of forest and other wood land (FOWL), of which forest and protected forest, and percentages of broadleaved, coniferous and mixed forest (source data: UNECE, 2000; CLC; COSTE4).

| | | FOWL | Forest | Protected FOWL | Share of land* | Broad Leaved | Conifer | Mixed |
|---|---|---|---|---|---|---|---|---|
| Code | Member State | Area in 1000 ha | | | % | % | % | % |
| AT | Austria | 3,924 | 3,840 | 57.1 | 46.8 | 12.2 | 68 | 19.7 |
| BE | Belgium | 672 | 646 | 26.3 | 22.0 | 50.7 | 42.4 | 6.9 |
| BG | Bulgaria | 3,903 | 3,590 | 368.6 | 35.2 | 67.4 | 22.1 | 10.5 |
| CY | Cyprus | 280 | 117 | 0 | 30.3 | 0.6 | 99.4 | 0 |
| CZ | Czech Republic | 2,630 | 2,630 | 200 | 33.3 | 13.2 | 31.2 | 55.7 |
| DE | Germany | 10,740 | 10,740 | 425 | 30.1 | 25.3 | 56.4 | 18.4 |
| DK | Denmark | 538 | 445 | 98.1 | 12.5 | 29.4 | 55.1 | 15.4 |
| EE | Estonia | 2,162 | 2,016 | 0 | 48.0 | 20.6 | 39.1 | 40.3 |
| ES | Spain | 25,984 | 13,509 | 3032.6 | 51.5 | 37.9 | 43.5 | 18.6 |
| FI | Finland | 22,768 | 21,883 | 3970 | 67.4 | 8 | 79.5 | 12.4 |
| FR | France | 16,989 | 15,156 | 194 | 30.9 | 63.8 | 27.2 | 9 |
| GR | Greece | 6,513 | 3,359 | 1093.7 | 49.4 | 57.5 | 42.5 | 0 |
| HU | Hungary | 1,811 | 1,811 | 374.1 | 19.5 | 77.7 | 10.1 | 12.2 |
| IE | Ireland | 591 | 591 | 11.5 | 8.4 | 12.8 | 82.7 | 4.5 |
| IT | Italy | 10,842 | 9,857 | 622.5 | 36.0 | 71.7 | 21.2 | 7 |
| LT | Lithuania | 2,050 | 1,978 | 0 | 31.5 | 36 | 46 | 18 |
| LU | Luxembourg | 89 | 86 | 0 | 34.4 | 61.6 | 36 | 2.3 |
| LV | Latvia | 2,995 | 2,884 | 0 | 46.1 | 18.5 | 39.1 | 42.4 |
| MT | Malta | 0.347 | 0.347 | 0 | 1.1 | 0 | 0 | 100 |
| NL | Netherlands | 339 | 339 | 21.5 | 8.2 | 43.1 | 42.2 | 14.7 |
| PL | Poland | 8,942 | 8,942 | 186.9 | 28.6 | 15.4 | 66.6 | 18 |
| PT | Portugal | 3,467 | 3,383 | 563.2 | 37.7 | 60.5 | 26.5 | 13 |
| RO | Romania | 6,680 | 6,301 | 527.1 | 28.1 | 69.7 | 30.3 | 0 |
| SE | Sweden | 30,259 | 27,264 | 1408.5 | 67.2 | 6.6 | 76.6 | 16.8 |
| SI | Slovenia | 1,166 | 1,099 | 81.4 | 57.5 | 37.4 | 30.1 | 32.5 |
| SK | Slovakia | 2,031 | 2,016 | 285.4 | 41.6 | 48.7 | 32 | 19.3 |
| UK | United Kingdom | 2,489 | 2,469 | 138.7 | 10.2 | 36.5 | 56.5 | 7 |
| **EU-27** | **EU-27** | **170,854** | **146,951** | **13686.2** | **39.5** | **32.1** | **52.8** | **15.1** |

* Share of land reflects ratio of FOWL to total land area per MS and for EU-27.

The most important area to consider in further biomass calculations is the forest area available for wood supply (FAWS) since all felling statistics and national forest inventories detailing biomass production and forest product harvesting refer to this surface area. Nearly 1.26 Mha or 73.5% of the forest and other wooded land is available for wood supply. The Member States with the largest FAWS area (in decreasing order of importance SE, FI, FR, ES, DE) account for 77 Mha or 61% of the EU-27 total FAWS area. Nearly 55 % of the EU-27 FAWS area is covered with conifer forests, 31% with broad leaved forest and the other area with a mixture of the former.

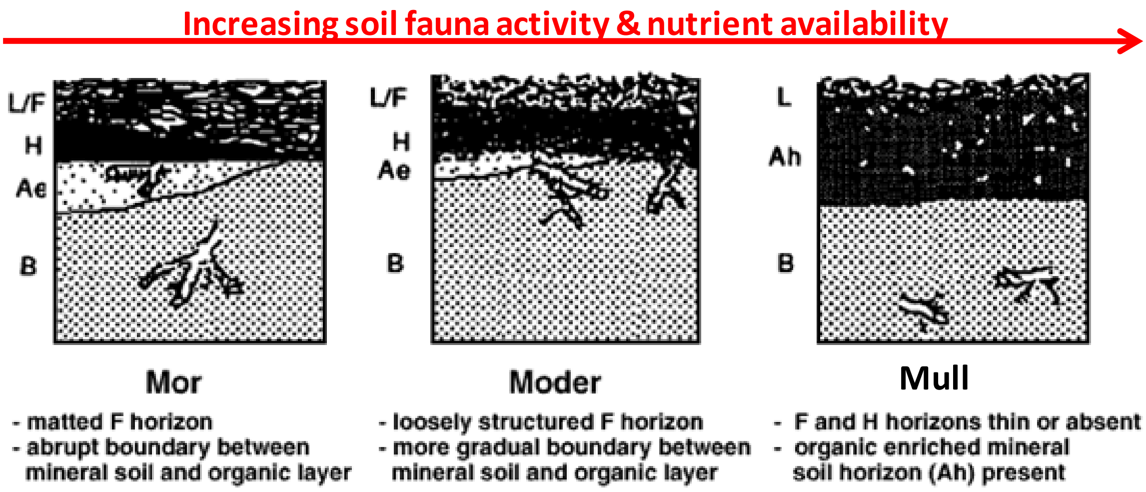

Data source: Forest Data and Information Centre, JRC; Pekkarinen et al., 2009.

Figure 14 The 2000 forest cover map

### 6.3.2 Status of soil organic matter under forests

The majority of the forest soils are derived from sedimentary rocks. European forest soils are often marginal for agriculture, due to physical limitations such as stoniness, sandy texture and high carbonate content. There are large differences in the morphology of humus beneath the various forest cover types and even beneath the same forest types growing on different soil types.

The organic layers (LFH) develop under well- to imperfectly drained conditions on top of the mineral soil layers (ABC layers). Their characteristics determine the humus form and are important for sustaining forest productivity. Seven morphological humus types are recognized according to the Forest Soil Coordinating Centre (FSCC) (Van Mechelen et al., 1997): mull, moder, mor, peat, anmor, raw (roh) and other. Mor and mull are most common with moder being a transition between them (Figure 15). Soils with peat or mor humus accumulate large amounts of plant organic material at the soil surface, as a result of a slow decomposition rate, and are dominant in northern Europe. Mull humus, characterised by a fast turnover rate and an intimate mixture with mineral soil materials, is the most frequently observed humus type in southern Europe. Next to climate, soil nutrient availability influences the decomposition rate of organic matter and the distribution of humus types. In climatic zones, where both mor and mull types are found, mor humus is usually found on nutrient-poor soils, containing organic matter with a wide range in C/N ratio, whereas C/N values are lower and nutrient contents are higher in soils underlying mull humus. The term raw (roh) humus indicates a thick accumulation of undecomposed litter, but is not often used. Anmor humus is used for dark mixtures of mineral soil and organic materials accumulated under hydromorphic conditions. Peat is composed of peat forming plant species such as Spaghnum mosses and sedges that accumulated under low base conditions and high water content.




After: Lavender et al. (1990) - LFH are organic layers, AB are mineral layers.

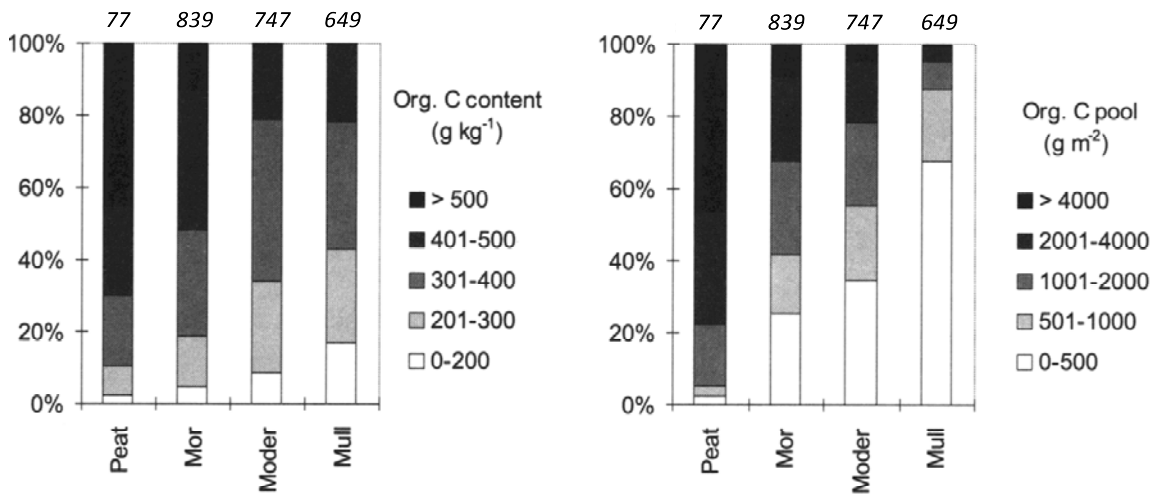
Figure 15 The distinguishing characteristics of mor, moder and mull humus forms in forest soils

***L: Litter*** *- relatively fresh organic residue with virtually no evidence of decomposition, the original structure is discernible (e.g., needles) but does not include rotting wood.*

***F: Fermented*** *- moderately decomposed organic residue, still identifiable as to its origin.*

***H: Humus*** *- well-decomposed organic residue dominated by fine substances in which the original vegetative material is not usually discernible. Rotting wood humus is included because it is considered more "active" nutritionally.*

The organic fraction of the forest soil ranges from untransformed or slightly transformed plant remains to humic substances, for which the cellular organisation of plant material is not recognisable under a light microscope. The majority of organic layers (LFH) has an organic carbon concentration between 20 and 50 %. Organic layers that are saturated with water for prolonged periods during the year, accumulate much more organic matter, exceeding 20 t/ha (2 kg m-2) in 66% of forest soils, as compared to organic layers with a drier moisture regime where less than 1 kg/m$^2$ of organic carbon is accumulated. Peat layers contain more than 40% organic carbon and store between 1 and 30 kg/m$^2$. Organic carbon pools in mor and moder layers vary within a wide range of values, but are about 10 times lower; from less than 0.1 to more than 3 kg/m$^2$ (Figure 16). Organic carbon pools in mull layers are generally lower than 1 kg/m$^2$.




Data source: Van Mechelen et al., 1997; numbers in italic represent number of samples.

Figure 16 Organic carbon content (left) and organic carbon stock (right) as related to humus type in forest plots

## 6.4 Results

### 6.4.1 Trends in forest areas at Member State level

Data from the reporting submissions to the UNFCCC (LULUCF) indicate that the forest area in EU-27 has increased by 369000 ha per year between 1990 and 2007 or on average 0.38% per year (Table 9). The largest increase in forest area is reported by Ireland (2.5% per year), whereas Belgium reported a decrease of forest area (-0.2% per year).

Table 9 Change in forest area (based on UNFCCC reporting) for the period 1990 to 2007

| Member State | average forest area change (1990 to 2007) 1000 ha/year | average rate of forest area change (1990 to 2007) % /year | forest area change for C-rich option 1000 ha/year | rate of forest area change for C-rich option % /year |
|---|---|---|---|---|
| Austria | 0 | 0 | 0.0 | 0.0 |
| Belgium | -1.18 | -0.19 | -1.2 | -0.2 |
| Bulgaria | 45.82 | 1 | 46.7 | 1.5 |
| Cyprus | 11.00 | 0.47 | 11.2 | 0.5 |
| Czech Republic | 1.25 | 0.05 | 1.3 | 0.1 |
| Denmark | 2.77 | 0.48 | 2.8 | 0.5 |
| Estonia | 2.92 | 0.14 | 3.0 | 0.1 |
| Finland | 15.82 | 0.07 | 16.1 | 0.1 |
| France | 62.11 | 0.40 | 63.4 | 0.4 |
| Germany | 18.85 | 0.18 | 19.2 | 0.2 |
| Greece | 2.95 | 0.05 | 3.0 | 0.1 |
| Hungary | 14.82 | 0.88 | 15.1 | 0.9 |
| Ireland | 14.46 | 2.46 | 14.7 | 2.5 |
| Italy | 72.78 | 0.90 | 74.2 | 0.9 |
| Latvia | 1.18 | 0.04 | 1.2 | 0.0 |
| Lithuania | 10.93 | 0.56 | 11.1 | 0.6 |
| Luxembourg | 0.29 | 0.32 | 0.3 | 0.3 |
| Malta | 0.00 | 0.00 | 0.0 | 0.0 |
| Netherlands | 0.66 | 0.17 | 0.7 | 0.2 |
| Poland | 19.64 | 0.22 | 20.0 | 0.2 |
| Portugal | 4.59 | 0.13 | 4.7 | 0.1 |
| Romania | 3.26 | 0.06 | 3.3 | 0.1 |
| Slovakia | 0.66 | 0.03 | 0.7 | 0.0 |
| Slovenia | 6.59 | 0.59 | 6.7 | 0.6 |
| Spain | 40.30 | 0.31 | 41.1 | 0.3 |
| Sweden | 1.48 | 0.01 | 1.5 | 0.0 |
| UK | 14.64 | 0.62 | 14.9 | 0.6 |
| **EU-27** | 369.00 | 0.38 | 376.4 | 0.4 |

### 6.4.2 Distribution of soil organic carbon content in the surface horizon of forests

The soil organic carbon content (expressed as percentage of total dry matter in the surface horizon of arable land) is much higher in Scandinavia (>8%), because of the prevalence of forests on peat soils. In Southern Europe there are many regions where the soil organic carbon content is less than 4% (Figure 17). In general the soil organic content of forest soils is higher than grasslands (Figure 7) and arable soils ( Figure 8).

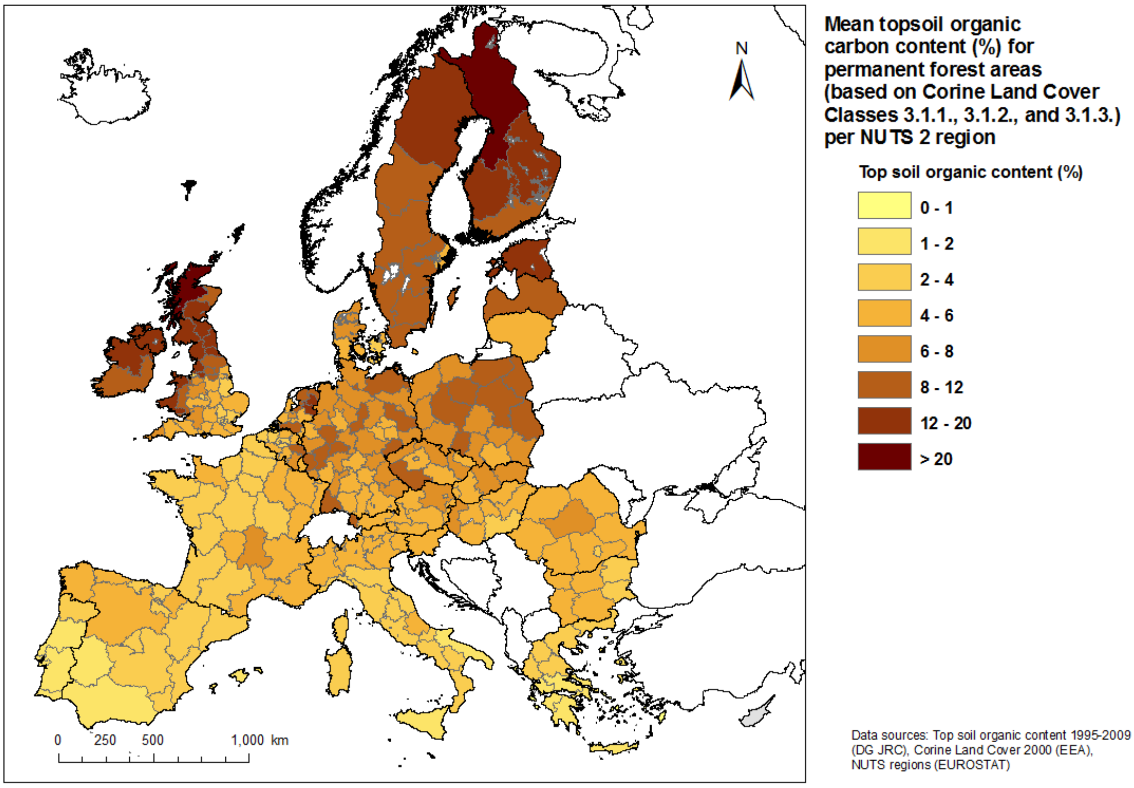



Figure 17 Topsoil organic carbon content (%) for forest soils per NUTS2 region.

### 6.4.3 Impact on soil organic carbon stocks of converting UAA to forest

On the basis of the difference between the average topsoil organic carbon content of arable and forests at the NUTS 2 level and the land use change, it is possible to assess the change of carbon stock when converting UAA to forests in a given Member State. This assumes that the average SOC stocks in the respective land uses reflect equilibrium. The analysis was carried out at the national level, to be in line with the "maintenance of grassland" and the "use of set-aside" scenarios. At the EU-27 level there is on average 47 tonnes/ha of SOC stock gain due to conversions of UAA to forest land. The distribution of these losses at the Member State level shows that the difference between SOC stock in arable and forest soils is much larger in Central European Member States as compared to Southern European Member States (Figure 18). For example, in Ireland converting UAA to forest would result in a potential soil organic carbon gain of more than 120 tonnes/ha, whereas in Portugal the same conversion would result in a gain of only 6 tonnes/ha. The large increase in SOC stock between arable and forest is probably due to forest areas growing on carbon rich soils in certain Member States (Figure 18), notably in Central Europe. This results in the finding that converting UAA to forests in Central Europe has a greater beneficial impact of sequestering additional carbon into existing SOC stocks than in southern Europe where topsoil organic carbon content is already low.

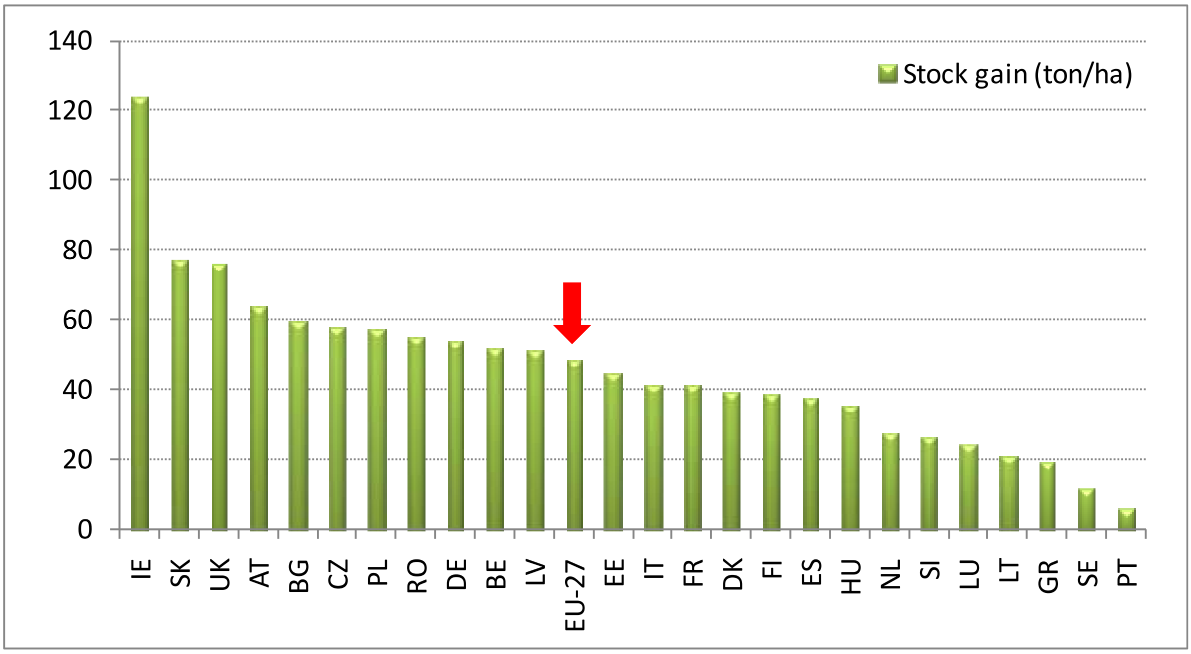



Figure 18 Stock gain in tonnes C/ha as a result of afforestation of arable land on the basis of the topsoil organic carbon map

#### 6.4.4 Change in forest areas and the change in the forest share due to the scenario adopting the current change of UAA in favour of forests (BAU 2030) and adopting a faster decrease of UAA in favour of forests (C-Rich 2030)

The highest share of forest areas are found in Scandinavia (more than 50%), and Central Europe (more than 40%), where as in most other MS the share is less than 30% (Figure 19). This influences the impact of the scenario for adopting the current change of UAA in favour of forests (BAU 2030), because it results in a large range of forest area changes across Europe (Table 10). For example in Sweden and Finland, where the percentage share of forest in 2000 is high (more than 60%) the relative change in forest due to conversions from UAA to forest is low for BAU 2030 (less than 2%). On the other hand the countries with a low percentage share of forest the relative changes are high (more than 50%). For example Ireland, with a relatively small forest share of 4% is expected to more than double its forest land for BAU 2030.

The impact of adopting a faster decrease of UAA in favour of forests, which will increase the soil carbon balance stock (C-Rich 2030) has a positive impact for all MS and results at the EU-27 level in a 13% increase in forest area in comparison to the 8% increase of BAU 2030. At the MS level the range of % changes is wide for the C-Rich 2030 scenario: 142% increase in Ireland and only 1% increase in Sweden.

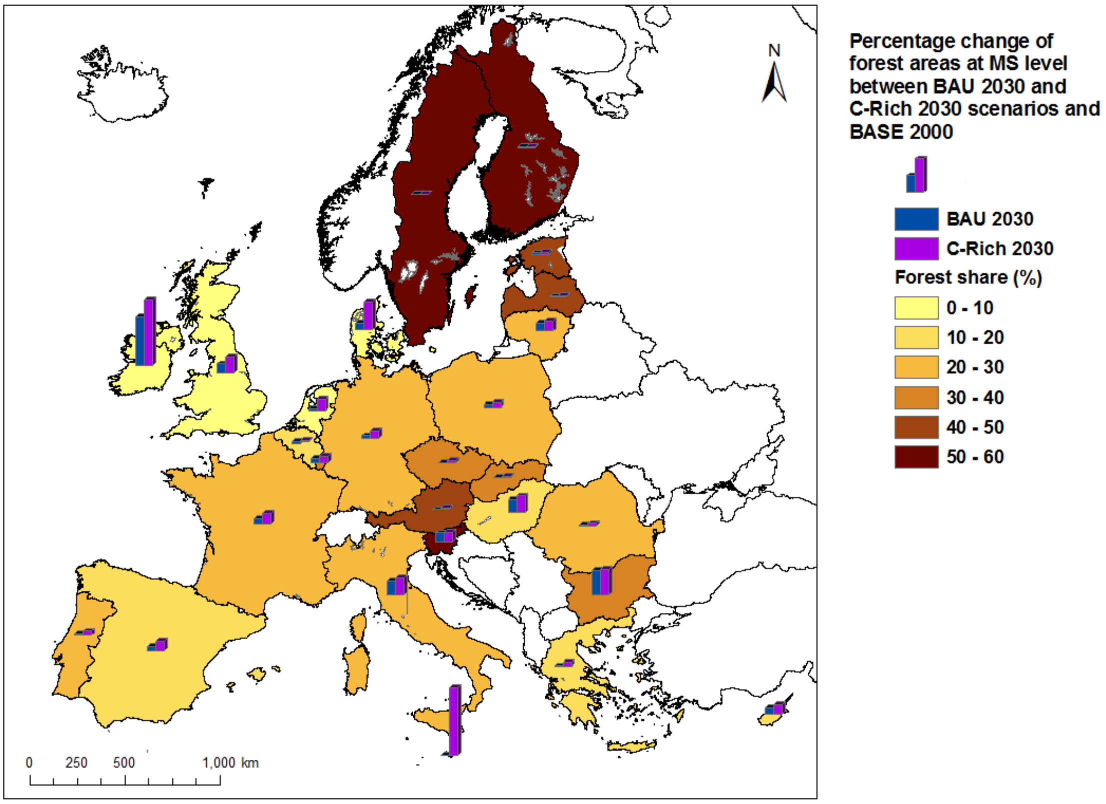



Figure 19 Forest share of MS in the baseline year (BASE 2000) in the background (shade of brown), with the percentage change in forest area for the scenario all set-aside changing to arable (BAU 2030 – blue bars),and for the scenario adopting a faster decrease of UAA in favour of forests (C-Rich 2030 – purple bars)

The information contained in a visual form in Figure 19 above is presented in a tabular format in Table 10 below.

Table 10 Forest areas and forest area changes per Member State for BASE 2000, BAU 2030 and C-Rich 2030

| MS | BASE 2000 (ha) | BAU 2030 (ha) | C-Rich 2030 (ha) | % change BASE to BAU 2030 | % change BASE to C-Rich 2030 |
|---|---|---|---|---|---|
| AT | 3,789,100 | 3,789,100 | 3,870,300 | 0 | 2 |
| BE | 615,700 | 581,600 | 624,200 | -6 | 1 |
| BG | 3,458,000 | 5,294,400 | 5,409,300 | 53 | 5 |
| CY | 156,500 | 180,100 | 188,900 | 15 | 21 |
| CZ | 2,558,200 | 2,596,900 | 2,688,200 | 2 | 5 |
| DE | 10,401,500 | 10,978,100 | 11,743,300 | 6 | 13 |
| DK | 760,600 | 878,200 | 1,122,800 | 15 | 48 |
| EE | 2,071,600 | 2,160,400 | 2,190,400 | 4 | 6 |
| ES | 9,213,900 | 10,110,400 | 10,895,700 | 10 | 18 |
| FI | 19,296,200 | 19,705,600 | 19,835,700 | 2 | 3 |
| FR | 14,463,900 | 16,304,100 | 17,370,800 | 13 | 20 |
| GR | 2,347,900 | 2,383,400 | 2,535,200 | 2 | 8 |
| HU | 1,719,000 | 2,235,800 | 2,362,700 | 30 | 37 |
| IE | 294,200 | 609,900 | 712,200 | 107 | 142 |
| IT | 7,912,600 | 10,352,700 | 10,743,200 | 31 | 36 |
| LT | 1,854,300 | 2,192,500 | 2,272,400 | 18 | 23 |
| LU | 91,900 | 101,100 | 104,500 | 10 | 14 |
| LV | 2,727,500 | 2,760,400 | 2,817,100 | 1 | 3 |
| MT | 200 | 200 | 500 | 0 | 150 |
| NL | 309,800 | 326,000 | 383,200 | 5 | 24 |
| PL | 9,153,700 | 9,777,500 | 10,180,200 | 7 | 11 |
| PT | 2,414,400 | 2,510,400 | 2,617,700 | 4 | 8 |
| RO | 7,008,400 | 7,135,700 | 7,405,500 | 2 | 6 |
| SE | 25,390,200 | 25,466,500 | 25,622,300 | 0 | 1 |
| SI | 1,134,000 | 1,352,900 | 1,366,700 | 19 | 21 |
| SK | 1,918,300 | 1,935,600 | 1,982,800 | 1 | 3 |
| UK | 1,982,200 | 2,386,000 | 2,672,000 | 20 | 35 |
| EU-27 | 133,043,800 | 144,105,500 | 149,717,800 | 8 | 13 |

#### 6.4.5 Change in soil organic carbon stock loss (tonnes/ha) due to the scenario adopting the current change of UAA in favour of forests (BAU 2030) and adopting a faster decrease of UAA in favour of forests (C-Rich 2030)

The impact of different scenario options for adopting the current change of UAA in favour of forests (BAU 2030) and adopting a faster decrease of UAA in favour of forests on SOC stock is assessed. The conversion from UAA to forests will have a positive effect on soil carbon stocks, as shown in Figure 7 and Figure 17. In incorporating the impact of the scenarios we have assumed that the change in SOC stock will correspond to the average SOC stock difference in the topsoil between current arable and forest land; the SOC stock is subsequently weighted by the land use change area. The outcome of this analysis is that the average SOC stock change for EU-27 is +18.2 tonnes/ha for the BAU 2030 scenario, compared to + 20 tonnes/ha for the C-Rich scenario (Figure 20). The highest relative SOC stock gains for the C-Rich scenario are in Slovakia and the Czech Republic, + 54.8 tonnes/ha and +43 tonnes/ha, respectively for the C-Rich scenario. Finland and Sweden on the other hand have very minor relative gains of less than 6 tonnes/ha, because in these Member States the share of UAA is very small as compared to forest area. **At the EU level an increase of the afforestation rate by 2% compared to business as usual would result in a 10% increase in carbon stock levels by 2030**.

In a second analysis we weighted the total SOC stock change for the entire forest area per Member State. The average relative change of SOC stock for EU-27 is +4 tonnes/ha for the BAU 2030 scenario, compared to + 5.7 tonnes/ha for the C-Rich scenario (Figure 21). The highest SOC stock gains for the C-Rich scenario are in Ireland (+73.1 tonnes/ha), UK (+33.5 tonnes/ha), and Bulgaria (+21.9 tonnes/ha) reflecting in all three cases a predominant increase in forested area. In Italy and Hungary, + 11 tonnes/ha and +10 tonnes/ha, respectively for the C-Rich scenario rates are much lower and reflect moderate rates of forest area change. Finland and Sweden on the other hand have very minor gains resulting in less than 0.2 tonnes C/ha, because in these Member States the share of UAA is very small as compared to forest area. The impact of both scenario options in Belgium are negative because the current forest area trends are decreasing and therefore result in a decrease in soil carbon stock.

The relative importance of converting UAA to forest areas is small when weighted by forest area (Figure 21), because the forest area is comparatively large in most Member States, with the exception of Ireland. When weighted by land use change area (Figure 20) the relative contribution of reforestation to all land use changes becomes apparent. Weighting by land use change area provides for a sound approach to compare the effect of land use changes on SOC stocks between Member States.

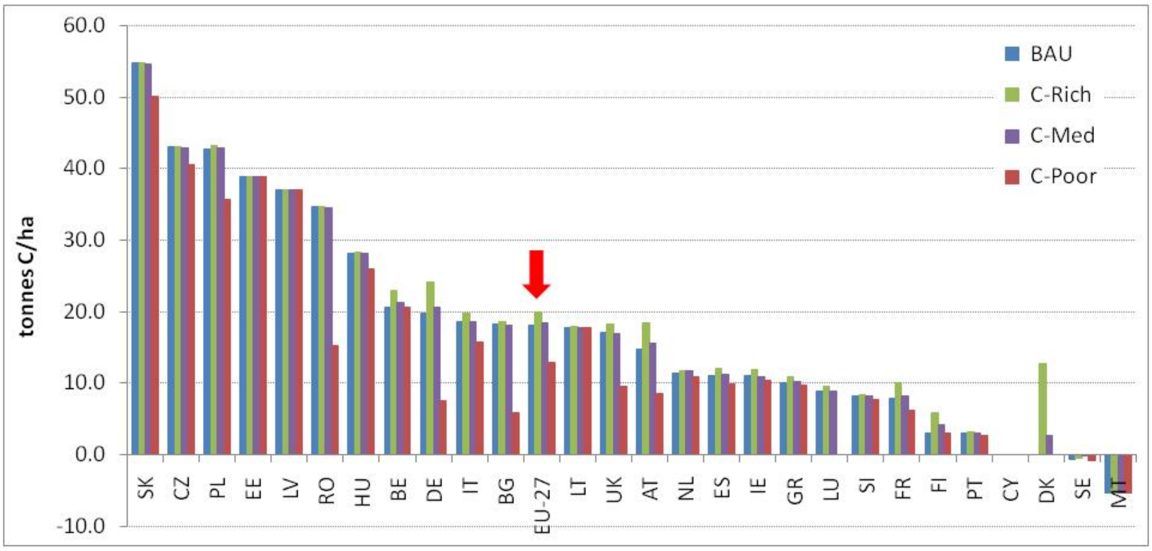



Figure 20 SOC stock changes (in tonnes/ha) due to conversion from arable land to forest, weighted for the total area of land use change under BAU, C-Poor, C-medium and C-rich scenarios.

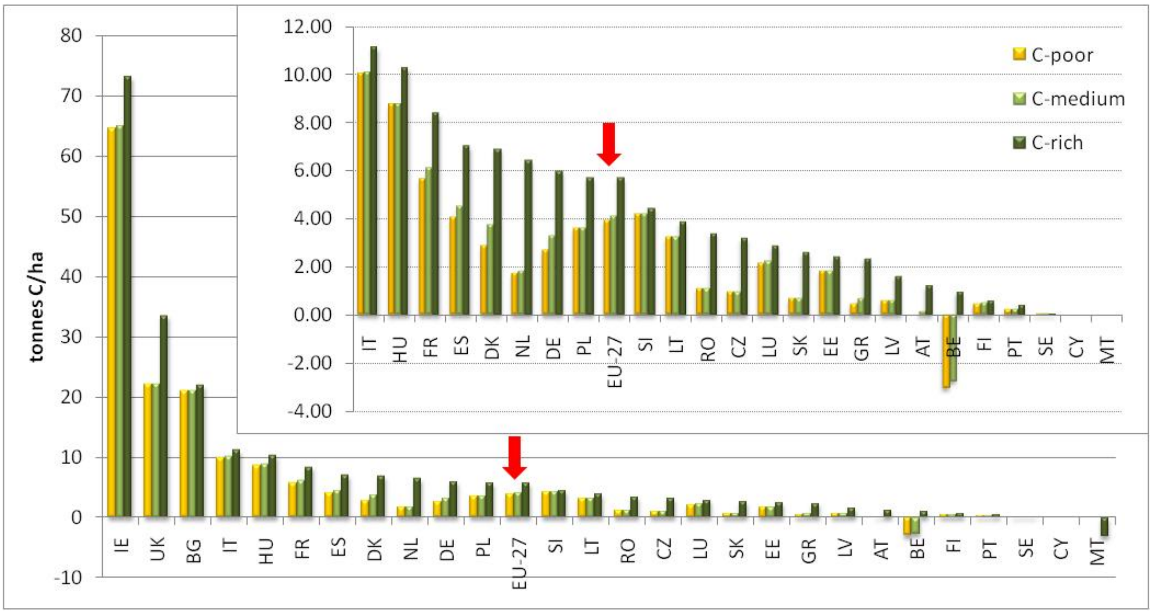



Figure 21 SOC stock changes (in tonnes/ha) due to conversion from arable land to forest, weighted for the entire forest area under C-poor, C-medium, C-rich scenarios.

# CHAPTER 7 USE OF CROP RESIDUES AND STRAW

## 7.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Agriculture – resource management issues*** | | | | | |
| *Use of crop residues and straw* | 10% crop residues and straw to bio-energy | 10% crop residues and straw to bio-energy | 10% crop residues and straw to bio-energy | **30% crop residues and straw to bio-energy** | **50% crop residues and straw to bio-energy** |

The scenario on the use of crop residues and straw examines the impact of the use and management of crop residues and straw on soil organic carbon fluxes. Depending on the climatic conditions and soil type, the amount of crop residue produced may vary from place to place and over time. The scenario analyses the impact of crop residue management for cereal, sugar beet, oilseed and grass, as these represent the major crops that can are cultivated for bio-energy production.

## 7.2 Scenario Approach and Method

LUMOCAP yield data are used to estimate the total above and below-ground biomass. Subsequently the biomass is modelled for cereal, sugar beet, oilseed and grass that enters the soil in the form of agricultural crop residue. The baseline year is set to 2000 and projections are made until 2030, when climate change and further technological developments are assumed to affect yield.

In the C-Rich, C-Medium and BAU scenario options 10% of the crop residues are assumed to be used for bio-energy. In the C-Low scenario and in the C-Poor scenario options, 30% and 50% of the residues are harvested, respectively; an extreme scenario option of all crop residues harvested (100%) is also assessed. In each case it is assumed that a minimum organic matter standard is maintained through the incorporation of roots, stubble and chaff or pods into the soil. For sugar beet only fine roots remain in the soil under all scenarios. The following scenarios are carried out for cereal, sugar beet, oilseed and grass:

1. Baseline: year 2000 – 0% residues harvested
2. BAU: year 2030 - 10% residues harvested
3. C-Low: year 2030 - 30% residues harvested
4. C-Poor: year 2030 - 50% residues harvested
5. Worst case: year 2030 - 100% residues harvested

The following steps are taken in the analysis:

1. LUMOCAP yield data are used to estimate the total above and below-ground biomass for cereal, sugar beet, oilseed and grass for the baseline 2000 and 2030;
2. REGSOM uses crop harvest indices (HI) to determine the above-ground crop biomass and crop root:shoot ratios (RSR) to determine below-ground crop biomass. The total crop biomass generated by a crop equals the above ground plus the below-ground crop biomass. A humification function is then included to derive humified organic carbon (HOC) per hectare. The relevant parameters for these terms for cereal, sugar beet, oilseed and grass are listed in Table 11.
3. REGSOM derives humified organic carbon (HOC) per hectare maps for each of the specified scenarios to compare the impact of different resource management options on regional soil organic carbon fluxes from crop residues and straw.

### 7.3 Regional organic matter balance for crop residues

Agricultural potential for organic matter sources depends on residue production such as crop residues from annual and perennial crops and manure application. Agriculture in Europe has a high technical potential for biomass production. In particular cereal straw, which is most often returned to the soil in arable cropping systems, is of renewed interest as a potential source of bio-energy. However, the sustainability of this practice which implies systematic removal of above ground biomass of cereal crops is a controversial issue, particularly in soils already having a low soil organic carbon content. We have therefore concentrated on calculating the regional organic matter balance for cereal production across EU-27.

The biomass and organic matter potential from crop production is derived from cultivated area and crop yield. Biomass production data are census data available from the Farm Structure Survey (Eurostat) and have been coupled to land use from the Corine Land Cover map. The year 2000 is used as the baseline by the LUMOCAP model. The LUMOCAP model projects future crop yields for cereals, rice, oilseeds, sugar beet, potatoes, fodder, tobacco, vegetables, grass, fruit, vineyards and olives (for more information see Annex I).

