SCIENTIFIC SUPPORT PLAN FOR A SUSTAINABLE DEVELOPMENT POLICY

SPSD II

MODELLING ECOSYSTEM TRACE GAS EMISSIONS (METAGE)

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ATMOSPHERE AND CLIMATE

MARINE ECOSYSTEMS AND BIODIVERSITY

PART 2

GLOBAL CHANGE, ECOSYSTEMS AND BIODIVERSITY —— TERRESTRIAL ECOSYSTEMS AND BIODIVERSITY

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SCIENTIFIC SUPPORT PLAN FOR A SUSTAINABLE DEVELOPMENT POLICY
(SPSD II)

Part 2:
Global change, Ecosystems and Biodiversity

FINAL REPORT

Modelling Ecosystem Trace Gas Emissions

(METAGE)

EV14

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January 2006

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D/2007/1191/34
Published in 2007 by the Belgian Science Policy
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Wetenschapsstraat 8
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LIST OF ABBREVIATIONS

A1FI	Global change scenario: world markets and fossil fuel intensive (Nakicenovic et al., 2000)
ALSU	Alternative LSU
ATEAM	Eu project 'Advanced Terrestrial Ecosystem Analysis and Modelling' (for details see: www.pik-potsdam.de/ateam/)
B1	Global change scenario: global sustainability (Nakicenovic et al., 2000)
BAU	Business as usual
BD	Bulk density (kg m-3)
BEF	Biomass expansion factor
BIO	Microbial biomass
BOC	Biomass organic carbon
C:N ratio	Carbon to nitrogen ratio
CAP	Common agricultural policy
CDM	Clear development mechanism
CI95	95% confidence interval
CLC-1990	Corine land cover for the year 1990
CO_2-eq	Carbon dioxide equivalents: i.e same global warming potential as a ton of CO2
COP	Conference of Parties
DPM	Decomposable plant material
EFOBEL	Forest biomass dynamics simulation model (Perrin, 2005)
EOH	Ectorganic horizons
EU15	European Union countries before the extension in 2005
FORSITE	Soil profile data base for forests in Flanders
FYM	Farmyard manure
GCTE SOMNET	Global network for modelling and calibration of soil organic carbon (www.rothamsted.bbsrc.ac.uk/aen/somnet)
GHG	Greenhouse gas
GIS	Geographical information system
GPG	Good Practice Guidance
GWP	Global warming potential compared to a ton of CO2
HADCM3	Global circulation model (Gordon et al., 2000)
HIBBOD	Soil profile data base for forests in Flanders
HUM	Humified organic matter
IMAGE 2,2	Global supply and demand model for agricultural goods and services
IOM	Inert organic matter
IPCC	Intergovernmental panel on climate change
LOI	Loss on ignition
LSU	Landscape unit
LU	Livestock unit
LULUCF	Land use, land use change and forestry
Mcrops	Empirical model for N2O emission from cropland (Roelandt et al., 20005)
Mgrass	Empirical model for N2O emission from grassland (Roelandt et al., 20005)
MIRA-T	Milieu rapport vlaanderen (T: annual evaluation)
N fert	Nitrogen fertilisation
N_2O	Nitous oxide
NIS	National Istitute for Statistics
NUE	Nitrogen use efficiency
NVZ	Nitrate vulnerable zone
OM	Organic matter (%)

P summer	Summer precipitation
RMSE	Root mean square error
RothC-26,3	Soil carbon dynamics simulation model (Coleman et al., 1996)
RPM	Resistant plant material
SD	Standard deviation
SOC	Soil organic carbon, expressed either in tC ha-1 or in Mt C
SRES	Special report on emission scenarios
T spring	Spring temperature
T summer	Summer temperature
T winter	Winter temperature
TOC	Total organic carbon
UNFCCC	United Nations Framework Convention on Climate Change
VMF	Bulk density of the mineral fraction (kg m-3)
VOM	Bulk density of organic matter (kg m-3)
YASSO	Soil carbon dynamics simulation model for forests (Liski et al., 2002)

ABSTRACT

The main aim of the METAGE (Modelling Ecosystem Trace Gas Emissions) project was to develop a spatially explicit approach to estimate greenhouse gases (GHGs) from terrestrial ecosystems under global change scenarios. The first part of the project deals with the determination of carbon stocks and GHG fluxes in (semi) homogenous spatial units covering the non-built up area of Belgium (81 % of the total surface). The data collected for these spatial units is then used in the second part of the project to evaluate the performance of carbon dynamics models at the regional scale. Finally, maps of the climate and land use for the period 2020 and 2050 are constructed under one of the IPCC global change scenarios. GHG fluxes are then calculated for the spatial units of these future climate and land use scenarios. The following GHG fluxes were considered: CO_2 fluxes as a result of changes in soil organic carbon (SOC) or biomass organic carbon (BOC) in forested areas and N_2O fluxes from agricultural soils.

The spatial units, **landscape units** (LSUs), were obtained from an intersection of the Corine land cover data set of 1990 and the soil association map. These units are characterised by rather homogeneous soil, climate, land use and land management characteristics. All fluxes of GHGs are consistently reported for these units. Multiple matching procedures were applied to characterise the LSUs from heterogeneous data sets. The following datasets were used: soil profile observations (1950-1970), routine soil analysis (1990-2000) and forest inventories (1984 and 2000).

Overall the **SOC** stock (0-30cm) of terrestrial ecosystems in Belgium increased from 138 ± 1.2 Mt C in 1960 to 157 ± 0.003 Mt C in 1990. From 1990 to 2000 the total SOC stock remains more or less constant. However, the evolution of the total SOC stock does not allow to detect trends according to land use or region. Grasslands sequestered CO_2 between 1960 and 1990, and became a CO_2 source from 1990 to 2000. Croplands emitted CO_2 in Wallonia (1960-2000) and in Flanders (1990-2000), but sequestered CO_2 in Flanders from 1960 to 1990. With regard to agriculture, changes in application practices of animal manure, as driven by economic factors and policy, are found to be the major explanatory factor for the observed changes. Using the regional scale manure input data in a soil carbon simulation model (RothC) the important increase in average SOC stock for grassland was correctly predicted. However, the model did not correctly predict the slight decrease in average SOC for croplands from 1960 to 2000 and neither the decrease from 1990 to 2000 for both grasslands and croplands. Overall the deviation between modelled and observed data (RMSE) is restricted to 1.5-1.6 % of the SOC stock in an LSU. From a confrontation of SOC data with historical land use change data from the National Institute for Statistics, it is suggested that recent land use changes do not unacceptably distort the results. All those findings must be put in the context of a possibly important source of error in SOC inventories: the different OC measurement techniques applied and their associated correction factor.

Total **BOC** stocks in forested LSUs amount to 57.8 ± 9.1 Mt C in 2000, with the highest per ha stocks residing in broadleaf forest (100 t C ha^{-1}) and the lowest in mixed forest (87 t C ha^{-1}). Forest in Flanders contains on average less BOC than in Wallonia and this may

be explained by the lower average forest age in Flanders. A forest growth model (EFOBEL) was developed for Belgium and the simulations indicate that the BOC stock will increase with 1.5 % to 2.0 % each year under a business as usual scenario. BOC stocks will then increase from 93 t C ha^{-1} in 2000 to 110 t C ha^{-1} in 2012.

Multivariate linear relationships were developed that link **N_2O** emissions to climate, land use and N-fertilisation. Two empirical models were presented for cropland (MCROPS) and grassland (MGRASS) emission patterns. The analysis demonstrated that seasonal climate has an impact on the annual N_2O emissions. Emissions from crops were found to be sensitive to spring temperature and summer precipitation. The grassland-emission pattern is, however, driven by nitrogen-fertilisation and winter temperature. Compared against independent data sets, the empirical models were able to explain 35% of the variance in annual N_2O emissions for croplands and 48% of the variance for grasslands. Based on a larger data set, MCROPS and MGRASS improve the statistical reliability compared to the IPCC default methodology. When these empirical models were applied to the LSUs, clear spatial patterns occurred with higher emissions in Flanders. As a result of the inter annual climate variability, N_2O emissions ranged from 1.1 - 2.2 kg N_2O-N ha^{-1} y^{-1} in Walloon grasslands to 3.2-6.6 kg N_2O-N ha^{-1} y^{-1} in croplands.

The implications of one of the IPCC **climate and land use scenarios** (A1 FI) on GHG emissions were assessed using simulation models (RothC, EFOBEL, MCROPS and MGRASS). The A1FI scenario is a globally and economically oriented scenario also referred to as 'world markets-fossil fuel intensive'. Land use change was taken into account from 2012 onwards by means of new LSUs based on a predicted land use map for 2050. According to the A1FI scenario, the areas of forest (20 % of the total surface) and built up land (22 %) will increase compared to the Corine 1990 land cover data set, while grassland (8 %) and cropland (42 %) will decrease and biofuels (5 %) will appear. The net increase in SOC and BOC stocks results in an annual sequestration of 0.95 Mt CO_2 from 2000 to 2012 (before the land use change) and 0.52 Mt CO_2 from 2012 to 2050 (after the land use change). This does not include an additional sequestration of c. 2.52 MtCO_2 yr^{-1} as a result of fossil fuel substitution by the burning of biomass. The C-sequestration potential for the period 1990-2000 was estimated at 2.85 MtCO_2 yr^{-1}

Since all GHG fluxes are calculated for the LSUs, a **GHG synthesis** can be produced expressed in CO_2 equivalents by using the global warming potential of the different gases. Overall a limited net sink of 0.66 Mt CO_2eq y^{-1} was registered from 1990 to 2000. SOC stocks in grassland and cropland decrease and these ecosystems also emit N_2O. Forest biomass and soils under forest sequester C, leading to a net sink under forest. Clear regional differences in GHG emissions occur with GHG emissions in the northern and middle part of the country dominated by intensive agriculture and net C sequestration restricted to the areas dominated by forest and grassland. N_2O emissions remain important under global change scenario and are in the same order of magnitude as C- sequestration (1.9-2.24 Mt CO_2eq. yr^{-1}). Under the A1FI-BAU scenario, terrestrial ecosystems gradually become a net source of GHGs after 2012 (–1.72 Mt CO_2eq. yr^{-1}). The decline in C-sequestration over time can largely be attributed to the slowing down of the growth rates of the forests.

The spatial inventory system based on the LSUs allows calculating stocks and fluxes for specified areas, administrative regions or land use type. A GIS based inventory system was developed, which allows the user to query the METAGE datasets and produce inventories according to criteria selected by the user. This system is available from the link on the METAGE website (www.geo.ucl.ac.be).

Key words: Greenhouse gas emissions, soil organic carbon, biomass organic carbon, land use, GIS, global change scenarios, emission factors, carbon sequestration, heterogeneous data sets, regional GHG inventories

1 INTRODUCTION

1.1 A SPATIALLY EXPLICIT APPROACH TO GREENHOUSE GAS INVENTORIES

Numerous studies have contributed to the understanding of the processes determining greenhouse gas (GHG) fluxes both at the global and local scale. However, policy-makers require information at intermediate scale levels (i.e. the national or regional scales) in order to make informed judgments about the effectiveness of different strategies for sustainable management of terrestrial ecosystems.

Currently, the inventory of the fluxes of the major GHG, i.e. CO_2, N_2O and CH_4, emitted by agriculture, land-use change and forestry is already carried out by the regions of Belgium under its obligations to the Kyoto Protocol. They use an internationally accepted standard methodology (IPCC, 1997). In the beginning, the fluxes calculated from these guidelines had a number of limitations and uncertainties related to the aggregation at regional scale the use of world-wide default values parameters, the calculations based on simple empirical expressions and the insensitivity to changes in climate, land-use and land management and atmospheric deposition. Fortunately, the guidelines have evolved in particular in the area of Land Use, Land Use Change and Forestry (LULUCF) and are currently under further revision.