The harvest index (HI in Table 11) of the crop, i.e. the ratio of harvested product such as grain to above-ground crop biomass, determines the amount of above-ground crop residues. The root:shoot ratio (RSR in Table 11) determines the below-ground crop biomass. The total crop biomass generated by a crop equals the sum of the above-ground and below-ground biomass. For tuber crops, the harvest index is the ratio of tuber harvested to the below-ground biomass and the root:shoot ratio determines the above-ground biomass that is regarded as crop residue. Different crop varieties will have different values for HI. The amount of total residue produced will vary from year to year depending on variations in *inter alia* weather, water availability, soil fertility and farming practices. The rooting system, root:shoot ratio and residue management ultimately determine the level of agricultural crop residue that can be left on the field to contribute to soil organic matter. The residue left on the field equals the total crop biomass, both above ground and below ground, minus the harvested products. For cereals the harvested products may be grain and straw. The residues can be calculated using the harvest index, the root:shoot ratio and the yield.

$$H = R.(e^{-Akt})$$

where R is the amount of residue added to the soil, expressed in tonnes C/ha, with 0.5 as C:OM ratio. H is the amount of organic carbon that humifies after one year. k is the decay factor or humification rate (Table 11) and t is year. In subsequent modelling steps k is corrected for temperature, moisture and plant development (*A*); and different decay rates (*k*) are introduced per organic matter compartment.

Table 11 Average crop parameters for organic matter production on agricultural land

| | Crop parameters | | | |
|---|---|---|---|---|
| Crop | Harvest Indices (HI) | Root-Shoot Ratio (RSR in DM) | Humification rate (k in year$^{-1}$) | C:OM |
| Cereal | 0.62 | 0.41 | 0.31 | 0.5 |
| Sugar beet | 0.99 | 14.29 | 0.29 | 0.5 |
| Oilseed | 0.29 | 0.18 | 0.31 | 0.5 |
| Grass | 1.00 | 0.80 | 0.26 | 0.5 |

Large quantities of residues are generated every year by agriculture. Cereals, grass, sugar beet, potatoes and oilseed rape are arable crops that generate considerable amounts of residues. In aggregate, figures of the total amount of residues look very attractive if not staggering. A distinction, however, has to be made between residues remaining in the field and those generated after harvesting and during processing. Field residues occur in smaller quantities, are spread over large(r) areas and remain in the field; examples are stubble, straw, stalks and leaves depending on the crop and the farming practice. Biomass and harvested residues are used for many often site-specific purposes: food, fodder, feedstock, fibre, fuel and further use such as compost production. These purposes are often not mutually exclusive; for example, straw can be used as animal bedding and thereafter as fertiliser. After processing residues can be concentrated which make their further use for compost production and soil amelioration easier.

Agricultural field residues constitute a major part of the total annual production of biomass and the residues are an important source of soil organic matter. The biomass residues that effectively contribute to the soil organic matter stock depend on the effective organic matter or the amount of organic matter left after one year of decay. Roughly this will be 25% of the freshly introduced organic material left on site, but for each crop the decay rates are different. Depending on the decay rates, the effective organic matter will be converted to stable humified organic matter. We converted the humified organic matter to humified organic carbon (HOC) assuming a 50% ratio of carbon to organic matter.

The data on harvest indices, root/shoot ratios and effective organic carbon content are combined with cropping areas and crop production such that the amount of agricultural residues generated can be calculated. The yield data are extracted from the LUMOCAP framework and rely on national and international statistics collected by Eurostat. The LUMOCAP framework, however, does not make a distinction between irrigated and non-irrigated crops such that yield and therefore organic matter input in Mediterranean areas may be overestimated. The meteorological influence in these regions, however, assumes a hot and dry environment for decay of the organic matter input.

We compared two residue management options for cereals, sugar beet and oilseed (described below). For both options we assumed that roots, stubble and chaff (cereal); roots, stubble and pods (oilseed) or fine roots (sugar beet) were left as residue on the field. For permanent grassland we assumed productive grassland with regular harvesting, i.e. yearly removal of biomass. For comparison, we also assumed the hypothetical practice of ploughing all grass biomass into the soil, a practice which is common to temporary grassland. We also assumed that all other residues left the field and did not return in another form (e.g. compost).

The amount of humified organic carbon added to the soil depends first of all on the yields as these directly relate to residue, and secondly on the prevailing climate with cold temperatures and dry moisture regimes being less favourable. For cereal production two extreme management options are presented: straw left as residue on the field and straw harvested (Figure 22, Figure 26). The practice of leaving cereal straw in the field has the potential of doubling the effective organic matter input. Member States with a high production such as BE, NL, IE, DK, UK, DE, LU and FR have a higher potential for sequestering carbon into the soil than the average for EU-27 at 0.86 tonnes/ha for all straw incorporated and at 0.44 tonnes carbon/ha for all straw harvested (Figure 22). The regional distribution further confirms this with southeast UK, Northern France, Northern Belgium, the Netherlands and Northern Germany displaying the largest humified organic carbon input of Europe (Figure 26).

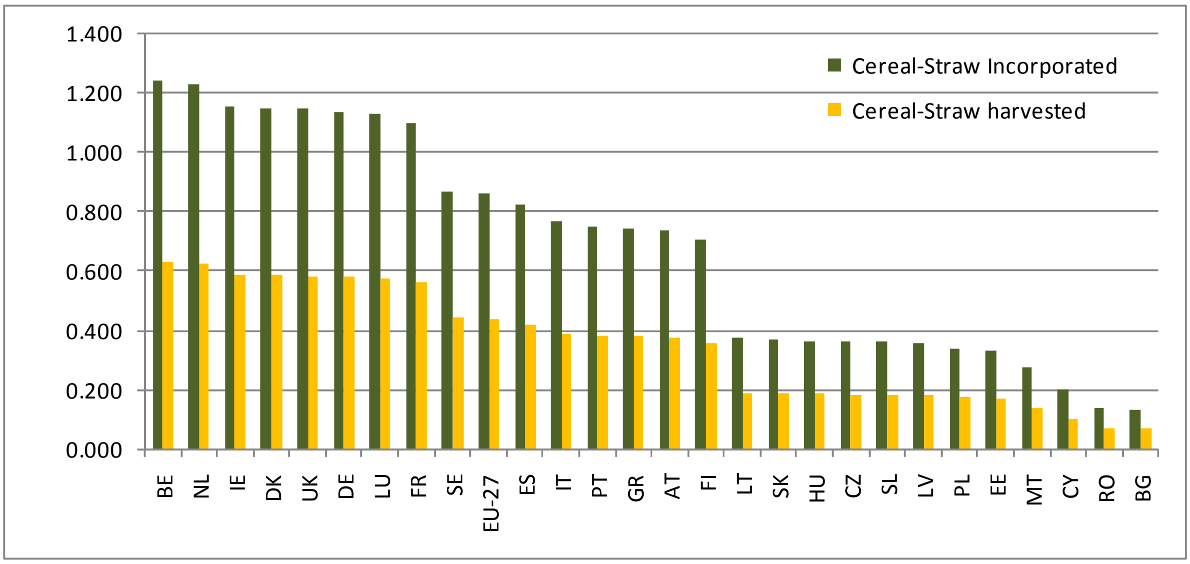



Figure 22 Comparison of average humified organic carbon production (tonnes/ha) under cereal with straw incorporated into the soil (green) and straw harvested (yellow).

Sugar beet only has half of the capacity for adding humified organic carbon (HOC) to the soil as compared to cereal. This is due to a large harvest index and large root:shoot ratio, leaving little residue on the field as compared to cereals. For sugar beet production the option of shoots incorporated into the soil is compared to combined root and shoot harvesting (Figure 23, Figure 27). Although the potential to introduce organic matter into the soil is lower as compared to cereals, residue management has a large impact. Root and shoot harvesting leaves little organic matter after cultivation: with an EU-27 average of 0.046 tonnes HOC/ha a factor 10 less as compared to residue incorporation into the soil (Figure 23), which is also reflected in maps presenting the regional distribution (Figure 27).

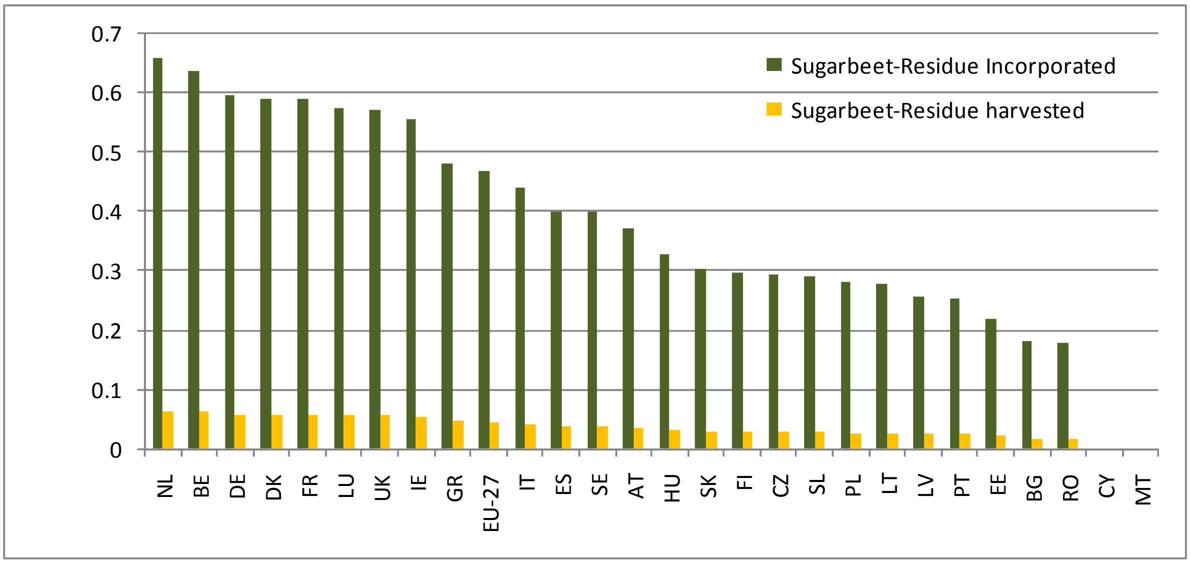



Figure 23 Comparison of average humified organic carbon production (tonnes/ha) under sugar beet with shoot & head incorporated into the soil (green) and shoot & head harvested (brown).

In the case of oilseed the option of straw harvesting is compared to straw incorporation into the soil (Figure 24, Figure 28). The ratio of grain to straw on a weight basis is lower for oilseed than for cereals and therefore results in an average 30% (EU-27) higher flux of humified organic carbon to the soil as compared to cereal. When oilseed straw is incorporated into the soil it results in an average flux of 1.12 tonnes HOC/ha (EU-27 in Figure 24) which is five times higher as compared to the practice of harvesting residues. The lowest fluxes are found in BG and RO, and the highest in BE and IE. The highest fluxes are found in Mid-Germany, Northern France, Northern Belgium, the Netherlands and Southern UK (Figure 28). The reasons for these differences are linked to the yield which in turn are related to the varieties used and the intensity of the cultivation system.

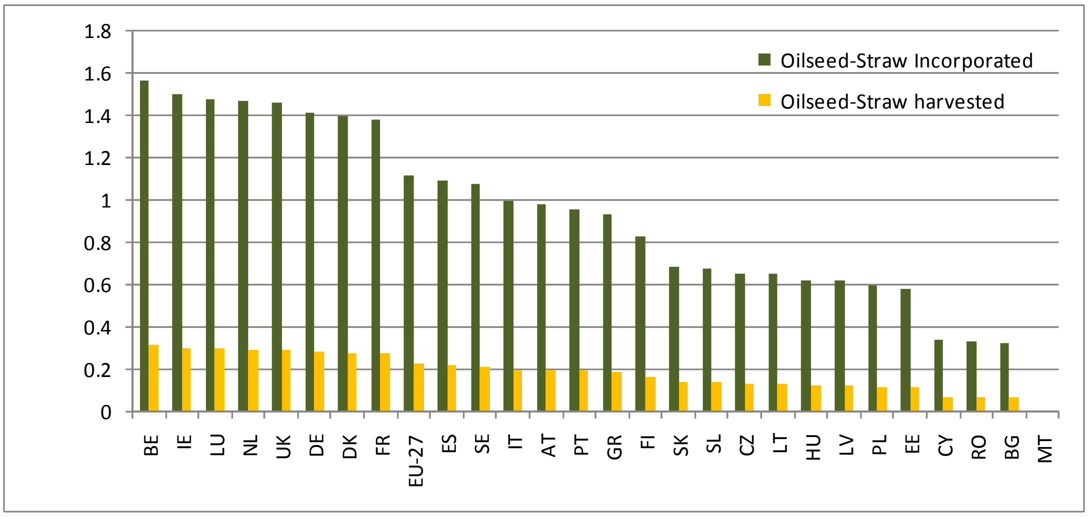



Figure 24 Comparison of average humified organic carbon production (tonnes/ha) under oilseed with straw incorporated into the soil (green) and straw harvested (brown).

Although fluxes under grassland relate to permanent pastures[13] for which regular harvesting is assumed, the case of grass ploughing is also considered here as it relates to a common farming practice for temporary grasslands and provides for a comparison with regular harvesting under permanent grassland (Figure 25, Figure 29). Grass ploughing provides for an instant large flux of organic matter into the soil that results in an average of 1.74 tonnes HOC/ha for EU-27. Regular harvesting of grass biomass ensures an average of 0.43 tonnes HOC/ha added to the soil. Grass ploughing may realise up to 4.5 times the amount of tonnes HOC/ha as compared to regular harvesting. Since the addition provides for an instant fairly substantial flux, temporary grass is common in arable rotations of livestock farms. The carbon sequestration potential of this practice, however, should be further evaluated against the fluxes of subsequent arable crop growth in common rotation schemes and against the protection of the existing soil carbon reserve, which is beyond the scope of the current project.

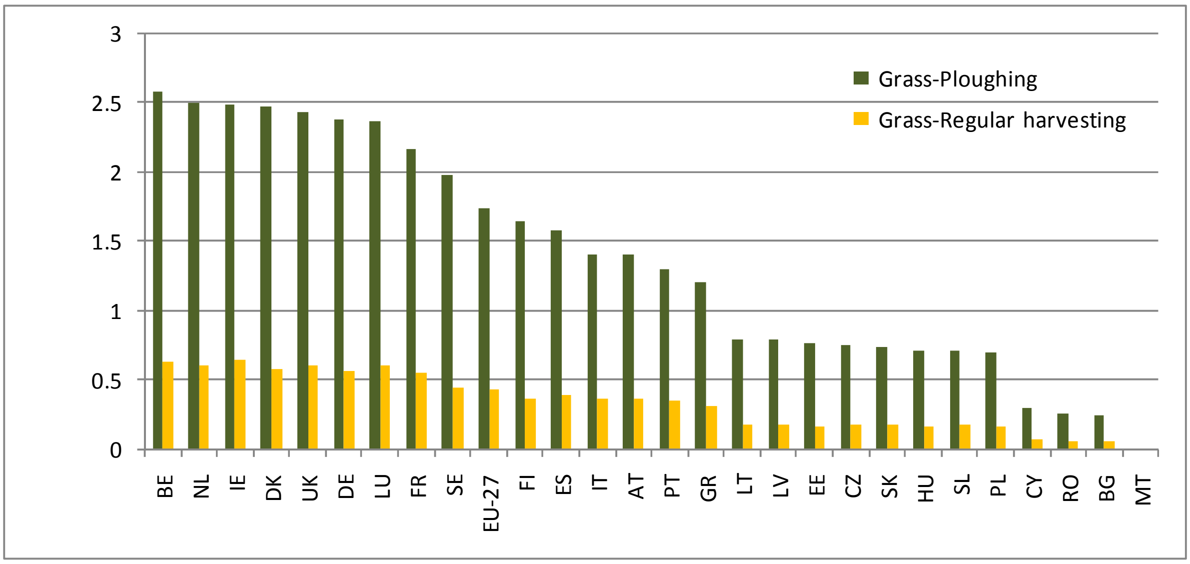



Figure 25 Comparison of average humified organic carbon production (tonnes/ha) under grass with grass incorporated into the soil (green) and grass harvested (yellow).

[13] The definition of permanent pastures is “land that has been under grass for at least 5 years and has not been ploughed for other crops in that time”.

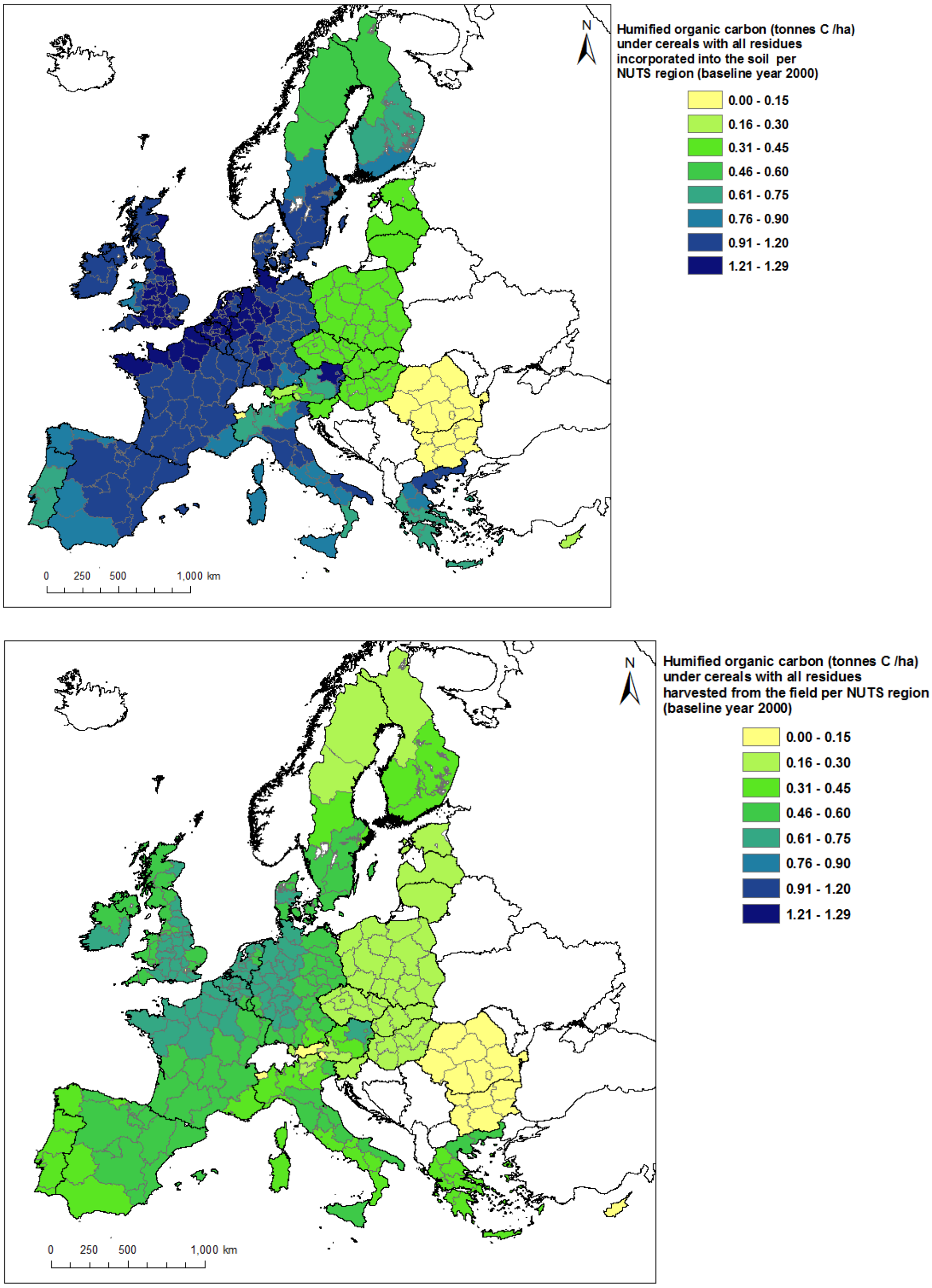
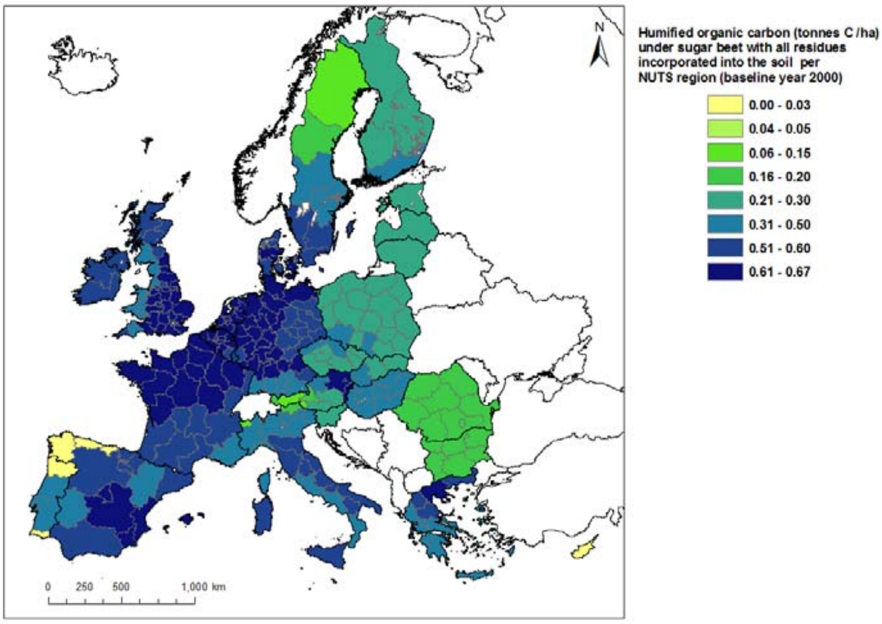



Figure 26 Humified organic carbon production (tonnes/ha) under cereal with straw incorporated into the soil (top) and straw harvested (bottom) across Europe



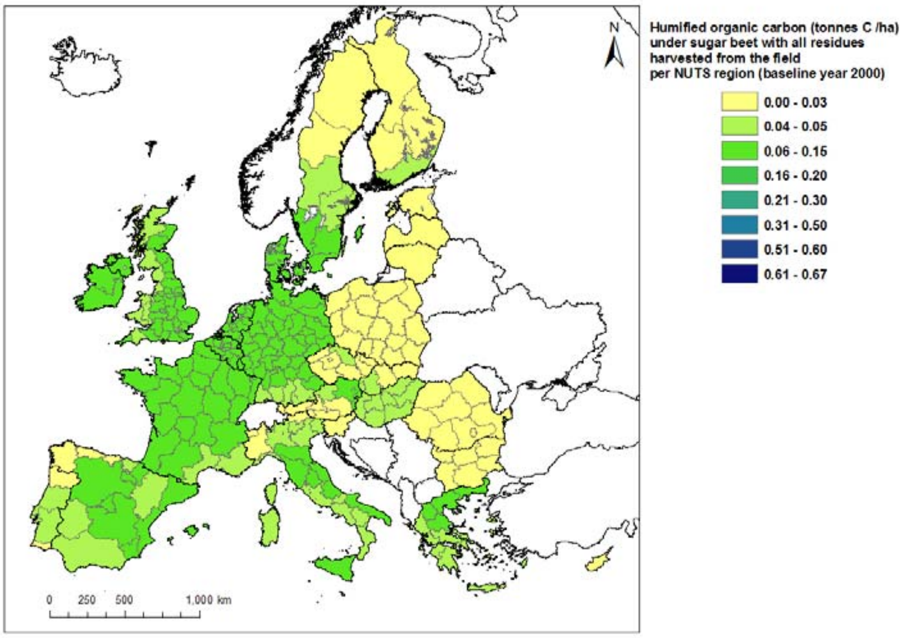
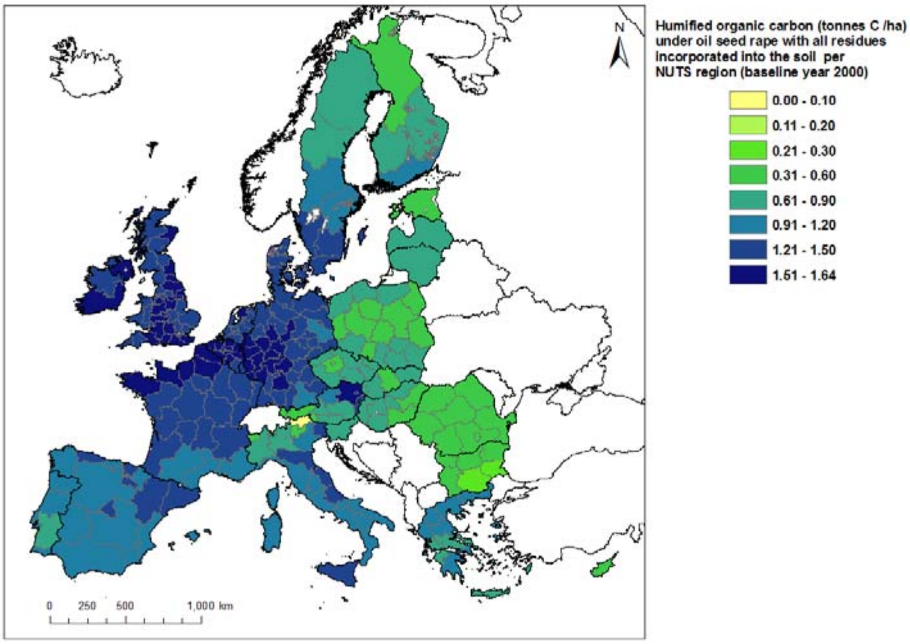




Figure 27 Humified organic carbon production (tonnes/ha) under sugar beet with shoots & heads incorporated into the soil (top) and shoots & heads harvested (bottom) across Europe



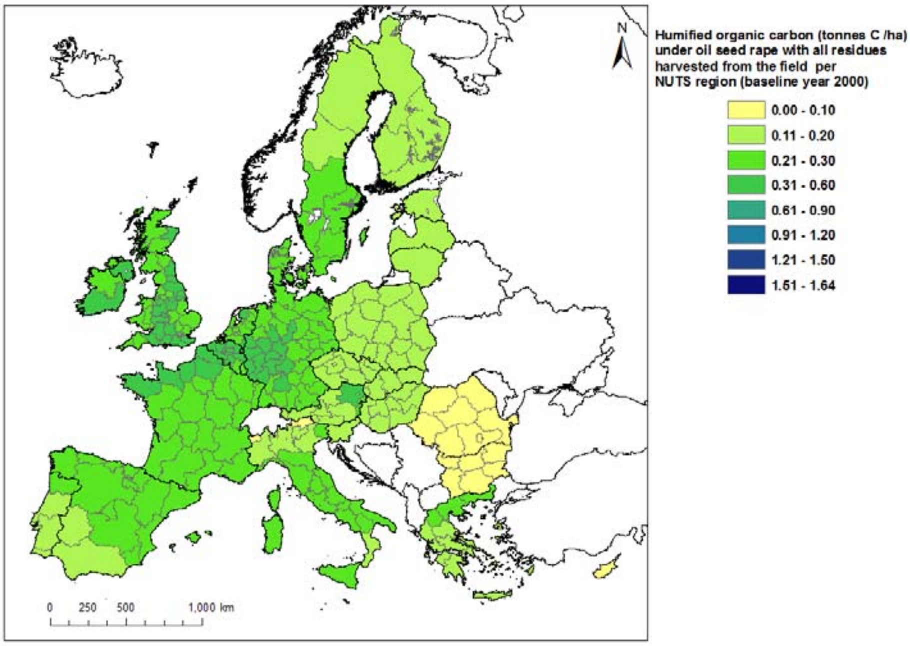
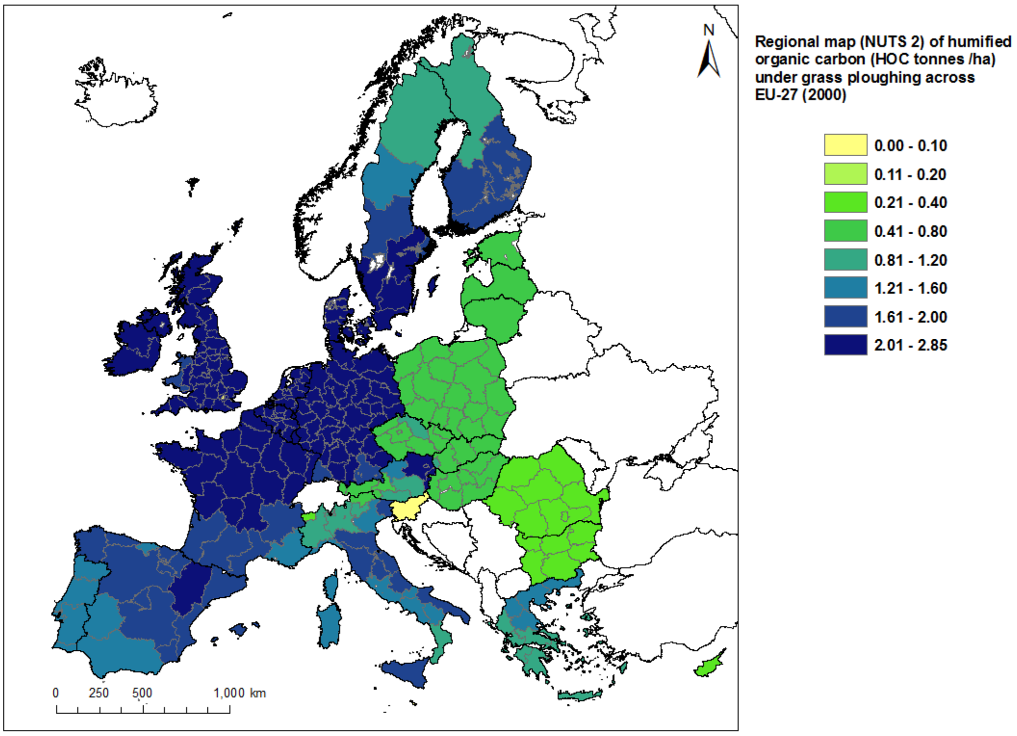




Figure 28 Humified organic carbon production (tonnes/ha) under oilseed with straw incorporated into the soil (top) and straw harvested (bottom) across Europe



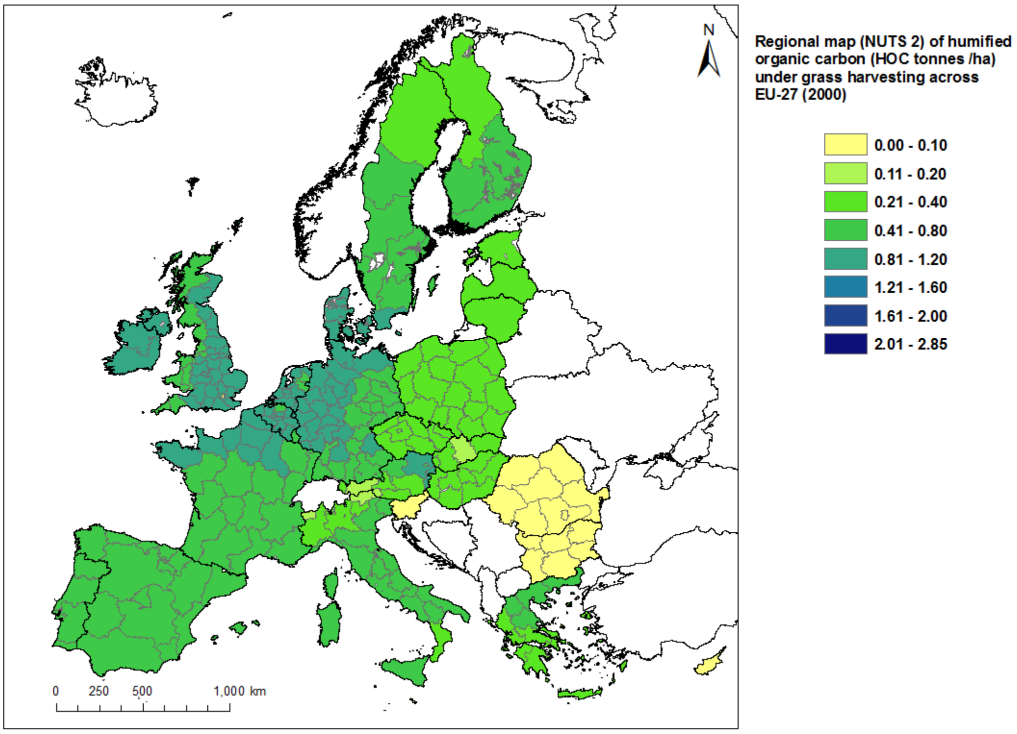




Figure 29 Humified organic carbon production (tonnes/ha) with grass ploughing (top) and grass harvesting (bottom) across Europe.

## 7.4 Results

The **impact of different resource management options on regional soil organic carbon fluxes from crop residues and straw** should be interpreted in view of projected yield, residue management and soil organic matter decay in different bio-geographic zones and climate change. The above-ground biomass provides for the most important source of organic carbon in agricultural soils. A yearly net removal of organic material (harvest) is responsible for decline in soil organic matter. LUMOCAP yields are projected to increase towards 2030 for combined reasons of technological development and $CO_2$ fertilisation effects. Although yield increases due to climate change are still debatable, the yield increases projected by LUMOCAP explain the gap between baseline 2000 and baseline 2030 with respect to addition of humified organic carbon to the soil. Humified organic carbon levels under cereal (Figure 30) are in some Member States higher under a worst case residue harvesting scenario in 2030 as compared to the baseline 2000 scenario with residue incorporated into the soil. Roots, stubble and chaff are directly related to the yield which is assumed to increase substantially in these Member States. Since these cereal residues are assumed to remain on the field and are incorporated into the soil, they contribute to a large portion of incoming humified organic carbon. Although less pronounced a similar effect con be observed for oilseed (Figure 32). Oilseed yields, as projected with LUMOCAP, are assumed to increase at lower rates than cereals. For sugar beet (Figure 31) a progressively smaller amount of humified organic carbon is added to the soil with more residues harvested. The worst case scenario of all residues harvested results in little addition of humified organic carbon to the soil.

The distribution of humified organic carbon added to the soil under different crops and for different scenarios of crop residues harvested shows large differences across the regions of Europe. High cereal production in Western European regions are responsible for higher additions of humified organic carbon to the soil (Figure 33). At the same time warmer and moister climate conditions by 2030 explain the increased ability to assimilate organic material in the form of humified organic carbon into the soil. For grass a comparison is made between grass harvesting and grass ploughing (Figure 34 and Figure 35). Although grass ploughing provides for an instant addition of large quantities of organic material into the soil and hence humified organic carbon, the soil reserve is more easily exposed to organic matter decline. This practices explains the benefit of incorporating grass into rotations as it provides for a large instantaneous flux. Under high productive conditions, this effect is more pronounced.

The projected areas for cereals, oilseed and sugarbeet in 2030, according to the LUMOCAP BAU scenarios, are 65 Mha, 10 Mha and 2 Mha, respectively. **This means that residue management of cereals has a much larger impact on carbon fluxes than oil seed and sugar beet.**

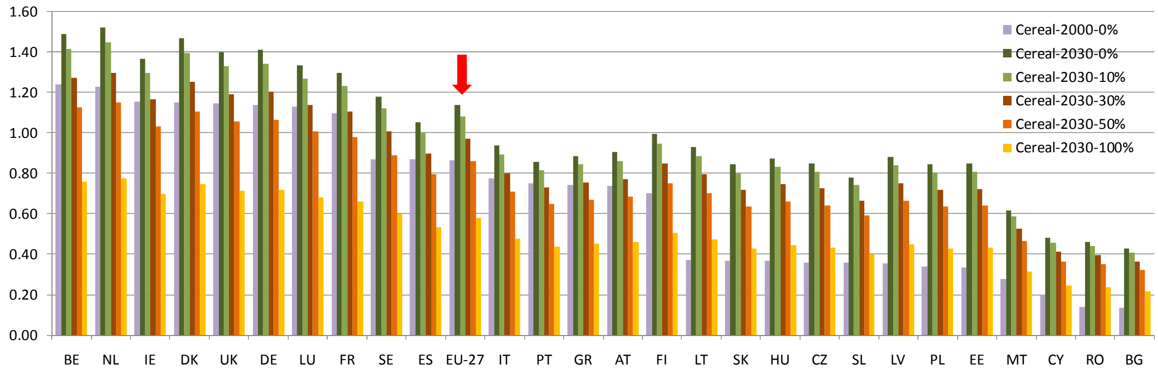



Figure 30 Humified organic carbon (tonnes/ha) under cereal production for different scenarios of residue management (0%, 10%, 30%, 50% and 100% of straw removed; 0% removed equals all residues incorporated into the soil)

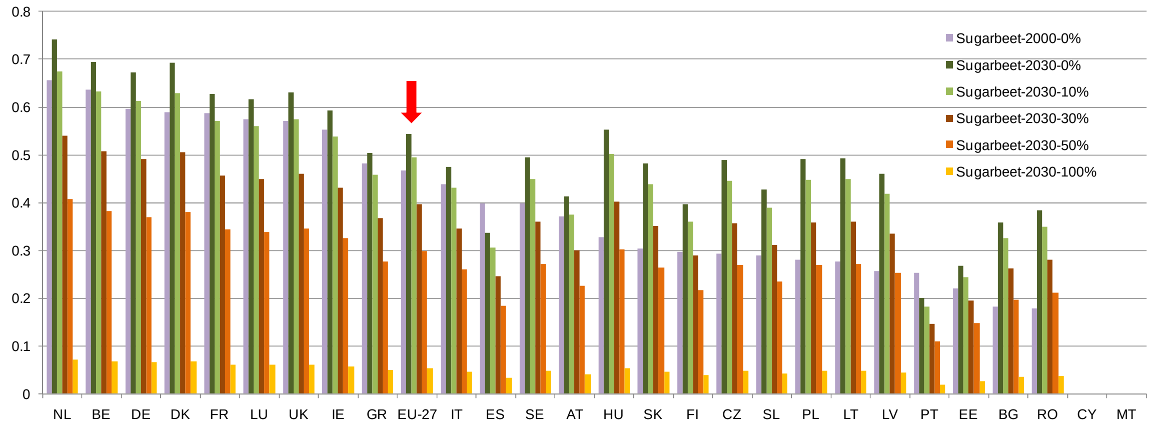



Figure 31 Humified organic carbon (tonnes/ha) under sugar beet production for different scenarios of residue management (0%, 10%, 30%, 50% and 100% of shoots & heads removed; 0% removed equals all residues incorporated into the soil)

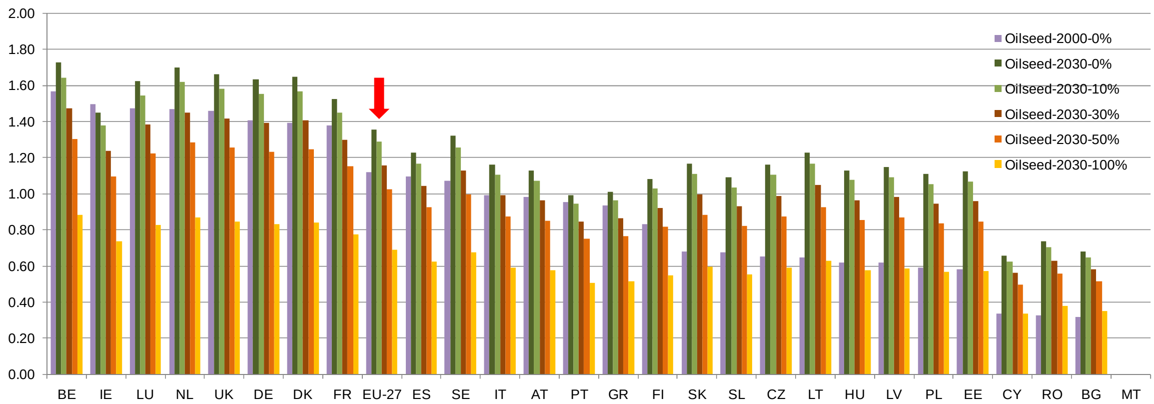



Figure 32 Humified organic carbon (tonnes/ha) under oilseed production for different scenarios of residue management (0%, 10%, 30%, 50% and 100% of straw removed; 0% removed equals all residues incorporated into the soil)

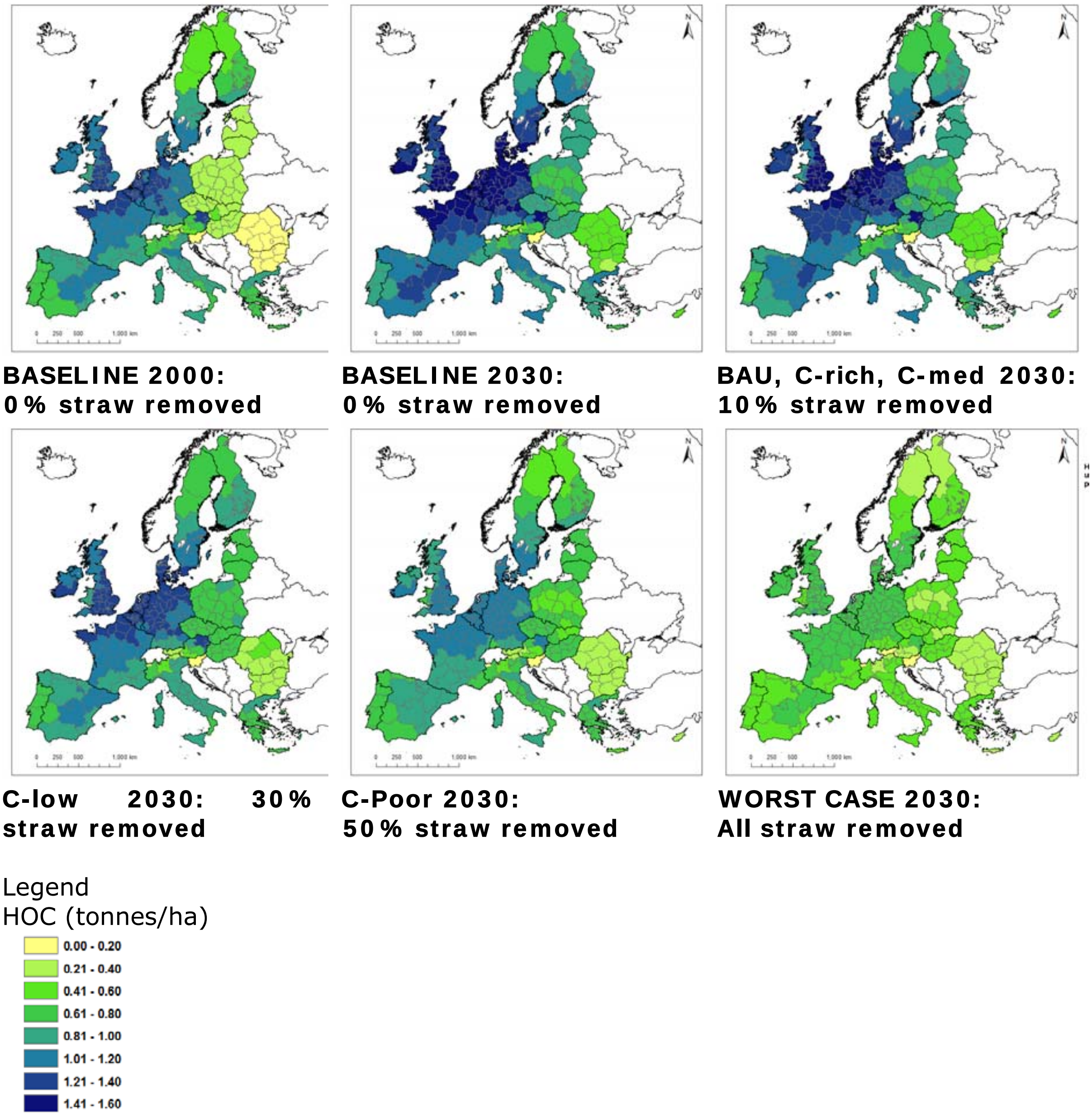
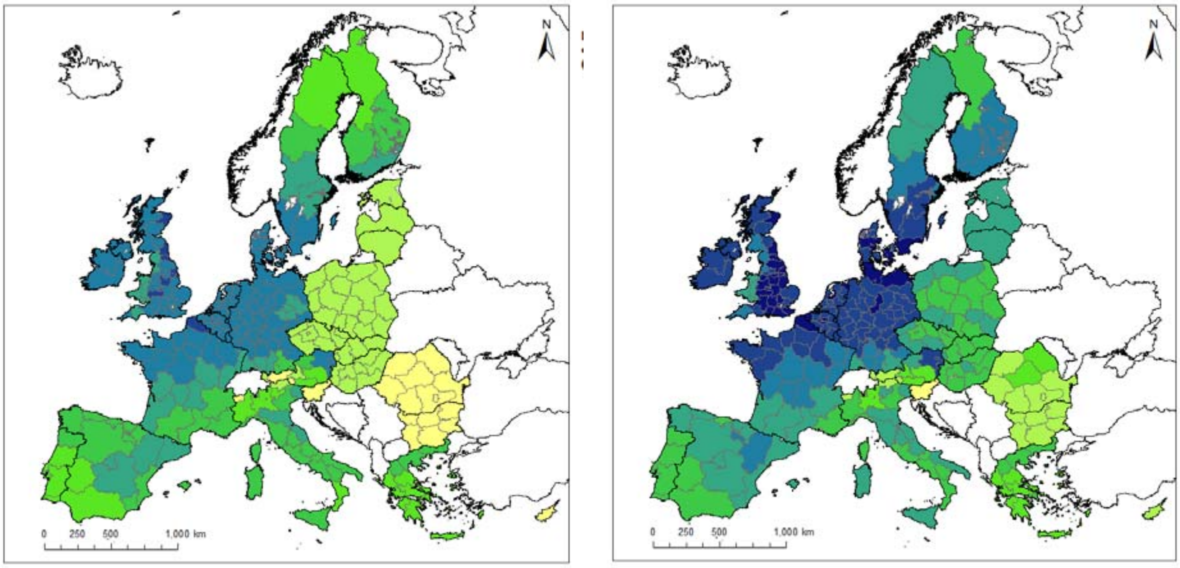
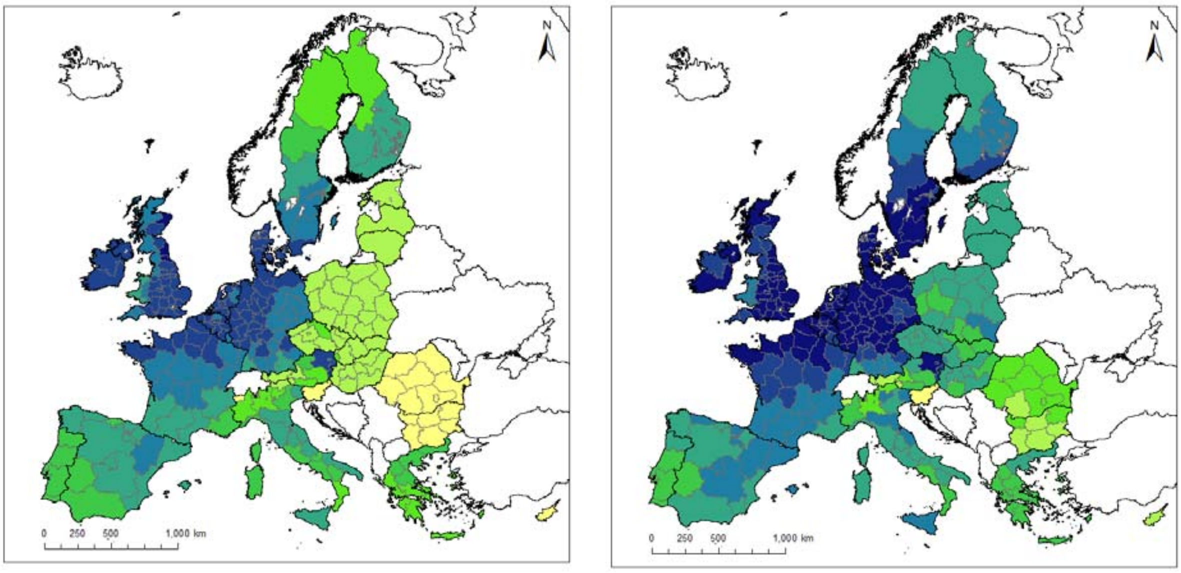



Figure 33 Distribution of humified organic carbon (HOC tonnes/ha) across EU-27 under cereal production with different levels of residue management




HOC 2000 Grass harvested HOC 2030 Grass harvested

Figure 34 Distribution of humified organic carbon (tonnes/ha)across EU-27 under grass with harvest in 2000 (left) and 2030 (right)




HOC 2000 Grass Ploughing HOC 2030 Grass Ploughing

Figure 35 Distribution of humified organic carbon (tonnes/ha)across EU-27 under grass ploughing in 2000 (left) and in 2030 (right)

# CHAPTER 8 USE OF MANURE AND COMPOST

## 8.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Agriculture – resource management issues*** | | | | | |
| *Use of manure and compost* | Current manure and **50% more compost** available for application | Current manure and **25% more compost** available for application | Current manure and compost available for application | **20% manure used for bio-energy** | **40% manure used for bio-energy** |

Both manure and compost are sources of organic matter to improve soils, but also as a source to produce bio-energy. The purpose of the "Use of manure and compost" scenario is therefore to assess the availability of organic carbon for agricultural land if some of the manure produced by farms and compost produced by the urban population is used for energy production instead.

Compost and livestock manure are further examples of fresh organic matter that can be added to the soil for the creation of soil organic matter or used to produce bio-energy. We explore the use of organic matter produced from compost in urban areas and the use of livestock manure to provide additional humified organic carbon to agricultural areas.

## 8.2 Scenario approach and method

Projecting the production of manure to 2030 (BAU 2030) is based on using the projected livestock population for 2030, estimated by the LUMOCAP model, and multiplying these predicted livestock numbers with current excretion coefficients to derive manure production (N Kg), and C:N ratios for different types of manure (see section 8.2.2 for  the calculation method). The potential rates of manure production are therefore determined by the livestock population, N excretion coefficients and agricultural area (UAA).

The potential compost production is based on Eurostat population projections to 2030 and the amount of compost produced per person for each NUTS2 area assumed to be 150 Kg per year (see section 8.2.1 for the calculation method). We adopt the following options for the use of manure and compost scenario:

1. Use current manure rates and 50% more compost than currently available (C-Rich);

2. Use current manure rates and 25% more compost than currently available (C-Medium);
3. Use current manure and compost rates currently available (BAU);
4. Use 20% of manure in 2030 for energy production (C-Low); and,
5. Use 40% of manure in 2030 for energy production (C-Poor).

The following steps are taken in the analysis:

1. Assess the projected trends in livestock manure production and potential compost production;
2. REGSOM derives humified organic carbon (HOC) per hectare maps for each of the specified scenarios to compare the impact of different resource management options on regional soil organic carbon fluxes from livestock manure and potential compost production.

### 8.2.1 Production of organic matter from urban areas

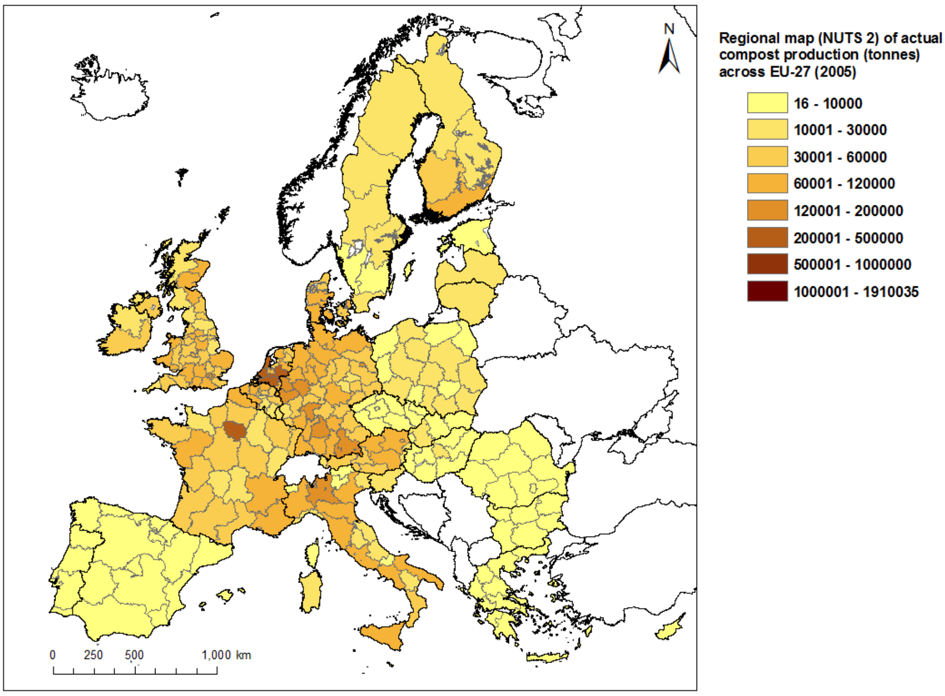
Households and the service sector (schools, hospitals, offices and shops) are potential providers of compost in urban areas. We calculated compost production in urban areas in two different ways:

1. We used statistics on the different contributors to compost production, i.e. **reported compost production**; and,
2. We used estimates of compost production based on production rates per capita, which are multiplied by population data to calculate **potential compost production** in urban areas.

Two types of compost are considered: kitchen compost made from vegetables, fruit and gardening waste (k-compost) and green compost made from made from prunings, branches, grass and leaf litter (G-compost).

### 8.2.2 Reported compost production

Report compost production is based on data provided by the European Compost Network (Table 12, Barth et al., 2008). We weighted the compost production by population to distribute MS values across the NUTS2 areas per Member State. This results in a map of reported compost production estimates at NUTS2 level (Figure 36).




Source: 2005 data from Barth et al., 2008

Figure 36 Regional map of reported compost production from urban areas (2005)

Not all countries have information on compost production (Table 12) so our approach was to fill in the gaps in reported compost production data with the compost production from neighbouring countries.

### 8.2.3 Potential compost production

Potential compost production is calculated on the assumption that 150kg of kitchen-compost and 120 kg of green-compost is produced per person per year (Barth et al., 2008). Coupling this estimate to population data at the NUTS 2 level means that it is possible to derive a regional map of potential compost production for EU-27 (Figure 37).

The population and areas with high population densities contribute most to the Member States compost production. With population increasing across EU-27 and increased urban growth, the potential for compost production will increase (Table 12).

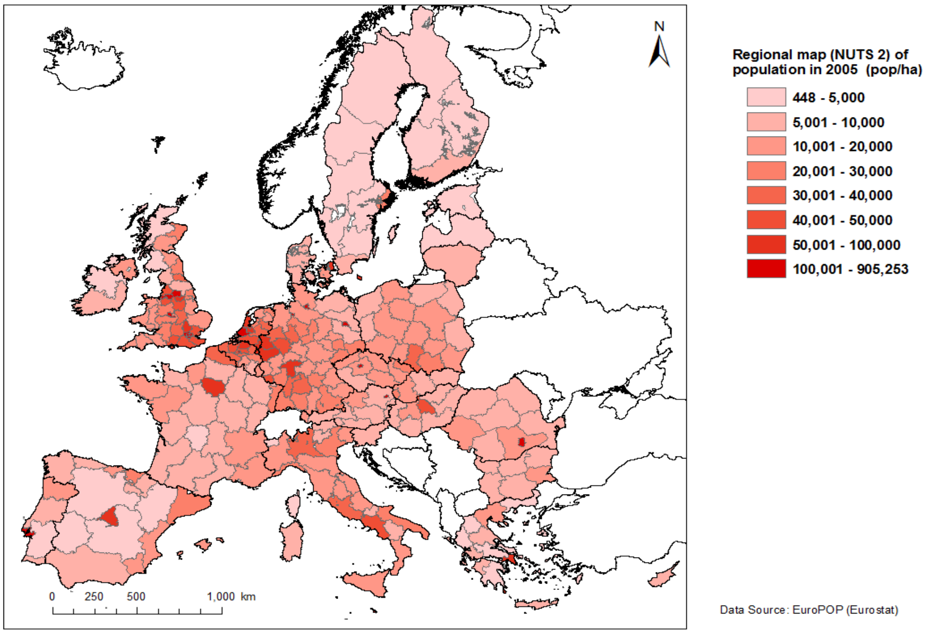



Figure 37 Population map of Europe (2005) at NUTS2 level

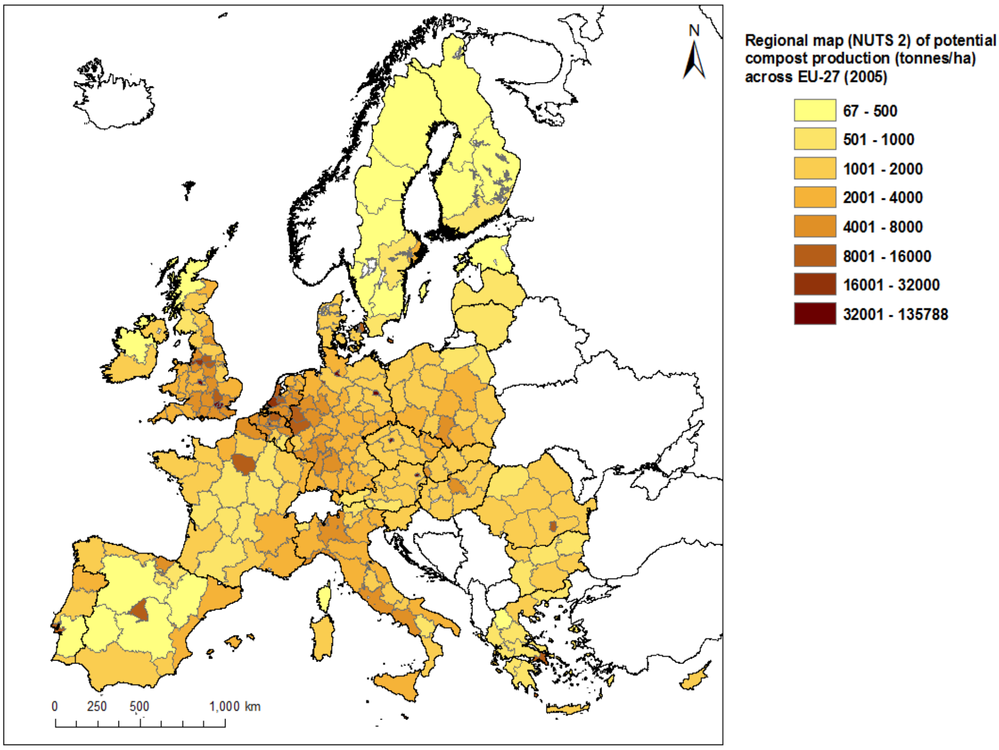


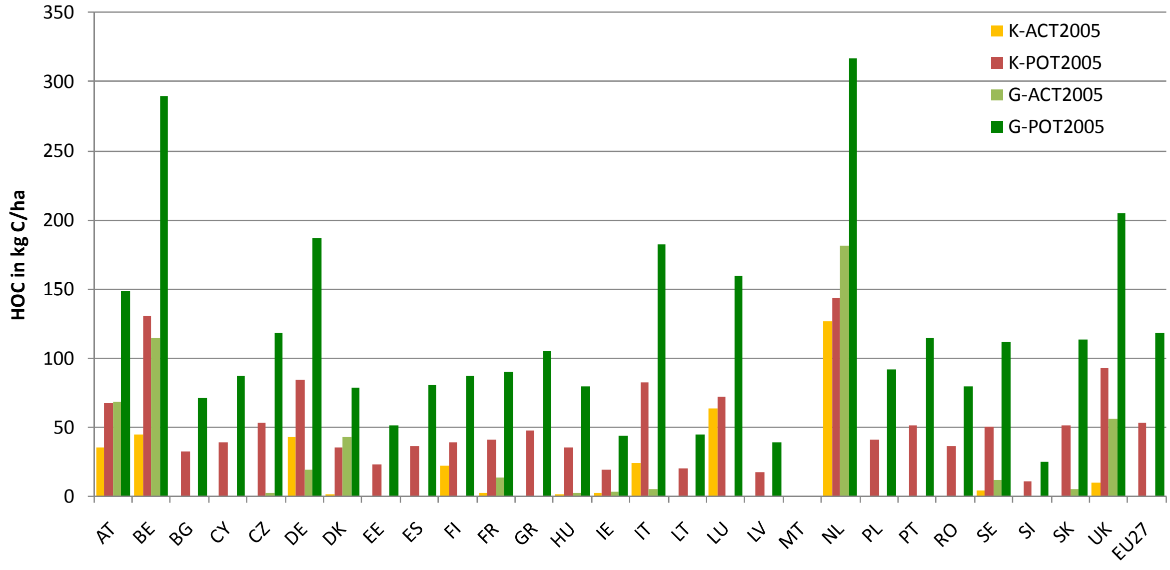

Figure 38 Regional distribution of potential compost production in 2005 for EU-27 based on assumptions by Barth et al. (2008)

Table 12 Reported and calculated potential compost production (tonnes/year) in 2005 across EU-27

| | | Reported compost production (from Barth et al., 2008) | | Calculated compost potential | |
|---|---|---|---|---|---|
| MS | Total | Kitchen compost | Green compost | Kitchen compost | Green compost |
| AT | 416,000 | 218,400 | 380,000 | 411,810 | 823,620 |
| BE | 342,000 | 103,000 | 239,000 | 523,940 | 1,047,880 |
| BG | No data | No data | No data | 386,990 | 773,980 |
| CY | No data | No data | No data | 37,890 | 75,780 |
| CZ | 77,600 | 4,000 | 21,600 | 511,790 | 1,023,580 |
| DE | 2,966,935 | 2,089,139 | 848,486 | 4,125,255 | 8,250,509 |
| DK | 350,000 | 15,200 | 294,800 | 270,970 | 541,940 |
| EE | No data | No data | No data | 67,305 | 134,610 |
| ES | 855,000 | 35,000 | No data | 2,073,355 | 4,146,710 |
| FI | 180,000 | 150,000 | No data | 262,300 | 524,600 |
| FR | 2,490,000 | 170,000 | 920,000 | 3,049,800 | 6,099,600 |
| GR | 8,840 | No data | 840 | 555,205 | 1,110,410 |
| HU | 50,800 | 20,000 | 30,800 | 504,350 | 1,008,700 |
| IE | 100,500 | 25,000 | 34,000 | 207,955 | 415,910 |
| IT | 1,200,000 | 850,000 | 180,000 | 2,930,350 | 5,860,700 |
| LT | No data | No data | No data | 170,715 | 341,430 |
| LU | 20,677 | 20,677 | No data | 23,260 | 46,520 |
| LV | No data | No data | No data | 115,025 | 230,050 |
| MT | No data | No data | No data | 20,175 | 40,350 |
| NL | 1,654,000 | 719,000 | 935,000 | 816,000 | 1,632,000 |
| PL | No data | No data | No data | 1,908,285 | 3,816,570 |
| PT | 29,501 | 2,086 | 1,730 | 503,150 | 1,006,300 |
| RO | No data | No data | No data | 1,081,715 | 2,163,430 |
| SE | 154,800 | 38,800 | 100,000 | 451,475 | 902,950 |
| SI | No data | No data | No data | 100,025 | 200,050 |
| SK | 32,938 | 1,836 | 27,102 | 269,350 | 538,700 |
| UK | 2,036,000 | 316,000 | 1,660,000 | 3,011,425 | 6,022,850 |

#### 8.2.4 Humified Organic Carbon of compost as spread on UAA

Humified organic carbon is calculated with the quite conservative assumption that all compost produced is spread on the entire utilised agricultural area for the year 2005. In addition the humification factor of green-compost is higher compared to kitchen-compost. This explains the factor 2 difference between HOC (kg/ha) between potential green-compost and potential kitchen-compost spread over UAA (Figure 39). Member States such as Belgium and the Netherlands have potential green composting rates of around 300 kg/ha, since they are heavily populated and have a relatively small UAA when expressed per person. In comparison the EU-27 average is 120 kg/ha.




Data source: UAA data from Eurostat and reported compost production from ECN (Barth et al., 2008). Malta not reported as the UAA is very low.

Figure 39 Humified organic carbon (in kg/ha UAA) from actual and potential Kitchen (K-) and Green (G-) compost in 2005

### 8.2.5 Organic carbon flux of livestock manure

The regional distribution of livestock units per area shows a high density in certain regions of Belgium, the Netherlands, Denmark, Italy, Spain and Slovakia (Figure 40). In the majority of these regions there exists a high level of farm specialisation and there is a trend towards larger farms. This in turn causes the production of huge amounts of manure and slurry in one region, whereas there may be a demand in neighbouring regions. Transportation of manure and slurry between the regions and Member States is not considered. The livestock units per Member State have an important influence on potential input of manure to the land as a source of carbon. In common farming practices, however, they are used as fertiliser because of their high nitrogen content. Therefore care has to be taken not to use excess manure as this can lead to eutrophication. Limitations set by the Nitrates Directive are taken into account when calculating carbon balances at NUTS2 level – whereby manure surpluses above the allowed application rates (including derogation levels) are assumed to be temporarily stored or used for bioenergy production.

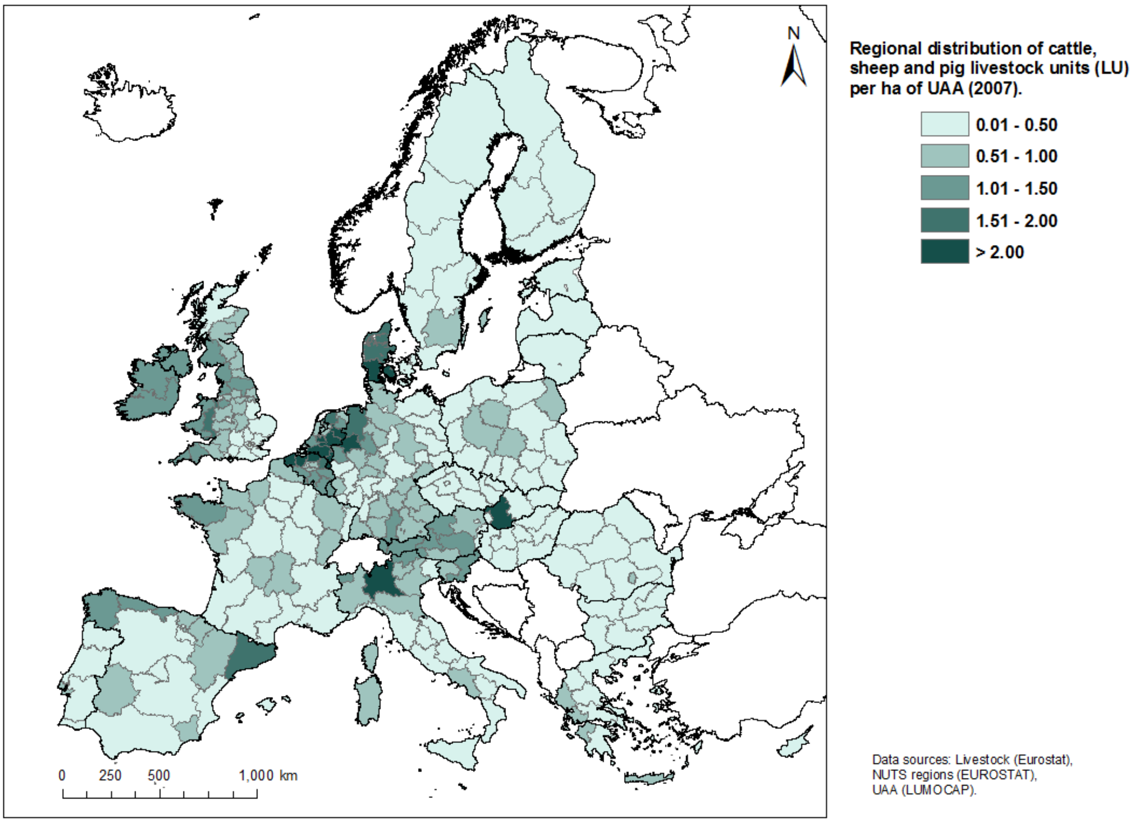



Figure 40 Regional distribution of cattle, sheep and pig livestock units (LU) per ha of UAA

The application of livestock manure, and in particular farmyard manure, to agricultural land is a source of carbon for increasing soil organic matter in the soil profile. Currently there is no statistical data available on livestock manure applications, in terms of manure type, storage practices, C:N ratios, other uses of livestock manure, field application rates and field application methods. In addition the manure application rates reported under the Nitrates Directive are not complete and have not been verified. The approach therefore to calculate the Humified Organic Carbon (HOC) due to the application of livestock manure on farm areas is based on:

- Livestock Manure (N kg) = Livestock population (bovines, dairy, pigs, sheep and goats, poultry, horses) * N content per livestock category (N Kg);
- Livestock Manure (C kg/ha) = Livestock Manure (N kg) * C:N ratio for different types of manure / Utilised agricultural area (ha); and,
- Livestock Manure (HOC kg/ha) = Livestock Manure (C kg/ha) * humification coefficient * decay function adjusted for climate.

The data sources used are as follows:

- Livestock population data (bovines, dairy, pigs, sheep and goats, poultry, horses) provided at NUTS 2 level (2005 and 2007) by EUROSTAT;
- Farmland areas is based on Utilised Agriculture Area (UAA) provided at NUTS 2 level (2005 and 2007) by EUROSTAT; and,
- N content per livestock category reported at national level, for EU Member States who are also members of the OECD (2004), downloaded from the OECD website.

The parameters used are as follows:

- C:N ratio for different types of manure based on fresh weight;

- Humification rate to estimate Effective Organic Content; and,
- Decay functions related to temperature and moisture.

Not all MS of EU-27 are members of the OECD – so we assume that N content coefficients per livestock category for:

- Estonia, Latvia, Lithuania are the same as Poland;
- Bulgaria, Romania, Slovenia are the same as the Czech Republic; and,
- Cyprus and Malta are the same as Greece.

N content coefficients can vary between Member States because of differing livestock varieties that have different metabolism and also differing bio-geography. The summary table of N content coefficients (Table 13) indicates that there is a considerable range of N coefficients reported for each livestock category.

Table 13 Mean nitrogen content coefficients, carbon to nitrogen ratios and humification coefficients different livestock categories for EU-27 based on OECD data (2004)

| Livestock Category | Max | Min | Mean | C:N | |
|---|---|---|---|---|---|
| | N Kg | N Kg | N Kg | ratio | UM |
| Bovine <1 year old – males (slurry) | 40.3 | 9.0 | 19.7 | 1.9 | 0.4 |
| Bovine <1 year old – females (slurry) | 29.8 | 15.5 | 23.5 | 1.9 | 0.4 |
| Bovine 1-<2 years – males | 70.4 | 36.0 | 48.3 | 4.7 | 0.4 |
| Bovine 1-<2 years – females | 82.4 | 36.0 | 48.5 | 4.7 | 0.4 |
| Bovine 2 years and older – males | 72.1 | 51.0 | 59.7 | 7.5 | 0.4 |
| Heifers, 2 years and older | 101.0 | 40.0 | 59.1 | 7.5 | 0.4 |
| Dairy cows | 126.2 | 60.0 | 89.1 | 11.2 | 0.5 |
| Other cows, bovine 2 years old and over | 98.0 | 42.0 | 66.1 | 11.2 | 0.5 |
| Pigs - piglets under 20 kg | 4.0 | 1.9 | 2.8 | 3.8 | 0.4 |
| Pigs – others | 15.2 | 9.7 | 12.2 | 5.4 | 0.4 |
| Pigs - breeding sows over 50 kg | 17.5 | 9.7 | 12.5 | 5.4 | 0.4 |
| Sheep / Goat | 23.4 | 7.0 | 11.2 | 10.5 | 0.5 |
| Poultry – broilers & other (per 1000) | 0.6 | 0.2 | 0.4 | 8.2 | 0.5 |
| Laying hens (per 1000) | 0.9 | 0.5 | 0.7 | 4.5 | 0.4 |

In relation to the Nitrates Directive we also assume that farmers do not apply more than 170 N kg/ha. For NUTS regions that are in Nitrate Vulnerable Zones (NVZ) we apply the derogation rates for manure applications as listed in COM(2010)47[14]. The communication states that the Nitrates Directive allows for the possibility for a derogation in respect to the maximum amount of 170 kg nitrogen per hectare per year for livestock manure, provided that it is demonstrated that the Directive's objectives are still achieved and that the derogation is based on objective criteria such as long growing seasons, crops with high nitrogen uptake, high rainfall or soils with a high denitrification capacity. Therefore two NUTS regions in Belgium and five NUTS regions in the Netherlands are allowed manure application rates of 250 N kg/ha.

The assumption is that all the produced livestock manure is applied to utilised agricultural area, with the maximum rate specified as 170 N kg/ha (or according to the derogation rate requested). Furthermore the livestock manure is not used for other

[14] COM(2010)47 On implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2004-2007.

purposes such as producing bio-energy, nor is it transported to another NUTS region and applied on farm areas elsewhere. However, excess manure above 170 N kg/ha (or the derogation rate requested) could be used for bio-energy production or stored. We distinguish between livestock categories for calculating the C content of manure. Table 13 indicates that the C:N ratio is highest for cows (11.2) and lowest for piglets (3.8).

The distribution of livestock manure production in terms of N kg/ha indicates that the major producers and users of livestock manure are the regions of Flanders, Brittany, southern Netherlands and West Denmark (Figure 41). In general southern Europe, Eastern Europe and Northern Europe are the lowest producers of livestock manure (less than 60 N kg/ha) and Western Central Europe are the highest producers of livestock manure (more than 150 N kg/ha).

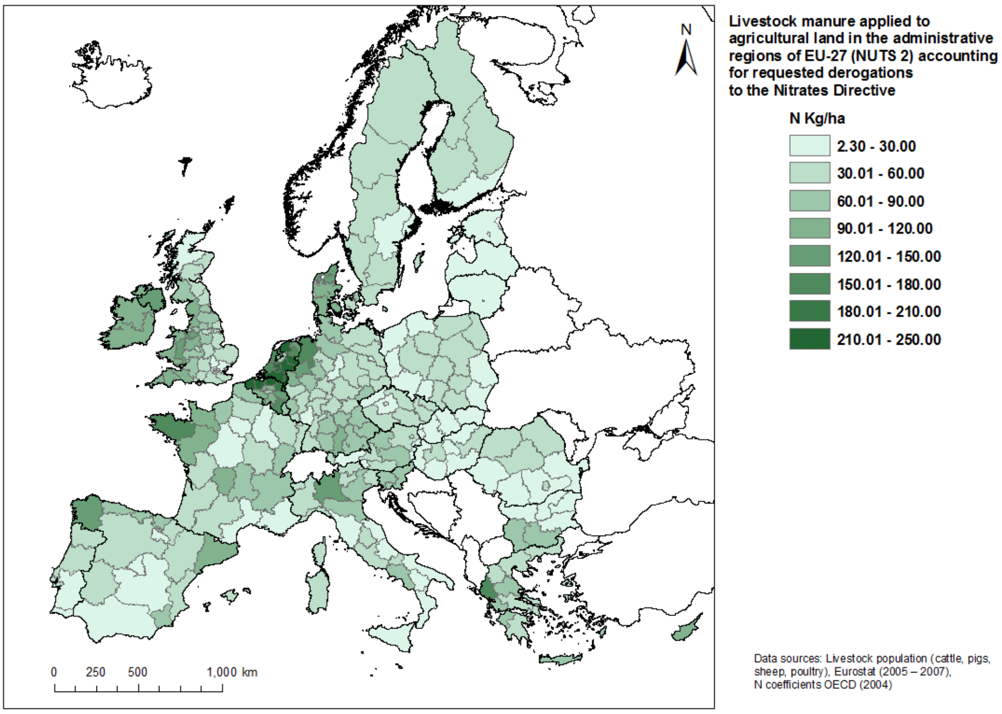



Figure 41 Livestock manure applied to agricultural land (N kg/ha)

Consequently the distribution of humified organic content (tonnes C/ha) follows the same pattern as the levels of livestock manure. The high livestock manure production areas have HOC levels of between 0.5 to 0.73 C tonnes/ha, whereas low livestock manure production areas have HOC levels below 0.2 C tonnes/ha.