It is now widely recognized that greenhouse gas fluxes from terrestrial ecosystems should be estimated for areas with sufficiently uniform climate, soils, land use and management. The proposed methodologies include inventories of stocks and stock changes to assess CO_2 fluxes from biomass and soils and focus on modelling efforts for assessment of emissions of N_2O and CH_4 from agricultural soils. Expected changes in climate and land use have important feedbacks on GHG fluxes from terrestrial ecosystems, which are however not taken into account in the current inventories. A framework is, therefore, required which on the one hand addresses the changes in climate, land use and management, and on the other hand allows for the application of GHG models sensitive to these changes. Care should be taken that all land units and GHG fluxes are evaluated in order to prevent leakage and possible compensation of sequestration by simultaneous emission of other GHGs.

The overall aim of the METAGE-project is to develop, apply and evaluate a procedure for the estimation of greenhouse gas fluxes from terrestrial ecosystems in Belgium under global change scenarios. The spatial units for which these fluxes are calculated have similar soils, climate and land use.

This report starts with a description of the spatial units used for the GHG inventories (Chapter 2) and of the approach to quantify the current and past fluxes from these spatial units: CO_2 from the soil (1960-2000), CO_2 from forest biomass (1984-2000), N_2O from agricultural soils (1990-2004). In Chapter 3, an attempt is made to integrate these fluxes expressed in CO_2 equivalents. The possibility of the spatial framework to estimate GHG emissions under one of the global change scenarios proposed by the IPCC in the Special Report on Emission Scenarios (SRES; NAKICENOVIC et al., 2000) is demonstrated in Chapter 4. Finally, recommendations for future research are given (Chapter 5). The accompanying website allows more detailed exploration of carbon stocks, CO_2 and N_2O fluxes for areas specified by the user (link on the Metage web site: www.geo.ucl.ac.be).

1.2 LINK WITH THE GOOD PRACTICE GUIDANCE FOR LULUCF

The United Nations Framework Convention on Climate Change (UNFCCC), one of three conventions of the United Nations resulting from the Rio Earth Summit, emerged in 1992. In 1997 some governments agreed to an addition to this treaty, called the Kyoto Protocol, which has more powerful (and legally binding) measures. The Kyoto Protocol envisages quantified engagements of greenhouse gas reduction, corresponding overall to 5% of the emissions of 1990 (baseline) during the first commitment period 2008-2012, for the Parties listed in Annex I (developed country subjected to an obligation of reduction). After the ratification by Russia in December 2004, the Kyoto Protocol will enter into force the 16 February 2005. The UNFCCC defines "sink" as "any process, activity or mechanism which removes a greenhouse gas, an aerosol or a precursor of a greenhouse gas from the atmosphere". The development of policy on "sinks" has evolved to cover emissions and removals of greenhouse gases resulting from land use, land-use change and forestry (LULUCF). Activities in the LULUCF sector can provide a way of combating climate change, either by increasing the removals by sinks of GHG from the atmosphere (e.g. by planting trees or managing forests), or by reducing emissions (e.g. by avoiding deforestation or no tillage of agricultural soils). For the UNFCCC, each Party must report annually on the GHG emission and/or removal by LULUCF activities. The methodology for this report is defined in the 1996 IPCC Guidance (IPCC, 1997) and the Good Practice Guidance for LULUCF activities (IPCC, 2005).

Under Article 3.3 of the Kyoto Protocol, parties must account for greenhouse gas removal through afforestation and reforestation since 1990. Conversely, deforestation activities will be subtracted from the amount of emissions that an Annex I Party may emit over its commitment period. This is mandatory; all Parties must report on Article 3.3. Through Article 3.4 of the Protocol, Parties decided on a voluntary basis that additional activities (forest management, cropland management, grazing land management and re-vegetation) should be added to the inventory for the first commitment period.

At COP 7 (Marrakesh Accord, LULUCF decision 11/CP.7, 2001), Parties were able to take a decision on LULUCF and related issues. These rules for LULUCF activities include three main elements: a set of principles to govern LULUCF activities; definitions for Article 3.3 activities and agreed activities under Article 3.4; and a four-tier capping system limiting the use of LULUCF activities to meet emission targets. With regard to the forest management, the Marrakesh Accord makes it possible to compensate the article 3.3 emissions by the article 3.4. C removals. Beyond this compensation, C sequestered by forest management can be entered to a maximum amount for each Annex I Parties, which is 0,03 MT of C per year during the first commitment period for Belgium. With regard to the management of the arable lands, it is allowed to enter C sequestered during the first commitment period, to amount of five times C sequestered by these activities during 1990. Lastly, with regard to the flexible mechanisms, C related to the projects of *Clean Development Mechanisms* CDM could be entered only for activities of afforestation or reforestation, reached a maximum to 1 % of the emissions of 1990. The definitions and methods of inclusion of afforestation activities in the CDM were adopted in Milan (COP 9, 2003) and Buenos Aires (COP10, December 2004).

The IPCC Good Practice Guidance (GPG) on LULUCF (IPCC, 2005) covers a broad range of questions relating to the preparation of the inventories of greenhouse gas emissions in terms of sources and sinks. The GPG were adopted as a whole at COP10 for the UNFCCC and the first commitment period. Their scientific basis was outlined in the Special Report on LULUCF (Watson et al., 2000). The objective of new Guidance is to complete the 1996 IPCC guidance and to propose new methodologies to meet the needs for reporting defined by the Marrakech accords.

Two main consequences of the Marrakech accords and GPG for Belgium must be highlighted: the forest definitions and the geographical location of elected activities. The definition of "forest " is based on criteria of surface and ground cover: ` a "forest" is a surface of territory ranging between 0,05 and 1 ha, whose minimal forest cover lies between 10 and 30 % and is ensured by trees likely to reach a minimal height from 2 to 5 m in maturity. One thus indicates by "forest " at the same time a dense forest formation, where trees form several stages and cover a strong proportion of ground, and an open forest formation, e.g. Mediterranean forest. The young plantations are also taken again under this forest definition, since they belong to the normal process of "forest management ". Each Annex I Party must, by December 31, 2006, adopt only one definition of forest.

The GPG defines six categories of land use: forest, arable land, pasture, wetland, built-up area, and other portions of territory. Each Party included in Annex I shall account for all changes in the following carbon pools: aboveground biomass, belowground biomass, litter, dead wood, and soil organic carbon. A Party may choose not to account for a given pool in a commitment period, if transparent and verifiable information is provided that the pool is not a source. The GPG requires that the Parties report for the six "categories " quoted above. The KP reporting imposes, moreover, the following subcategories: the forests which fall under Article 3.3, the lands which fall under the Article 3.4 and the not managed land territories.
The reporting of land areas within the context of Articles 3.3 and 3.4 must include information on the geographical limits of the portions of territories subjected to these activities. The GPG gives two reporting methods: method 1: the broad representation of geographical units including/understanding of multiple legal, administrative or ecological units subjected to the human activities; method 2:- the complete identification of each unit. We also discuss the feasibility of the application of these two methodologies in the concluding chapter.

2 SPATIAL FRAMEWORK FOR GHG MODELLING

2.1 DEFINITION OF SPATIAL UNITS

The Belgian territory was divided into landscape units by the topological intersection of the 1990 version of the Corine Land Cover (CLC) geo-dataset (European Commission 1993) and the digitized Soil Association map of TAVERNIER ET AL. (1972). The CLC geo-dataset has been produced by manual digitization of printed LANDSAT-images, taking into account a minimal mapping unit of 25 hectares. The 34 of the 44 possible classes of the original legend that occur in Belgium were aggregated into the 11 broader classes: (i) cropland, (ii) grassland (both permanent and temporary), (iii) broadleaf forest, (iv) coniferous forest, (v) mixed forest, (vi) fallow land, (vii) heath land, (viii) inland marshes, poplar in pasture, rush and reed vegetation, (ix) clay pits, mineral extraction sites and excavated soils, (x) peat bogs, (xi) not specified. The Soil Association map (1:500,000) represents broad zones with similar topsoil texture and drainage conditions in 64 soil associations. The overlay of both geo-datasets resulted in 567 landscape units (LSU), each characterized by one soil association and one land use class, scattered over 101,376 polygons. For 65 of the 567 LSUs, the land use class is "built up". Ten LSUs are situated in areas for which either soil type or land use type is lacking (mainly military domains). These 75 (65 + 10) LSUs cover 5,962 km^2 (19% of the total area) and are excluded from further study. The 203 LSUs characterized by the land uses fallow land (388 km^2, 1.3%), heath land (186 km^2, 0.6%), excavated soil (83 km^2, 0.3%) and inland marshes (92 km^2, 0.3%) are omitted from the study as well, except for the 1960 soil organic carbon (SOC) assessment, due to lack of SOC data. As a result, 492 LSUs are considered for the 1960 SOC assessment, and 289 LSUs are studied for the years 1990 and 2000. The former 492 LSUs cover 24,637 km^2, which is 81% of the Belgian territory. The average area of an LSU is 5,372 ha and the average polygon size is 30 ha. The latter 289 LSUs cover 24,042 km^2 (79% of the Belgian territory). In this case, the average area of an LSU is 8,229 ha and a single LSU polygon has a mean area of 33 ha. Whereas the SOC inventory considers all LSUs, the biomass organic carbon (BOC) inventory is limited to forested LSUs (159 LSU with a total area of 6,222 km^2). The N_2O-emission inventory applies to the agricultural LSUs only (130 in total, with a total area of 17,820 km^2).

Due to the generalised nature of the CLC geo-dataset and the Soil Association map, LSUs are pseudo-homogeneous with respect to soil and land use composition. This is especially true in highly fragmented landscapes, as those present in major parts of Belgium, and is illustrated by the fact that according to CLC, there is 1,782,028 ha of arable and grassland in Belgium while official net land use statistics show 1,400,300 ha (NIS, 2000). Moreover, a CLC compatible geo-dataset for 1960 and 2000 is not available. Therefore, LSUs derived from CLC-1990 are used for 1960 and 2000 as well. Hence, spatially explicit land use changes between 1960, 1990 and 2000 are not accounted for and no data can be generated on the effect of land use change on SOC stocks as derived from multi-temporal assessments. This assumption of absent land use changes may be fairly realistic with regard to the overall delineation of the LSUs. The total agricultural area for example, increased by less than 3% between 1990 and 2000, as indicated by agricultural statistics (NIS, 1990). However, the increase of within-LSU heterogeneity of land use, e.g. the obvious growth of rural residential areas, mostly at the expense of agricultural land fragments, is disregarded.

2.2 ATTRIBUTING PROPERTIES TO THE LSU

2.2.1 Soil series as a go-between

Soil and land use characteristics are intrinsic properties of the LSU, since they are at the basis of the delineation of the LSU boundaries. For soils, it was decided to further characterize the soil *associations* by typical soil *series*. The soil associations can indeed be considered an ensemble of soil series. The soil series classification provides a link with extensive datasets containing the results of SOC measurements made in the context of soil profile studies between 1950 and 1990. Those profiles have been geographically referenced but have also been classified in terms of morphogenetic or geomorphic soil series, according to the Belgian soil classification system (TAVERNIER ET AL., 1972). The same is true for the soil classification system used in the Flemish forest soil inventory of 2000. The N_2O fluxes depend on soil characteristics as well.