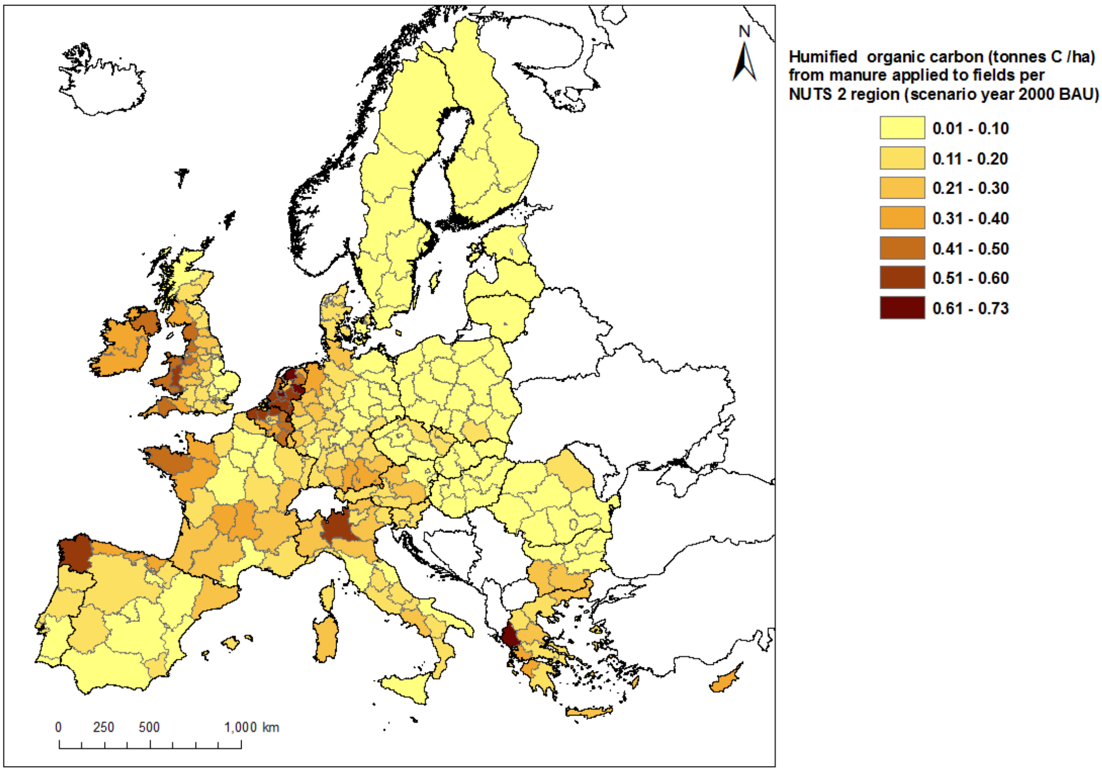



Figure 42 Distribution of humified organic carbon from livestock manure applied to agricultural areas (C tonnes/ha)

## 8.3 Results

### 8.3.1 Projected trends in livestock manure production

As shown in Table 14 and according to LUMOCAP projections there is an increase in pig populations (115%) between 2000 and 2030 in EU-15, but a decline in the other livestock categories. In the EU-10 there is a large increase in sheep (154%) and a large decline in dairy cows (51%) between 2000 and 2030. In the EU-2 there is a general decline in all livestock categories between 2000 and 2030 – the largest decline is in dairy cows (74%). In general the BAU for 2030 will therefore see a reduction in manure that is available for increasing soil organic matter – apart from in the regions with intensive pig farming (ie regions of NW France, N Belgium, NE Spain and W Denmark).

Table 14 Percentage change in livestock population between 2000 and 2030 based on LUMOCAP projections

| **Livestock category** | **EU-15**[15] | **EU-10**[16] | **EU-2**[17] |
|---|---|---|---|
| Cattle population trend | 81% | 67% | 85% |
| Dairy cows population trend | 55% | 51% | 74% |
| Sheep population trend | 96% | 154% | 84% |
| Pig population trend | 115% | 99% | 91% |

### 8.3.2 Projected trends in potential compost production

NUTS areas with a high population density will consequently have much larger values (Figure 43). Compared to 2005 actual compost rates, determined as 2005 compost production divided by UAA, the results of projected compost production reflects a large potential for increased compost production across EU-27 (Figure 44). Obviously in regions with large projected increases in population density the effect is even more pronounced. The location of the bio-waste source (mostly cities), however, is usually not the same as the location of demand (farmers' fields) such that transportation costs may hamper widespread application. In addition, other considerations need to be made such as introduction of contaminants.

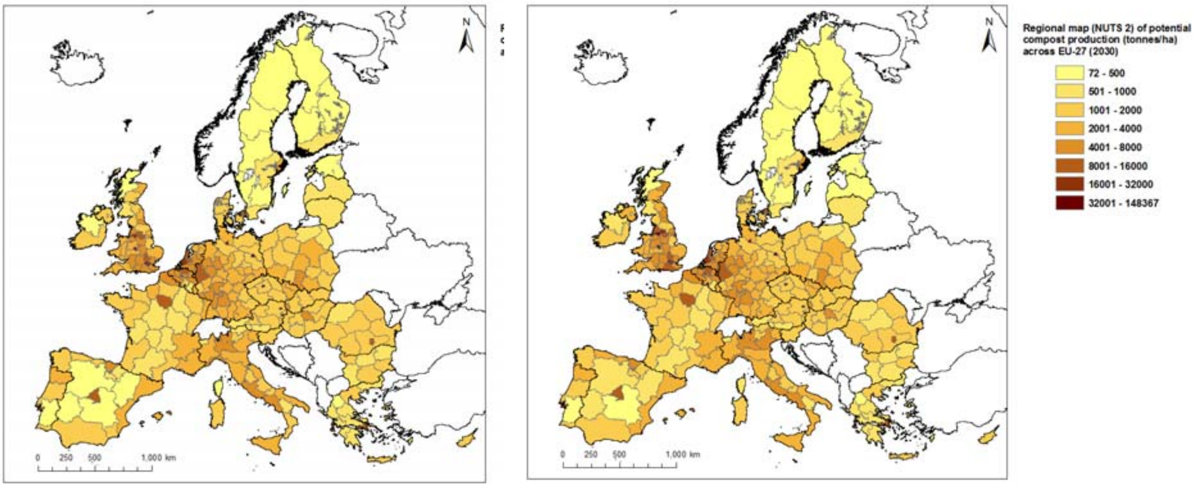



Figure 43 Regional map of potential compost production (tonnes/ha) in 2005 (left) and 2030 (right) across EU-27.

[15] European Union - 15 (EU-15): Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden, United Kingdom.
[16] European Union – 10 (EU-10): Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia and Slovenia.
[17] European Union – 2 (EU-2): Romania, Bulgaria

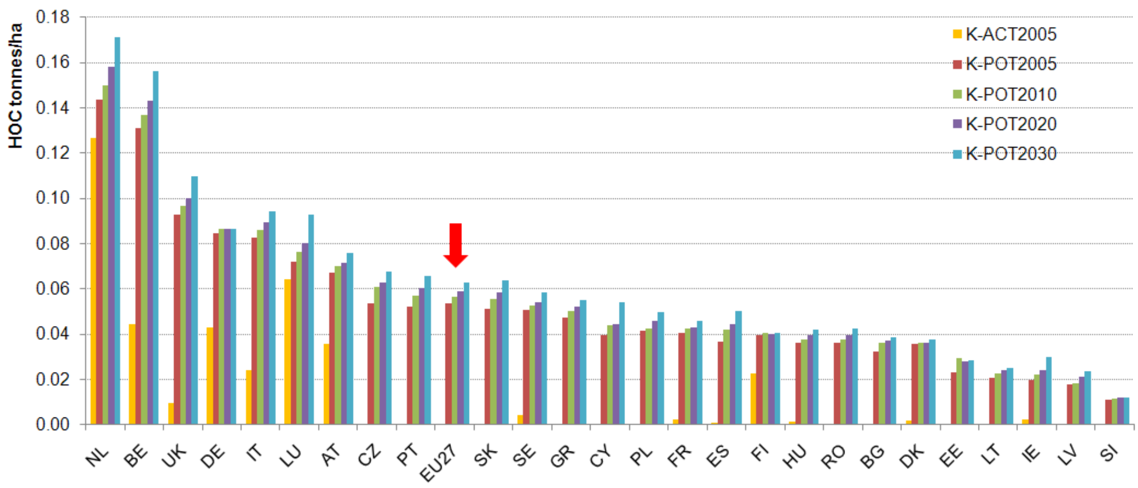



Figure 44 Evolution of potential kitchen compost until 2030 as compared to current actual compost as spread over the Utilised Agricultural Area per Member State

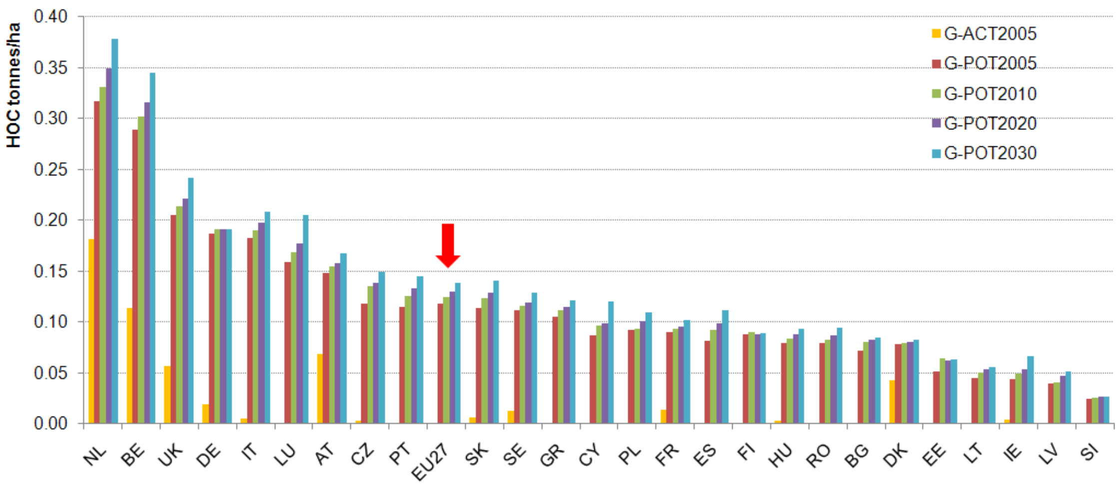



Figure 45 Evolution of potential green compost until 2030 as compared to current actual compost as spread over the Utilised Agricultural Area per Member State

### 8.3.3 Impact of different resource management options on regional soil organic carbon fluxes from livestock manure and potential compost production

For the C-Rich and C-Medium scenarios manure application rates are maintained at the current potential level, whereas 50% more compost and 25% more compost are applied, respectively. The C-Low scenario assumes that 20% of manure produced is used for energy production and the C-Poor scenario assumes that 40% of manure produced is used for bio-energy production, meaning 80% and 60% of livestock manure being made available for application to agricultural land, respectively..

There is a distinct regional effect indicated by the contrasting resource management options for livestock manure (Figure 46 and Figure 47). In the regions where there is intensive pig farming and a large production of livestock manure (e.g. regions of NW France, N Belgium, NE Spain and W Denmark), there is a clear difference between the

C-Poor 2030 and BAU 2030. In these regions there is a clear reduction in the humified organic carbon from livestock manure between BAU 2030 levels (>0.5 tonnes/ha) and C-Poor levels (0.3 to 0.5 tonnes/ha). So livestock manure not used for improving the soil organic carbon balance is used instead to produce bio-energy, therefore reducing HOC levels. In general the practice of using concentrated manure first for bio-energy, and then applying the residues used as a "compost" is beneficial for the soil and reduces the risk of applying excessive nutrients.

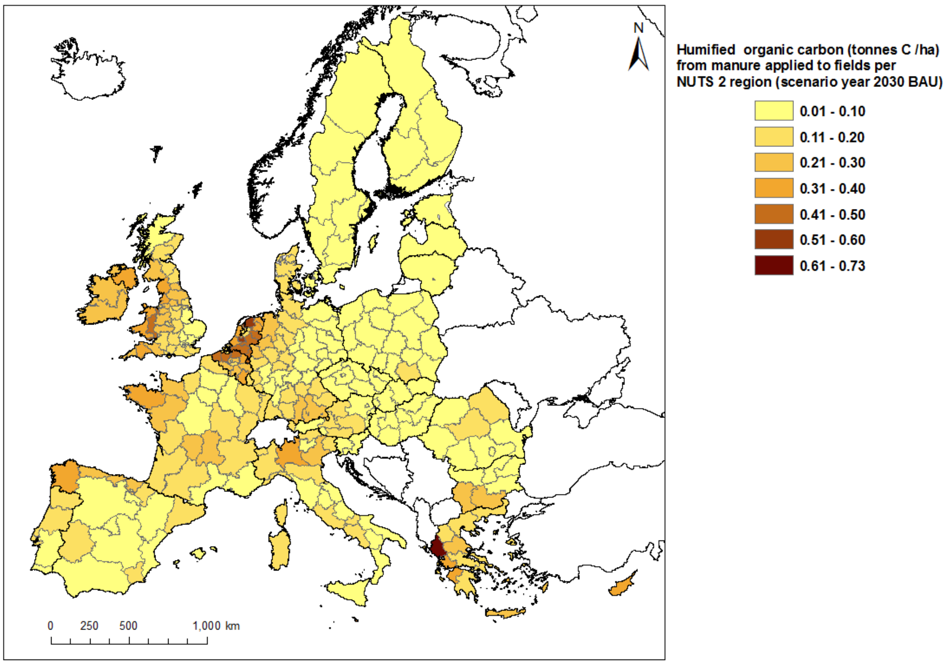



Figure 46 Humified organic carbon (tonnes C/ha) from projected manure production applied to the UAA per NUTS 2 region (BAU 2030)

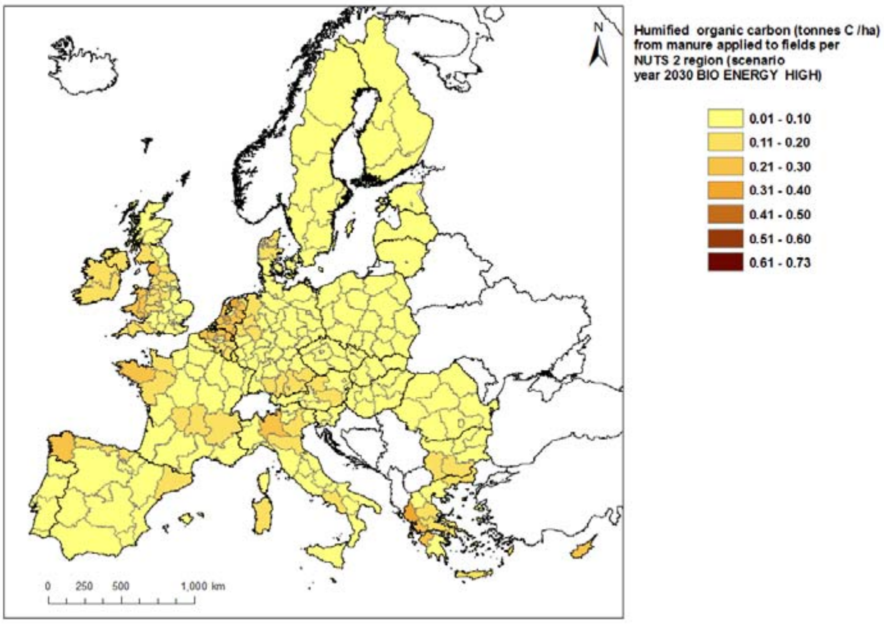



Figure 47 Humified organic carbon (tonnes C/ha) from 60% of projected manure production applied to the UAA per NUTS 2 region (C-Poor 2030)

### 8.3.4 Lifestock manure

At the EU-27 level stable organic carbon input levels drop from 0.19 tonnes/ha to 0.15 tonnes/ha between BASE 2000 and BAU 2030. This 21% reduction is due to the expected reduction in livestock levels between 2000 and 2030. The input levels in 2030 are further reduced to 0.12 and 0.09 tonnes/ha, by using 20% (C-Low) and 40% (C-Poor) of the available manure for bio-energy, respectively. This means that at the EU-27 level C-Low 2030 and C-Poor 2030 scenarios result in 40% and 60% reductions in humified organic carbon levels compared with BAU 2030. For Member States such as the Netherlands, Belgium and Ireland, where livestock production is important, the differences between the scenario options are greater, but the C-Poor 2030 humified organic carbon levels are still double the Base 2000 levels of many Scandinavian and Eastern European Member States. This indicates that manure as a source to provide more organic matter to soils is highly variable across Europe, as well as being highly variable between regions of some of the larger Member States of EU-27 (e.g. France and Spain).

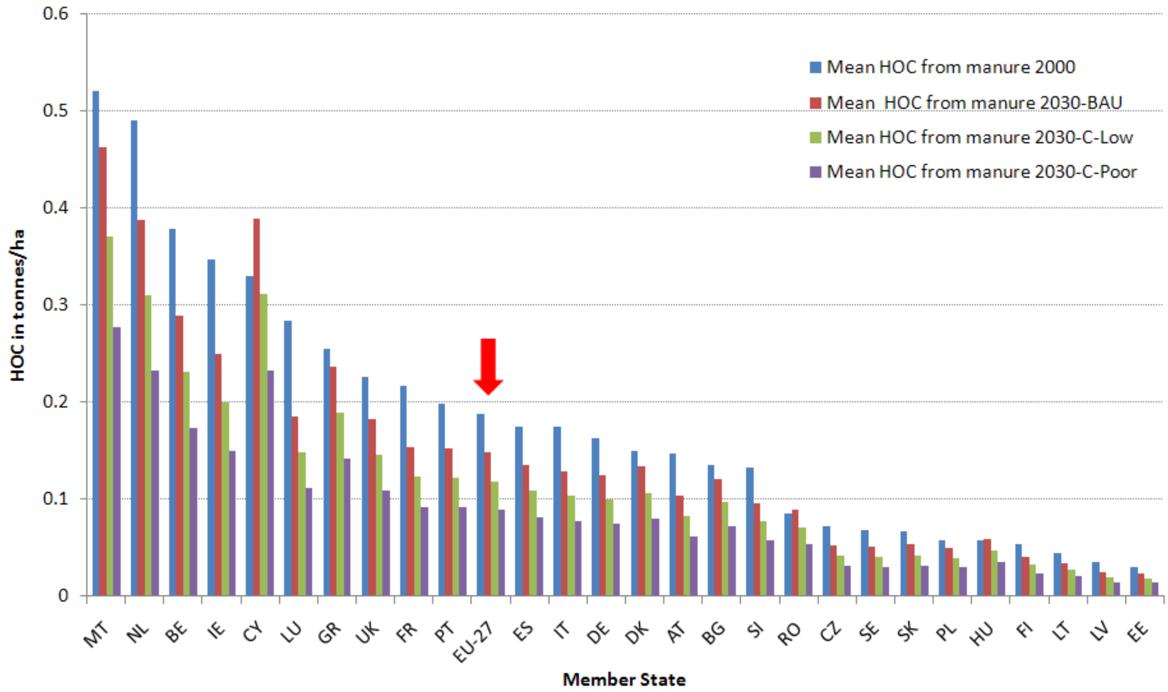



Figure 48 Comparison of mean humified organic carbon (HOC in tonnes/ha) for different manure management options (Base 2000, BAU 2030, C-Low 2030 and C-Poor 2030) for MS

### 8.3.5 Compost

At the EU-27 level the compost scenario options indicate an increase in humified organic carbon from 0.05 tonnes/ha (BASE 2005) to 0.07 tonnes/ha (BAU 2030) (Figure 49), representing an increase of 40%. The levels of humified organic carbon are increased to 0.08 tonnes/ha and to 0.095 tonnes/ha for 25% (C-Medium 2030) and 50% (C-Rich 2030) increases in compost generation, respectively. This represents an increase of 14% and 36%, respectively, compared to BAU 2030. Highly populated Member States, such as the Netherlands and Belgium could reach up to 0.25 and 0.23 HOC (tonnes/ha), respectively, if potential compost generation is increased by 50% (C-Rich 2030), whereas for low populated Member States, such as Estonia, Lithuania, Ireland, Latvia and Slovenia increasing the potential compost generated by 50% would still result in HOC levels of less than 0.05 tonnes/ha.

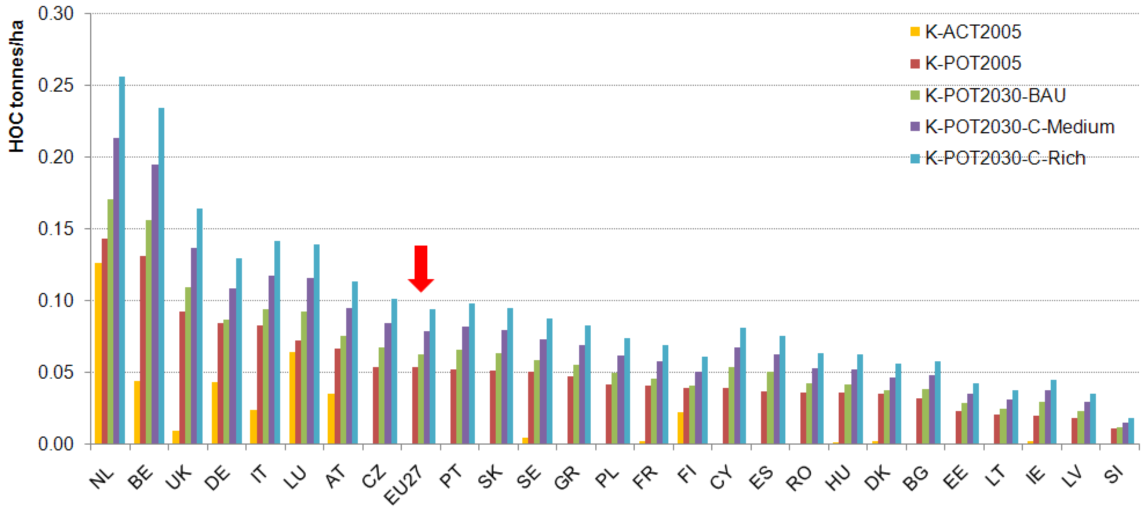



Figure 49 Comparison of mean humified organic carbon (HOC in tonnes/ha) for different kitchen compost management options (Actual 2005, Potential 2005, BAU 2030, C-Medium 2030 and C-Rich 2030) for MS

The distribution of potential Green compost production across Europe (Figure 50) shows a similar pattern to kitchen compost, but the potential for stable carbon assimilation is more or less double. This means that if the organic matter resources are added together, the C-Rich scenario will provide nearly 0.3 tonnes/ha at the EU-27 level. For NL and BE the addition of Kitchen and Green composts for the C-Rich scenario option results in 0.8 and 0.75 tonnes/ha, respectively.

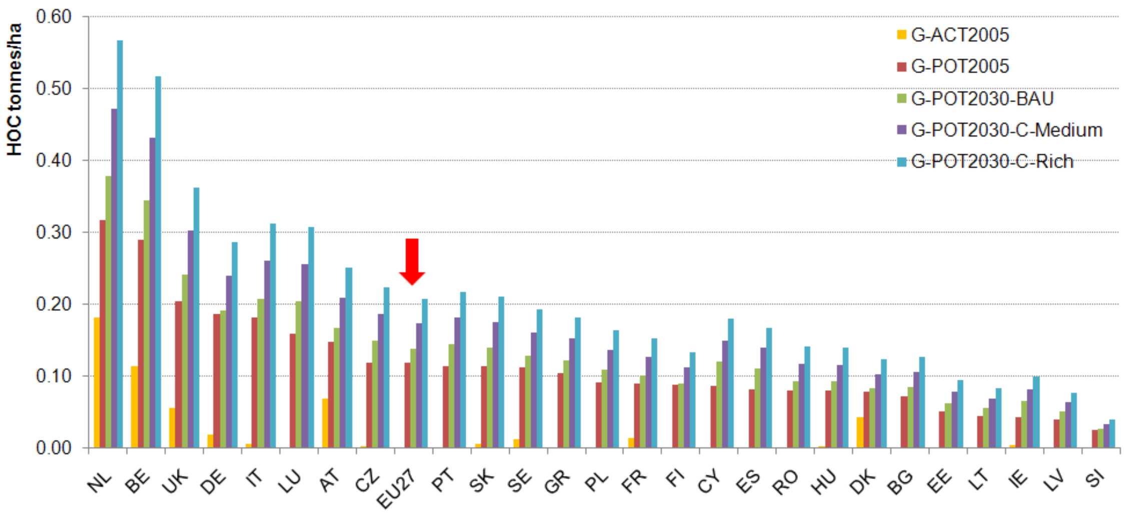



Figure 50 Comparison of mean humified organic carbon (HOC in tonnes/ha) for different green compost management options (Actual 2005, Potential 2005, BAU 2030, C-Medium 2030 and C-Rich 2030) for MS

# CHAPTER 9 USE OF FOREST RESIDUES

## 9.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Forests – resource management issues*** | | | | | |
| *Use of forest residues* | **No forest residues removed for bio-energy** | 10% forest residues removed for bio-energy | 10% forest residues removed for bio-energy | **20% forest residues removed for bio-energy** | **25% forest residues and 10% area stumps removed for bio-energy** |

The general carbon balance under forests depends primarily on forest biomass production. The organic matter that contributes to soil organic carbon depends on the interaction between the two major stocks: the forest biomass (both above and below ground biomass) and the forest soil reserve. Fluxes to and from the soil reserve are therefore influenced by litterfall, natural fellings, deadwood, logging residues, roots and disturbances. In this chapter we focus on estimating the regional carbon fluxes from forests taking into account broad forest categories and different resource management options.

The scenario on the use of forest residues examines the impact of different forest residue management options. The particular interest is to examine whether residues from branches and roundwoods that are removed from forests for bio-energy production have a detrimental effect on soil organic matter levels.

## 9.2 Scenario approach and method

All scenarios represent projections until 2030; the baseline represents the situation in the year 2000. Forest residues considered are roundwood residues from both stemwood and large branches, residues from branches (brash) and stump residues from stumps and coarse roots. Wood residues are defined here as residues from stemwood, large branches and brash but exclude stump, coarse roots and fine roots. Differences between scenarios consider systems with roundwood harvesting only, with roundwood and additional wood residue harvesting, and with roundwood, wood residue and stump harvesting. Officially protected forest areas are excluded from the analysis since they are not commercial accessible.

In the C-Rich 2030 scenario, all forest residues are left on-site. In C-Medium and BAU 2030 10% of forest residue from branches and roundwood is removed for bio-energy purposes in addition to regular roundwood removal. In C-Low 2030 scenario this removal amounts to 20% and in the C-Poor 2030 scenario 25 % of forest brash is being removed in addition to the removal of stumps which is assumed to occur on 10% of the forest surface area. A maximum residue removal scenario (or Worst Case 2030) is also considered since the use of brash and, to a lesser extent, of stumps is rapidly increasing. Despite the poor understanding of the impacts of stump removal and collecting of logging residue on the nutrient cycle, microbiology, restocking, carbon balance, and succession of vegetation, current certification schemes (FSC and PEFC) require that a certain share of biomass must be left on the forest floor for soil protection and biodiversity purposes. In the maximum residue removal scenario (or Worst Case 2030), we assume 70% wood residue removal and 25% stump removal. The analysis assumes that in all scenarios foliage (needles and leaves) is left on the forest floor and is not available for energy use but decays faster in the case of stump removal, because stump removal exposes foliage to more rainfall and temperature changes.

The following scenarios are carried out for both coniferous and broadleaved forests:

1. Baseline year 2000 – 0% residues harvested
2. Baseline, C-Rich: year 2030 - 0% residues harvested
3. C-Medium and BAU: year 2030 – 10% residues harvested
4. C-Low: year 2030 - 20% residues harvested
5. C-Poor: year 2030 - 25% residues harvested plus 10% stumps removal
6. Worst case: year 2030 - 70% residues harvested plus 25% stump removal

The following steps are taken in the analysis:

1. UNECE/TBFRA data are used to estimate the total above and below-ground biomass for broadleaved and coniferous forests for the baseline 2000 and 2030;
2. REGSOM uses allometric parameters to determine the above-ground biomass, the below-ground biomass and the litterfall. Different turn-over and humification functions are subsequently included to derive the flux of humified organic carbon (HOC) to the forest soil per hectare; and,
3. REGSOM derives humified organic carbon (HOC) maps (tonnes/ha) for each of the specified scenarios to compare the impact of different resource management options on regional soil organic carbon fluxes from forest residues.

## 9.3 Production of organic matter from forests

### 9.3.1 Forest biomass production

In order to link biomass production to forest surface area, several databases had to be connected. Land use/cover maps represent data closer to forest and other wood land (FOWL) (Figure 51), whereas inventories and statistics relate to forest area available for wood supply (FAWS) provide information on stocks and potential management practices. For biomass production we relied on national felling statistics and inventories which ultimately enable determining the different components of the forest soil organic carbon balance. Subsequently, the biomass production had to be coupled to the LUMOCAP land cover map. Time series of felling statistics and national inventories mostly apply to forests available for wood supply (FAWS) as the growing stock of these type of forests are monitored closely. The TBFRA (UNECE, 2000), however, reported on forest and other wood land (FOWL) areas, protected forest and other wood land (OWL) areas and the share of FAWS to FOWL for the year 2000. We linked these to the LUMOCAP 2000 baseline map which only includes the class 'forest'. Areal distribution into coniferous and broad leaved forest was based on Corine Land Cover (CLC) and area statistics. The relative share of FAWS to FOWL was linked to CLC areas and baseline LUMOCAP map. All biomass components are modelled with REGSOM; all surface areas are modelled with LUMOCAP.

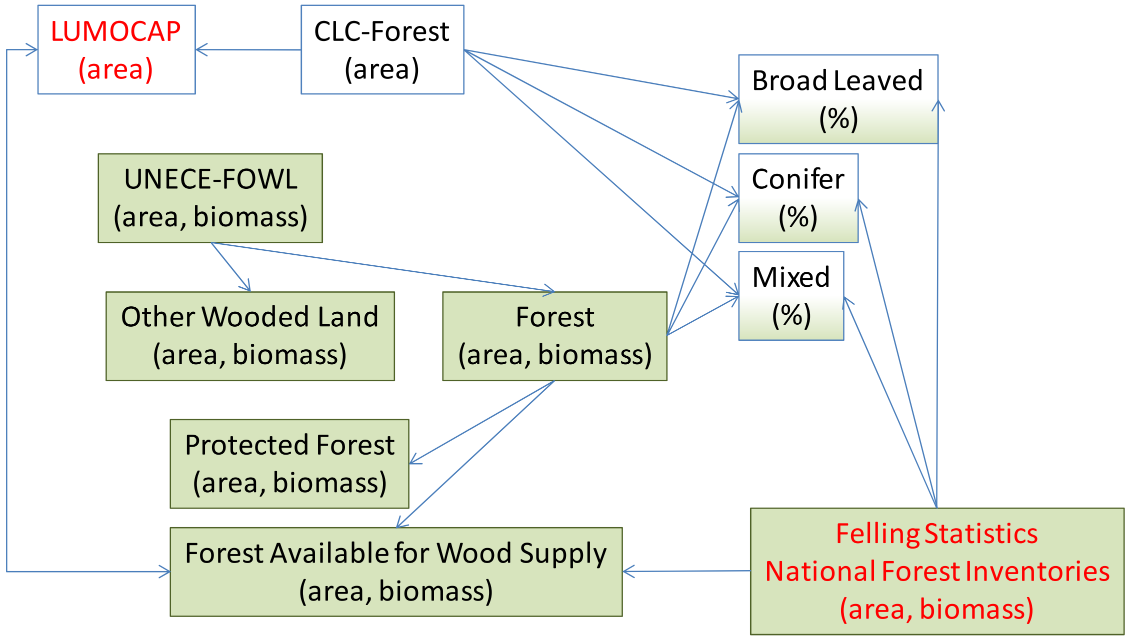



Figure 51 Different available databases for linking forest surface area and biomass production

The data sources used to calculate the baseline organic matter potential for the EU-27 include the 2000 statistics reported in FAOSTAT (2000), EFSOS (European Forest Sector Outlook Study), TBFRA-2000 and FAO's global Forestry Resources Assessment (FRA2005). As much as possible we used baseline statistics from the UNECE TBFRA 2000 report.

The volume of growing stock, annual fellings and removals are important to understand biomass production under forested land. The total biomass equals the standing volume above stump, the stump and the roots (Figure 56). The growing stock volume is the above-stump volume of living trees measured over bark to the tree tops (see Box 7 below). This can be expressed by volume or by weight. The latter requires conversion with the average density of the forest stand. In the statistics databases of UNECE, FAO-FRA and Eurostat the variables are reported in an aggregated form displaying one figure per variable at the Member State level.

All the growing stock and biomass variables, however, are dependent on forest age, species type, environmental growth factors and management type. When reported in aggregated form at the Member State level, the resulting figures display a fairly large range. To illustrate this we analysed the EFISCEN inventory database that is composed of country data per forest type. A forest type is "the forest that can be distinguished according to region, owner class (management level), structure, site class (environmental characteristics) and tree species". The level of detail between different countries varies, as not all have presented their information at all possible levels (e.g. region, owner class, tree species, etc.). For each forest type and age class, the forest area, the total and mean volume, the total annual increment and the current annual increment were retrieved from the EFISCEN Inventory database. Such data are available for all countries which have an even-aged forest structure. Despite the higher level of detail in terms of forest age, species type and management level, the EFISCEN inventory database does not contain all forests in a Member State and therefore represents considerable uncertainty in aggregated form when compared to international statistics.

Based on allometric[18] data in the EFISCEN database, an average standing biomass (m³/ha) was computed per forest type (coniferous and broadleaved) and per country (Figure 52, Figure 53). The difference according to maturity of a forest and management level was not taken into account. In general mature coniferous forests have a higher growing stock than broad leaved forests. We have disaggregated all international statistics to two broad classes: coniferous and broad leaved. A further disaggregation of the standardized national databases to species type is not possible since the current national aggregation of parameters such as wood density or stump/roots to above-ground biomass introduces a large uncertainty. Despite a comparable EU-27 average growing stock per hectare for coniferous (163 m³/ha) and broadleaved (165 m³/ha) forest (Figure 52, Figure 53), there is a larger spread in coniferous growing stock (STD 94 m³/ha, range: 28-391 m³/ha, Figure 53) as compared to broadleaved stock (STD 80 m³/ha, range: 35-336 m³/ha, Figure 52). SK, DE and CZ have a coniferous growing stock larger than 300 m³/ha, whereas GR and ES have a growing stock below 70 m³/ha. DE (336 m³/ha) and PL have the largest broadleaved growing stocks, with ES and PT having the lowest growing stock.

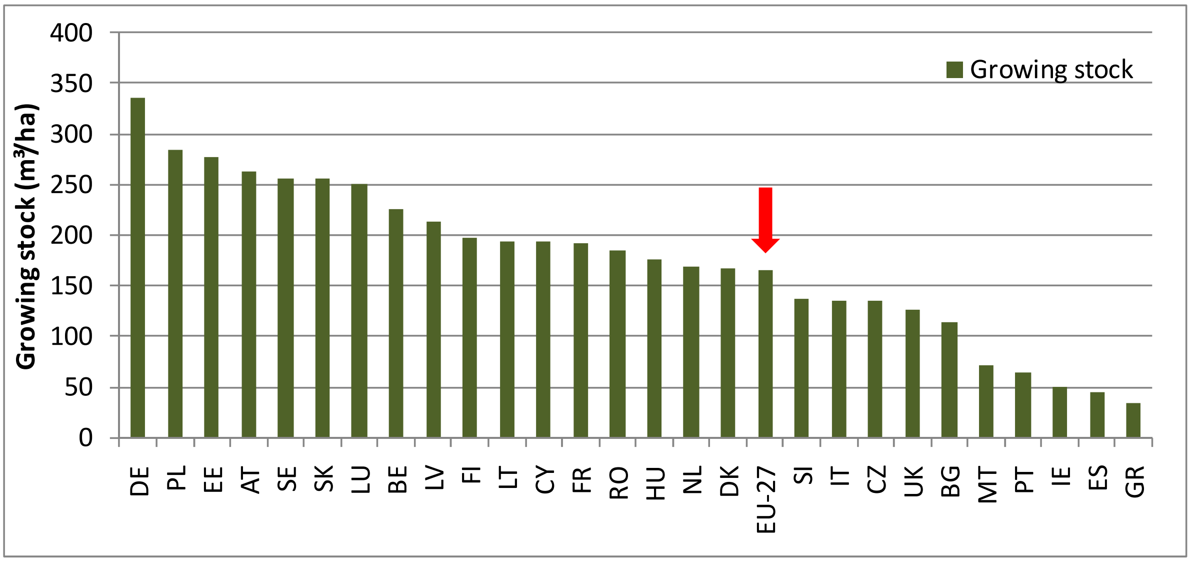



Figure 52 Growing Stock (in m³/ha) of broadleaved forests based on data from UNECE (2000)

[18] Allometry is statistical shape analysis, i.e. the study of the relationship between size and shape. Tree allometry establishes quantitative relations between key characteristic dimensions of trees (usually fairly easy to measure, e.g. diameter at breast height) and other properties (often more difficult to assess, e.g. growing stock).

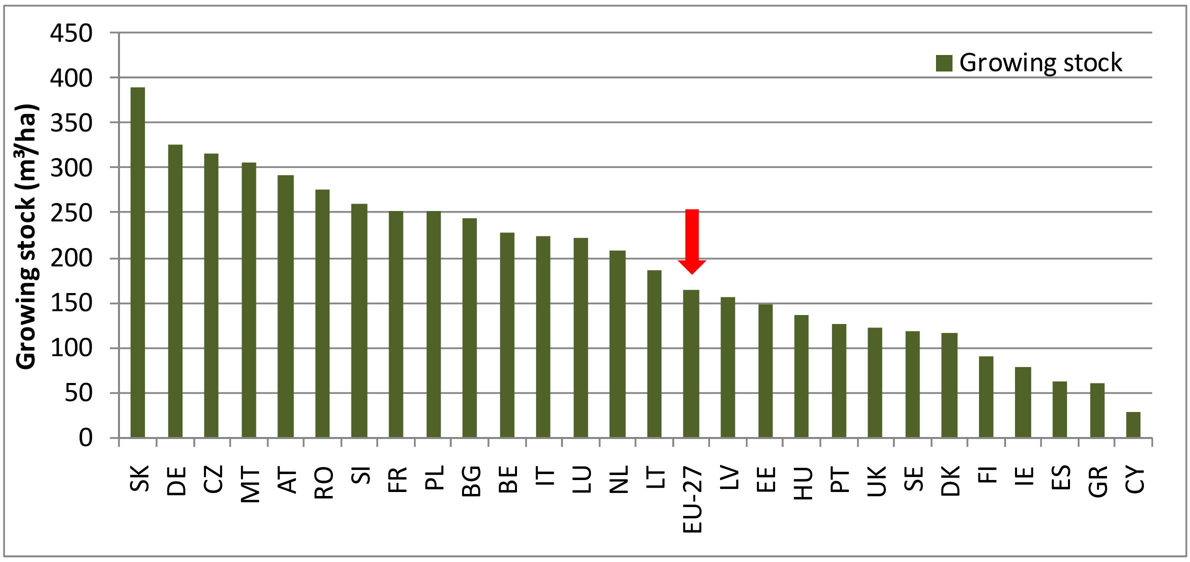



Figure 53 Growing Stock (in m³/ha) of coniferous forests based on data from UNECE (2000)

### 9.3.2 Forest biomass compartments

Both forest inventories and wood consumption statistics provide for data on forest biomass dynamics. The conversion to biomass was done for all the volumetric statistics for forest, i.e. the volume of growing stock, gross annual increment, net annual increment, natural losses, felling, removals and unrecovered fellings.

| | |
|---|---|
| *Standing volume* | *growing stock + dead trees* |
| *Growing Stock* | *standing growing wood volume, growing stem wood volume; depending on minimum threshold value for diameter, starting point of stem volume (above stump, above ground) and end point of the stem volume included (minimum top diameter).* |
| *Gross Annual Increment* | *average annual volume of increment over the reference period of all trees, measured at a minimum diameter at breast height.* |
| *Fellings* | *average annual standing volume of all trees that are felled over the reference period, also including logging residues.* |
| *Natural fellings* | *Natural losses* |
| *Net Annual Increment* | *Gross Annual Increment – Natural losses* |
| *Removals* | *fraction of fellings that is taken from the forest* |
| *Unrecovered fellings* | *Fellings - Removals* |
| *Bark* | *wood removals over bark – wood removals under bark*<br>*Wood removals under bark are a measure for roundwood production, found in* wood production statistics |

Box 7 Important biomass production definitions from forest inventories (as reported in UNFCCC, FAO, UNECE, 2000)

Most Member States have data on forest products, fellings and/or wood removals for the majority of forests but not for forests under protection, the latter not being considered for modelling. Forests under protection are not considered further in the modelling. The wood removals overbark are 82 - 97% of the fellings for broad leaved forest (Figure 54) and 85 - 99% for coniferous forest (Figure 55) depending on the region in Europe. Where possible we used felling statistics to avoid unnecessary conversions, for MT and CY we used the regions average percentage. Definitions of wood production statistics and felling statistics are provided in Box 7.

Volumetric statistics of growing stock and fellings were converted to weight using wood density or specific gravity of wood (in tonnes bone dry matter/m$^3$). The latter differs considerably with species type, maturity and region. Wood from broad leaved forest has a higher density (e.g. oak 0.75, beech 0.8, birch 0.65) as compared to coniferous wood (e.g. pine: 0.35-0.5, spruce: 0.45, fir: 0.53). In aggregated form additional variation is introduced with species composition per forest, regional management and environmental growth conditions. The volumetric statistics are important for scenario work as these provide the basis for establishing the growing stock changes.

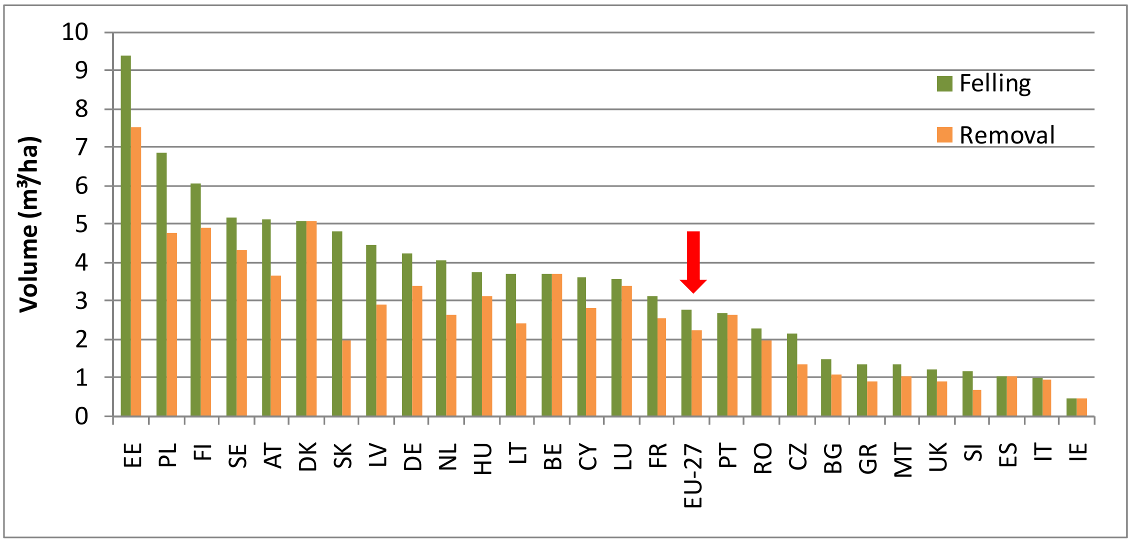



Figure 54 Felling and removal (in m³/ha) in broadleaved forests based on data from UNECE (2000)

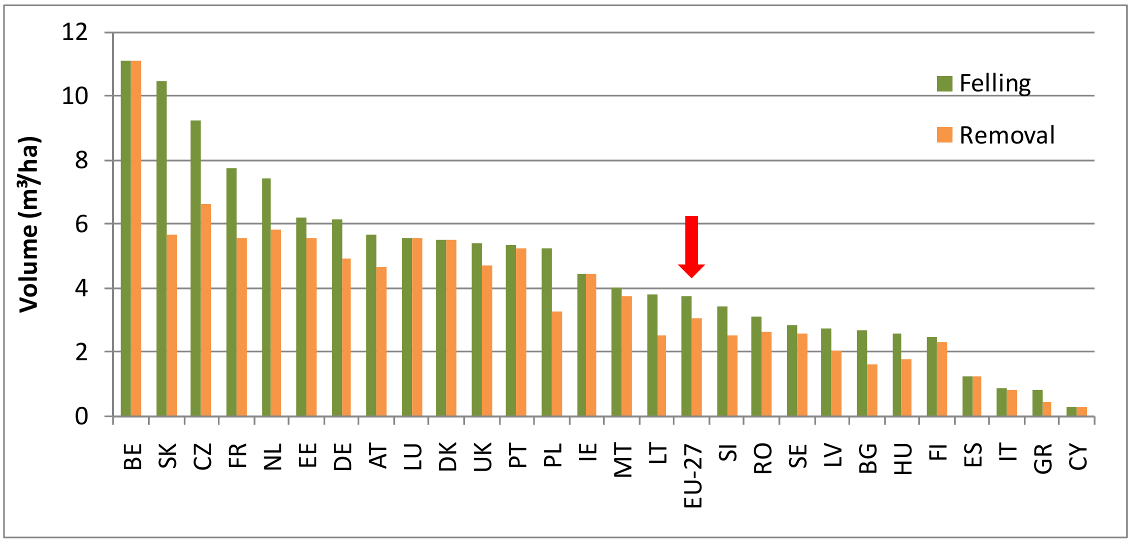



Figure 55 Felling and removal (in m³/ha) in coniferous forests based on data from UNECE (2000)

Five compartments are considered for modeling biomass production: foliage, branches & top, stem, trunk & coarse roots and fine roots (Figure 56). Only for the stem compartment statistics are available since it represents the commercially most interesting product. Conversion factors (Table 15) are therefore used to relate the other four compartments to the stem compartment, first by volume using volumetric proportions and later by weight through multiplication with specific weight (density).

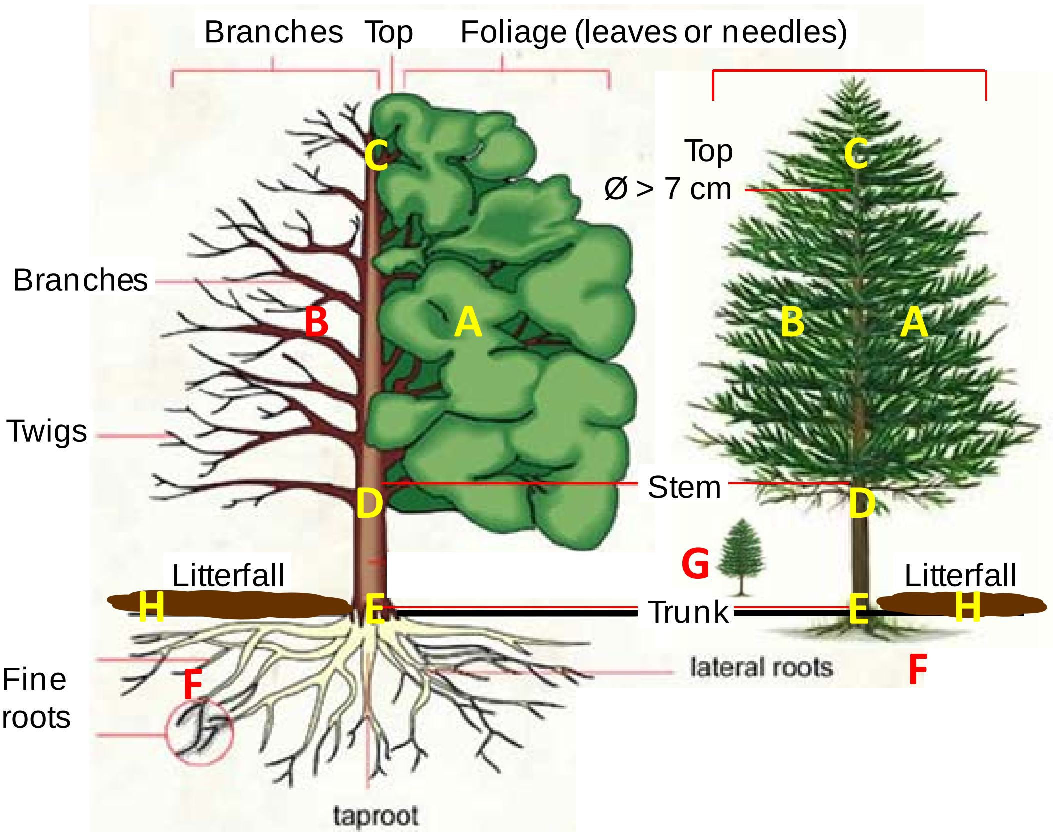



Figure 56 Structure of a tree and relation to biomass sources (A Foliage, B Branches, C Top, D Stem, E Trunk & roots, F Fine roots, G Small trees, H Litterfall)

### 9.3.3 Fluxes from living forest biomass to the soil

Litterfall is a key parameter in the biogeochemical cycle linking standing biomass to the water and soil component of the forest ecosystem. Litter decomposition is a major pathway of nutrient fluxes and determines the organic matter input to forest soils. Both the litter biomass and its chemical content are needed to quantify the annual return of organic matter to the soil. Changes in litterfall are responses to phenological development of the stand, to biotic disturbances such as pests/diseases, to anthropogenic effects, to environmental factors such as weather extremes and climate variability. Litterfall determines soil carbon cycling and sequestration in the forest soil.

Four major types of litterfall can be distinguished (Perruchoud, 1996): foliage litter, fine root litter, branch litter and coarse woody litter (e.g. stem, stump and coarse roots). The biomass of litter fall from living trees was calculated for each biomass compartment by multiplying the biomass of the growing stock by compartment-specific turnover rates, whereby each compartment is expressed as a mass fraction of the stemwood (Table 15).

Table 15 Turn-over rate, mass fractions, decomposition and humification rates for each compartment and for coniferous (CON) and broadleaved (BL) forest as used in REGSOM.

| Compartment | Turnover rate[1] | | Mass fraction[2] | | Decomposition[3] | | Humification[4] | |
|---|---|---|---|---|---|---|---|---|
| | CON | BL | CON | BL | CON | BL | CON | BL |
| Foliage | 0.2000 | 1.0000 | 0.250 | 0.350 | 0.250 | 0.350 | 0.510 | 0.510 |
| Branches | 0.0270 | 0.0250 | 0.220 | 0.220 | 0.220 | 0.220 | 0.450 | 0.450 |
| Stump* | 0.0270 | 0.0250 | 0.028 | 0.120 | 0.028 | 0.120 | | |
| Fine roots | 0.8680 | 0.8680 | 0.250 | 0.350 | 0.250 | 0.350 | 0.270 | 0.270 |
| Coarse litter | 0.0043 | 0.0087 | 0.028 | 0.120 | 0.028 | 0.120 | 0.450 | 0.450 |

* Stump parameters are only relevant for felling; [1] rate of carbon gains and losses, [2] weight fraction of stemwood per compartment; [3] organic material broken down into simpler forms of matter; [4] organic matter reaching a state of stability

### 9.3.4 Fluxes from felled biomass to the soil

Data on harvesting losses are mostly missing, while these are important for organic matter supply to the forest soil as modeled by REGSOM. Therefore they had to be estimated on the basis of forest survey statistics. Residue rates after logging vary considerably depending on local conditions. A 50/50 ratio is found for spruce forests e.g. for every cubic meter of log removed, a cubic meter of residue remains in the forest including the less commercial species. Values of up to one cubic meter of residue for three cubic meter of log extracted may be valid for European broadleaved forests. Pine forests produce a ratio of circa 57/43, i.e. 5.7 cubic meters of logs versus 4.3 cubic meters of residues remaining in the forests. The 43% consists of 7% stem wood loss and felling damage, 15% branches, 4% needles, 2% top and 16% stump and root losses. The logging residues are considered to remain in-situ and will decompose together with roots, stump and litterfall as organic matter into the soil, leaving bacteria and carbon dioxide. All the other forest products are removed from the forest to the wood industry.

Table 16 Proportions of tree components to standing volume (adapted from Eggers, 2002; Marklund, 1988)

| | Stem+bark | Wood loss | Branches | Needles | Tops | Stump & root |
|---|---|---|---|---|---|---|
| SPRUCE | 45 | 7 | 20 | 9 | 2 | 17 |
| PINE | 56 | 7 | 15 | 4 | 2 | 16 |
| CONIFER | 50 | 7 | 18 | 7 | 2 | 16 |
| BROADLEAVED | 66 | 7 | 10 | - | 1 | 16 |

### 9.3.5 Forest organic carbon balance

The on-site residue production consists of below-ground biomass (stumps, roots) and above-ground biomass (needles or leaves, tops, branches, fine roots). The humified organic carbon (tones/ha) for broad leaved forests is 1.75 times that of coniferous forest because of higher decay and assimilation rates. Broad leaved forest can assimilate up to 1.7 tonnes C/ha yearly; coniferous forests up to 1.1 tonnes C/ha (Figure 57, Figure 59). The differences can be attributed predominantly to the share of needles or leaves and fine roots. Needles and fine roots represent between 68% and 94% of total residues. the majority of wood is of commercial interest leaving 6% to 32% of coarse predominantly woody residues behind. The regional distributions of HOC for broadleaved forest (Figure 58) and for coniferous forest (Figure 60) not only show the differences in organic matter assimilation into the soil but also show the differences between countries as related to their prevailing climatic conditions. Countries with wet and cold climates have low assimilation rates and will therefore have less humified organic carbon assimilated into the forest soil (Figure 15, Figure 58). Under temperate mild and moist climatic conditions assimilation rates are optimal and a lot of organic matter is being assimilated into the forest soil. Under these conditions mull can be formed. In hot and dry climates organic material may not have the chance to assimilate into the soil (Figure 58).

Unfortunately there seems to be a bias in the statistics between countries as can be seen in the maps (Figure 58, Figure 60) – however it is still the most reliable data to use. The bias can be explained by differences in measuring and reporting on growing stock and felling, by uncertainties in area occupied by a certain forest type, but also by forest composition (age distribution, tree species), management and environmental factors. More detailed forest data are required for analyzing the effects of all the explained differences.

Litterfall from broadleaved trees is richer in nutrients and decomposes more rapidly as compared to coniferous litterfall and explains the differences between coniferous and broadleaved assimilated carbon into the forest soil (Figure 58, Figure 60). The relationships between litterfall and climatic factors show that in the temperate areas, broadleaved forests had higher litterfall than coniferous ones, whilst the opposite was found for boreal forests. Litterfall in broadleaved forests increases faster with temperature and precipitation than that in coniferous forests.

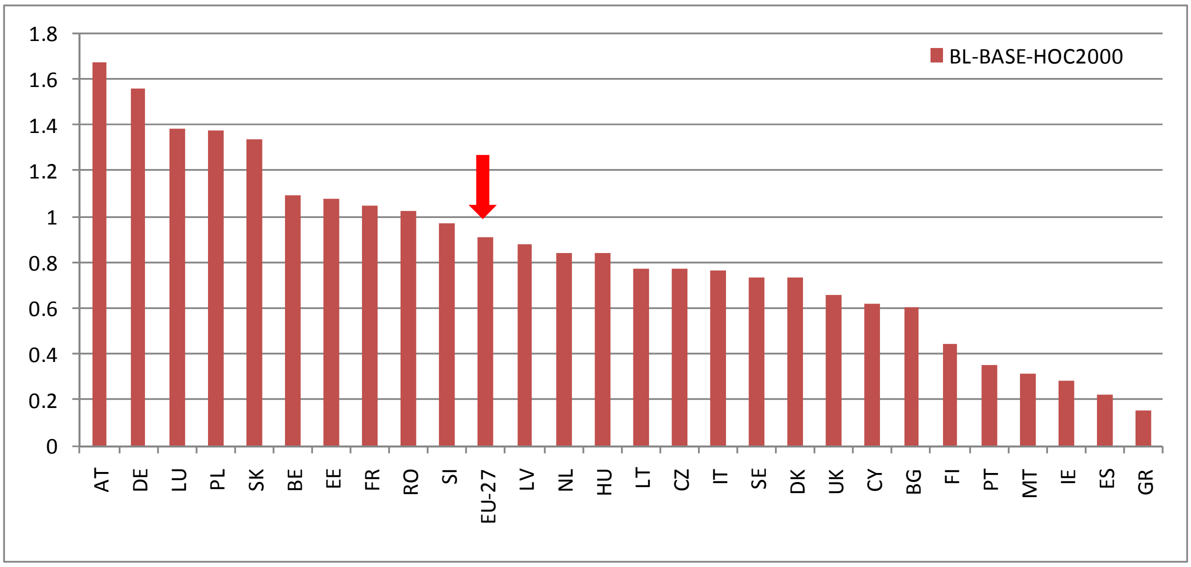



Figure 57 Humified Organic Carbon from broadleaved forest (tonnes HOC/ha) per Member State

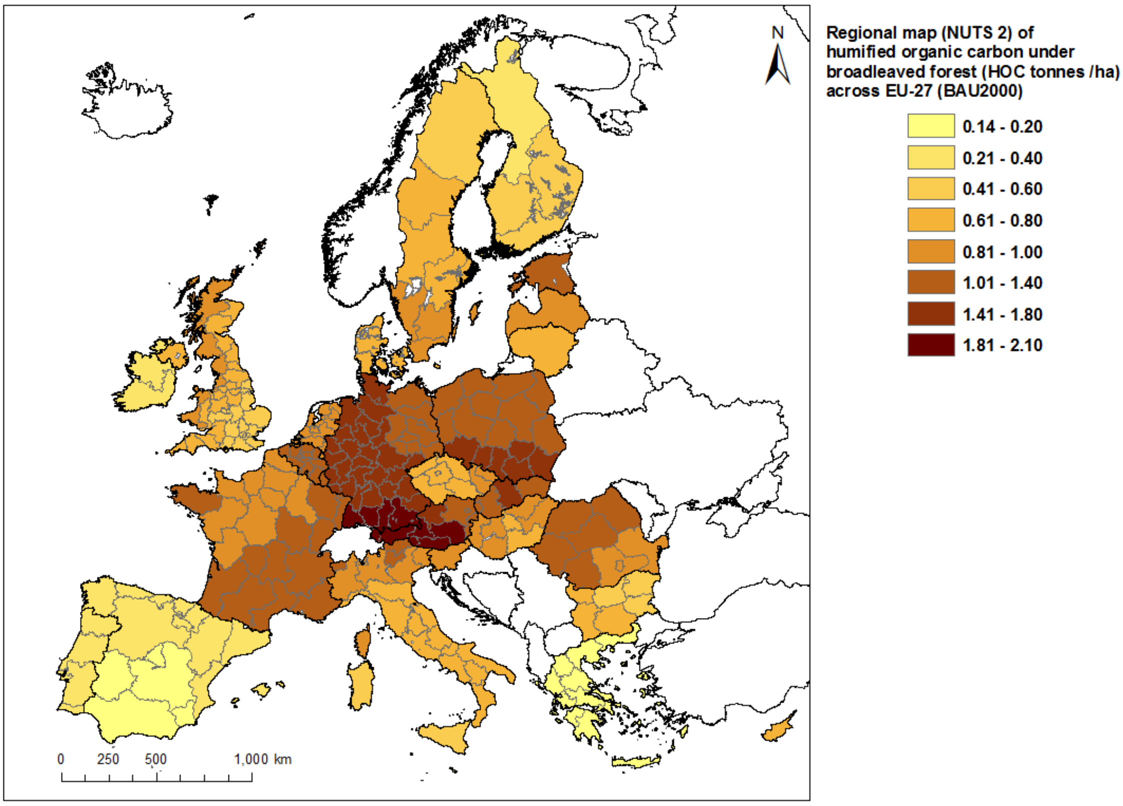



Figure 58 Distribution of Humified Organic Carbon from broadleaved forest (tonnes HOC/ha)

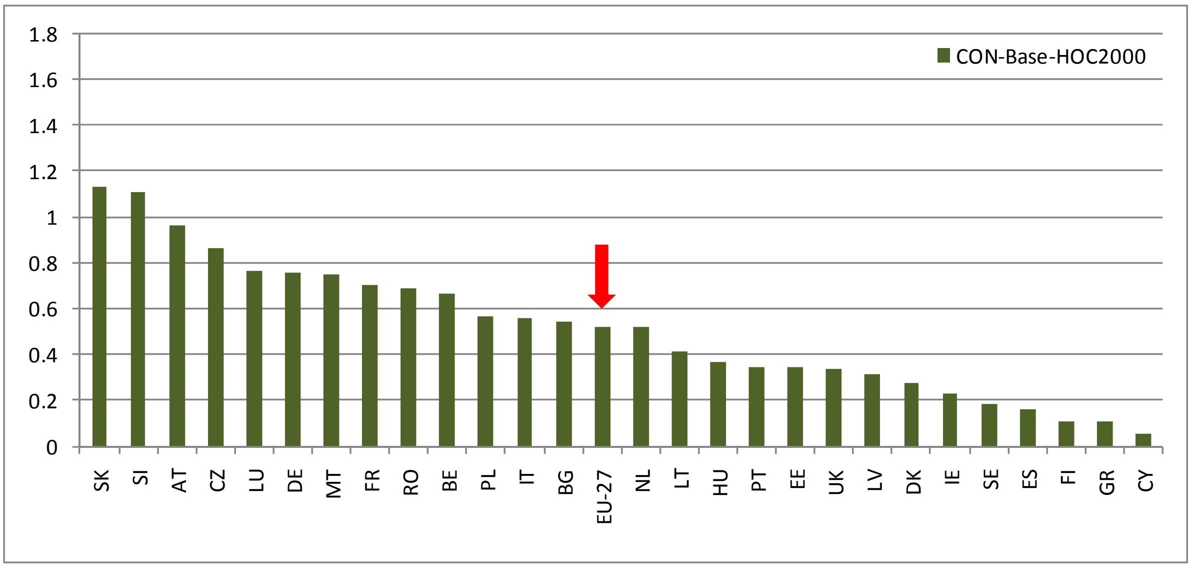



Figure 59 Humified Organic Carbon from coniferous forest (tonnes HOC/ha) per Member State

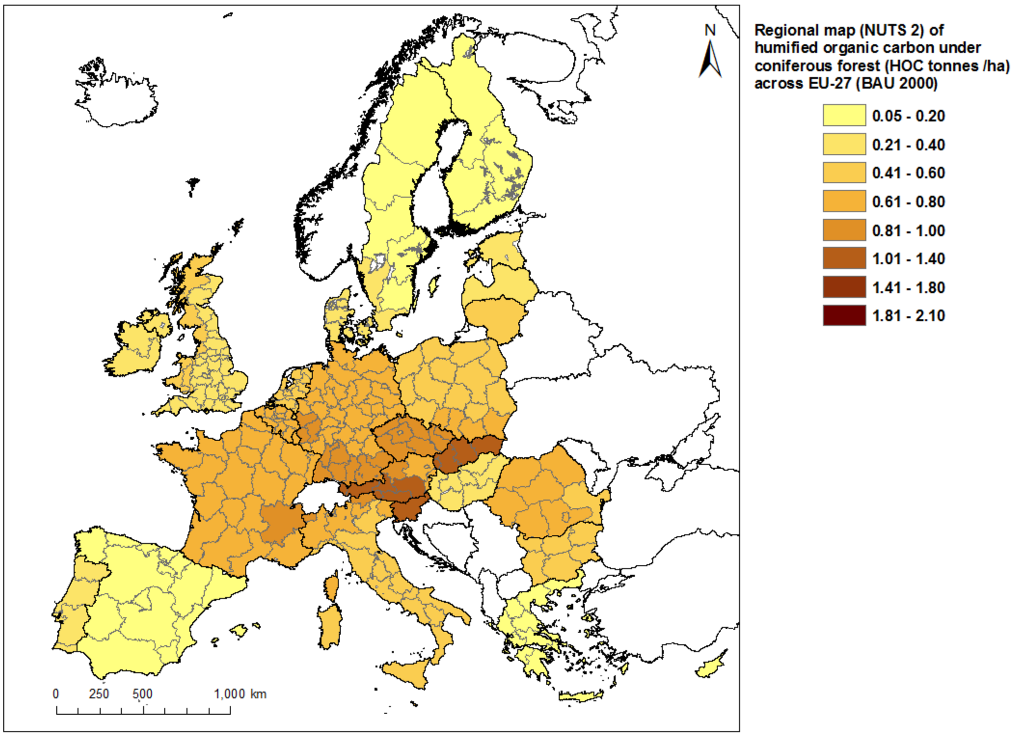



Figure 60 Distribution of Humified Organic Carbon from coniferous forest (tonnes HOC/ha)

## 9.4 Loss of organic matter from forest products

**Residues from the wood industry** such as saw milling and plywood production are considered as losses. Recovery rates related to saw milling vary with local practices as well as species. After receiving the logs, about 12.5% is residue in the form of bark. A bark fraction of 11% for coniferous wood and 13% for broad-leaved tree species was used (Haygreen and Bowyer, 1989). Slabs, edgings and trimmings amount to about 34% while sawdust constitutes another 12% of the log input. After drying the wood, further processing may take place resulting in another 8% waste (of log input) in the form of sawdust and trim end (2%) and planer shavings (6%). For calculation purposes a yield factor of 50% has been used (38% solid wood waste and 12% sawdust).

*Roundwood production = Industrial Roundwood Production + fuelwood*
*Industrial Roundwood Production = pulpwood + sawnwood + plywood/veneer + paricle board + other industrial roundwood*

Plywood making is a large-scale operation that involves debarking, cutting the logs to the length required, slicing off the veneer and gluing and hot-pressing into plywood sheets. All of these processes result in wood residues that amount to 50-55% of the total harvested wood volume. A factor of 50% has been used for calculating the potential organic matter residue left after saw milling or plywood making. The majority of these residues, however, are used in particle board production. During the process of particle board production an estimated 10% of residues are produced in the form of sawdust. The sawdust residues from the wood industry can be used for bio-energy purposes or composting.

The off-site organic matter production consists of bark and wood waste. Countries with an important wood industry such as FI, SE, DE and FR obviously produce most of the off-site dry matter. These values may represent the amount of wood waste that could be used for bio-energy purposes, thereby displacing the need for harvesting felling residues that are valuable to building up soil organic matter.

Estimated losses of organic matter from the wood industry should be used as the prime source for bio-energy purposes rather than harvesting logging residues and disturbing forest ecosystems.

## 9.5 Results

The scenario results present the flux into the forest soil of humified organic carbon from forest residues. Removing forest residues reduces the carbon stock of the forest compared with conventional stem-only harvest and hence reduces the carbon flux into the soil. With higher demands for bio-energy and subsequent increased forest residue removal, carbon fluxes into the soil become smaller as clearly shown in the scenario analysis (Figure 61, Figure 62). On average for EU-27, fluxes under coniferous forest (Figure 61) are half those under broadleaved forest (Figure 62) since coniferous forest residues decays less rapidly as compared to broadleaved forest residues.

The difference between the baseline (2000) and the C-rich scenario (2030) represents the climatic effect on carbon turnover to humified organic carbon in the soil (Figure 61, Figure 62). For coniferous forests, the climate effects result in up to 7.4% decrease in humified organic carbon added to the soil as compared to baseline for Southern European Member States and up to 9.7% increase for Northern European Member States. For broadleaved forests, the climate effects are up to 6.3% decline (S-Europe) and 9.2% increase (N-Europe). The decrease links to a drier moisture regime in Southern Europe, whereas the increase in Northern Europe relates to warmer temperatures. Climatic change influences organic matter decay factors and has the largest impact on easily decomposable forest residues, i.e. mainly foliage and fine roots.

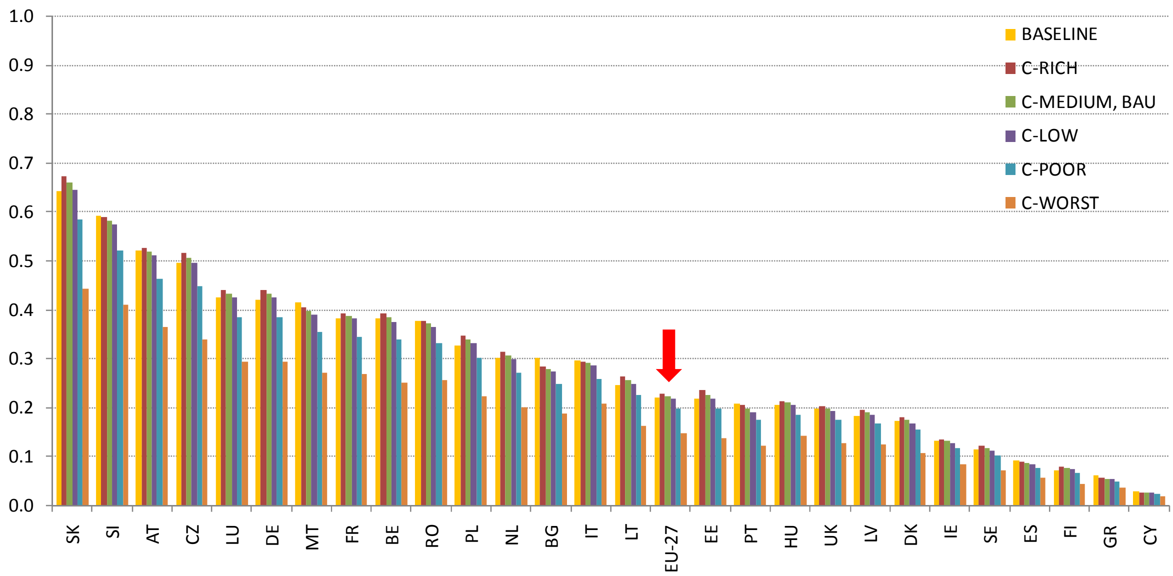



Figure 61 Contribution of forest residue to Humified Organic Carbon (tonnes/ha) in coniferous forest across Europe according to different scenarios

The composition of forest residues determines the flux into the soil. The contribution of woody residues represents a slow flux into the soil, but provides for an important carbon reserve that adds to the overall forest carbon stock. Woody residues contribute, depending on the scenario, less than one third of the humified organic carbon into the soil (Table 17). Consequently, the influence of forest residue management across the different scenarios is most noticeable for Worst Case 2030 with an EU-27 decrease of 35.6% for coniferous forests and 33.6% for broadleaved forests (Table 17). The scenarios C-Poor and C-Low are not significantly different for the contribution of woody residue to HOC due to a double effect of increased wood removal and increased foliage removal with stump harvesting. The increased foliage removal, however, results in a significant difference in carbon flux decline into the soil. Soil carbon assimilation rates for broad leaved forests are on average 1.6 times higher than for coniferous forests.

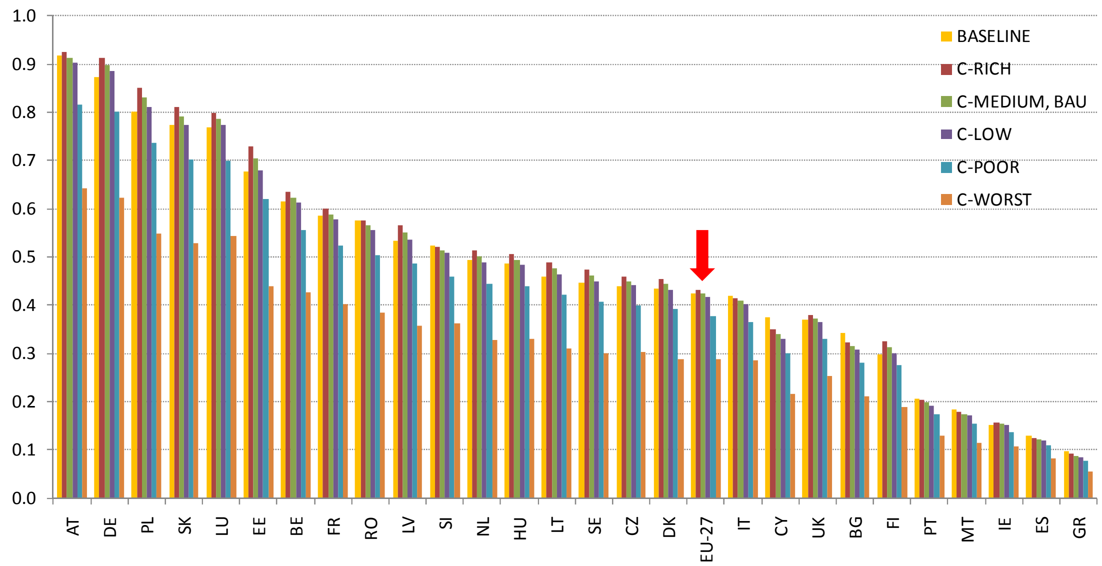
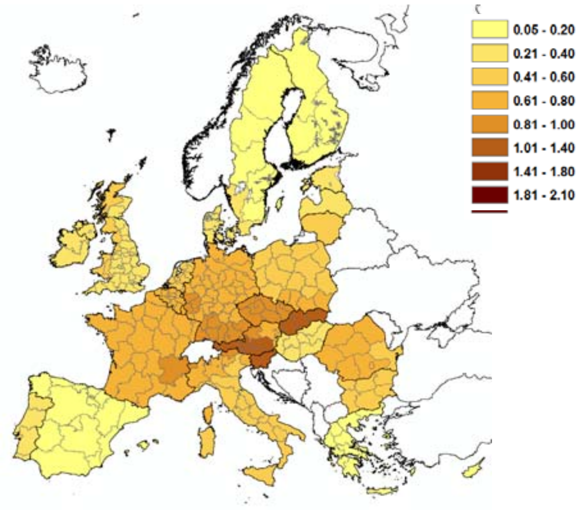



Figure 62 Contribution of forest residue to Humified Organic Carbon (tonnes/ha) in broadleaved forest across Europe according to different scenarios

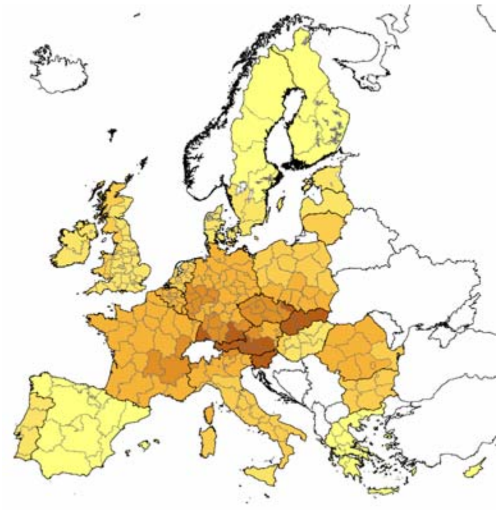
Table 17 Contribution of woody residue (stemwood, branches and stump) to humified organic carbon into the soil and decline due to residue harvesting. Figures in italic show ranges based on Member State values. Baseline is in 2000, all scenarios are in 2030.

| Forest type | BASELINE | C-RICH | BAU, C-MED | C-LOW | C-POOR | WORST CASE |
|---|---|---|---|---|---|---|
| | 2000 | 2030 | | | | |
| | Contribution (%) of woody residue to HOC | | | | | |
| Conifer | 26.8 *(11-49)* | 25.9 *(11-45)* | 24.1 (10-42) | 22.2 *(9-40)* | 22.8 *(9-40)* | 13.6 *(6-26)* |
| Broad leaved | 27.3 *(14-52)* | 26.8 *(14-47)* | 25.0 *(13-45)* | 23.3 *(12-43)* | 23.9 *(12-44)* | 15.3 *(7-32)* |
| | Decline (%) in HOC due to residue harvesting | | | | | |
| Conifer | | | 2.4 *(1-4)* | 4.7 *(2-8)* | 13.5 *(11-16)* | 35.6 *(29-43)* |
| Broad leaved | | | 1.9 *(1-4)* | 3.8 *(2-7)* | 12.9 *(12-16)* | 33.6 *(30-42)* |

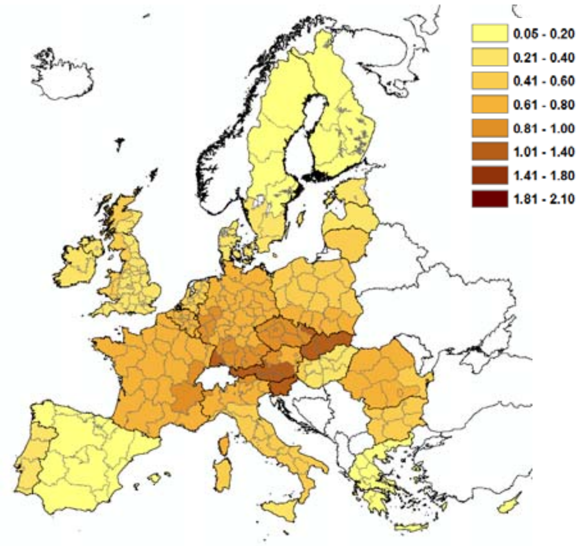
In the nation-wide forest soil carbon balance the impact of forest residue removal may not be very visible (Figure 63, Figure 64), but the general trends show that in the C-Rich scenario more carbon is humified as compared to the baseline in 2000 for both coniferous and broadleaved forests. The regional distribution should be compared by Member State across the different scenarios, since the underlying growing stock data differ enormously between Member States as these depend on forest management, environment, species and age distribution. High and maximum residue harvesting lead to a serious decline in carbon fluxes into the soil, an effect that is very pronounced in coniferous forest.