Typical soil series were chosen for each association by overlaying the Belgian soil association map (soil association legend) and the Belgian soil map (soil series legend). For Flanders both maps are available in digital format, therefore the area of each typical series could be computed. The soil series with the largest relative area (or representing a group of series with the largest relative area) within an association was selected as the dominant series. Similarly, the second most important (group of) series was selected as the associated series. Finally a typical but not necessarily widespread (group of) series was retained as the included series. The relative areas of these series within the association were computed for use as a measure of the importance of a certain soil series within an association. Since the soil map was not available in digital format for the regions of Brussels and Wallonia at the time of this exercise, three typical series were chosen for each association by analysing the paper maps. The area of each series within an association could only be estimated. We decided to attribute the fractions 0.6, 0.35 and 0.05 as weighing factors for dominant, associated and included series respectively.

2.2.2 Climate

The climate conditions prevailing for the centroïd of each LSU for the 1990 to 2000 period were extracted from the CRU TS 2.1 dataset (MITCHELL ET AL., 2004). This dataset comprises observed monthly climate data for the period 1900-2000 and covers the European land surface at a 10 minute spatial resolution. The grid is converted to a shape file and intersected with the landscape unit geo-dataset. The climatic characteristics of an LSU are not necessarily homogeneous, since polygons of some LSUs (e.g. LSUs on the alluvial soil association) occur throughout the country. However, most LSUs are restricted to a fairly limited and hence climatically rather homogeneous part of the country.

2.2.3 Activity data

Activity data comprise data on crop types and crop areas, animal types and numbers, fertiliser types and amounts. Activity data are required to estimate manure and mineral fertiliser application and related carbon and nitrogen inputs to the LSU. These data are generally retrieved from the NIS. They are mostly totals or averages per municipality. Like it is the case for climate, the geographical scatter of the LSU (based on geophysical factors) may not always coincide with the geographical scatter of the activity data (based mainly on administrative and regional practice). For instance fertilization rates differ strongly throughout the country and may therefore vary within one LSU. This discrepancy is difficult to circumvent. The obvious solution is to make the LSUs smaller, but that would compromise data availability.

3 ESTIMATES OF PAST AND CURRENT CO_2 AND N_2O EMISSIONS FROM SPATIAL UNITS

3.1 GENERAL APPROACH

The METAGE project uses the stock change method for estimating CO_2 fluxes from the LSUs i.e. SOC and BOC stocks of different years are compared. BOC stocks are considered for forested LSUs only. It is assumed that the per-LSU and total CO_2 flux is equal to the observed change in SOC or BOC stock in CO_2 equivalents over a certain time span and that the per-LSU-fluxes can be aggregated to yield total fluxes at regional or national levels. SOC stocks for LSUs are computed for the years 1960, 1990 and 2000 and BOC stocks are assessed for 1984 and 2000. The SOC/BOC estimations are based on a number of heterogeneous databases and modelling efforts. Three cases can be distinguished when computing per-LSU SOC/BOC values.

- When elementary point measurements are available, they are attributed to the LSU in a process called matching (VAN ORSHOVEN ET AL., 1993). Through matching, points are attributed to the LSU either based upon their location within the boundaries of the LSU ("geomatching") or based upon corresponding soil and land use characteristics as the LSU ("classmatching"). Classmatching may be completely independent of the point's location. In our approach classmatching was restrained by a stratification by soil association.
- With regard to agriculture, a number of data sources provide an average SOC-percentage per municipality or other type of administrative unit. These data can be considered to be indirectly geo-referenced to the administrative units, functioning as alternative LSUs (further termed ALSU) that do not correspond spatially with the LSUs to which we want to attribute the data. Therefore, the measurements are first disaggregated to the intersection of the ALSU and the LSU and then re-aggregated to the LSU.
- For LSUs for which no measurements are available, the stocks can be estimated using a mechanistic model. This has been done for the 1990-stocks of forested LSUs.

For the computation of the N_2O emissions, a different approach is applied i.e. the fluxes are empirically modelled directly per LSU.

3.2 CO_2 FLUXES AS A RESULT OF CHANGES IN SOIL ORGANIC CARBON STOCKS

3.2.1 Data sources

3.2.1.1 Measured SOC data for 1960

Between 1950 and 1970, a comprehensive soil survey was undertaken in Belgium. Soil units were mapped and soil profiles scattered throughout the administrative regions were described, sampled and analyzed by pedogenetic horizon. Since sites of special pedologic or agricultural interest were explored more intensively, the soil profile data collection cannot be considered as a fully random nor systematic sample. This two-decade effort resulted in an operational database "Aardewerk" with 13,033 profile descriptions (including soil series, map coordinates and land use class) and analysis results for 69,600 horizons (VAN ORSHOVEN ET AL., 1993). For each horizon, depth and thickness, textural fractions and class, volume percentage of rock fragments (or stoniness) and organic carbon content (percentage C by mass, measured according to WALKLEY AND BLACK (1934)) are available for the total soil depth to a maximum of 120 cm.

For the northern part of the country Aardewerk was complemented with data about forest ectorganic horizons from a second database, HIBBOD (LEROY ET AL., 2000). Only forested LSUs are assumed to have ectorganic horizons. Peat soils are considered separately. Apart from the Aardewerk and the HIBBOD profile-databases, a dataset of ca. 15,000 soil surface (upper 20 cm of mineral soil) samples was used. These data were collected approximately at the same time but independently of the profile data. A subset of 3,134 samples situated under forest in Flanders was geo-referenced in the HIBBOD project. The samples were analysed for organic C, textural fractions and soil series, using the same methods as for the profile data. This geo-referenced soil surface dataset has been used in a validation exercise.

3.2.1.2 *Measured SOC data for 1990 and 2000*

The datasets used for assessing the SOC content of the selected LSU in 1990 and 2000 are listed in TABLE 3.1. As a consequence of Belgium's federal state structure, the three administrative regions (Brussels Capital region, Flanders and Wallonia) have the authority for agricultural, forestry and environmental policies. Therefore, most datasets cover only one of the regions. For the almost completely built up Brussels region no data are available. For Flanders and Wallonia, the following databases were used:

- For LSU under forest in Flanders, the ForSite soil profile database of the Institute for Forestry and Game Management (unpublished data) is used. For those forests, covering 1,153 km^2 according to CLC-1990, data from 290 geo-referenced soil profiles are available. The profiles are located on the systematic 1,000 x 500 m grid used in the Flemish Forest Inventory, consisting of 3,294 grid points. Horizon-based sampling to a maximum depth of 120 cm took place between 1997 and 2002. The data are considered to be representative for the year 2000.
- For agricultural land in Flanders, the database of the Belgian Soil Service (unpublished data) is exploited. This dataset contains the average SOC percentage of samples analysed over a period of 3 years by administrative unit (municipality) and agropedological zone (14 broad zones within Belgium reflecting dominant soil texture and climate). In this study, the periods 1989-1991 and 1998-2000 are used for the inventories of 1990 and 2000. A distinction is made between arable land and grassland. For the former the upper 23 cm has been sampled while for the latter only the upper 6 cm was investigated.
- In Wallonia, SOC data for agricultural land are collected and managed at the provincial level. Therefore, several databases are joined to cover all the LSUs under agriculture: the Requasud database (unpublished data) and the databases of the Centres d'Information Agricole in the (i) Province de Luxembourg, (ii) Province de Liège, (iii) Province de

Hainaut (unpublished data). They all contain average SOC values per sub-municipality (i.e. a municipal district) for a fixed depth of 15 cm for, among others, the years 1989, 1990, 1991 (used for the year 1990) and 1999, 2000 and 2001 (used for 2000).

- For Walloon forests a dataset compiled by the Centre d'Information Agricole, Province de Luxembourg was used (unpublished data), containing 390 samples with the municipality name as the only geo-reference, collected in the years 1996-2000 for the upper 20 cm of mineral soil. These data were complemented with data from 16 geo-referenced profile studies of the Division de la Nature et des Forêts du Ministère de la Région wallonne (unpublished data). In order to maintain conformity with the other two Walloon forest datasets, the geographic coordinates of these profiles were omitted and replaced by the name of the municipality in which they occurred.

The heterogeneity of the datasets with respect to the degree of aggregation, sampling scheme and depth is obvious. However, differences also pertain to the techniques used to determine SOC concentration, as reflected by the conversion factor or equation shown in Table 3.1.

Table 3.1. *Main characteristics of the datasets for the SOC inventory of Belgium for the years 1990 and 2000 (NA = not available; sdev = standard deviation; n = number of observations; o.m.% = organic matter percentage; W&B = SOC measurement by*

	Belgian Soil Service	Institute for Forestry and Game Management	Centre d' Information Agricole, province de Liège et Hainaut	Centre d' Information Agricole, province de Luxembourg	Requasud - Région Wallone	Projet Chênes
Inventory year 1990	1989-1991	NA	1990	1989 - 1991(agriculture)	1990	NA
Inventory year 2000	1998-2000	1997-2002	NA	1999 - 2001(agriculture) 1996 - 2000 (forest)	2000	2000
Land use	cropland / grassland	forest	cropland / grassland	cropland / grassland / forest	grassland / cropland	forest
Organic carbon	weight% (modified W&B)	weight% (LOI)	weight% (W&B)	weight% (W&B)	weight% (W&B)	weight% (W&B)
Conversion from measured to total SOC	factor 1.14	TOC = -1.2785 + 0.5783*LOI	factor 1.33	factor 1.33	factor 1.33	factor between 1.82 and 2.21, depending on the o.m. %
Sampling by horizon	NA	up to 20 cm	NA	NA	NA	up to 200 cm
Sampling by fixed depth	23 / 6 cm	NA	15 cm	15 cm (agriculture) 20 cm (forest)	15 cm	NA
% Stones	NA	NA	NA	NA	NA	measured on site
Texture	NA	derived from soil map	NA	NA	NA	measured on site
Bulk density	estimated	measured	estimated	estimated	estimated	estimated
Administrative region	Flanders	Flanders	Wallonia	Wallonia	Wallonia	Wallonia
Aggregation level	1674 averages, sdev and n per 3-year period, municipality and agropedological zone	Lambert coordinates of 290 individual profiles are known	367 averages, sdev and n per sub-municipality	agriculture: 11977 and forest: 306 individual observations; both per sub-municipality	579 averages, sdev and n	Lambert coordinates of 16 individual profiles are known

Walkley and Black; TOC = total organic carbon; LOI = loss on ignition).

3.2.2 Modelling tools

3.2.2.1 SOC model for forest soils - The YASSO model

The YASSO-model describes decomposition and dynamics of soil carbon in well-drained soils. YASSO is calibrated to describe the total stock of soil carbon without distinction between soil layers. The model can be applied for both coniferous and deciduous forests. It has been shown to describe appropriately the effects of climate on decomposition rates of several litter types in a wide range of ecosystems from arctic tundra to tropical rainforest (LISKI ET AL., 2002). The soil module consists of three litter compartments and five decomposition compartments (Figure 3.1).