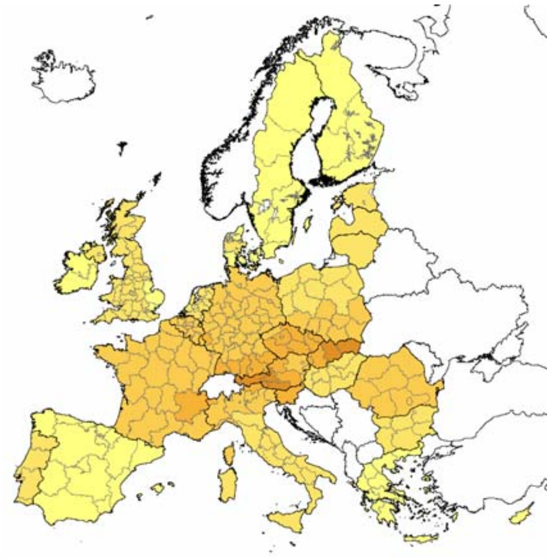

**BASELINE: coniferous**
**0% wood residues removed**



**C-RICH: coniferous**
**0% wood residues removed**

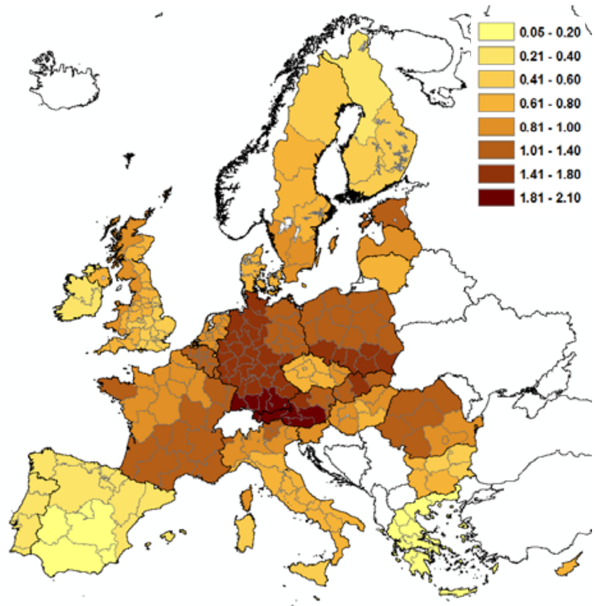



**C-POOR: coniferous**
**25% wood residues & 10% stumps removed**

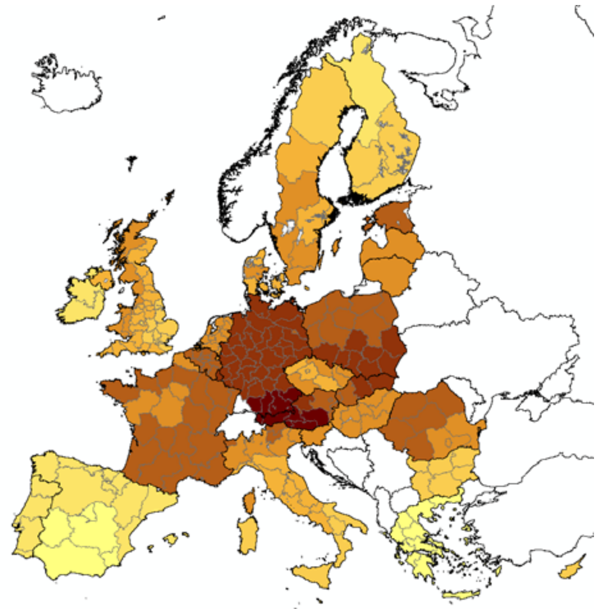


**WORST CASE: coniferous**
**70% wood residues removed & 25% stumps removed**

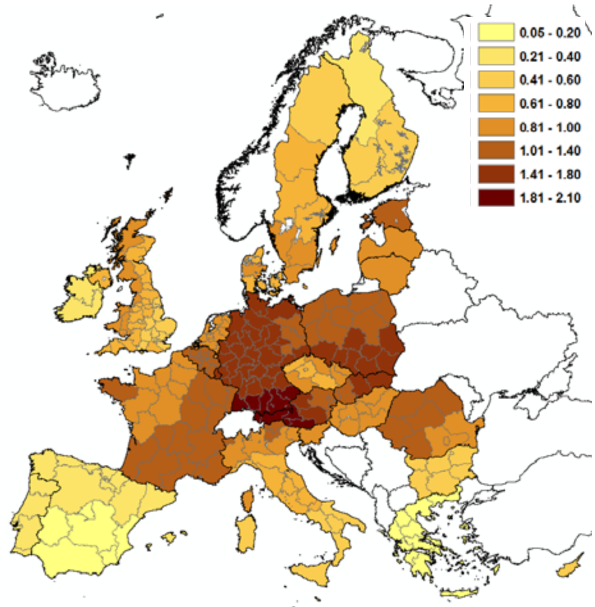
Figure 63 Distribution of humified organic carbon (tonnes/ha) across EU-27 under coniferous forest with different levels of forest residue management



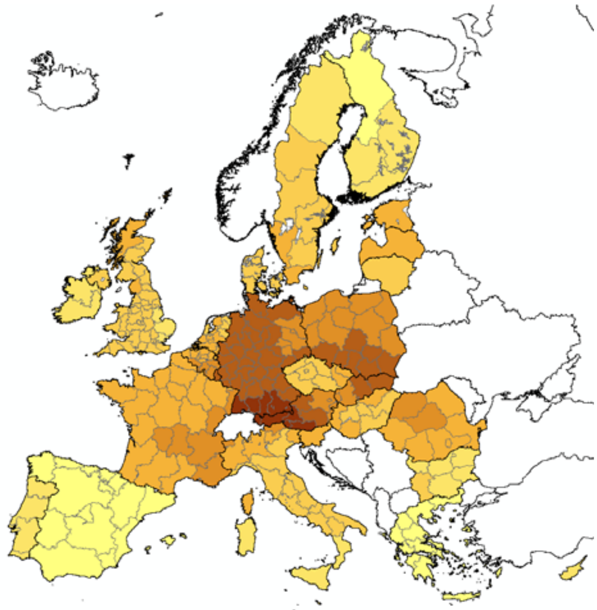

**BASELINE: broadleaved**
**0% wood residues removed**



**BAU, C-RICH: broadleaved**
**0% wood residues removed**




**C-POOR: broadleaved**
**25% wood residues & 10% stumps removed**



**WORST CASE: broadleaved**
**70% wood residues removed & 25% stumps removed**

Figure 64 Distribution of humified organic carbon (tonnes/ha) across EU-27 under broad leaved forest with different levels of forest residue management

# CHAPTER 10 CONSERVATION OF PEATLANDS

## 10.1 Introduction

| *Environmental policy /resource management issue* | C-Rich | C-Medium | BAU | C-Low | C-Poor |
|---|---|---|---|---|---|
| ***Peatlands – conservation*** | | | | | |
| *Conservation of Peatlands* | **No further drainage of peatlands allowed** | **50% reduction of historical rates (1980-2000) for peat drainage** | Continuation of historical rates (1980-2000) of peatland drainage | Continuation of historical rates (1980-2000) of peatland drainage | Continuation of historical rates (1980-2000) of peatland drainage |

In this chapter we assess the factors that determine soil organic carbon stock and fluxes under peatlands, and examine the impact of scenario options to conserve peatlands. The total restoration of peatlands to a pristine state can take 1000s of years, but the rewetting of peatlands by stopping drainage is a measure to reduce carbon emissions and partially restore peatlands. Therefore we look at the impact of contrasting scenarios to conserve peatlands.

Peat is the accumulated remains of dead organic material, and it forms in growing peatlands where the activity of decomposing organisms is suppressed in waterlogged conditions (Lappalainen 1996). 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. Peat layer growth and the degree of decomposition depend principally on its composition and on the degree of waterlogging. Peat formed in very wet conditions accumulates considerably faster, and is less decomposed, than that in drier places. The average regrowth of a single peat bog, meaning complete restoration to pristine peatland after peat extraction, could take as much as 1,000 to 5,000 years. Rewetting current peat bogs, however, could stop carbon losses.

## 10.2 Scenario approach and method

The potential impact of land use changes on future carbon losses in peatland is estimated for the following scenarios:

- BAU 2030: continued trend in historical conversion rates (1980 to 2000) of natural peatlands to drained soils for forestry, agriculture (grassland, cropland) and peat extraction;

- C-Medium 2030: 50% reduction in the historical conversion rates (1980 to 2000) of natural peatlands to drained soils for forestry, agriculture (grassland, cropland) and peat extraction; and,
- C-Rich 2030: no further drainage of peatlands allowed.

The baseline balances discussed in the previous section are considered the reference situation for 2000 – BASE 2000 (see results in Table 21). Off-site emissions due to peat combustion are not included. The average rate of peatland losses is estimated from historical and actual peatland area data reported by Joosten & Clarke (2002). The historical data refer to original peatland areas prior to any intensive use and exploitation of natural peatlands. For each country, the average loss rate is translated to land use conversion rates taking into account the actual distribution of land use on peat soils.

For C-Rich, carbon emission estimates are made for 25%, 50%, 75% and 100% restoration of peatlands currently used for forestry, agriculture and peat extraction (see baseline in previous section).

The following steps are taken:

1. The area of peatlands is taken from Byrne et al. (2004) and Joosten and Clarke (2002). Emission factors are averaged from Byrne et al., 2004; IPCC, 2006; Strack, 2008; Schils et al., 2008; Couwenberg, 2009;
2. Land use changes are implemented as described above: continued historical trend, 50% reduction in the historical conversion rates and no further drainage of peatlands allowed; and,
3. In addition several options of restoring peatlands and the impact on the GHG emissions are explored.

## 10.3 Soil organic carbon stock under peatlands

### 10.3.1 Surface area under peatland

There is a great deal of uncertainty in assessing the surface area of peatlands and their carbon stocks, this is due to the fact that peat and peatlands have been defined differently depending on country, scientific discipline and linguistic problems in translating many peat-related terms (Joosten and Clarke, 2002). 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 land cover and Historical Climatology Network data (GHCN, Easterling et al., 1996). Montanarella et al. (2006) concluded that for most European countries the Map of OC in Top soils of Europe with a threshold of 25% OC gives the most accurate estimation of distribution and area of peatland (peat and peat-topped soils). According to this approach the area of peat and peat-topped soils with OC > 25% in Europe is about 29 Mha (Source: Montanarella et al., 2006

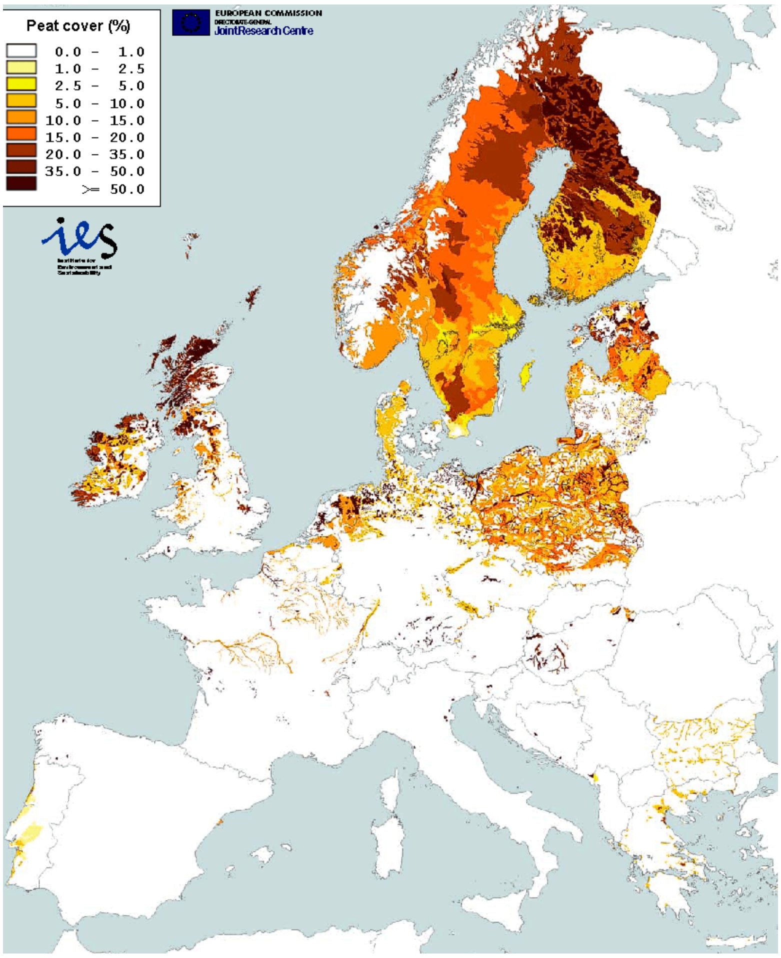
Figure 65).

Schils et al. (2008) assessed the current occurrence of peat in the EU Member States based on recently published information wherever available and using Lappaleinen (1996) and Montanarella et al. (2006) for the remainder. They estimated that the current area of peat occurrence in the EU Member States is more than 31.8 Mha. More than 50% of this surface is located in Finland, Sweden, and the United Kingdom. More than half of the peatland areas has been drained (Schils et al., 2008). Most of the undrained areas under pristine peatlands are in Finland and Sweden.

A third approach by Byrne et al. (2004) estimates the area of peatland in EU-27 at 23.2 Mha (Based on: Byrne et al., 2004 and Joosten & Clarke, 2002

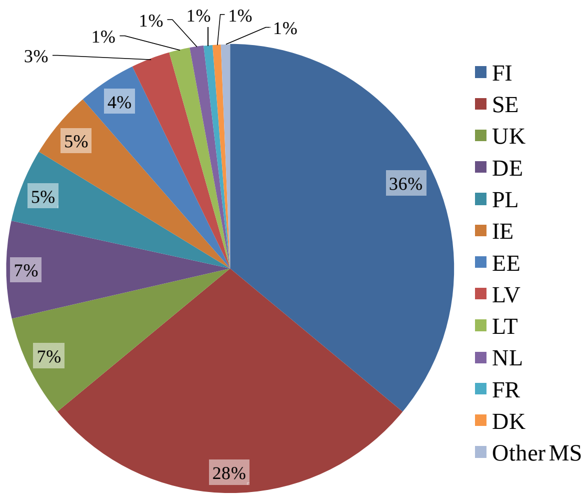
Figure 66). In this case peatland is an area with or without vegetation with a naturally accumulated peat layer at the surface, including mires drained for forestry, agriculture, horticulture and energy production. A mire is a peatland where peat is currently being formed (Joosten & Clark, 2002).

Although there are gaps in the available data on land use in peatlands, Schils et al. (2008) estimated that 20% of the European peatland area has been drained for agriculture, 28% has been drained for forestry and less than 1% is used for peat extraction. Of this extracted peat, some 85% is burnt as fuel, mainly in Ireland and Finland, and the rest is used in horticulture, agriculture, land reclamation and smaller scale uses such as a bio-filtration medium. Only half of the peatlands are reported to exist in their natural state, which of course depends on the reference or baseline year taken for comparison. Thus according to Schils et al. (2008), there are about 16 Mha of undrained peatlands in the EU and a similar area of peatlands that have been drained for agriculture and other uses. The fact is that pristine peatlands are scarce in EU-27.




Source: Montanarella et al., 2006

Figure 65 Relative cover (%) of peat and peat-topped soils in the Soil Mapping Units (SMUs) of the European Soil Database




Based on: Byrne et al., 2004 and Joosten & Clarke, 2002

Figure 66 Relative contribution of peatland areas in the EU Member States to the total EU-27 peatland area

### 10.3.2 Current state of soil organic carbon under peatlands

Peatlands in EU-27 Member States contain an estimated carbon stock of 17.2 Gt (Table 18). The vast area of peatlands in Finland and Sweden accounts for nearly 60% of this carbon storage and another 20% is stored in peatlands in UK and Ireland (data summarized by Byrne *et al.*, 2004). The average carbon stock of peatlands in Europe is 741 tonnes/ha to a depth of 1.5 m. With bulk densities varying from 0.05 to 0.5 for peatland, the average SOC stock can vary between 51.7 and 517 tonnes SOC/ha for the top 20 cm.

Table 18 Estimates of European carbon storage in peatlands. Rough carbon storage estimates for the entire Russian and Canadian peatlands included for comparison

| Region | Area (Mha) | Peat Depth (m) | Dry Bulk Density (g/dm$^{-3}$) | C content (%) | C stock (Gt) | Stock (tonnes/ha) |
|---|---|---|---|---|---|---|
| Russia | 140 | 2.3 | 91 | 51.7 | 152 | 1086 |
| Canada | 110 | 2.3 | 91 | 51.7 | 119 | 1082 |
| Europe | 51.5 | 1.7 | 91 | 51.7 | 41.8 | 812 |
| EU-15 | 19.9 | 1.6 | 91 | 51.7 | 14.9 | 749 |
| EU-27 | 23.2 | 1.6 | 91 | 51.7 | 17.2 | 741 |

Source: Based on Byrne et al., 2004

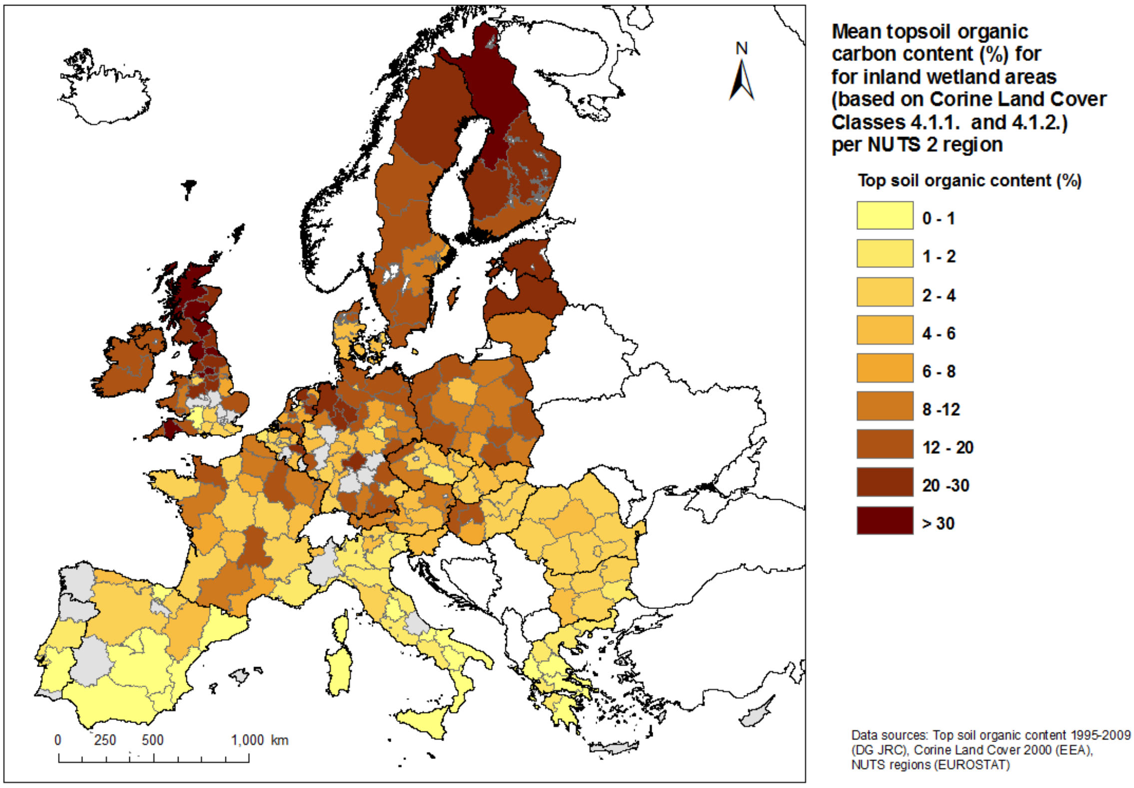
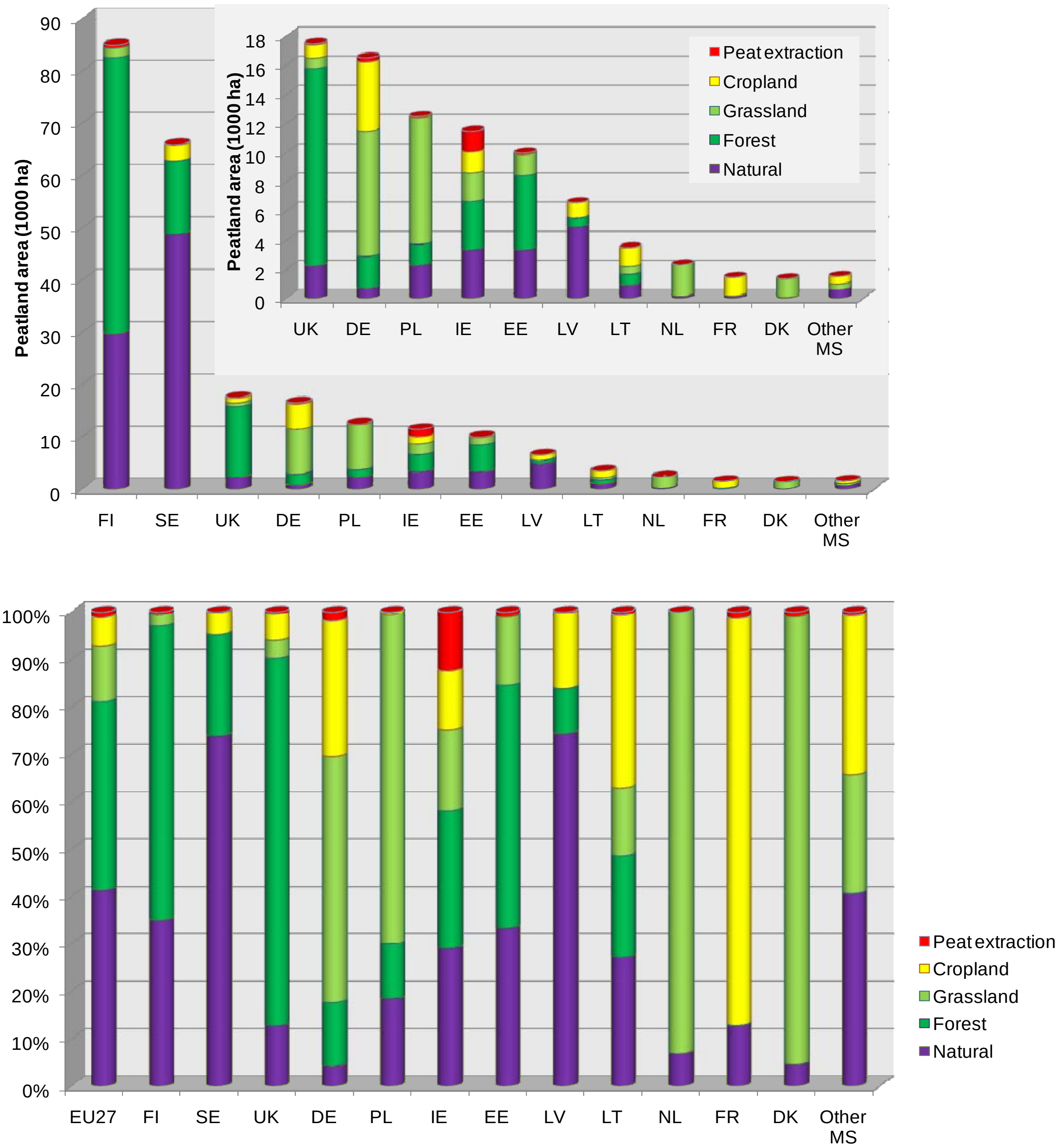



Figure 67 Mean topsoil organic carbon content (%) for inland wetland areas

A geographic analysis of topsoil organic matter in wetland areas (as defined by Corine Land Cover) provides a good estimate of carbon content in waterlogged soils, such as peatlands (Figure 67). The conversion of carbon content in % to tonnes per ha, however, requires information on bulk density and the depth, both of which are not always available for peatland. Topsoil carbon contents above 25% are equivalent to SOC stocks of between 25 and 250 tonnes C/ha for the upper 20 cm layer. This range is very wide and reflects the uncertainty in estimating carbon content in waterlogged soils and locating them adequately on a EU-27 level map.

The distribution of different land uses on peat soils in EU-27 for those countries having considerable areas of peatland demonstrates that the majority of peatlands are no longer pristine (Figure 68). These surface areas have been used to calculate the scenarios. Less than 40% of peatlands is pristine in EU-27. Other common land uses are forest and grassland. Sweden, followed by Finland have the most peatland.




Data based on Byrne et al. (2004) and Joosten & Clarke (2002)

Figure 68 Land use on peat soils in EU-27 for those countries having more than 1400 ha peatland, inset

### 10.3.3 SOC stock loss under peatlands

**SOC stock losses due to peat extraction**

Since the middle of last century peat extraction for energy use has generally decreased but is still an important energy source mainly in rural districts of Finland, Ireland, Sweden, the Baltic states and Russia. Peat extraction used for energy is still the main use in Finland, Russia and Ireland (Byrne *et al.*, 2004; Joosten & Clarke, 2002). In most other European countries extracted peat is used as substrate in horticulture. The most important property of peat is retaining moisture in soil when it is dry and yet preventing the excess of water from killing roots when it is wet. Peat can also store nutrients although it is not fertile itself. Using alternatives to peat gardens such as compost will help to preserve carbon stores. Peat products are also used in chemical and medical/cosmetic industry and as insulation material in housing. Over the period 1990-2005, the average total peat extraction in all Eurpoean countries amounted to 13.5 Mt per year and varied from 7 to 18 Mt per year (UN statistics, Schils *et al*., 2008).

Peat harvesting for energy production, for fertiliser use in agriculture and for substrate in horticulture affects only a small part of the total European peatland area, but it represents a serious land use impact on the SOC stock of peat soils. Given a rate of SOC stock loss rate of 0.19 - 2.80 t C/ha/y due to peat extraction and an areal loss of 0.2 Mha of peatlands (Schils et al., 2008), the net rate of carbon emissions due to peat extraction in EU-27 is estimated to range between 0.15 and 2.26 Mt $CO_2$ equivalents per year, which is less than 0.05% of the total GHG emissions.

**SOC stock losses due to land use change and land management**

The waterlogged conditions in a peatland provide for an anaerobic environment in which plant material is inhibited from breaking down such that large amounts of carbon are stored. The conversion of peatland to agriculture and forestry results in a drier moisture regime, an increased decomposition of SOC and a subsequent net loss of carbon stock with $CO_2$ release to the atmosphere.

Large areas of organic wetland or peat soils are drained for agriculture, forestry and peat extraction (Table 19). Such changes in land use involve changes in the peatland hydrology due to drainage. The organic carbon that was built up over thousands of years and is normally under water, is suddenly exposed to the air. On average European peatlands loose 6.9 tonnes C/ha/yr due to conversion to agricultural land use (Table 19).

Table 19 Carbon loss (in million tonnes per year and in tonnes per ha per year) in peat soils under agricultural land use; surface areas are based on Byrne et al. (2004)

| Member State | AGRIC | ARABLE | GRASS | Carbon loss | |
|---|---|---|---|---|---|
| | km² | km² | km² | Mt/yr | tonnes/ ha/yr |
| BE | 252 | 25 | 227 | 0.15 | 6.0 |
| DE | 14133 | 4947 | 9186 | 10.41 | 7.4 |
| DK | 184 | 0 | 184 | 0.1 | 5.4 |
| EE | 840 | 0 | 840 | 0.46 | 5.5 |
| FI | 2930 | 0 | 2930 | 1.6 | 5.5 |
| IE | 2136 | 896 | 1240 | 1.65 | 7.7 |
| IT | 90 | 90 | 0 | 0.1 | 11.1 |
| LT | 1900 | 1357 | 543 | 1.78 | 9.4 |
| LV | 1000 | 1000 | 0 | 1.09 | 10.9 |
| NL | 2050 | 75 | 1975 | 1.16 | 5.7 |
| PL | 7600 | 55 | 7545 | 4.18 | 5.5 |
| SE | 2500 | 630 | 1870 | 1.71 | 6.8 |
| UK | 392 | 392 | 0 | 0.43 | 11.0 |
| EU-27 | 36007 | 9467 | 26540 | 24.8 | 6.9 |

### SOC stock losses due to natural hazards

Peat can easily burn under low moisture conditions. Peat fires are often smouldering fires that can burn undetected for very long periods of time. C loss rates due to burning were estimated at 0.26 t C/ha/y for EU-27 (peat surface and above-ground biomass loss). The 2010 Russian peat fires, caused by an unusually high heat wave, were responsible for covering Moscow with a toxic blanket of smog.

## 10.4 Soil organic carbon fluxes under peatland

The main processes affecting the carbon balance of peatlands are carbon accumulation due to peat formation, extraction due to human activity and losses due to conversion to different types of land use (unmanaged or natural peatlands, forests, grassland, arable land). Exploitation of peatlands for forestry, agriculture or peat extraction involves drainage of peatlands. As a result, the drained peat layer undergoes oxidation resulting in emissions of $CO_2$.

### 10.4.1 Positive SOC fluxes under peatland

In a natural state, peatlands accumulate carbon because the rate of biomass production is greater than the rate of decomposition. The accumulation of peat involves an interaction between plant productivity and carbon losses through the process of decay, leaching, peat fires and deposition of carbon into the mineral soil beneath peat layers. Most peat-forming systems consist of two layers: an upper aerobic layer of high hydraulic conductivity in which the rate of decay is high; and the predominantly anaerobic underlying layer of low hydraulic conductivity with a lower rate of decay. The boundary between these layers is approximately at the mean depth of the minimum water table in summer (Clymo, 1983, 1984; Joosten & Clarke, 2002). Carbon is added to the surface of the peat through net primary production. Depending on the peat type and decay rates, carbon accumulation in boreal regions ranges from 0.10 to 3.0 t C/ha/y (Tolonen & Turunen, 1996). Boreal regions of the EU include most of Sweden and Finland, all of Estonia, Latvia and Lithuania and much of the Baltic Sea (Sundseth, K. 2009). The recent rate of carbon accumulation normally refers to young peat layers some hundreds of years old. The long-term apparent rate of C accumulation (LORCA) throughout the Holocene is calculated from the profile of dry bulk density from surface to bottom of the peat layer. Estimates for LORCA in Finland, Russia and Sweden (Byrne *et al*.,2004) were in the range of 0.15-0.25 t C/ha/y. Cannell *et al*. (1999) estimated accumulation rates in UK peatlands at 0.20 - 0.50 t C/ha/y. The true net rate of C accumulation (ARCA) can be determined by peat accumulation models, and has been estimated at 2/3 of LORCA (Tolonen & Turunen, 1996). Given an ARCA range of 0.100 - 0.333 t C/ha/y (2/3 of the LORCA range 15-50 t C/ha/y) and 31.8 Mha of peatlands, the net rate of carbon accumulation in EU-27 is estimated to range between 11 and 38 Mt $CO_2$ equivalents per year, which is less than 1% of the total GHG emissions.

### 10.4.2 Negative carbon fluxes for peatland

Historically, northern peatlands have functioned as a carbon sink, sequestering large amounts of soil organic carbon, mainly due to low decomposition in cold, largely waterlogged soils. Because of the high water-holding capacity of peat and its low hydraulic conductivity, accumulation of soil organic carbon raises the water table, which lowers decomposition rates of soil organic carbon in a positive feedback loop in a two-way interaction between hydrology and biogeochemistry. The feedback between the water table and peat depth increases the sensitivity of peat decomposition to temperature, and intensifies the loss of soil organic carbon in a changing climate (Ise *et al*., 2008). With climate change leading to a drier water balance and higher temperature regime in most regions, peatland becomes vulnerable to drainage and exposure to air which results in rapid organic matter decay and release of methane gas ($CH_4$) into the atmosphere as carbon dioxide ($CO_2$).

Schils *et al*. (2008) made a GHG balance for peat soils in agricultural use. Based on emission data for peatlands collected by Couwenberge *et al*. (2008), unpublished data from Van den Akker about fens in the Netherlands and the data summarised by Oleszczuk *et al*. (2008), Schils *et al*. (2008) estimated the carbon losses from peat areas used as grassland at 20 t $CO_2$/ha/year while losses from peat areas used as cropland are estimated at 40 t $CO_2$. The $N_2O$ emissions are assessed assuming 1.25% of the mineralized nitrogen to be converted into $N_2O$ (Mosier *et al*, 1998). Only EU countries with a substantial area of peatsoil in agricultural use were included in the approach. A more detailed balance was made by Byrne *et al*. (2004). They considered five types of land use: natural, forest, grassland, cropland and peat extraction. A distinction was made between bogs (ombrotrophic) and fens (minerotrophic) as in nutrient poor bogs the potential $N_2O$-production is limited, even under drained conditions, while in nutrient rich fens the potential for $N_2O$-emissions is much higher. The corresponding emissions rates for each land use type were derived from literature. All EU-27 countries were included in the approach.

According to Schils *et al*. (2008) the GHG emission from cultivated and drained organic soils in EU-27 (agricultural area, cropland and grassland) is approximately 100 Mt $CO_2$ equivalents per year. Estimates form Byrne *et al*. (2004) are about 30% lower, mainly because lower emission rates per hectare were used for grassland on bog peatland and for cropland on fen peatland. Compared to the total GHG emissions in EU-27 of ~5000 Mt $CO_2$ equivalents per year (EEA, 2010), the emissions from cultivated and drained peatlands for agricultural use estimated by Schils *et al*. (2008) have a contribution of about 2%.

Table 20 Average emission factors (kg C or N.$ha^{-1}.yr^{-1}$) based on measured fluxes from European bogs and fens under different land uses

| | $CO_2$ | $CH_4$ | $N_2O$ |
|---|---|---|---|
| Management Type | (kg C/ha/yr) | (kg C/ha/yr) | (kg C/ha/yr) |
| Bog | | | |
| Grassland | 2350 | 2.00 | 0.01 |
| Arable | 4400 | 0.00 | 0.00 |
| Restoration | 620 | 15.00 | 0.02 |
| Fen | | | |
| Grassland | 4120 | 0.40 | 5.05 |
| Arable | 4090 | -0.20 | 11.61 |
| Restoration | | 12.40 | 0.64 |

Source: modified Byrne et al. (2004)

From a range of drained nutrient poor bogs in Europe, Byrne *et al* (2004) estimated the average net GHG fluxes ($CH_4$ and $CO_2$) due to drainage in managed peatlands (forestry, agriculture, peat extraction) to amount 1.25 t $CO_2$-C eq/ha/y versus only 0.19 t $CO_2$-C eq ha/yr in natural bogs. Initial carbon losses from newly drained peatland are in the range 2 - 4 t C/ha/y (Hargreaves *et al*., 2003).

It is a common practice to afforest used peat bogs, leading to lower levels of organic matter storage than the original peat bog. SOC stock losses from peats under forest are sustained in the long term with loss rates as high as 2.50 - 5.00 t C/ha/y in Finland and Estonia (Minkkinen *et al*., 2007). Land use conversions on peatland lead to net SOC stock loss. Therefore, peatland afforestation cannot be considered an effective means of sequestering C.

## 10.5 Contribution of peatland to GHG balance

### 10.5.1 Contribution of peatland to GHG balance

In Finland, the contribution of peat to greenhouse gas emissions can exceed 10 million tonnes carbon dioxide per year, equal to the total emissions of all passenger car traffic in Finland. Compared to the emissions reported for IPCC sector 5 (LULUCF), however, peatlands have a contribution of 3 - 9 % to carbon sequestration (EEA, 2010). An intact hectare of wet peatland can sequester the equivalent of 10-15 tonnes of carbon dioxide annually. These are compelling reasons for calculating the contribution of peatland to the GHG balance.

The position of the water table is one of the most important factors influencing peat formation conditions and processes in organic soils. Increasing the water level in peat decreases emissions of $CO_2$ (by up to 20%) and $N_2O$, but increases emissions of $CH_4$ (Strack, 2008). An average estimation of peatland carbon and GHG emission balances was based applying different sets of emission factors for peat soils (Byrne et al., 2004; IPCC, 2006; Strack, 2008; Schils et al., 2008; Couwenberg, 2009) and using peatland area and land use data collected by Byrne et al. (2004) and Joosten & Clarke (2002) (Table 21). The resulting carbon and GHG balances are compared with the GHG emissions of EU-27 in 2008 as reported by the European Environmental Agency (EEA, 2010). The peatland emissions are weighed against the total GHG emissions (excl. sector 5) and against the emissions from land use & land use change and forestry (sector 5, LULUCF). Over the period 1990-2007, the average total peat extraction in the European Union amounted to 21 Mt per year. Off-site emissions are estimated to account for 6-16% of the summed GHG emissions from peatlands and peat use in EU-27. Compared to the total GHG emissions, the contribution of off-site emissions from extracted peat is less than 1%, but in some countries it causes substantial GHG emissions. In Finland, for example, peat combustion is estimated to generate about 15% of the country’s net GHG emissions (Lapvetainen et al., 2007).

The contribution of different land uses on peat soils to the peat GHG emission budget mainly reflects the distribution of the different land use types over the entire area of peatland per Member State. The differences in land use distributions between Member States explain the contribution to emissions: despite having similar areas of peatlands, the Germany has three times more carbon emissions from peatlands as compared to UK (Table 21) due to the high proportion of agricultural land use on peatlands compared to forests in UK (Figure 69). Uncertainties in the balance estimates are due to variation in the ratios between the land use specific emission factors, and in particular due to the emission factors related to peat extraction.

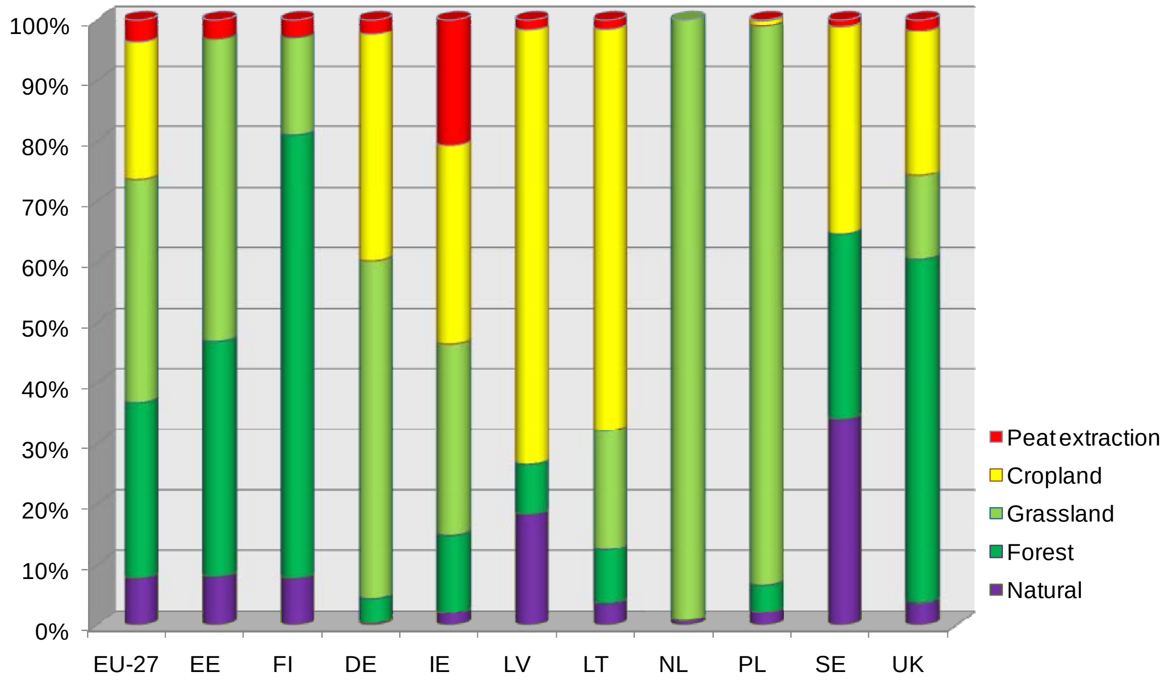



Figure 69 Relative contribution of different land uses on peat soils to the peatland GHG emission budget for EU-27 and for 10 selected countries with considerable peatland areas.

In some countries, peatlands (under different land uses) act on a regional scale as modest carbon sinks while others are carbon sources. However, all countries are net GHG emitters from peatlands. The obtained results are associated with considerable uncertainties regarding the distribution of land use types on peat soils, the estimation of emission rates and fluxes and the classification of peatlands. Nevertheless, some interesting trends and estimates can be derived from these regional balances. The main contribution to GHG emissions from peatlands in EU-27 originates from Germany, Poland, Sweden, Finland and UK. These countries account for 70-85% of the emissions. Compared to the total GHG emissions of EU-27 in 2008 (EEA, 2010), the overall contribution of peatland emissions is only 2-5% but there are huge regional differences. In countries with vast areas of peat soils, the contribution of peatlands to the national GHG budget is substantial. In comparison with the GHG emissions in the land sector (IPCC sector 5, LULUCF), peatlands can be considered an important source of GHG emissions related to land use. The estimated GHG budgets range between 20 and 50% of the emissions reported IPCC for sector 5 in 2008 (EEA, 2010).

### 10.5.2 Contribution of peatlands to GHG emissions

The contribution of off-site emissions due to peat combustion can be estimated from the total amount of peat extracted assuming the entire carbon content (50%) to be released during mineralisation. Given the amount of peat extracted in Europe (7-18 Mt per year), the $CO_2$ emission due to peat combustion ranges between 12 and 33 Mt per year, which corresponds to only 0.26 - 0.66% of the total GHG emissions. However, peat combustion has a substantial contribution to the GHG balance of some individual countries that use large amounts of peat for energy production. In Finland, for example, peat combustion alone is estimated to generate about 15% of the country's net GHG emissions (Lapvetainen et al., 2007).

Table 21 Peatland carbon and GHG balance and relative contribution of the national peatland GHG budget to the total GHG emissions per Member State.

| | **Peatland area[1]** | | **C balance[2]** | **GHG balance ($CO_2$, $CH_4$, $N_2O$) [3]** | | |
|---|---|---|---|---|---|---|
| | **$km^2$** | **% of EU-27** | **kton C $y^{-1}$** | **kton $CO_2$-C eq $y^{-1}$** | **% of EU-27** | **% of total GHG emissions** |
| Austria | 200 | 0.08 | -4.52 | 5.10 | 0.01 | 0.03 |
| Belgium | 160 | 0.07 | 77.12 | 85.63 | 0.20 | 0.25 |
| Bulgaria | 25 | 0.01 | -0.36 | 0.31 | 0.00 | 0.00 |
| Cyprus | 0 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Czech Republic | 200 | 0.08 | 63.84 | 80.43 | 0.19 | 0.22 |
| Denmark | 1400 | 0.59 | 547.18 | 656.68 | 1.65 | 4.03 |
| Estonia | 10000 | 4.24 | 1308.49 | 1795.19 | 4.03 | 34.47 |
| Finland | 85000 | 36.02 | 6911.49 | 9904.50 | 20.35 | 53.78 |
| France | 1500 | 0.64 | 432.25 | 600.11 | 1.41 | 0.44 |
| Germany | 16520 | 7.00 | 7468.06 | 8954.00 | 21.54 | 3.65 |
| Greece | 71 | 0.03 | 32.43 | 43.15 | 0.10 | 0.13 |
| Hungaria | 330 | 0.14 | 123.94 | 154.02 | 0.38 | 0.83 |
| Ireland | 11500 | 4.87 | 2272.88 | 3190.80 | 7.44 | 18.18 |
| Italy | 300 | 0.13 | 151.89 | 199.69 | 0.47 | 0.14 |
| Latvia | 6600 | 2.80 | 545.82 | 962.25 | 2.22 | 32.71 |
| Lithuania | 3520 | 1.49 | 1041.30 | 1394.71 | 3.22 | 22.41 |
| Luxembourg | 3 | 0.00 | 1.45 | 1.61 | 0.00 | 0.05 |
| Malta | 0 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Netherlands | 2350 | 1.00 | 1141.00 | 1294.63 | 3.26 | 2.46 |
| Poland | 12500 | 5.30 | 4643.21 | 5426.03 | 13.49 | 5.41 |
| Portugal | 20 | 0.01 | 10.13 | 13.31 | 0.03 | 0.07 |
| Romania | 71 | 0.03 | -1.96 | 2.37 | 0.01 | 0.01 |
| Slovakia | 26 | 0.01 | 5.42 | 7.30 | 0.02 | 0.06 |
| Slovenia | 100 | 0.04 | -3.09 | 3.85 | 0.01 | 0.10 |
| Spain | 60 | 0.03 | 23.11 | 31.45 | 0.07 | 0.03 |
| Sweden | 66000 | 27.97 | 2305.00 | 6028.95 | 13.20 | 39.28 |
| United Kingdom | 17500 | 7.42 | 2617.60 | 3112.86 | 6.70 | 1.87 |
| ***EU-27*** | **235956** | **100** | 31713.66 | 43948.93 | 100.00 | 3.48 |

1: From Byrne (2004) and Joosten & Clarke (2002); 2 & 3: own calculations with emission factors averaged from Byrne et al. (2004); IPCC (2006); Strack, (2008); Schils et al. (2008) and Couwenberg, 2009.
1 gigagram = 1000 ton (metric)

## 10.6 Results

### 10.6.1 BASE 2000: Impact of peat extraction rate on carbon stock gains and losses

The impact of $CO_2$ and $CH_4$ emissions from peatlands and peat extraction on soil carbon stocks for Base 2000 is estimated by weighting the carbon emission budget against the carbon stock estimates. For some countries, where peatlands are mainly under natural conditions, the carbon emission budget shows negative values (source) indicating carbon accumulation. In those countries there is still an increase of the national carbon stock in peatlands. In contrast, positive values in the carbon emission budget point to soil carbon losses.

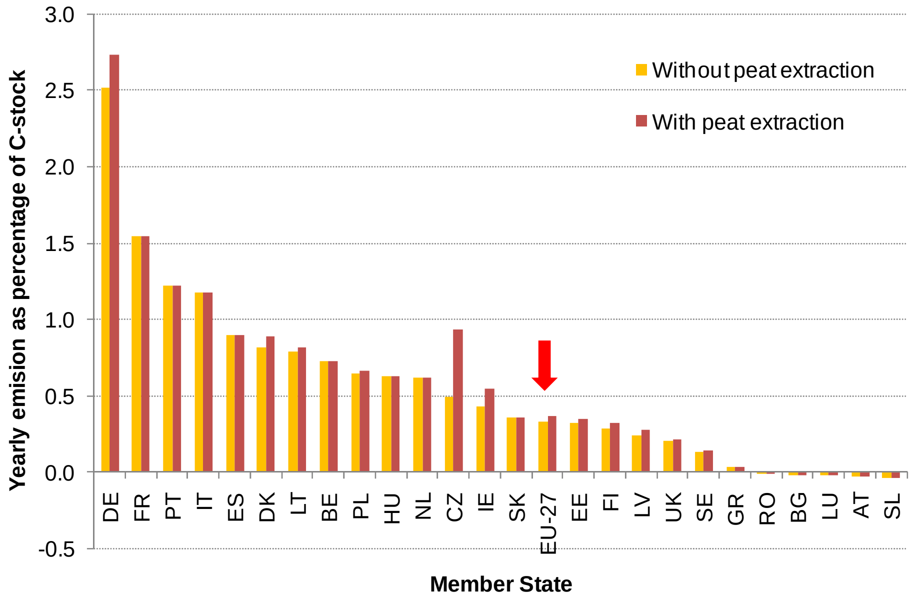



Figure 70 Annual carbon emission (i.e. $CO_2$ and $CH_4$) as % of estimated peatland C stock with and without current rates of peat extraction (unit is %/yr) – BASE 2000 (estimated C stock is the total soil carbon stock in peatland areas).

### 10.6.2 Scenarios

Yearly percentages of peatland carbon stock gains and losses are estimated for EU-27 Member States (Figure 70), and taking into account peat extraction or not. The estimates are based on carbon balances (only $CO_2$ and $CH_4$) using peatland area, historic land use conversion data and emission factors. Peat extraction emissions are estimated from peat production volumes reported in the Industrial Commodity Statistics Database of the United Nations (UNData, 2010) assuming the entire carbon content to be released. The estimated peatland carbon stock loss rates in EU-27 range between 0.13 and 0.36% per year which means that 13-36% of the current soil carbon stock in European peatlands might be lost by the end of this century. Large regional differences exist. In some Member States, all peatland carbon reserves may already be gone within a couple of decades. Obviously curbing current land use conversion rates will be necessary to safeguard the large carbon reserve of peatland soils.

For the 10 Member States with the highest peatland areas the relative increase of carbon emissions due to high rate land use conversion of peatlands are estimated (BAU

2030). Carbon losses in EU-27 peatlands are expected to increase by about 8% between 2000 and 2030 (Figure 71). At the MS level, the BAU 2030 scenario means that all peatlands the Netherlands are drained by 2010. In Poland, Lithuania, and Sweden the increase of emissions between 2000 and 2030 is 23%, 16% and 9%, respectively, whereas in Ireland, the UK, Estonia and Germany the increase of emissions between 2000 and 2030 is less than 5%.

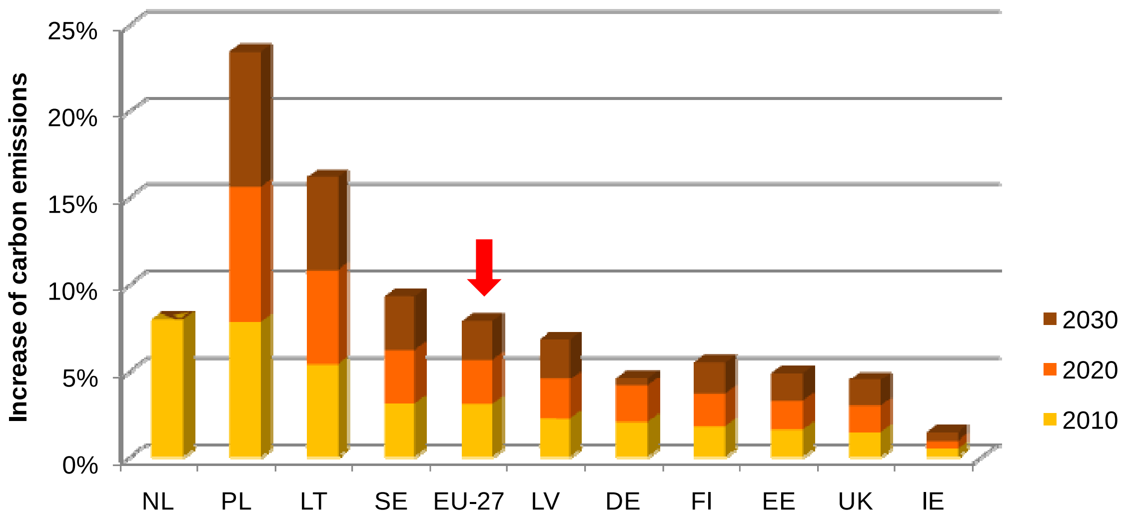



Figure 71 Relative increase of carbon emissions due resulting from the BAU 2030 peatland conservation scenario (continued trend in historical conversion rates)

For EU-27 and for the 10 selected countries with substantial peatland areas the relative increase of carbon emissions the carbon emissions in EU-27 are expected to increase by about 4% between 2000 and 2030 for the C-Medium scenario (Figure 72). A summary carbon balance for EU-27 following low rates of land use conversion of peatlands shows that carbon emissions are approximately half those of the BAU 2030 scenario.

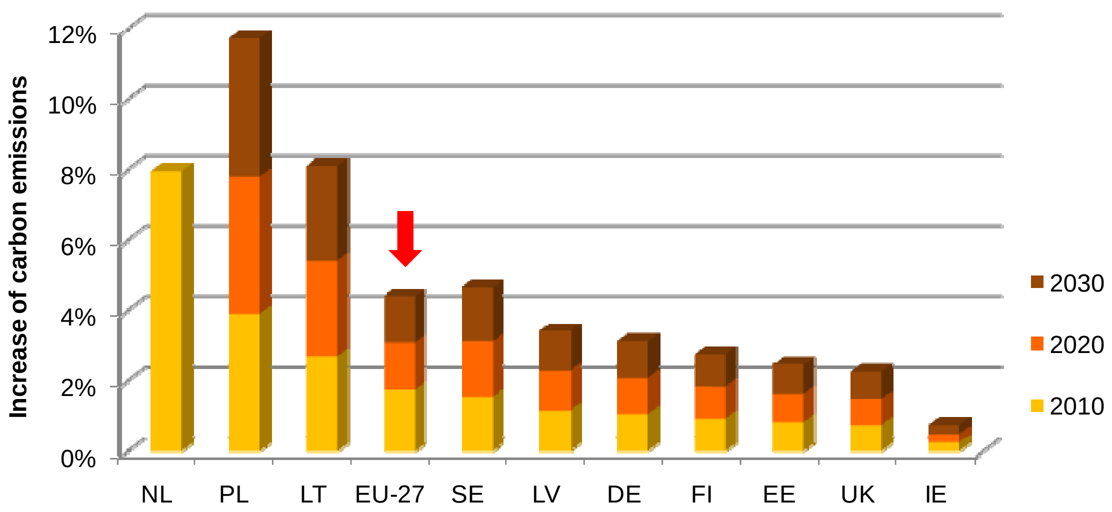



Figure 72 Relative increase of carbon emissions due resulting from the C-Medium 2030 peatland conservation scenario (50% reduction in historical conversion rates)

For the C-Rich 2030 scenario no further drainage of peatlands is allowed. The emission factors are the same as during the baseline period. Linking the rates to climate change is possible but additional information on waterlogging and flooding is needed to make an accurate analysis. We calculated the relative decrease of carbon emissions due to different rates of peatland restoration (Figure 73). The restoration rates may be thought of as restoration of flood and waterlogged areas in order to encourage peatland formation. Only at 100% restoration the restored peatlands become sources of methane but this natural emission is less than the carbon dioxide loss due to oxidation.

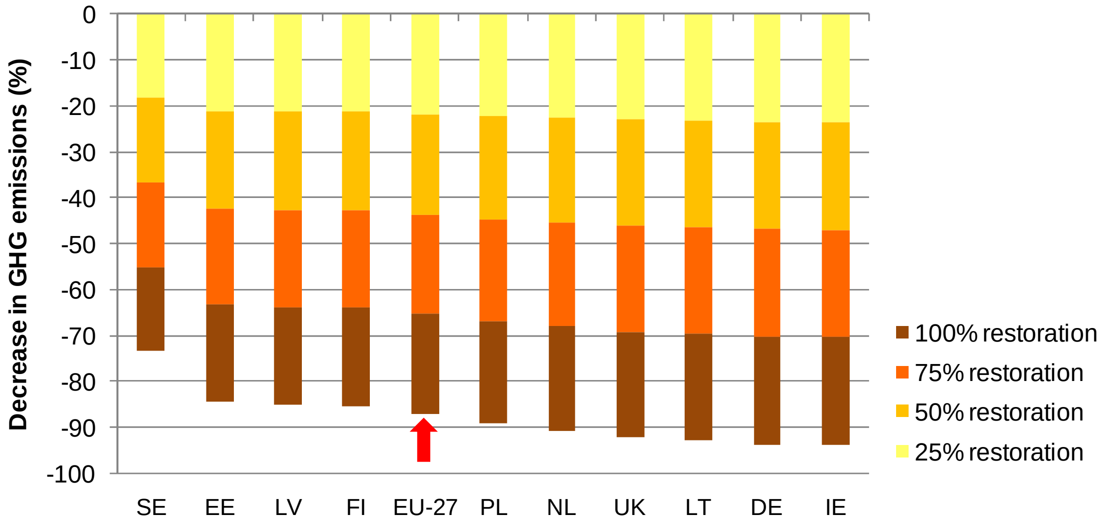



Figure 73 Relative decrease of carbon emissions due to different rates of peatland restoration.

# CHAPTER 11 SUMMARY AND CONCLUSIONS OF SOIL ORGANIC CARBON STOCK AND FLUXES ANALYSIS

In this study we make a clear distinction between soil organic carbon stocks and fluxes. Soil organic carbon stocks are based on soil type, long term climate and long term land use. We use spatial analysis to combine predicted land use changes (LUMOCAP) with the topsoil soil organic content map of Europe (Jones et. a., 2004) to assess the impact of land use changes on soil organic carbon stock. We make the assumption that the average soil organic carbon stock of the surface horizon reflects the equilibrium state; therefore, the differences between SOC stocks under different land uses reflect the change from one equilibrium state to another. Soil organic carbon fluxes on the other hand are like snapshots in time of the impact of resource management on the soil. We provide an estimate of humified organic content (REGSOM) from agriculture and from forests. Carbon fluxes are therefore snapshots of carbon input and cannot be directly compared or added on to carbon stocks. For Peatlands, another approach again is adopted whereby carbon stocks and GHG fluxes are assessed.

In this chapter we bring together the main results of the modelling and scenario work together and draw the most pertinent conclusions of the analysis.

## 11.1 Soil organic carbon stocks under agriculture and forests

We use three different scenarios to assess the impact of selected land use changes on soil organic carbon stocks under agriculture and forests, with C-Rich, C-Medium, BAU, C-Low, and C-Poor options (see Table 5 for scenario option details):

- Maintenance of Grassland;
- Use of Set-aside (for EU-15 only); and,
- Change from Utilised Agricultural Area (UAA) to forest.

We compare the outcome of the three scenarios in a 2-D plot for C-Poor (Figure 74) and C-Rich (Figure 75) options, in terms of soil organic carbon stock changes (tonnes/ha), and then combine all the scenarios and options into a 3-D plot (Figure 76).

For the C-Poor scenario option at the EU-27 level the UAA to forest scenario results in a gain of 12 tonnes/ha SOC stock, the use of Set-aside scenario results in a decline of 6 tonnes/ha SOC stock, and the maintenance of grassland scenario results in a decline of 18 tonnes/ha SOC stock (Figure 74). For the C-Rich scenario option at the EU-27 level the change from UAA to forest scenario results in a gain of 20 tonnes/ha SOC stock, the use of Set-aside scenario results in a decline of 2 tonnes/ha SOC stock, and the maintenance of grassland scenario results in a decline of 12 tonnes/ha SOC stock (Figure 75). The adding of the gains and losses results in a 9.7 tonnes/ha SOC stock loss for C-Poor and a 5.0 tonnes/ha SOC stock gain for C-rich (Figure 75).

At the MS level there are large variations of SOC stock losses and gains between Member States (e.g. UAA to Forest is 50 tonnes/ha SOC stock gain for Slovakia compared to 2 tonnes/ha SOC stock gain for Portugal), but between scenario options for a particular Member State there is often a small variation (e.g. Grass to Arable is 18 tonnes/ha SOC stock loss for C-Poor and 15 tonnes/ha stock loss for C-Rich). This explains why when one combines the impact on SOC stock of all land use changes and options (Figure 75) there is much greater differences between MS than between scenario options.

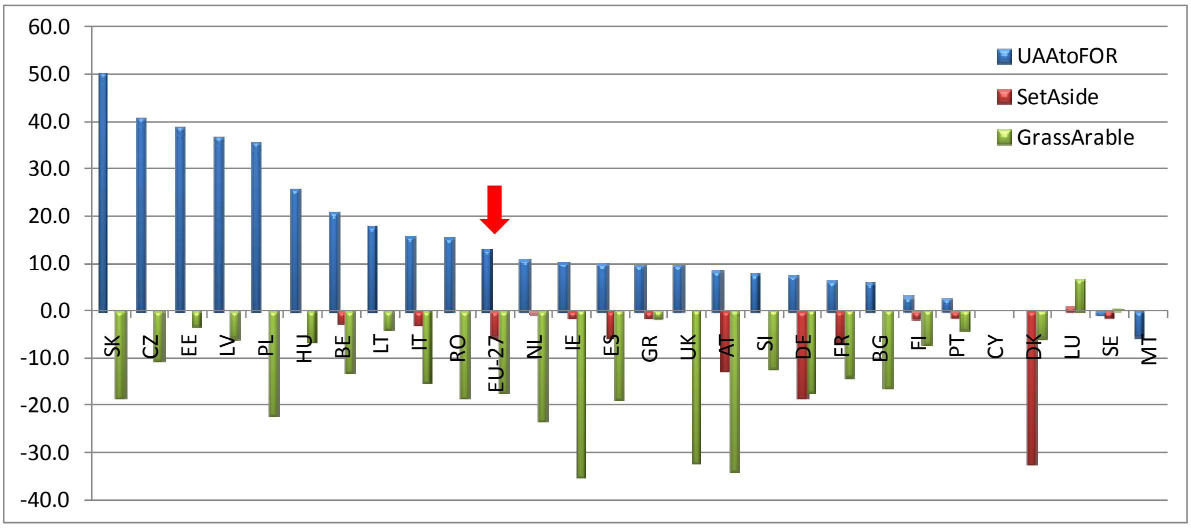



Figure 74 SOC stock changes (in tonnes/ha) due to land use conversions in a C-poor scenario

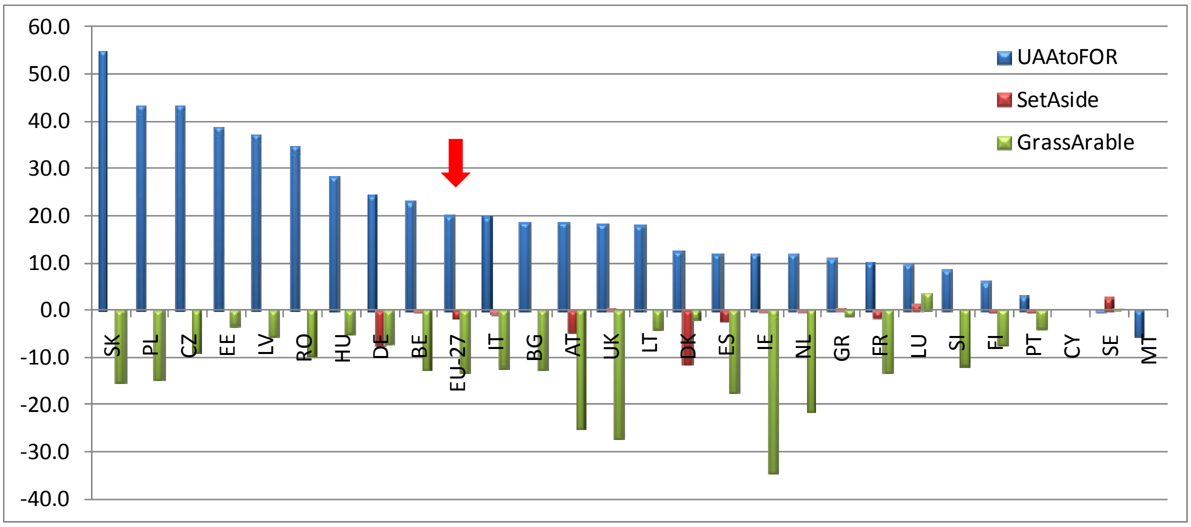



Figure 75 SOC stock changes (in tonnes/ha) due to land use conversions in a C-rich scenario

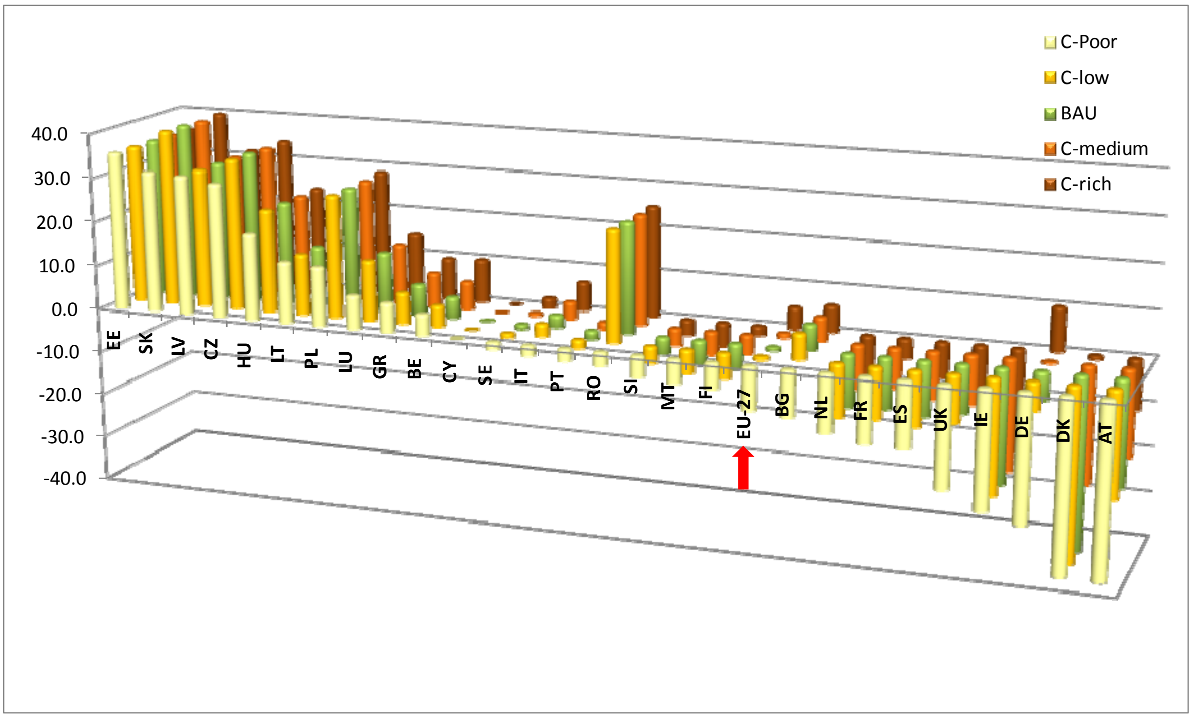



Figure 76 SOC stock changes (in tonnes/ha) due to land use conversions to and from agricultural land, i.e. UAA to forest, arable-grassland conversions and set-aside.

From the overall assessment of the soil organic carbon stocks under agriculture and forests we can draw the following conclusions:

- The mean top soil organic content for arable and grasslands areas indicates a clear gradient with temperature increasing and moisture regime decreasing from northern to southern Europe.
- At the EU-27 level there is on average 31 tons/ha SOC loss when grassland is converted to arable.
- Abolishing permanent pastures GAEC would have a negative effect on soil organic carbon stocks, estimated to be an average loss of 17.2 C tonnes/ha for EU-27.
- With the assumption that the majority of set-aside areas reverted back to arable production means that this policy has led to a negative effect on soil organic carbon stocks, estimated to be an average loss of 5.7 C tonnes/ha for EU-27.
- The combined effect of land use changes to and from agricultural land use in the different scenarios and for different Member States demonstrates an EU-27 average -9.7 tonnes/ha SOC stock loss for the C-Poor option and a +5.0 tonnes/ha SOC stock gain for C-Rich option.
- Converting agricultural land to forest has a positive effect on SOC stocks, but the range is between +54 tonnes/ha for Slovakia (in the C-Rich scenario) to only 2 tonnes/ha in Portugal (in the C-Rich scenario).
- In general the analysis confirms that forest sequesters more carbon than grass and arable; and that grass sequesters more carbon than arable. When considering all the addressed land use changes and respective scenario options we have shown that there are much greater differences between Member States than between scenario options.
- Regional factors, such as bio-geographic characteristics, have a greater impact on soil organic carbon than the presented scenario factors – meaning that policy decisions to improve soil organic matter need to be made at the regional level.

## 11.2 Soil organic carbon fluxes under agriculture

We use two different scenarios to assess the impact of resource management issues on soil organic carbon fluxes under agriculture, with C-Rich, C-Medium, BAU, C-Low, and C-Poor (C-Worst case) options (see Table 5 for scenario option details):

- Use of crop residues and straw; and,
- Use of manure and compost.

The use of crop residues and straw scenario compares the impact on soil organic carbon fluxes for grass, oilseed, cereal and sugar beet. These are presented in terms of Humidified Organic Content in tonnes per ha for BAU 2030 (10% crop residues and straw removed), C-Low (30% crop residues and straw removed), C-Poor (50% crop residues and straw removed) and C-Worst Case (100% crop residues and straw removed) at the EU-27 level (Figure 77).

For the BAU2030 scenario option the grass residues (1.58 tonnes/ha) can provide more than 3 times the levels of HOC than sugar beet (0.5 tonnes/ha). The Worst Case scenario option of removing all residues from the field, reduces the HOC from grass to 0.58 tonnes/ha, but this is still higher than the BAU2030 scenario option for sugar beet. When these crops are grown to provide material to produce bio-energy, the general practice is to remove all biomass from the field. The analysis shows that it is well worth investigating to what degree farmers should be obliged (through cross compliance) or volunteer (through agri-environmental measures) to retain a certain percentage of residues to replenish soil organic matter on the fields if crops are being produced for bio-energy.

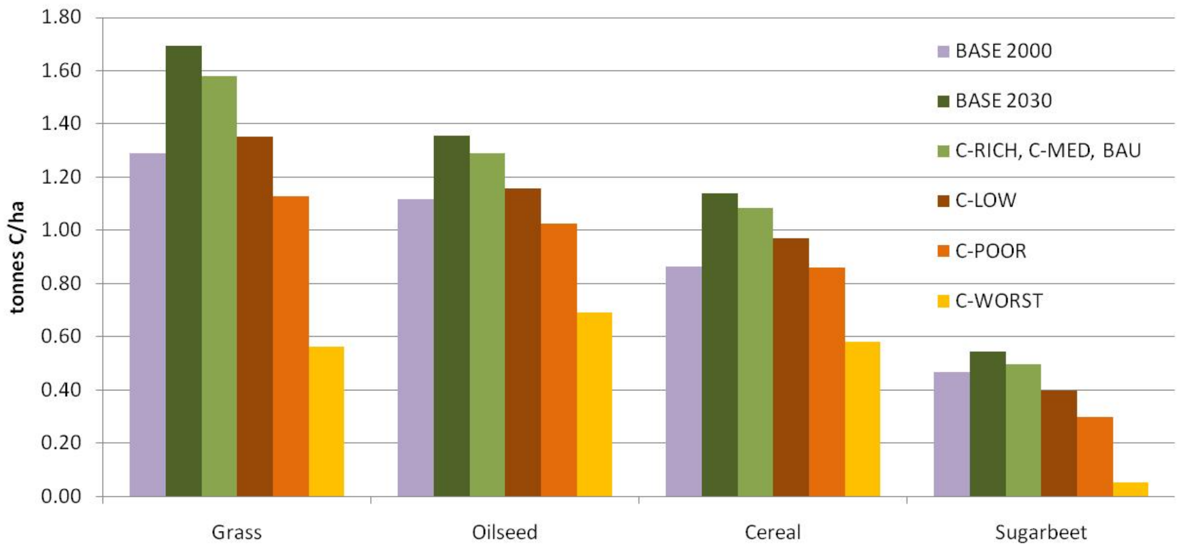



Figure 77 Flux of Humidified Organic Carbon (tonnes per ha) into the soil from grass, oilseed, cereal and sugar beet residues based on BAU 2030, C-Low, C-Poor and C-Worst Case scenarios at the EU-27 level (In Base 2000 all residues remain on the field, whereas in C-worst all residues are removed from the field)

The use of manure and compost scenario compares the impact on soil organic carbon fluxes for kitchen composts, green compost, and livestock manure. These are presented in terms of Humidified Organic Content in tonnes per ha for Base 2000 (compost and manure available), C-Rich 2030 (50% more compost), C-Medium 2030 (25% more compost), BAU 2030 (Compost and manure available), C-Low (20%

manure used for energy), and C-Poor (40% manure used for energy) at the EU-27 level (Figure 78).

In terms of the HOC available manure and green compost in BAU 2030 is comparable, just below 0.15 tonnes/ha, and green compost is more important than kitchen compost in providing organic matter to soils (Figure 78). The livestock manure available declines between 2000 and 2030 due to the general decline in livestock numbers at the EU-27 level. On the other hand the amount of compost available increases between 2000 and 2030 due to the increase in population and the decline in agricultural land. Obviously there remains regional differences with traditionally livestock intensive farming areas, such as Flanders, Britanny, Catalonia having a large supply of livestock manure in 2030, and highly urbanised regions having more access to compost than remote rural regions.

If we compare the amount of HOC generated by crop residues and composts/manures the crop residue management is more important (Figure 77) to maintain soil organic matter levels than the availability of composts/manures (Figure 78) as HOC levels are 10 times higher for BAU 2030. But of course we are assuming that the manure and compost produced is spread across all of the UAA, which is unlikely to be true. On the other it is not an option to apply more than 170 N kg/ha/yr (sometimes derogation allows 250 N kg/ha), because of nutrients leaching to the groundwater. Therefore, regional assessments need to be taken in terms of the availability of livestock manure and the best use of it for supplementing soil organic matter in the context of current legislation.

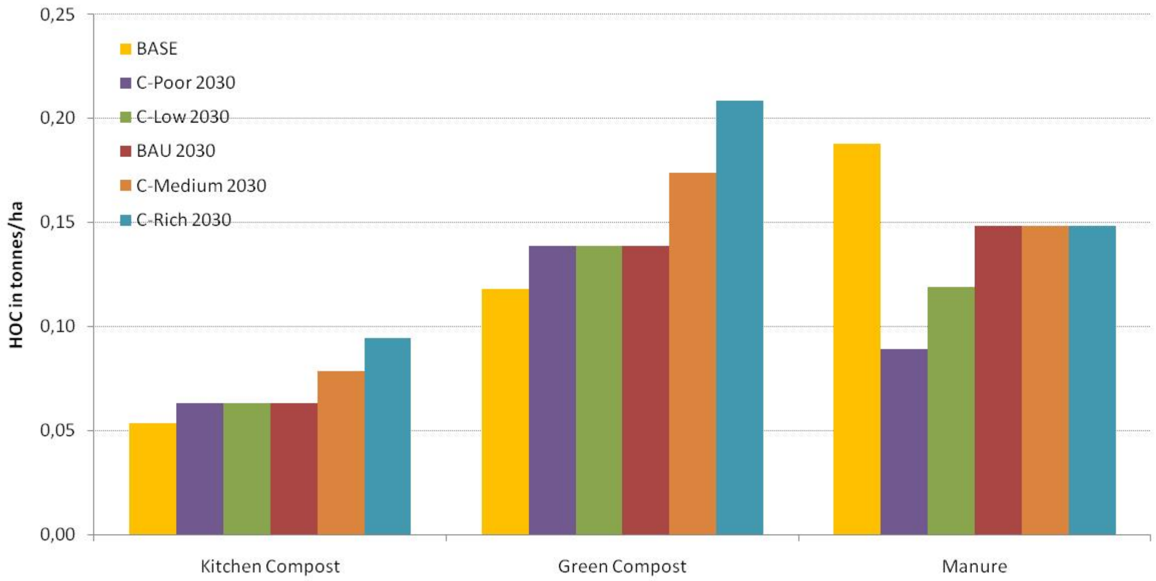



Figure 78 Humidified Organic Content in tonnes per ha from the application of kitchen compost, green compost and livestock manure based on Base 2000, C-Poor 2030, C-Low 2030, BAU 2030, C-Medium and C-Rich scenarios at the EU-27 level

The analysis assumes that manure and compost are applied to the UAA – but it is likely that only farmers close to urban sources will apply compost and farmers with livestock or close to mixed farms will apply manure – due to transportation costs.

### 11.2.1 Crop residues

From the overall assessment of the soil organic carbon fluxes under agriculture we can draw the following conclusions for crop residues:

- The practice of leaving straw from cereals in the field as a residue has the potential of doubling the effective organic carbon input to the soil.
- The ratio of grain to straw on a weight basis for oil seed is higher than for cereals, which results in a 30% higher humified organic carbon.
- Sugar beet has only half the capacity for adding humified organic carbon to the soil as compared to cereal.
- Grass ploughing may release up to 4.5 times the amount of tonnes HOC/ha as compared to regular biomass harvesting. Since the addition provides for an instant fairly substantial flux, temporary grass is a good practice to include arable rotation. However, although grass ploughing provides for an instant addition of large quantities of organic material into the soil and hence humified organic carbon, the soil reserve is more easily exposed to organic matter decline. These factors therefore have to be weighed up in the particular by bio-geographic zone.

### 11.2.2 Compost / manure

From the overall assessment of the soil organic carbon fluxes under agriculture we can draw the following conclusions for compost and manure:

- Densely populated regions have the potential to provide compost for improving the soil organic status of the surrounding farm areas (Belgium more than 600 kg/ha, compared to 120 kg/ha for EU-27), and compost is a good addition to the soil in agricultural land close to urbanised regions, but probably not economically feasible in more secluded areas.
- In general southern Europe, Eastern Europe and Northern Europe are the lowest producers of livestock manure (less than 60 N kg/ha) and Western Central Europe are the highest producers of livestock manure (more than 150 N kg/ha).
- The "use of manure and compost" scenario shows a clear difference between the extreme scenarios of the potential application rate and the reduced application rate when 50% of the livestock manure is used for bio-energy purposes, resulting in reduced soil organic matter levels. Manure and compost is not as important as crop residues in maintaining soil organic matter, however they remain important supplements to the soil becasue they release nutrients slowly to crops and help to improve soil structure.
- Comparing the HOC levels it is clear that at the EU-27 level manure and compost production is not as important as crop residues for maintaining SOM levels. However, at the regional level where in some regions manure and compost production is high (e.g. Northern Belgium, Southern Netherlands) – these sources might be important supplements to the soil, especially if crops and crop residues are being produced for bio-energy. Even more so, as one as to remember that it has been assumed that compost is spread on the entire utilised agricultural area. This is an oversimplification made necessary by the lack of more specific information on compost use possibilities at the regional level. In any event, care has to be taken that nutrient applications do not exceed specific application rates set by legislation.

## 11.3 Soil organic carbon fluxes under forests

We the use of forest residues scenario to assess the impact of resource management issues on soil organic carbon fluxes under forests, with C-Rich, C-Medium, BAU, C-Low, C-Poor, and C-Worst Case options (see Table 5 for scenario option details).

The use of forest residues scenario compares the impact of resource management options on soil organic carbon fluxes for coniferous and broadleaved forests. These are presented in terms of Humidified Organic Content in tonnes per ha for Base 2000 (10% forest residues removed), C-Rich (no forest residues removed), BAU 2030 (10% forest residue removed), C-Low 2030 (20% forest residues removed), C-Poor 2030 (25% forest residues and 10% area stumps removed) and C-Worst Case 2030 (70% forest residues and 25% area stumps removal) at the EU-27 level (Figure 79). At the EU-27 level broadleaved forests provide almost double the amount of HOC made available than conifer forests. In both cases the worst case scenario of using 70% forest residues and 20% area stumps for bio-energy has a large impact on available HOC in 2030, resulting in a 30% reduction in HOC.