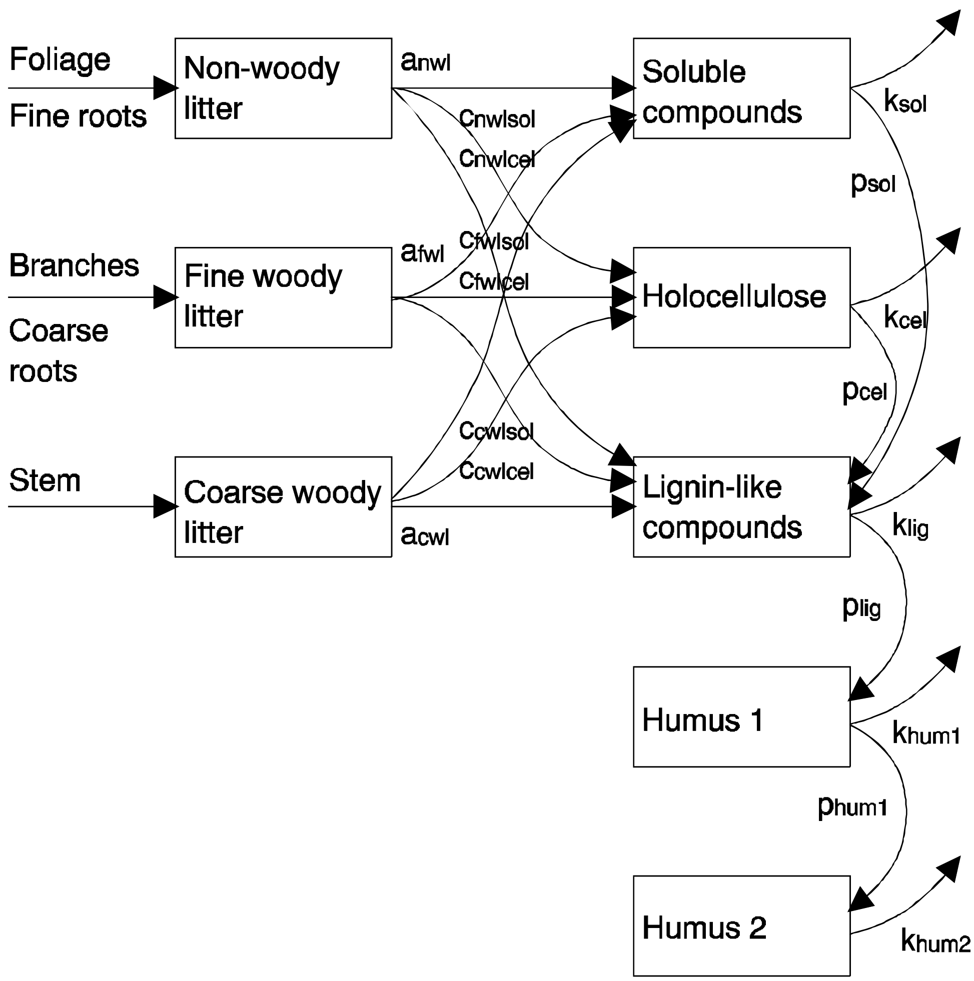


***Figure 3.1**: The YASSO model*
(Liski et al, 2005)

Input data on litter comes from the EFOBEL-model (see 3.3.3) through biomass turnover (see PERRIN, 2005). For the soil carbon module, the litter is grouped as non-woody litter (foliage and fine roots), fine woody litter (branches and coarse roots) and coarse woody litter (stems and stumps). Since the biomass module makes no distinction between fine and coarse roots, root litter is separated into fine and coarse roots according to the proportion between branch litter and foliage litter. Each of these litter compartments has a fractionation rate determining the proportion of its contents released to the decomposition compartments in a time step. For the compartment of non-woody litter, this rate is equal to 1 which means that all of its contents are released in one time step, whereas for the woody litter compartments this rate is smaller than 1. Litter is distributed over the decomposition compartments of compounds with low molecular weight, celluloses and lignin-like compounds according to its chemical composition. Each decomposition compartment has a specific decomposition rate, determining the proportional loss of its contents in a time step. Fractions of the losses from the decomposition compartments are transferred into the subsequent decomposition compartments having slower decomposition rates while the rest is removed from the system. The fractionation rates of woody litter and the decomposition rates are controlled by temperature and water availability.

The equations and the parameters values are presented in LISKI ET AL. (2002). Within the framework of our work, we did not have experimental data at large scale to carry out a calibration and a validation of YASSO on the collected data. So, we decided to use the default values, which already were the subject of a procedure of validation on large spatial and temporal scale. The model computes the annual evolution of soil carbon stocks for each sample point. These were subsequently integrated at the regional scale based on the LSUs.

3.2.3 Computation of SOC-stocks

3.2.3.1 1960

The SOC content of an LSU is the average SOC content of a selection of soil profiles typical of this LSU. To attribute the profiles to the LSU, the two matching procedures are applied. The point data of a certain site *geomatch* with a given LSU when its geographical position is within (one of the polygons of) the LSU. The concept is illustrated in Figure 3.2a for SOC. Due to map impurities, measurement points may geomatch with an LSU of which the soil and land use class differ from the ones of the measurement point. Hence by applying the geomatching procedure, variability of soil and land use characteristics and of the related SOC and BOC content within the LSU is included in the results. However, since the measurement locations have not been selected on a random or systematic basis, but rather to represent 'typical' and 'extra-ordinary' situations, the statistical value of the thus assessed variability is rather weak. Through *classmatching*, point measurements are attributed to LSU only if both the soil and the land use class is equal (or similar) for LSU and point measurement.

Classmatching can be performed regardless of the point location. It can be expected that by classmatching, variability is underestimated since point measurements of a soil-land use combination which is not present on the LSU-map but may be present in reality, cannot match and are excluded from further use. Classmatching requires that the same classification system of soil type and land use is used for the point data and the LSU. For land use, this condition was met by the reclassification of the 34 CLC land cover classes into 11 aggregated classes (see 2.1). With regard to soil, typical soil series per association are defined to enable classmatching between point measurement and LSU (see 2.2.1).

To compute the SOC-content of a profile, the available SOC weight percentages per horizon are first converted into amounts of carbon (t C ha^{-1}). Considering the reported texture class and the organic matter content of the horizons (as derived from the carbon content), the bulk density BD (10^6 g m^{-3}) of the non-stony horizon material was estimated according to the formula of RAWLS (1983), as calibrated by BOON (1984; eq. 3.1):

$$BD = \frac{100}{\frac{OM}{VOM} + \frac{100 - OM}{VMF}} \tag{3.1}$$

Where OM = % organic matter = %C * 4/3 * 2 (4/3 is the Walkley-Black correction factor for incomplete oxidation and the factor 2 converts carbon to organic matter); VOM = bulk density of the organic matter = 0.224 10^6 g m^{-3}; VMF = bulk density of the mineral fraction (10^6 g m^{-3}) per texture class (for values see BOON (1984)).

For stony horizons, the mass of carbon per horizon or per fixed depth layer is corrected for the volume of stones. The mass of carbon is computed by horizon and then summed to attain the mass of carbon in the profile. Equation 3.1 is also used for the ectorganic horizons since the texture class of the topsoil is known. For peat soils a bulk density of 0.31 10^6 g m^{-3} (BATJES, 1996) is applied. The mass of carbon (SOC) for the fixed-depth layers is computed and used according to eq.2:

$$SOC = D * \Sigma f_i * C\% * BD * (1 - V_{st}) * 100 \tag{3.2}$$

Where: SOC = mass of carbon per fixed depth layer (t C ha^{-1}); D= total fixed depth for which SOC is computed (m); f_i = fraction of the i-th horizon present in the fixed-depth layer, derived from depth and thickness of the horizon (dimensionless); BD = bulk density of the horizon (10^6 g m^{-3}), according to equation 3.1; V_{st} = volume of stones in the horizon (dimensionless). SOC content of each horizon is calculated by multiplying the carbon content, the bulk density and the non-stony volume of soil. The SOC content per fixed depth layer is computed as the sum of the SOC contents per horizon, insofar the horizon is within the lower boundary of the total fixed depth, multiplied with the total fixed depth. The SOC content of an LSU is the average SOC content of a selection of soil profiles typical of this LSU and is expressed by depth increments of 10 cm.

For 1960, the selection of soil profiles per LSU is done either by geomatching or by classmatching. The choice for geomatching or one of the classmatching variants is based upon the number of matching profiles and the degree of similarity between land use and soil type of profile and landscape unit. For part of the 492 LSU initially considered for the year 1960, it is impossible to determine the average SOC content by the methodology described above due to lack of data. In order to estimate and present the total stock of organic carbon in rural LSU in Belgium, we have allocated default estimates to these uncharacterised LSU. The average SOC content per land use class and region (Flanders or Wallonia) is attributed to them. From the LSU specific SOC contents and the area of the LSU, the total SOC stock (Mt) in Belgium for the year 1960 is computed by summation.

3.2.3.2 Computation of SOC stocks for 1990 and 2000

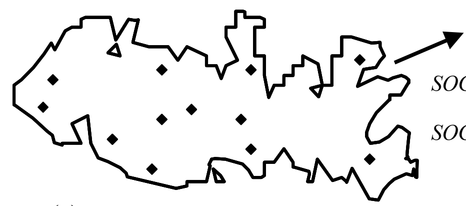
The geomatching procedure has been used to assess the SOC content of LSU in 2000 under forest in Flanders using the data of the 290 soil profiles of the ForSite dataset. For the conversion of SOC% to SOC content, equation 3.3 was applied:

$$SOC = C\% * BD * D * 100 \qquad 3.3$$

Where SOC= mass of carbon per fixed depth layer (t C ha^{-1}); BD= bulk density of the horizon (10^6 g m^{-3}), according to equation 3.1; D= sampling depth (here 0.2 m).

As a result of the applied geomatching procedure, the SOC content of an LSU is the average SOC content of all profiles located within this LSU (see Figure 3.2a). The Flemish forest database contains information on ectorganic horizons as well. They have been processed the same way as the mineral soil data, with the exception of the bulk density, which is not estimated with equation 3.1 due to lack of texture data, but approximated by values found in literature, namely 0.04 10^6 g m^{-3} for broadleaf forest, 0.12 10^6 g m^{-3} for coniferous forest and 0.08 10^6 g m^{-3} for mixed forest (VEJRE ET AL., 2003).

The other databases available for 1990 (agriculture in Flanders and Wallonia) and 2000 (agriculture and forest in Flanders and Wallonia and forest in Wallonia) contain average SOC concentrations for a surface layer of specified depth within a (sub-) municipality per land use category (arable land, grassland, broadleaf, coniferous and mixed forest) and (for the Belgian Soil Service database only) per agro-pedological zone. Hence the SOC data are linked to the intersection of the CLC-map, the map with the (sub-) municipalities and, for Flanders only, the agro-pedological zones map. These portions of land are considered as ALSU (alternative landscape units). The bulk density is estimated according to equation 3.1. However, since no textural class is available for each SOC value in the datasets, the dominant textural class of the agro-pedological zone, covering the full or largest part of the municipality is used. The average SOC % for each ALSU is then converted into SOC content (t C ha^{-1}) by multiplying the SOC% with bulk density (10^6 g m^{-3}) and fixed depth (m). To link the ALSU to the LSU, the procedure illustrated in Figure 3.2b has been applied. The SOC values for the ALSU are disaggregated according to their spatial coincidence with the agricultural LSU for which we need the values. Thus, it is assumed that the area-weighed average SOC value of the ALSU that geographically intersect the LSU represent the SOC of the latter. This assumption will introduce error, because although land use type is equal for ALSU and target LSU, soil type is not necessarily so. However, we apply this LSU-centred approach since we consider the LSU to be an essential part of the proposed spatially explicit approach, which facilitates (i) interpretation of the results, (ii) the comparison of SOC stocks with earlier inventories and (iii) the application of global change scenarios that depend, among other variables, upon current land use and soil type. In Figure 3.2b the studied LSU is covered by parts of 4 ALSU. Suppose that the average SOC values are available for 3 of the 4 ALSU, then these values are weighted according to the area of the ALSU within the LSU. ALSU with missing SOC value are not accounted for. The SOC content for incremental layers of 10 cm up to a total depth of 100 cm is obtained in Figure 3.2 by multiplying the SOC values for the depths as available from the input datasets with an LSU specific depth inter- or extrapolation factor derived again from the Aardewerk database.

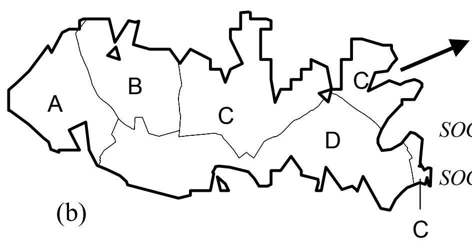


	Profiles					
	1	2	3	4	…	12
SOC	SOC_1	SOC_2	SOC_3	SOC_4	…	SOC_{12}

$$SOC_{LSU,uncorr} = \frac{1}{12}\sum_{i=1}^{12} SOC_i$$

$$SOC_{LSU} = SOC_{LSU,uncorr} * DEF_{LSU} * (1 - S_{LSU})$$

(a)



	Municipality			
	A	B	C	D
SOC	SOC_A	N.A.	SOC_C	SOC_D
Area%	16	16	33	35
weight	19	0	39	42

$$SOC_{LSU,uncorr} = \sum_{i=1}^{4} SOC_{ALSU_i} * weight_i$$

$$SOC_{LSU} = SOC_{LSU,uncorr} * DEF_{LSU} * (1 - S_{LSU})$$

(b)

FIGURE 3.2 *Illustration of the matching procedures. (a) Geomatching computes the average soil organic carbon of all profiles within the landscape unit (LSU). (b) Disaggregation computes the area-weighted average of all alternative landscape units (ALSU) that intersect with the landscape unit. Both matching procedures require a depth extrapolation (DEF_{LSU}) and a stone content (S_{LSU}) correction factor per landscape unit.*

For part of the 289 LSUs initially considered in the 1990 and 2000 assessments, it is impossible to determine the average SOC content due to lack of data. In order to compute the total SOC stock in Belgium, the same methodology as for 1960 is applied. The average SOC per land use class within respectively Flanders and Wallonia is attributed to the uncharacterised LSUs. From the LSU-specific SOC data and the area of the LSU, the total SOC stock (Mt) in Belgium for the year 1990 and 2000 and for the depths of 20, 30 and 100 cm is then derived by summation.

3.2.3.3 Computation of SOC stocks for forest soils in 1990 (Wallonia)

No data were available on SOC contents in forest soils for 1990. The availability of the 1984 forest inventory data for the region of Wallonia allowed to model the stocks instead. The growth of the forest stands as described by the forest inventory of 1984 was simulated using the EFOBEL model. The input of litter to the soil compartment was then used in the YASSO model in order to estimate the carbon stocks.

3.2.4 Assessment of uncertainty of SOC stocks

3.2.4.1 Uncertainty of SOC stocks for 1960

The SOC contents of matching profiles are considered as 'repeated measurements' on the same LSU. The matching procedures yield at least one but mostly multiple SOC observations per LSU and per depth layer. This "unbalanced design" allows for the assessment of variability and uncertainty. The software package SAS (SAS Institute Inc., Cary, NC, USA) is used for this purpose. The influence of the matching procedure (see section 3.2.3) is tested separately. Flanders and Wallonia are tested separately as classmatching with all series is not possible in Wallonia (due to unavailability of a digital soil series map).

Average SOC contents of LSUs were pair-wise compared through analysis of variance between en within land use classes, soil associations and their interaction terms (i.e. the LSU). The number of possible pair-wise comparisons of n LSU is equal to n(n-1)/2. Insofar the data are normally distributed (after transformation if necessary), parametric analysis of variance and Tukey-adjusted multiple comparison was considered appropriate. The parametric model considers the factors land use and soil type and, for tests concerning the LSU, also their interaction factor. If the data were not normally distributed, the non-parametric Kruskal-Wallis test) was preferred. Significance of the parametric model, the model factors (association and land use) and their interaction term is accepted at the significance level α = 0.01. Significance of differences between LSUs, land use classes or associations in the parametric or non-parametric tests is accepted at significance level $\alpha = 0.05$. Like for the computation of SOC-stocks using class-matching, the PROC GLM is weighted by a factor that expresses the importance of the profile. This weighing factor is only used for classmatching with all series and with three series and is related to the area of the soil series (to which the profile matches) within the association (of the LSU).

The consistency of the results is tested with the soil surface samples available from the HIBBOD project. Those samples are situated under forest in northern Belgium. The SOC contents of the upper 20 cm obtained for forested LSU in north Belgium are compared with the SOC contents computed from the HIBBOD data for the same LSU, using the same matching approach. Results are compared with a t test since they are considered as dependent 'measurements' of the C content of the LSU.

3.2.4.2 Uncertainty of SOC stocks for 1990 and 2000

Whereas matching of multiple point data to the forested LSU provides for each LSU an average SOC content with standard deviation and number of observations, a different approach must be taken with the average SOC-values available for 1990 and 2000. Since the available sets of aggregated data all come with standard deviation and number of observations in addition to the average values, we exploited this information based on simple statistical rules (GOODMAN, 1960; NETER ET AL., 1996) in order to accompany the SOC values for LSU characterised by dis- and re-aggregation by measures of variation as well (eqs. 3.4-3.8):

$$VAR(Y) = [STD(Y)]^2 \quad 3.4$$
$$VAR(aY) = a^2\ VAR(Y) \quad 3.5$$
$$VAR(Y_1 + Y_2) = VAR(Y_1) + VAR(Y_2) + 2COV(Y_1, Y_2) \quad 3.6$$
$$VAR(Y_1 * Y_2) = [M(Y_1)^2 * VAR(Y_2)] + [M(Y_2)^2 * VAR(Y_1)] - [VAR(Y_1) * VAR(Y_2)] \quad 3.7$$
$$VAR\ (M(Y)) = VAR\ (Y) / n \quad 3.8$$

Where STD = standard deviation; VAR = variance; COV = covariance; a = constant;
X, Y = independent and homoscedastic variables (therefore: COV (X,Y) = 0);
M(Y) = average of Y; n = number of observations.

For forests in Flanders, application of the correction factor for the stone content and depth extrapolation factor, both themselves averages with a standard deviation derived from a number of observations, requires rules (3.7) and (3.8). For all other datasets, the steps explained earlier are applied (paragraph 3.2.3.2). The first step (from % to t C ha^{-1} by multiplication with BD) is the calculation of SOC content per ALSU. The rules (3.4) and (3.5) are used. Bulk density is considered constant. This is a simplification that is justified through the application of the Delta Method (RICE, 1995). This method, which estimates the standard deviation of transformations of variables, demonstrates that the standard deviation does not increase after multiplying the SOC% with the bulk density (itself dependent upon the SOC%). The second step is the computation of the average SOC content per LSU by weighted averaging (Figure 3.2b) and uses equations (3.5), (3.6) and (3.8). The third step is the application of the correction factor for the stone content and depth extrapolation factor, with rules (3.7) and (3.8). Because of the very high numbers of observations and following the Central Limit Theorem we assumed that the average SOC content per LSU has a normal distribution. Since the average SOC content for each LSU is accompanied by a standard deviation, the confidence interval can be calculated according to Tukey (NETER ET AL., 1996) to determine the level of significance of the observed differences. A significance level $\alpha = 0.05$ is imposed.

3.2.5 Results

3.2.5.1 Cover of the matching procedures in 1960, 1990 and 2000

After combining the matching procedures for the year 1960, 347 of the 492 LSU have at least one matching profile for the upper 30 cm. The 145 LSUs without a match represent only 0.5% of the inventoried, non-built up area of Belgium. For the year 1990, there are no forest data available, hence only the 130 (17,820 km²) agricultural LSUs are included in the analysis. For all of them an average SOC value could be determined based upon the available SOC data per municipality. Finally, for 74 of the 289 considered LSUs (554 km^2 of 24,042 km^2 - 2%) in 2000, no average SOC value could be determined for the year 2000 based upon the available SOC datasets. All of those 74 are forested LSUs.

3.2.5.2 SOC stocks of the mineral soil in 1960, 1990 and 2000

For the year **1960**, the transformation z = ln(y) is applied to reduce the pronounced tails in the distribution of the data and stabilize the error variance. This results in approximately half of the LSUs having normally distributed SOC values. The influence of the matching procedure was found to be not significant (P-value for the intercept is 0.17 for Flanders and 0.19 for Wallonia so that the intercept can be omitted from the regression model). Nevertheless, it was judged useful to keep the different matching procedures in order to ensure large numbers of observations and avoid missing values. The effect of soil association, land use and their interaction all contribute significantly to the explanation of the variability of the SOC content of the LSUs. The Tukey test and non-parametric analysis show that 6% (3,453) of the pairs of LSUs have significantly different SOC values for the upper 30 cm and 5% (2,459 pairs) for the upper 100 cm.

For the upper 20 cm, 41% (856) of the association pairs and 53% (24) of the land use pairs have significantly different SOC. Broadleaf forest has significantly lower SOC than coniferous or mixed forest. Cropland has significantly less SOC than all land use types, except excavated soil and fallow land. Grassland and peat have significantly higher SOC than broadleaf forest. Very high SOC contents are observed for the land use classes peat bogs (96 t C ha^{-1}; 95% confidence interval CI95 = ± 12 t C ha^{-1}) and inland marshes (61 ± 7 t C ha^{-1}). SOC content under coniferous forest (54 ± 1 t C ha^{-1}), grassland (54 ± 0.8 t C ha^{-1}) and heath land (48 ± 3 t C ha^{-1}) are also high (Figure 3.3). The soil association dominated by peat soils has a C content of 91 ± 9 t C ha^{-1} in the upper 20 cm of mineral soil. Other associations where peat soils occur also have high C contents (data not shown; 64 and 59 t C ha^{-1}). Moreover, associations with high SOC values are often poorly drained. Examples are association 15 (wet sand to loamy sand soils with humus and/or ferralic B horizon) with 53 t ± 1 C ha^{-1} and association 21 (wet sand to loamy sand soils with anthropogenic humus A horizon) with 54 ± 3 t C ha^{-1}. They also tend to have high clay contents, e.g. association 49 (clay and stony loam soil with admixture of schist) contains 57 ± 3 t C ha^{-1} and association 57 (clay soils with structure B horizon) is characterised by 64 ± 6 t C ha^{-1}. Sandy soils in the north of Belgium often have high SOC contents compared to other sandy soils, for instance association 15 or 22 (non differentiated sandy soils on a sand substrate) with 53 ± 5 t C ha^{-1}.