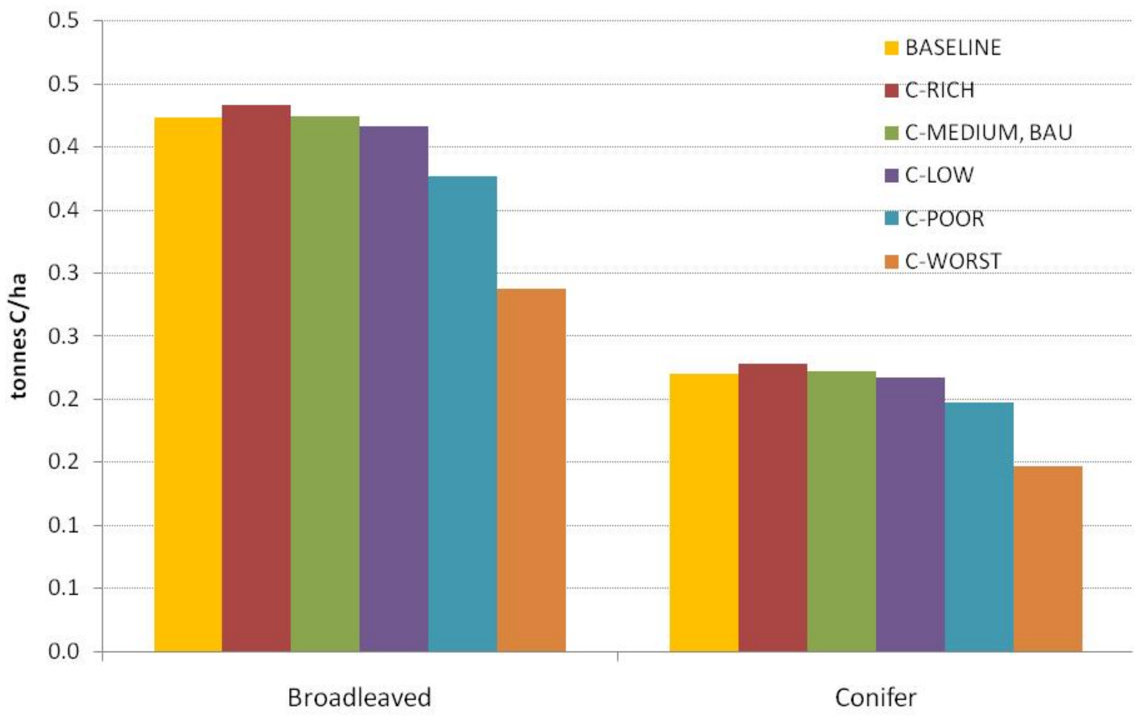



Figure 79 Flux of Humidified Organic Content in tonnes per ha from broadleaved and conifer forest residues to the soil based on Base 2000, C-Rich 2030, BAU 2030, C-Low, C-Poor and C-Worst Case 2030 scenarios at the EU-27 level.

From the overall assessment of the soil organic carbon fluxes under forests we can draw the following conclusions:

- The humified organic carbon (tonnes/ha) for broad leaved forests is 1.75 times that of coniferous forest because of higher decay and assimilation rates. Broad leaved forest can assimilate up to 1.7 tonnes C/ha yearly; coniferous forests up to 1.1 tonnes C/ha. The differences can be attributed predominantly to the share of needles or leaves and fine roots.

- The regional distributions of HOC for broadleaved forest and for coniferous forest not only show the differences in organic matter assimilation into the soil but also show the differences between countries. There seems a bias between the statistics per country. The bias can be explained by differences in measuring and reporting on growing stock and felling, by uncertainties in area occupied by a certain forest type, but also by forest composition (age distribution, tree species), management and environmental factors.
- Countries with an important wood industry such as FI, SE, DE and FR obviously produce most residues from the wood industry and represent the amount of wood waste that could be used for bio-energy purposes. This displaces the need for harvesting felling residues that are valuable to building up soil organic matter, meaning that wood residues could be used instead of felling residues to produce bio-energy, and thereby letting felling residues to be left in situ to enhance soil organic matter build up.
- The composition of forest residues determines the flux into the soil. The contribution of woody residues represents a slow flux into the soil, but provides for an important carbon reserve that adds to the overall forest carbon stock. Woody residues contribute, depending on the scenario, less than one third of the humified organic carbon into the soil.

## 11.4 Soil organic carbon stocks and fluxes under peatlands

There is a great deal of uncertainty in assessing the surface area of peatlands and their carbon stocks, this is due to the fact that peat and peatlands have been defined differently depending on country, scientific discipline and linguistic problems in translating many peat-related terms.
The assessment of the soil organic carbon fluxes under peatlands leads to the following conclusions:

- The estimated peatland carbon stock loss rates in EU-27 range between 0.13 and 0.36% per year which means that 13-36% of the current soil carbon stock in European peatlands might be lost by the end of this century.
- Expressed in C loss tonnes per hectare per year, current carbon losses are around 1.6 tonnes C per hectare and include peat extraction (Figure 80). The scenarios assume no further peat extraction but assume a continued conversion to forest and agricultural land at current rates and at lower rates until the year 2030. No further conversion (C-Rich) results in a loss of 1.4 tonnes C per ha. If 50% of the original peatlands were rewetted, then yearly losses are 0.58 tonnes per ha for EU-27. In the case of 100% restoration 0.23 tonnes of carbon can be sequestered annually.
- Over the period 1990-2007, the average total peat extraction in the European Union amounted to 21 Mt per year. Off-site emissions are estimated to account for 6-16% of the summed GHG emissions from peatlands and peat use in EU-27.
- The BAU 2030 scenario indicates that on the basis of the same historical trends in peatland drainage there will be a 4% decrease in greenhouse gas emissions by 2030, compared to BASE 2000 for EU-27. This compares to a 8% decrease in greenhouse gas emissions by 2030 if the current trends are reduced by 50% (C-Medium 2030) and a 12% decrease in greenhouse gas emissions by 2030, if no further peatlands drainage is allowed (C-Rich 2030).
- For the Best Case scenarios, whereby up to 100% of peatlands are restored, greenhouse gas emissions are further reduced until peatlands become a sink for GHG emissions rather than a source.

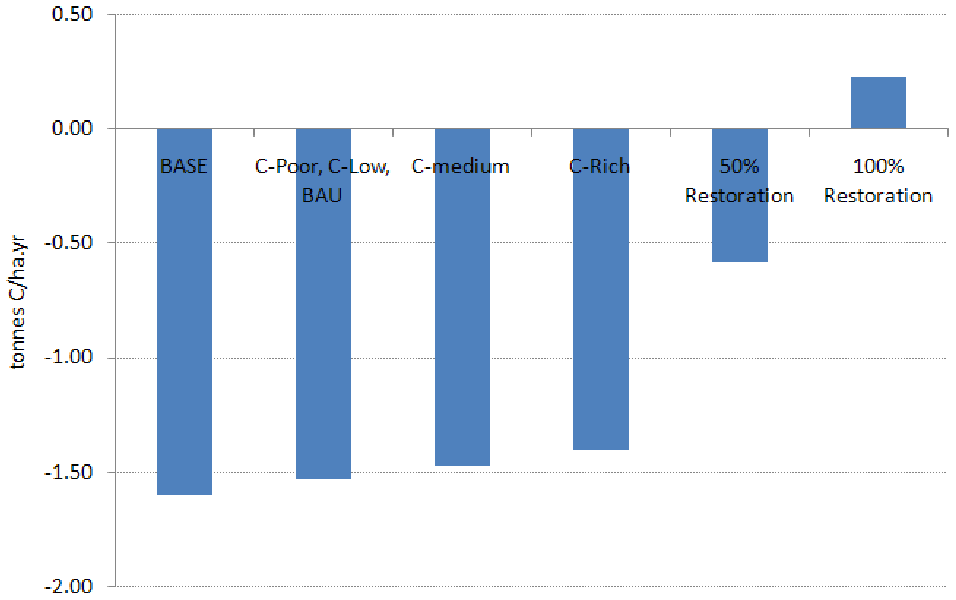



Figure 80 Carbon fluxes from the soil in tonnes per ha per year from peatlands during Base 2000, C-Poor 2030, C-Low 2030, BAU 2030, C -Rich 2030, and 50% and 100% restoration scenarios at the EU-27 level (Positive values are gains, negative values are losses).

# CHAPTER 12 IDENTIFICATION OF BEST PRACTICES IN RELATION TO SOIL ORGANIC MATTER MANAGEMENT ON THE BASIS OF SELECTED CASE STUDIES

## 12.1 Selection of case studies

The project identified 8 case studies addressing different issues concerning soil organic matter from different regions of Europe. Using a standard template and some structured questions to experts with knowledge of the particular case studies an in-depth analysis was undertaken of each case study. The objective was to identify best practices from the case studies that could be relevant at the EU, national, regional and local levels.

The following case studies were analysed:

- The effect of long term crop rotations on soil organic matter status in North Eastern Italy;
- Long term effect of reduced tillage systems on soil organic matter in Northern France;
- Evaluation of crop residue management options on soil organic matter levels in Jutland (Denmark);
- Production and management of compost in Northern Belgium;
- Production and management of sugar-beet composts (vinasse) in South Western Spain;
- Effects of afforestation on arable land in Northern Europe;
- Conservation of mires in Latvia; and,
- Restoration of bogs in Ireland

The in-depth analysis of each case study is provided in Annex III – the analysis is summarised in the following sections, along with identified best practices that could be relevant for other regions (

Table 22).

Table 22 Summary information concerning the selected case studies

| Case Study | University / Institute | Period | Crops represented | Practice |
|---|---|---|---|---|
| The effect of long term crop rotations on soil organic matter status in North Eastern Italy (Experimental Farm of Padova University) | University of Padova, Padova / Veneto (Italy) | 1962 - 2010 | wheat, maize, silage maize, permanent grass, ryegrass, sugar-beet, alfalfa, oat-faba bean-pea, soybean | Continuous maize system |
| Long term effect of reduced tillage systems on soil organic matter in Northern France (Boigneville) | Agronomic National Institute Paris-Grignon & KU Leuven | 1972-2004 | Cereals (maize and wheat) | Effect of convent-ional tillage / superficial tillage / non tillage on soil several parameters |
| Evaluation of crop residue management options on soil organic matter levels in Jutland (Denmark) (Roenhave, Askov and others) | Aarhus University | Varying from 1970s to after 2000 | Spring Barley (and catch crops) | Removal, burning or incorporation of straw into soil |
| Production and management of compost in Northern Belgium | Flanders Compost Network | 1995 - 2005 | Vegetable and arable crops | Two compost trials on vegetable crops and three compost trials on arable crops to analyse the effect on yields but also soil quality |
| Production and management of sugar-beet composts (vinasse) in South Western Spain (Guadalquivir Valley) | Institute for Natural Ressources and Agrobiology of Seville | 1993-1995 | Sunflower, sugarbeet and corn | Vinasse compost applications in field experiments |
| Effects of afforestation on arable land in Northern Europe (Sweden, Denmark, the Netherlands) | Swedish University of Agricultural Sciences (SLU), Dept. of Forest Soils | 1999-2003 | Two differently aged oak stands (Quercus robur L.) and four differently aged Norway spruce (Picea abies (L.) Karst.) stands | Farmland afforestat-ion and the effect on N deposition, nitrate leaching, water balance, C sequestration and retention of SOM |
| Conservation of mires in Latvia ( 4 mire sites) | Latvian Fund for Nature (NGO) | 2004-2008 | Peatland, natural vegetation, afforestation | Practices to protect and manage peatlands according to the Mire Habitat Management Plan for Latvia |
| Restoration of bogs in Ireland (20 blanket bog sites , 14 raised bog sites) | Coillte Teoranta (The Irish Forestry Board) | 2004 – 2008 | Peatland and natural vegetation- | Restoration of raised bog habitats. |

## 12.2 The effect of long term crop rotations on soil organic matter status in North Eastern Italy

The case study focuses on the observations made during a long term (more than 40 years) experiment carried out by the University of Padova. The case study focuses on the findings of the crop rotation trials, and the effects of crop rotation on soil organic matter. Different crop rotations are assessed with maize as the principle crop (6-year, 4-year, 2-year and monoculture. Monoculture resulted in a significant loss of C, and thus had a negative effect on the SOM status. However, the longer and more complex rotations with the addition of farm yard manure had beneficial effects on the SOM levels. This indicates that crop rotations should be combined with other beneficial farm management practices to improve soil organic matter levels. It also shows that many factors that must be taken into account (soil type, organic fertiliser available and its quality (Farm Yard Manure/slurry), crops feasible to cultivate) when trying to predict the effect of an improved crop rotation management system.

**Best practice identified:** crop rotations with the addition of organic fertilisers will increase soil organic matter levels.

## 12.3 Long term effect of reduced tillage systems on soil organic matter in Northern France

The case study focuses on assessing the long term effect (over 30 years) of reducing reduced tillage systems on soil organic matter levels in Northern France. The aim of this study was to quantify the differences in C and N pools when using reduced tillage systems for producing cereals in Northern France. The work focused mainly on: (1) conventional tillage with mould board ploughing to 20 cm depth (CT) and (2) a no-tillage (NT) system. Over a 30 year period using similar N fertilisations maize and wheat crops developed more slowly in the early growing stages in the NT system but the final cereal yields were not statistically different for the two tillage systems. During this period the no-tillage system resulted in 5-15% larger C stocks and 3-10% larger N stocks compared to conventional tillage in the superficial layers of the soil, but these concentrations decreased with increasing depth compared to conventional tillage where they were relatively homogenous through the plough layer. These larger C and N stocks are mainly attributed to an enhanced macro-aggregate formation in the 0-5 cm layer due to higher soil organic matter content and a better protection of this SOM in the 5-20 cm layer due to a larger proportion of small pores and lack of soil disruption by tillage or climate.

Since cereal yields were not negatively affected by long-term no-tillage, the organic C content was higher and this agricultural practice consumes less energy and time compared to the conventional tillage, no-tillage appears to be a cost-saving choice for maize and wheat production under the temperate environmental conditions of Northern France. Other benefits of no-tillage identified included reduced field erosion, cleaner run-off water and decreased soil evaporation due to a crop residue cover. However no tillage systems also entail several disadvantages compared to conventional tillage. First of all, due to the need for weed control, more herbicides are used in no tillage systems. Secondly, the surface mulch of residues promote the presence of parasite and damaging slugs. Soil compaction associated with NT systems cause problems for the establishment or emergence of some crops – and thereby increase runoff. Finally it has to be realised that there is a real requirement for the soil to be ploughed – meaning that NT is entirely unsuitable. Other experimental work indicates that the flux of organic matter under NT is superficial and does not reach more than 5 – 10 cm depth. This means that benefits of reduced tillage systems have to be further assessed.

**Best practice identified:** on appropriate soils reduced tillage systems will increase soil organic matter.

## 12.4 Evaluation of crop residue management options on soil organic matter levels in Jutland (Denmark)

The case study focuses on crop residue management options for barley on the basis of field trials over a 30 year period in Jutland (Denmark). The study evaluates various crop residues management options on soil organic matter levels. Field trials on crop residue management showed that where straw was incorporated, the soil C content is significantly higher compared to the trials where straw is removed and where straw is burned), regardless of the soil type. The annual incorporation of 4-5 t/has straw led to an average increase in soil carbon storage of 13% (over a 30 year period). Straw combined with catch crop growing gave more SOM than when straw alone was incorporated while the addition of pig slurry contributed little to OM accumulation. Compared to straw removal, the annual incorporation of 4, 8 and 12 t straw ha-1 over a period of 18 years caused a relative increase in the SOM level of 12, 21 and 30 %, respectively. In another experiment straw combined with catch crop growing gave more SOM than straw incorporation alone. This has implications for the use of cereals cultivated for biofuels. Conventionally all biomass produced for biofuels is removed from the soil – therefore there are no crop residues to be reincorporated. This means that crops grown for biofuels are detrimental to soil organic matter levels – meaning that alternative organic amendments need to be added to the soil to stop the decline. Crop residue incorporation needs to be combined with other management practices such as adding organic manures, using catch crops (in between the main crops) and having a suitable crop rotation (if possible) to enhance its effects.

**Best practice identified:** incorporating crop residues coupled with organic amendments will increase soil organic matter.

## 12.5 Production and management of compost in Northern Belgium

The study focuses on the production and management of compost on vegetables and arable crops in Northern Belgium. It uses data from the Flanders compost Network for the period 1995 to 2005. The case study evaluates the effect of compost applied as fertiliser on crops (quality and quantity) and on soil quality in Northern Belgium. Compost makes a good soil amendment and an excellent source of organic matter. Compost is the product resulting from the controlled biological decomposition of organic material that has been sanitised through the generation of heat and processed to further reduce pathogens (PFRP), homogenised and stabilised to the point that it is beneficial to plant growth. VFG-compost is compost made from vegetables, fruit and garden waste. Green compost is made solely from prunings, branches, grass and leaf litter. In the Flanders Region compost is allowed to contain a maximum of 25% industrial bio-waste but is in that case regarded by law as fertiliser or animal manure instead of VFG- or green compost. Well prepared compost is free from weed seeds and disease organisms and it offers a well-balanced slow release supply of nutrients. Results showed that an additional 0.15 to 0.33% SOC content can be expected after 10 years of compost application at a rate of 10-15 t/. The carbon content in soils doubles after 10 years at higher application rates and increases with 50% at lower rates. More earthworms occur as indicator for increased soil biological life and water retention increases as well. Yearly applications of small doses (e.g. 15 t/ha/a) have a superior effect on C-sequestration as compared to an accumulated application every 3 years (e.g. 45 t/ha/3a), because the compost is better incorporated in the soil. However the majority of compost produced is used for urban green spaces as farmers are reluctant to use composts not produced on their own farms because of concerns about the addition of trace elements (particularly heavy metals) or other substances (such as pesticide residues) to farmland.

**Best practice identified:** applying compost increases both the quanity and quality of vegetables and arable crops. Yearly applications of compost are more efficient at C-sequestration than the same total amount applied every 3 years. Need to improve compost quality guarantee labels to increase farmer uptake.

## 12.6 Production and management of sugar-beet composts (vinasse) in South Western Spain

The case study focuses on the production and management of sugar-beet composts (vinasses) in South Western Spain. Crop trials of vinasse applied to sunflower, sugar-beet and maize were conducted over a two year period. The case study evaluates the effect of three vinasse composts applied as a deep fertiliser on crops and the effect on some chemical properties of the soil. Vinasse is a dark brown effluent with high organic matter content generated during distillation of alcohol from sugar beet. Experiments showed that the use of vinasse compost increased both the soil organic matter and the yields significantly in comparison to a conventional inorganic fertliser treatment. Despite the salinity and the sodium content of the composts, no sign of salinisation or sodification of the soils was observed after two years of experiments – but this of course is an insufficient period to assess this process. It is therefore recommended to continue to monitor the soil of these fields to be sure that no problem would appear after a decade or more. The scientists involved in the experimental work are very positive about using vinasse as a compost – at the moment it is generally free as it is considered a waste product, but only farmers close to sugar refineries can access vinasse. Presently farmers are still suspicious about using it on their fields – probably because, as in Belgium, farmers are reluctant to apply composts/wastes that have not been generated on their own land, which still remains their most valuable capital.

**Best practice identified:** need to improve compost quality guarantee labels to increase farmer uptake.

## 12.7 Effects of afforestation on arable land in Northern Europe

The case study focuses on the effects of effects of afforestation on former arable land in terms of carbon storage and soil organic matter (SOM) levels in Northern Europe. The case studies are in Sweden, Denmark and the Netherlands – monitoring oak and spruce stands over a 4 year period. A land-use change from cropland to forestry may provide benefits to SOM; the decomposition rate of organic matter is expected to decrease as a result of changed temperature and moisture conditions, the introduction of specific litter types, and the cessation of frequent soil cultivation. The sites investigated included two oak (Quercus Robur) and four Norway spruce (Picea Abies) stands in chronosequences aged 1 to 90 years in Sweden, Denmark and the Netherlands. For the total forest compartment including tree biomass, forest floor and mineral soil, C sequestration was evident at all the sites. Out of the total C increase for the chronosequences, two thirds were observed to be accumulated in the tree biomass. Between 0 and 31% of total C was accumulated in the soil compartment (forest floor and mineral soil plough layer). With an annual average sequestration of 0.8 Mg ha-1*yr, the C sequestration in the soil compartment ranged from minus 0.24

(Vestskoven, DK) to plus 1.26 (Oak/spruce, NL) Mg ha-1*yr. In all sites there was an increase in biomass and forest floor C levels following afforestation of agricultural land. For the total forest compartment (including tree biomass, forest floor and mineral soil) C sequestration was evident at all sites. Of the total C increase two thirds were observed to be accumulated in the tree biomass, with the rest being accumulated in the soil compartment. Key factors for the success of afforestation involve the right choice of management methods, the choice of wood species and the soil type, as well as the existence of a market for the wood products (including bio fuels). The changes in nutrient status as well as the economy of afforestation depend also on the land use before the afforestation (marginal land versus pasture or high-quality arable soil). In terms of afforestation uptake by farmers – this depends on local conditions such as climate and soil type, and most importantly, on availability of suitable land and land resources local economic conditions. One advantage for the farmer may be a long term economic benefit if making use of marginal, less productive land, since forest can generally grow on less fertile soils than agricultural crops. Barriers to afforestation for the individual farmer may also include: a lack of knowledge, since forestry, similar to farming, takes specific practical and theoretical skills; the high initial cost for equipment, plant material and labour; and traditions and cultural factors. The benefits of afforestation for farmers are therefore not so obvious for farmland that is highly fertile and productive. It is consequently most likely to be a candidate for land use change on farmland that has been abandoned because it is is marginal and unproductive. Successful farmers are unlikely to make the switch to being foresters unless afforestation for biofuels is the goal (e.g. short rotation coppices) and this represents a move to diversify farm income.

**Best practice identified:** Afforestation has the largest impact on increasing soil organic matter levels when carried out on marginal lands with low organic matter levels.

## 12.8 Conservation of mires in Latvia

The case study aims to make an overview of the best practices occurring in projects being carried out in Latvia to conserve important mire sites. The Cena Mire was the second largest bog in Latvia covering 10 600 ha before peat extraction began. Drainage activities began at the site in 1933 and peat extraction was initiated in 1940. Currently, only a fifth or 2 133 ha remain as a protected area. The conservation of this remaining part has ensured raised water levels which have helped to reintroduce functioning hydrological cycles and the re-establishment of wild species, thus restoring to a certain extent the biodiversity. The raising of stakeholder awareness around the mires, including municipalities, private companies, forest services, etc. was an important step. The step enabled them to become interested in the project and recognize the benefits for biodiversity. The rewetting of mires by the construction of small dams to retain water was needed to ensure the protection and conservation of their functions. Carbon monitoring has not been carried out during the project – as it is very slow and costly to implement.

**Best practice identified:** the raising of stakeholder awareness around mires, including municipalities, private companies, forest services, etc. is an important step to generate interest and support for conservation developments.

## 12.9 Restoration of bogs in Ireland

The case study aims to make an overview of the best practices occurring in projects being carried out in Ireland to restore raised/blanket bogs. Peatlands in Ireland take the form of bogs and fens. The bog is one of Ireland's most characteristic features. Bogs cover about one sixth of Ireland (1 200 000 ha). In Ireland industrial scale exploitation of bogs for fuels has greatly contributed to the reduction in bog area. Two distinct types of bogs exist: blanket bogs, expansive, generally formed in wet or upland areas, typically found in western Ireland and mountainous areas; and raised bogs, smaller, generally formed in lowland areas, and found almost exclusively in central Ireland (the Midlands). In most cases blanket bogs have been afforested so the two main steps are thus the removal or felling of tree plantations and the blocking of forestry drains to restore water levels and facilitate the growth of bog vegetation. So here we have an example of forests actually being removed and bog vegetation being restored. Different techniques were tried out during the blanket bog restoration project. As regards tree felling windrowing was considered to be the best technique. In terms of blocking drains and raising water levels peat dams were more cost effective than plastic dams.

**Best practice identified:** the most effective way to ensure that carbon is stored in peatland soils is to reduce land-use changes (to agriculture and forestry) and the extraction of peat (for energy and horticulture).

# CHAPTER 13 RECOMMENDATIONS

Soil is a fundamental natural resource - it provides many essential services on which we rely including food production, water management and support for valuable biodiversity and ecosystems. As a large store of carbon it also plays a vital role in the fight against climate change. Farmers, foresters and other land managers manage the majority of our soils therefore these recommendations need to be discussed with stakeholders and the representatives of our soil managers to provide incentives to improve the management of soil organic matter in soils.

On the basis of the analysis undertaken during the project in terms of the literature review of soil organic matter issues, the estimation of the baseline levels of soil organic matter across Europe for the main sectors (agriculture, forest, urban fabric and peatlands), analysing the effect of different scenarios to promote soil organic matter in soils or encourage the use of biomass for energy production, and the information garnered from selected case studies concerning generally local, region specific issues related to soil organic matter levels, we have come up with the following policy recommendations:

1. Bio-geography and pedology are important factors in determining the levels of soil organic matter across Europe, showing that **practices need to be adapted to regional conditions to be most effective**. Policy decisions at the regional level have to take this into account.

2. We have shown that crops or forests grown for bio-energy production, whereby all residues are removed, is detrimental to the soil, resulting in a reduction of soil organic carbon stocks and an increase of carbon dioxide concentrations in the atmosphere. Therefore, we recommend that **a (significant) minimum percentage of residues should be retained in soils for crops and forests grown for bio-energy**. Further work needs to be done to set such minimum percentage values, which could vary between bio-geographic regions as well as crop and forest types. These standards could be introduced through cross-compliance measures or standards for good agricultural and environmental conditions (GAECs) for crops under the Common Agricultural Policy and/or the use of standards or labels for crop and forestry products used for bio-energy production.

3. The policy implications for compost and livestock manure are also highly regional. Densely populated regions have the potential to provide compost for improving the soil organic status of the surrounding farm areas, however the cost implications of transporting urban produced compost need to be taken into account. Livestock manure can only be used for bio-energy production in highly intensive livestock rearing regions. In these regions, **bioenergy production can be seen as an added environmental benefit for manure** that has to otherwise be kept in storage facilities that are built to reduce N emissions. Indeed farmers should be encouraged to use liquid manure for producing bio-energy and then **transforming this bi-product into a compost rather than spreading or injecting liquid raw manure into the soil.** The case study work indicates that farmers are not keen to add composts to fields when they are not confident of the quality – so improved standardization or quality labels need to be

introduced. However, for both manure and composts, care still has to be taken that nutrient applications do not exceed specific application rates set by legislation.

4. Concerning peatlands we see that the current land use conversion and peat extraction rates enhance drainage and decomposition, thus increasing greenhouse gas emissions, and that the restoration of peatlands turns them from a carbon source into a carbon sink. This means that **the conservation, restoration and management of peatlands should be an important environmental policy concern in terms of both retaining peatlands as a key land use to reduce or even reverse carbon dioxide and also methane emissions**. It is clear, therefore, that peatland drainage, for example for agriculture and forestry, needs to be stopped and reversed, to prevent further emissions. This has implications for Climate Change policy and negotiations, but also for policy measures in the Common Agricultural Policy and NATURA 2000.

5. There is a need to increase the understanding of complex relationships in the soil carbon cycle. There are significant challenges in coming up with cost effective techniques to measure soil organic carbon changes efficiently. Climate change but especially – as this report shows - land use practices and land use changes are likely to have a significant influence on soil carbon stocks and will make it more difficult to predict the sequestration potential of soils and its permanence. **Soil monitoring is therefore vital to provide evidence on the state of, and change, in our soils**, underpinning policy development and allowing to evaluate its effectiveness. This means developing a set of soil quality indicators and new biological indicators of soil quality.

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# Annex I Description of LUMOCAP

The LUMOCAP Policy Support System (PSS)[19] is an operational tool for assessing how different policy scenarios will impact the land use and landscape in the 27 Member States of the European Union (EU-27). It focuses on the relations between EU policies, such as the Common Agricultural Policy (CAP), and landscape changes and emphasizes the spatial and temporal dimension of this process. Because of the inherent complexity of land use change processes, policies at the European level targeted at one sector (e.g. agriculture) have their effect not only on developments in this sector, but also on for instance regional ecological coherence and socioeconomic dynamics of rural areas. This means that a model for policy impact assessment should reach beyond EU agricultural policies and include policies and processes at other levels and sectors such as local zoning regulations, infrastructure planning and interaction between sectors as well as external factors like climate change and socio-economic drivers. The LUMOCAP PSS allows investigating the relation between EU policies, agricultural economics, land suitability and land use dynamics through dynamic simulation. It incorporates an integrated model, tools to set up scenarios for (a combination of) policy measures and external factors and tools to visualise and analyse indicators (Figure 81).

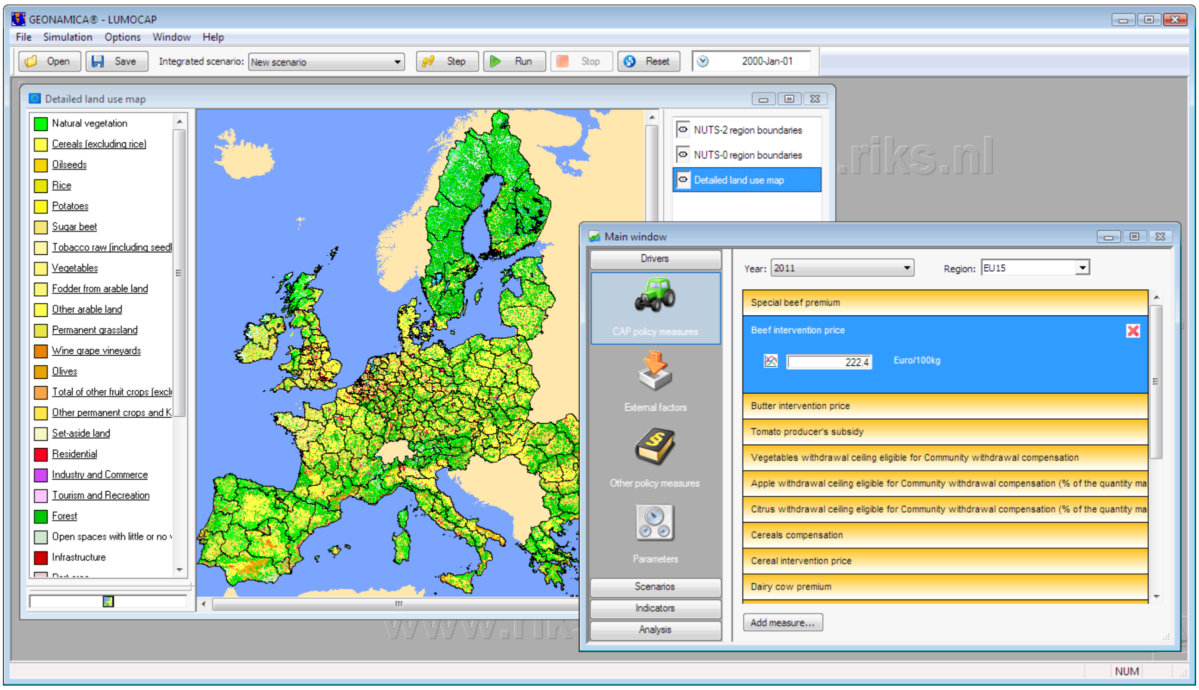


Figure 81 Screenshot of the LUMOCAP PSS

In Figure 81 the measures can be entered for specific years and regions. Depending on the spatial level policies impact on, instruments are entered as numerical values or maps. Results can be found on the land use map shown in the back and through the indicator section in the main window.

The core of the LUMOCAP PSS consists of a selection of models, all linked into a single integrated model simulating the linked bio-physical and socio-economic developments in the entire European Union (EU-27) up to 30 years into the future. To capture processes occurring at different spatial scales, the system includes models operating at three different levels: EU-27, country (the Member States of EU-27), and local. At local

[19] For more information on LUMOCAP: http://www.riks.nl/projects/LUMOCAP

level the system operates at a 1 km grid covering the area of the entire EU. The temporal resolution of the system is a year, its temporal horizon 2030. The different models and their linkages are schematised in the system diagram in Figure 82. A short overview of the models and their interactions is provided below. A more elaborate description is provided in Van Delden et al. (2010)[20].

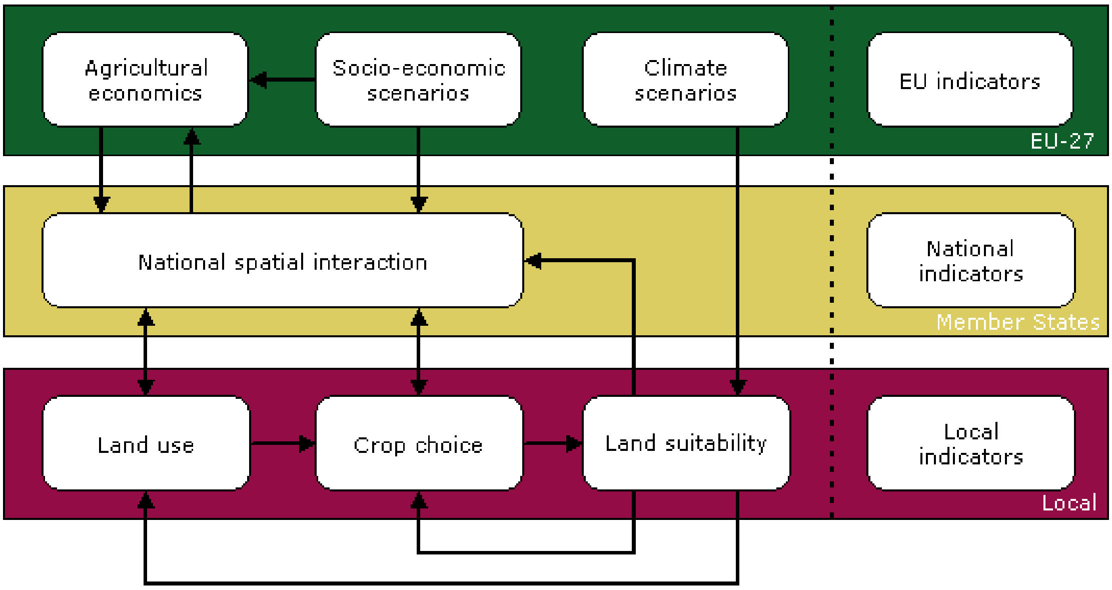


Figure 82 The LUMOCAP system diagram.

*The integrated model*
As can be seen from the system diagram, models are incorporated in LUMOCAP at various spatial levels:

- At the highest level of the model, the entire European Union, an agricultural economic model calculates the acreages per crop type, yields and production. Since crop area in main regions is driven by different mechanisms, different equations for EU-15, EU-10 and EU-2 are used to reflect this. Furthermore expectations regarding growth of population, GDP and jobs are seen as external driving forces and can as such be entered and/or adapted in the socio-economic scenario component. Finally, climate change scenarios are included at the highest spatial level.
- A spatial interaction model distributes population, jobs and hectares for different crop types from EU level to the individual Member States. The relative attractiveness of the Member States plays a crucial role in the migration and distribution of activities and the allocation of crop areas. At national level, activities are converted to land use demands.
- Within the Member States a constrained cellular automata model (Metronamica[21,22]) allocates the demand for the different land use functions to cells of 1x1 km$^2$. At this level there are 3 models available and interacting with each other:

[20] Van Delden, H., Stuczynski,T., Ciaian, P., Paracchini, M.L., Hurkens, J., Lopatka, A., Shi,Y., Gomez Prieto, O., Calvo, S., Van Vliet, J. and Vanhout, R. (2010). Integrated assessment of agricultural policies with dynamic land use change modelling. Ecological Modelling.
[21] www.metronamica.nl
[22] RIKS, 2009. Metronamica – Model descriptions. RIKS, Maastricht, The Netherlands.

- a land use model that allocates the broad land use categories, such as residential, industry & commerce, forest, agriculture and natural vegetation to the grid cells. The allocation takes place based on spatial policies, existing practices, neighbouring land uses, physical characteristics and accessibility;
- a crop choice model that allocates the total demand for each crop type from Member State level into agricultural cells, based on existing practices, physical characteristics of each location and spatial policies such as the Less Favoured Areas policy. Agricultural cells can be occupied by a combination of crops and crop shares per cell are calculated;
- a dynamic suitability model that calculates the land suitability of each 1 km cell based on static factors and the impact of climate change. Physical suitability is combined with a country specific technology component (GDP is used as a proxy) to incorporate differences in management practices in different EU Member States. Aggregate suitability information is used as one of the factors determining the attractiveness of countries in the spatial interaction model at national level and in the agricultural economic model at EU level. In these models it is - together with technology factors - used as a proxy for yield. The suitability of locations is used in the land use and crop choice models for the land use and crop allocation respectively.

*Policy support and usability*
LUMOCAP has the characteristics of a Decision or Policy Support System (DSS/PSS): it is a flexible, transparent, PC-based analytical system, enabling to interactively choose, via a user-friendly interface, policy options under a specific set of natural and socio-economic conditions, as external driving forces, to formulate potential land use scenarios, and to assess their impacts on the quality of rural landscapes through selected landscape indicators. Consequently, the developed modelling framework allows the identification of areas of adverse land use changes caused by non-sustainable agricultural systems.

To ensure the policy-relevance of the LUMOCAP system, policy measures at different levels are included which can be entered or adapted by the user. Examples of these policies are CAP measures from pillar 1 at EU level, Rural Development Policies at national level and Less Favoured Areas, Natura2000 and construction of infrastructure at local level. Since the impact of policy interventions depends heavily on drivers that cannot be defined by policy makers, LUMOCAP also enables the user to select different external factors such as climate change and socio-economic scenarios to analyse the sensitivity of the policy options under different conditions. On the output side the model results are converted into policy-relevant social, economic and environmental indicators.

To facilitate the use of the system for different users with different needs, the graphical user interface (GUI) is split into a policy interface for impact assessment and a modeller interface for updating data and calibration parameters and for fine-tuning the models incorporated in the system. The policy interface only gives access to a limited number of drivers and indicators; the modeller interface gives access to all data and parameters and to most of the equations.

*Acknowledgements*
The LUMOCAP PSS has been developed as part of the LUMOCAP project (Dynamic land use change modelling for CAP impact assessment on the rural landscape), a research project financed under the Sixth Research Framework Programme of the EU on Sustainable Development, Global Change and Ecosystems.

LUMOCAP is developed using the Geonamica software environment that is specifically designed for the development of spatial decision support systems that integrate a number of non-spatial and spatially explicit models[23].

[23] Hurkens, J. Hahn, B.M. and Van Delden, H. (2008). Using the GEONAMICA® software environment for integrated dynamic spatial modelling. In: M. Sànchez-Marrè, J. Béjar, J. Comas, A. Rizzoli and G. Guariso (Eds.) Proceedings of the iEMSs Fourth Biennial Meeting: "Integrating Sciences and Information Technology for Environmental Assessment and Decision Making". International Environmental Modelling and Software Society, Barcelona, Spain, 2008.

# ANNEX II CASE STUDIES

Annex II is a separate document titled: “Soil organic matter management across the EU – best practices, constraints and trade-offs Annex II Case Studies”.

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