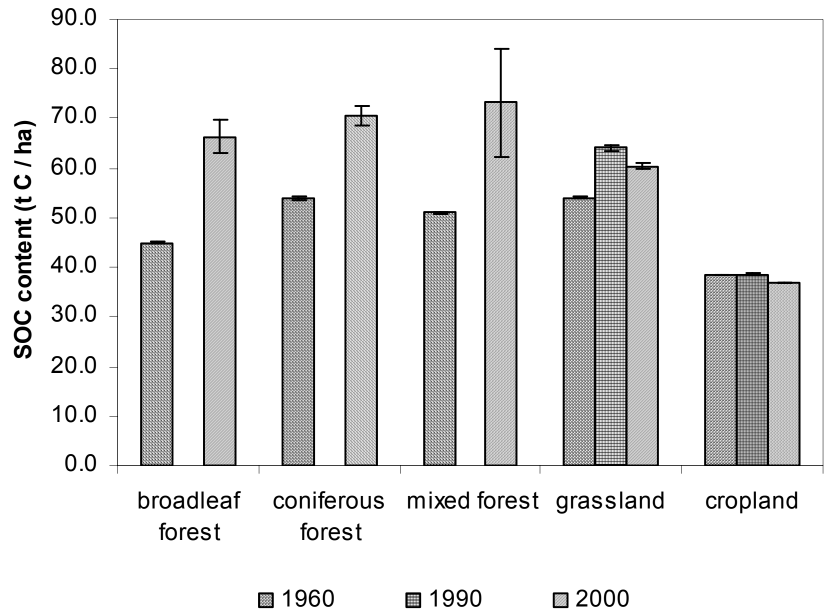


Figure 3.3. *Average SOC content and 95% confidence intervals (t C ha^{-1}) for 0-20 cm for the land use types broadleaf forest, coniferous forest, mixed forest, grassland and cropland for 1960, 1990 and 2000.*

The geographical distribution of SOC stocks in the upper 100 cm is illustrated (Figure 3.4). In northern Belgium, soils in the Kempen and in the Ardennes with extensive land use have a higher C content than soils under intensive agricultural land use. The Loam Belt is divided in an eastern carbon rich region and a western part with low SOC stocks. Similar low stocks occur in the Condroz. Dark spots mostly indicate recent alluvial soils or peat soils (south of Sambre-Meuse). The light area in the southernmost tip of the country is dominated by sandy soils and has again lower SOC values, with the exception of some wet alluvial soils (dark lines).

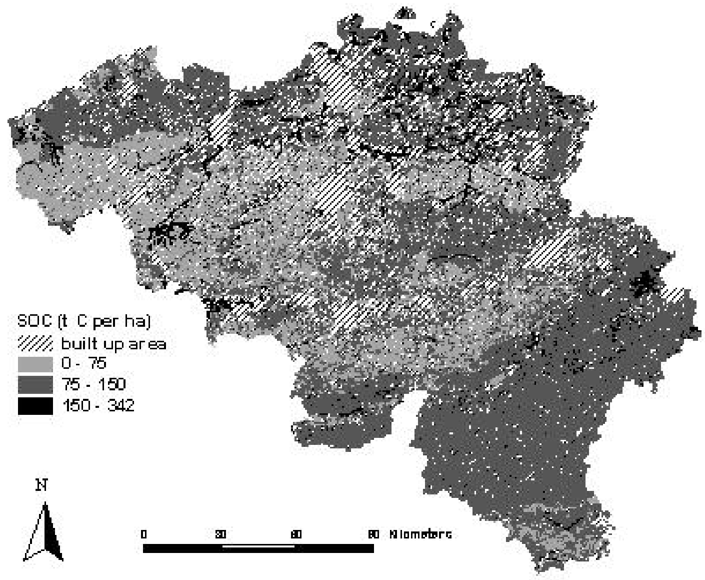


Figure 3.4. *SOC stock (t C ha^{-1}) of landscape units in Belgium in 1960 in the upper 100cm of mineral soil*

The change of the SOC content with depth averaged over all LSUs reveals that the upper 10 cm of mineral soil contains 24 t C ha^{-1} and the upper 120 cm 105 t C ha^{-1} on average (data not shown). The cumulative increase of SOC with depth follows a logarithmic curve ($R^2 = 0.999$). The change of SOC with increasing depth is also shown for the LSU 19-3 (grassland on wet sand to light sandloam soils of northern Belgium) and 55-4 (cropland on sand to sandloam soils of southern Belgium). These examples show that the behaviour of individual LSUs may deviate considerably from the average.

For the upper 20cm, 30% of all the LSU (225*224/2 pairs) have mutually significantly different SOC in **2000**, but this number decreases to 11% for the upper 100 cm. Differences in SOC for 0-100 cm are often smaller and the error of the depth extrapolation term is larger than for the upper 20 cm. The result is a lower number of significant differences. SOC is significantly lower in cropland compared to the four other land use types for all soil depths. Coniferous and broadleaf forest both have significantly higher SOC content (71 ± 2 t C ha^{-1} and 66 ± 3 t C ha^{-1}) than grassland (60 ± 1 t C ha^{-1}) for the 0-20 cm layer. The SOC contents of the three forest types do not differ significantly, but SOC under mixed forests is highest (73 ± 11 t C ha^{-1}). The difference with grassland is, however, not significant due to large standard deviations. The average SOC content (t C ha^{-1}) and 95% confidence intervals per land use type are shown in Figure 3.3. For the upper 20 cm, 1,009 of the 2,080 possible pairs of soil associations (49%) have a significantly different SOC value. For the upper 100 cm this number decreases to 547 (26%). High SOC values occur in soil associations with pertinent presence of peat and clay soils. Sandy, often podzolic soils in northern Belgium are also characterized by a high SOC content in 2000. Loamy associations have low SOC values, possibly due to the widespread occurrence of arable land (with typical low SOC values).

For the year **1990**, 26 % (0-100 cm) and 52% (0-20 cm) of the agricultural LSU have a significantly different SOC content. Similar trends exist as for 2000. Grassland has a significantly higher SOC content (0-20cm: 69 ± 1t C ha^{-1}) than cropland (43 ± 0.2 t C ha^{-}1; Figure 3.3.). Podzolic and clayey soils of Northern Belgium have a higher SOC value than the loam soils of central Belgium, which are mainly occupied by cropland.

3.2.5.3 SOC stocks of the ectorganic horizons in 1960 and 2000

With respect to the carbon contained in the ectorganic layers in 1960, the ANOVA shows that the overall model is significant with association as the only significant effect (R^2 = 0.27). Peat soils have important ectorganic layers that contain large amounts of C (data not shown; 133 t C ha^{-1}). SOC values of the other associations vary mostly between 30 and 60 t C ha^{-1}. There is no clear influence of texture or drainage class. With regard to the land uses, soils under coniferous forests store 70 t C ha^{-1}, broadleaf forests 46 t C ha^{-1} and mixed forests 55 t C ha^{-1} in their ectorganic layers. These values are 49%, 55% and 62% of the SOC content in the upper 100 cm of mineral soil respectively.

While for 1990, no data are available for ectorganic horizons, the ForSite database contains data for the ectorganic horizons of Flemish forests in 2000. OC values of 10 t C ha^{-1} (LSU under broadleaf forest), 20 t C ha^{-1} (mixed forested LSU) and 35 t C ha^{-1} (LSU under coniferous forest) are obtained. These values are 7, 13 and 22% of the SOC content in the upper 100cm.

3.2.5.4 Total SOC stocks in 1960, 1990 and 2000

Considering the area of each LSU, the total organic C stock in 1960 amounts to 236 ± 3.5 Mt C in the upper 100 cm of mineral soil for the land use types cropland (129 Mt C in 14,136 km²), grassland (42 Mt C in 3,684 km²) and forest (65 Mt C in 6,222 km²), together 24042 km² or 79% of the Belgian territory (Table 3.1). The total SOC stock in mineral soil in 2000 in rural Belgium amounts to 264 ± 0.007 Mt C for the 0-100 cm layer. The total stocks per land use class are given in Table 3.2. Forests contain 95 Mt C, grassland 48 Mt C and cropland 121 Mt C. The total agricultural SOC stock for 1990 in 0-100 cm is 178 Mt C of which 52 Mt C under grassland and 126 Mt C under cropland.

Table 3.2. *Area and total soil organic carbon stock (Mt C) per land use type for the years 1960, 1990 and 2000, for three depth intervals.*

Land use	Year	Area (ha)	SOC 20 cm (Mt C)	SOC 30 cm (Mt C)	SOC 100 cm (Mt C)
Broadleaf forest	1960	199986	9	12	18
	2000		13	17	29
Coniferous forest	1960	140669	8	10	16
	2000		10	13	22
Mixed forest	1960	281522	15	19	31
	2000		21	27	44
Grassland	1960	368403	20	25	42
	1990		24	31	52
	2000		22	29	48
Cropland	1960	1413624	56	75	129
	1990		54	74	126
	2000		52	70	121
Total	1960	2404206	108	119	236
	2000		118	156	264

3.2.5.5 *Validation of the 1960 stock assessment*

The independent surface sample dataset of forest soils in Flanders contains SOC values for LSU representing 8% of the total area or 30% of the forested area in Belgium. The per-LSU-difference between the SOC-results derived from this dataset on the one hand and the Aardewerk dataset on the other hand was tested by means of a t-test and no significant differences were found ($P = 0.62$). This strengthens the validity of the results.

3.2.5.6 *Evolution of SOC stocks between 1990 and 2000*

Due to the absence of forests in the 1990 assessment, the SOC content of only 130 agricultural LSU can be compared between the years 1990 and 2000 by means of a pair-wise comparison. The SOC stock in the 0-20 cm layer of 87 LSUs (74% of the total agricultural area) decreased between 1990 and 2000. For 25 (48% of the total agricultural area) of these LSU, the decrease is significant at the 95% confidence level. Those LSUs are mostly under cropland and all of them occur in northern or central Belgium. Most of the remaining 43 LSUs (26%) with increased SOC content are situated in southern or central Belgium. Also the 1 LSU (3%) that shows a significant increase of SOC content is located in southern Belgium.

On average, the SOC content of both the land use type grassland and cropland decreased significantly between 1990 and 2000 (Figure 3.3.). Grassland SOC decreased from 64 t C ha^{-1} in 1990 to 60 t C ha^{-1} in 2000 (0-20 cm). Cropland soils stored 38 t C ha^{-1}in 1990 and 36 t C ha^{-1} in 2000. If only the soil associations are considered, 39 of the 65 associations have lost SOC between 1990 and 2000. They are situated in the coastal zone and in northern and central Belgium. The 16 associations with a significant decrease in SOC are restricted to northern and central Belgium. The 4 associations with a significant increase in SOC are all situated in southern Belgium.

Finally, the total SOC stock in agricultural land was compared between 1990 and 2000. In 1990 this stock amounted to 178 Mt C in the upper 100 cm and for 2000 it decreased to 169 Mt C. This decrease is significant at the 95% confidence level.

3.2.5.7 Evolution of SOC stocks between 1960 and 2000

For 180 LSUs, covering an area of 23,384 km² (76% of the national territory of Belgium), the average SOC content and standard deviation are available for both 1960 and 2000. For the remaining 109 LSU (3% of the national territory) the average SOC content could not be computed for either 1960 or 2000. The SOC content increases for 131 LSUs (52% of area) for the upper 20 cm and for 133 LSUs (52% of area) for the upper 100 cm of mineral soil. These increases were significant for 34 LSUs (17% of area) in the top 20 cm of mineral soil and 18 LSUs (12% of area) in the top 100 cm. Significant decreases are found for 7 LSUs (17%) in the top 20 cm and 3 LSUs (11%) in the top 100 cm. As illustrated in Figure 3.3, the observed changes differ per land use type. For grassland and forest, SOC content increases strongly and significantly between 1960 and 2000. The largest changes occur under mixed forest (+ 22 t C ha^{-1} for 0-20 cm) and the smallest under grassland (+ 6 t C ha^{-1}). For arable land, SOC content decreases slightly (but significantly) from 38 t C ha^{-1} in 1960 to 37 t C ha^{-1} in 2000. All the LSUs with a significant decrease in SOC are under cropland. The soils with significant increases in SOC occur under all land use types, but mostly on grassland. The direction and significance of SOC changes in the 0-20 cm layer is further illustrated in Figure 3.5. It is apparent that LSUs of which SOC content significantly increased occur mainly in northern Belgium, while LSUs with SOC content that significantly decreased are concentrated in the central Loam Belt. LSUs with non-significant changes are present throughout the country.

13 of the 65 soil associations show a significant increase in SOC (19% of the total area), in contrast with only 3 (11% of the total area) associations that have a significantly lower SOC content in 2000 compared to 1960 for the upper 20 cm (data not shown). Most of the associations with a significant increase are situated in northern Belgium, predominantly sand to sandloam soils. The three associations with decreased SOC content are covered mainly by cropland and are situated in the central Loam Belt. By aggregating the SOC contents of the 180 LSUs, we conclude that the 13% C stock increase in the upper 100 cm of mineral soil (from 234 Mt C in 1960 to 264 Mt in 2000) is significant.

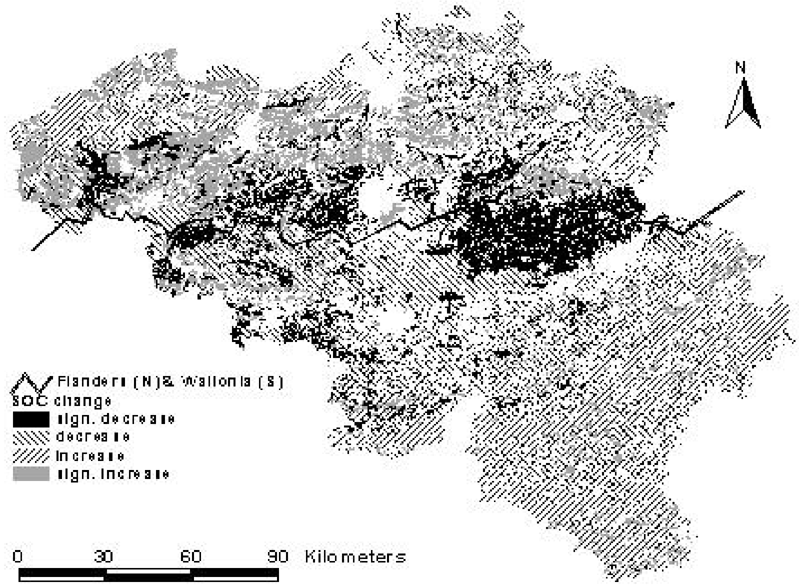


Figure 3.5 *SOC change of landscape units in Belgium between 1960 and 2000 in the upper 20 cm of mineral soil. Hatched areas represent non-significant changes*

3.2.5.8 The reference year 1990 within the 1960-2000 period

Due to lack of data for forest soils, SOC contents for 1990 could be computed for agricultural LSUs only. For the grassland LSU, 1960 SOC content in 0-20 cm is 54 t C ha^{-1}. A peak (64 t C ha^{-1}) in SOC content appears in 1990 followed by a decrease that leads to an intermediate (between 1960 and 1990) SOC value of 60 t C ha^{-1} (Figure 3.3). The decrease between 1990 and 2000 seems to have happened at approximately the same rate (0.4 t C ha^{-1} yr^{-1}) as the increase between 1960 and 1990 (0.3 t C ha^{-1} yr^{-1}). For cropland the SOC stock remains constant between 1960 and 1990 around 38 t C ha^{-1}. Between 1990 and 2000 a slight, but significant decrease of 2 t C ha^{-1} occurs. Not all but only 87 of the 130 agricultural LSU (68 % of the total considered area) lose SOC between 1990 and 2000. For 25 of these LSU, the decrease is significant. Most significant decreases occur in cropland soils of northern and central Belgium, while the increases (of which only 1 significant) are situated in southern or central Belgium.

3.2.6 Discussion

3.2.6.1 SOC stocks of land use types and soil types

ARROUAYS ET AL. (2001) estimate that 43 t C ha^{-1} is stored in the upper 30 cm of arable land in France. For the whole of Europe, SMITH ET AL. (2000) gave a SOC value of 53 t C ha^{-1} for arable land in 0-30 cm. ARROUAYS ET AL. (2001) found a SOC content of 70 t C ha^{-1} (30 cm) for permanent grassland and forested land. According to VEJRE ET AL. (2003), Danish forest soils contain 97 t C ha^{-1} (100 cm). This corresponds well with the values we estimated for Belgium for 1960, 1990 and 2000 (Figure 3.3.).

SLEUTEL ET AL. (2003) estimated that in Belgian arable land, the 1990 SOC content was 84 t C ha^{-1} in the 0-100 cm layer. This value compares well with the values proposed here. Nevertheless, their total SOC stock under cropland is much smaller than the value we propose (49 Mt C) due to entirely different estimate of the cropland area. They use the data from the National Institute for Statistics (NIS), which reflect the cropland area reported by farmers (5,881 km²). We use the area for cropland derived from the Corine Land Cover (14,196 km²), which is the only explicitly spatial and homogeneous land use information available for the whole of Belgium. The legend classes "complex cultivation patterns" and "land principally occupied by agriculture, with significant areas of natural vegetation" (European Commission, 1993) constitute almost half of the cropland area. It can be concluded that on the one hand we indeed overestimate the cropland area. On the other hand, our approach reflects the spatially fragmented nature of cropland in large parts of Belgium and as such is in agreement with the spatially explicit landscape unit concept. Landscape units under a certain land use should rather be interpreted as dominated by this land use. The proposed geomatching approach is appropriate for the assessment of carbon contents for these heterogeneous landscape units since it fully takes account of 'soil and land use impurities' within the LSU.

The largest SOC stocks are found in peat soils, poorly drained areas and clay soils, as expected (ARROUAYS ET AL., 2001; POST ET AL., 2000). The estimate for total SOC of peat soils depends strongly upon their bulk density. In this study, peat soils were attributed a default bulk density of 0.31 10^6 g m^{-3} (BATJES, 1996). This value lies between 0.11 10^6 g m^{-3} (MILNE ET AL., 1997) and 0.35 10^6 g m^{-3} for blanket peat and up to 0.50 10^6 g m^{-3} for deeper layers (HOWARD ET AL., 1995). The northern sandy soils show large SOC contents as well. This may be explained by the occurrence of carbon rich podzols (VAN ORSHOVEN ET AL., 1991; ARROUAYS ET AL., 2001) and "plaggen" soils which have been enriched with manure for several centuries and by the relatively high proportion of forested land. VEJRE ET AL. (2003) observed the same phenomenon in nutrient poor Spodosols in Danish forests. Both the O horizon and the spodic horizon act as C sinks.

3.2.6.2. Changes of SOC stocks versus management of LSU

The size and significance of the SOC change in the upper 20 cm of mineral soil between 1960 and 2000 is shown in Figure 3.5. The two are not always correlated. Large absolute changes may prove non-significant while smaller changes are significant. Due to the extremely large number of observations, the confidence interval becomes very narrow for large spatial units. For instance, 37,405 observations determine the total SOC stock of Belgium in 2000 at 264 ± 0.07 Mt C. Similar observations are made in other studies. WANG ET AL. (2003) compute the total SOC stock in China in the 1980s at 92 ± 3 Pg based on 2,442 observations. GARTEN ET

AL. (1999) argue that for a given variance, the minimum detectable difference decreases non-linearly with the number of observations.

The largest **SOC increases on agricultural land** between 1960 and 1990 occur in Northern Belgium (Flanders) (Table 3.3). Grassland SOC content increases by 33% in Flanders (0-100 cm) and 18% in Wallonia. SOC under cropland increases by 6% in Flanders and decreases by 10% in Wallonia, leading to a net loss of SOC in Belgian cropland. The observed regional rise between 1960 and 1990 correlates with the increased spreading of animal manure and other organic fertilizers in this region, following a dramatic growth in livestock density. In the western part of Flanders for example, pig numbers increased strongly, from 886,501 in 1960 to 4,690,153 in 1990 and 5,160,129 in 2000. VAN WESEMAEL ET AL. (2005) demonstrate that in Flanders the production of manure and slurry per area of agricultural land has more than doubled between 1958 and 1990, from 1.5 t C ha^{-1} to 3.5 t C ha^{-1}. In 2000 application rates are estimated to be slightly lower, namely 3.1 t C ha^{-1}. Based on the same assumptions, manure application rates in Wallonia have remained stable between 1960 and 2000 around 1.2 –1.3 t C ha^{-1}. SOC content of Belgian agricultural land decreases between 1990 and 2000 for most LSU, with the exception of Walloon cropland, for which SOC levels remain stable. This may be explained –at least partly- by the reduced application of farmyard manure and the recent legal restrictions on the quantity of animal manure that can be applied to the field, thus preventing the excessive rates of manure application that often occurred in the past.

Whereas the increased application of organic amendments may explain to an important extent the rising SOC levels of grassland and cropland in Flanders and the more slowly rising SOC levels of grassland in Wallonia between 1960 and 1990, the reason for the observed decrease of SOC in Walloon cropland in general and in the central Loam Belt in particular remains unclear. Livestock density has always been low in the Loam Belt, leading to high application rates of synthetic fertilizer, often in combination with the $CaCO_3$ rich by products of the sugar refinery that is typical for the region. Both the absence of organic fertilizer and the application of $CaCO_3$ to the field are known to cause increased activity of microorganisms and thus accelerate SOC decomposition (CHAN ET AL., 1999; ROGASIK ET AL., 2004). Spatial redistribution and loss of upper soil and contained OC through erosion, which is an important issue in the central Loam Belt, may also be part of the explanation. Intensification of agriculture, increasing field size, removal of hedges and trees, combined with the high erodibility of loam soils do indeed go along with high erosion rates. VAN OOST ET AL. (2000) find erosion rates between 14.2 and 18.0 t ha^{-1} yr^{-1} in 1990 for test areas in the Loam Belt. VAN MUYSEN ET AL. (2000) predict even higher current erosion rates of up to 30 t ha^{-1} yr^{-1} in the Loam Belt if tillage is carried out in the up and down slope direction. If 30 t ha^{-1} yr^{-1} disappears during 40 years and an average soil bulk density of 1.16 10^6 g m^{-3} is taken into account (average BD for the top 30 cm of European soils derived from Smith *et al.*, 1997), the upper 10 cm of soil would be carried away. Since the top mineral soil contains the highest amounts of SOC (according to this study: 0-10 cm: 19 t C ha^{-1}; 10-20 cm: 18 t C ha^{-1}; 20-30 cm: 12 t C ha^{-1} in the cropland LSU of the Loam Belt), soil losses in this layer would incur more than proportional C losses.

Table 3.3 *(a) Absolute (t C ha^{-1}) and (b) percentage SOC change on average in Belgium, Flanders and Wallonia in the upper 100 cm of mineral soil for the land use types cropland, grassland and forest for the years 1960, 1990 and 2000.*

(a)

		Grassland (368403 ha)			Cropland (1413624 ha)			Forest (622178 ha)	
		1960	1990	2000	1960	1990	2000	1960	2000
SOC top 100cm Belgium	*(t C ha^{-1})*	117	140	132	89	89	85	103	151
SOC top 100cm Flanders	*(t C ha^{-1})*	119	159	148	93	99	91	112	135
SOC top 100cm Wallonia	*(t C ha^{-1})*	110	130	123	88	79	79	101	158

(b)

		Grassland			Cropland			Forest
		1990-1960	2000-1990	2000-1960	1990-1960	2000-1990	2000-1960	2000-1960
SOC change Belgium	*(%)*	20	-6	13	0	-5	-4	47
SOC change Flanders	*(%)*	33	-7	24	6	-8	-3	20
SOC change Wallonia	*(%)*	18	-5	12	-10	0	-10	56

The SOC content of forest soils also increased over the 40-year period. The current age-structure of forests shows that large areas of forest have been planted after World War II (AFDELING BOS & GROEN, 2001). This implies that around 1960 forests were on average younger and thus contained less living biomass than in 2000. The average carbon stock in biomass per hectare for a certain age class has increased as well, due to high forest productivity in Belgium (PERRIN ET AL., 2000; LAITAT ET AL., 2004; PERRIN, 2005) Increased carbon stocks in biomass may lead to increased SOC levels. LISKI ET AL. (2002) who find a carbon sink in western European forest soils for the year 2000, attribute it mainly to increased litter fall from living trees. Their inventory data show an increasing growing stock between 1950 and 2000. A second and maybe more important source of SOC is belowground biomass. The total carbon flux from roots may be twice as high as the C from litter fall in mature forests (DAVIDSON ET AL., 2002). Total belowground C allocation (root mortality and carbon exudates) increases with forest productivity. Besides the increased C stocks in biomass, larger amounts of residues of harvests and natural disturbances also have a positive feedback on soil C content as (JOHNSON ET AL., 2001). Following guidelines for sustainable management, or simply with a view to reduce costs, Belgian foresters nowadays tolerate more dead wood in their forests than in the 1960s (AFDELING BOS & GROEN, 2001).

3.2.6.3. Issues of computational methodology

Comparing data from inventories with a 40-year difference incurs a number of difficulties. This is especially true for agricultural soils, since technology has changed considerably, notably leading to an increase in plough depth that is difficult to incorporate in the computations. In 1960, a tillage depth of 22 cm is reported (VAN MEIRVENNE ET AL., 1996; VAN OOST ET AL., 2000). From 1970 onwards, as a result of mechanization, tillage depth increased and stabilized around a region-specific depth. VAN OOST ET AL. (2000) reported 25 cm in the Loam Belt and VAN MEIRVENNE ET AL. (1996) measured 36 cm in North Western Belgium. When tillage depth increases, the top soil with a relatively high SOC content is mixed with subsoil containing less SOC, leading to a decrease of SOC content in the original topsoil but also preserves a constant SOC content of the now deeper topsoil layer. VANMEIRVENNE ET AL. (1996), found for north-western Belgium that, between 1951 and 1990, the plough depth increased from 22 cm to 36 cm, while the SOC % decreased from 1.41% to 1.39%. Since the carbon rich topsoil gained 14 cm (from 22 cm to 36 cm), while losing only 0.02% of SOC (from 1.41 to 1.39%), the net effect was an increase in SOC for the upper 36 cm. Other authors argue that increasing the plough depth leads to increased oxidation of organic matter, and more intensive erosion, thus possibly decreasing total C stocks of the profile (JANZEN, 2005; FREIBAUER ET AL., 2004). In the present study, most carbon measurements in cropland reach 20 cm or 23 cm, possibly leading to errors in the estimation of the SOC content in 1990-2000 if the depth extrapolation factors derived from the 1960 data are applied. This effect is assumed negligible in this study and certainly does not play a role for the results of the upper 20 cm.

The SOC analysis techniques are a second methodological point of interest. For most databases, the original WALKLEY AND BLACK (1934) procedure was applied, with a correction factor for incomplete SOC determination of 1.32 (WALKLEY AND BLACK, 1934; GILLMAN ET AL., 1986). However, research has indicated that the correction factor may differ depending on climate, soil and land use (DROVER ET AL., 1975; DIAZ-ZORITA, 1999; BRYE ET AL., 2003). Moreover, apparently small details of the analytical procedure may strongly influence the results. For instance, SKJEMSTAD ET AL. (2000) found that the use of smaller flasks during Walkley and Black analysis increases temperature to such an extent that a correction factor becomes superfluous. All laboratories but two, providing data for this study, used the same original Walkley and Black procedure. However, the SOC values under agriculture in Flanders for 1990 and 2000 were analysed using a modified Walkley and Black procedure, which consists of boiling the soil-bichromate-sulfuric acid mixture for 5 minutes. We derived and applied a specific conversion factor of 1.14 instead of 1.32 to correct measured SOC (LETTENS ET AL., 2005). A second exception is the forested LSU in Flanders, for which SOC concentration values are obtained by loss-on-ignition (LOI). The SOC values are estimated by regression equations based on a 20% calibration subset of all LOI determinations in comparison with TOC measurements (DE VOS ET AL., submitted). A full ring test including all laboratories would be useful to prove homogeneity. However, not only is it costly to undertake such a test, but also some laboratories that analysed the 1960 data have ceased to exist.

Several authors have stressed the important contribution of the ectorganic horizons (EOH) to the total SOC stocks of (semi-) natural ecosystems (e.g. GUO AND GIFFORD, 2002; TATE ET AL., 2003). We have data on depth and OC concentration of ectorganic layers at our disposal for all forests in 1960 and for forests in Flanders in 2000. For 2000, SOC values of 10 t C ha^{-1} (LSU under broadleaf forest), 20 t C ha^{-1} (mixed forested LSU) and 35 t C ha^{-1} (LSU under coniferous forest) are obtained. In 1960, Flemish EOH stored 29 t C ha^{-1} (broadleaf forest), 46 t C ha^{-1} (mixed forests) and 31 t C ha^{-1} (coniferous forests). The discrepancy between both years may be attributed to the different BD that has been used. Equation 3.1 was applied for 1960, while computation in 2000 used a fixed value from literature that was 5 to 13 times smaller. Application of equation 3.1 probably caused an overestimation of the EOH bulk density. Using the literature value for BD, Flemish EOH in 1960 contain 4 t C ha^{-1} (broadleaf forest), 7 t C ha^{-1} (mixed forests) and 7 t C ha^{-1} (coniferous forests). It then becomes clear that SOC stocks in EOH increased strongly, which agrees well with the observed increases in mineral SOC of forests.

3.3 CO_2 FLUXES AS A RESULT OF CHANGES IN FOREST BIOMASS

3.3.1 Data sources

3.3.1.1. Forest inventories

Belgium has a temperate maritime climate, with moderate temperature variability, prevailing westerly winds and well-distributed rainfall. The distribution of forests in Belgium is shown in Table 3.4. Forests covered 6931 km² or 22,7 % of the national territory in 2000. Deciduous and coniferous species covered respectively 51 and 49 % of the area. Spruce (*Picea abies* (L.) Karst) is the more important species, followed by oaks (*Quercus* sp) and pines (*Pinus* sp). After these inventories, Belgium has the second highest net annual biomass increment (after Germany) of the 48 countries considered, of which most are Annex I countries(UNECE/FAO, 2000).

Table 3.4: *Forest cover in Belgium*

Region	Total area	Forest area [1]	Forest cover	Part in the Belgian forest
	(km^2)	(km^2)	(%)	(%)
Wallonia	16845	5448	32.3	78.6
Flanders	13521	1447	10.8	21.1
Brussels Capital	162	20	12.3	0.3
Belgium	**30528**	**6931**	**22.7**	**100.0**

[1] *total forest area including productive (stands harvested for commercial wood and non productive surfaces like ponds, roads, tree nursery, ...)*

The principal data come from the Walloon and Flemish forest inventories. They were conducted both in the two regions using similar inventory techniques. The inventories are drawn up by sampling to determine the surfaces by categories of property (Private or Public: State, Province, Community), type of forest, species, age, size and quality. They also provide estimates of the volume of standing timber and of their growth, harvest and future potential yield for species of economic relevance. The sampling plots are selected according to a 1 000 m x 500 m grid directed from the East to the West on the National Geographic Institute maps at a scale of 1/25 000. A sample plot (a circle with a diameter of 0.1 ha) is laid out on each grid intersection wherever a forest is indicated on the map.

The first Walloon forest inventory was completed in 1984. The current permanent systematic sampling started in 1994 and covers each year 10 % of the approximately 11,000 sampling points (LECOMTE ET AL., 1994). In 2000 (reference year for this study), 50 % of the sample points of the second inventory were measured. In Flanders, 2,665 plots were sampled in the framework of the first forest inventory over the period 1997-1999 (AFDELING BOS & GROEN, 2001). It is foreseen to repeat this regional inventory every 10 years, to allow e.g. the calculation of growth rates in the Flemish forests.

With more than 13,000 plots over a territory of 30,528 km^2, forest inventories in Belgium have one of the highest sampling densities in Europe. Compared to other countries or regions, the Belgian sampling grid, with each sampling point representing 50 ha of forest, is very dense (LAITAT ET AL., 2000; DIETER ET AL., 2002). In comparison, one plot represents 2400 ha of forestland in the U.S. (BROWN, 2002).

The forest inventories results are shown in Table 3.5. The Walloon and Flemish Administration provided the data used within the framework of this study. They are the ages and the solid wood for each sample plot. The geographical co-ordinates of the central point are also provided. The complex treatment process of the forest inventories is described in PERRIN (2005) It included the calculation of volumes for young plantations and the growth increments per age classes.

***Table 3.5**: Forest inventories results in 2000. The areas represent the productive forest only.*

	Wallonia			**Flanders**		
Species	**Area (ha)**	**Volume (1000 m^3)**	**% of total volume**	**Area (ha)**	**Volume (1000 m^3)**	**% of total Volume**
Pine	14800	3743.4	3.0	63550	12867.2	39.9
Douglas fir	10800	2387.2	1.9	1280	371.0	1.2
Larch	8200	2081.2	1.7	3060	782.3	2.4
Spruce	171700	52502.8	41.8	2860	527.1	1.6
Other coniferous	19600	4955.4	3.9	910	174.0	0.5
Total coniferous	225100	65669.9	52.2	71660	14721.5	45.7
Beech	42200	12278.0	9.8	7790	2500.5	7.8
Oak	81600	20372.4	16.2	14320	3696.4	11.5
Mixed noble	57100	15041.4	12.0	10250	2357.0	7.3
Poplar	9500	2703.9	2.2	19060	5217.2	16.2
Other deciduous	43200	9661.7	7.7	21650	3753.1	11.6
Total deciduous	233600	60057.3	47.8	73070	17524.1	54.3
TOTAL	**458700**	125727.1	100.0	144730	32245.5	100.0

3.3.1.2 Conversion factors

The calculation of the amount of carbon stored in the biomass of trees is usually based on biomass expansion factors *s.l.* (NABUURS ET AL., 1993 ; BROWN, 2002; GRACIA ET AL., 2002). For each dominant species, we transformed (Tables 3.6 and 3.7; Fig. 3.6):

- Volume of solid wood in total dry mass multiplying by the infra-densities (WD)
- Solid wood total dry mass in total above-ground dry biomass (biomass expansion factor 1 or BEF 1)
- Above-ground dry biomass in total dry biomass (roots included, biomass expansion factor 2 or BEF2)
- Total dry biomass in carbon quantities (carbon content or CC).

Some explicit conditions were applied for the selection of biomass expansion factors s.l. from the literature. For the expansion factors s.s., foliage had to be included, in accordance with the IPCC-methodology (IPCC, 2005). The analysis was limited to data reported for Austria, Belgium, Denmark, France, Germany, Great Britain, Ireland and The Netherlands. Values were selected for 'coniferous' and 'broadleaf' species separately, but also for the most important tree species in the Belgian forests: pine (*Pinus* sp.), Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), larch (*Larix sp.*), Norway spruce (*Picea abies* (L.) Karst.), beech (*Fagus sylvatica* L.), oak (*Quercus robur* L. and *Q. petraea* L.), mixed 'noble' species (including maple (*Acer pseudoplatanus* L.), elm (*Ulmus* sp.), ash (*Fraxinus excelsior* L.), red oak (*Quercus rubra* L.)) and poplar (*Populus* sp.).

We established the frequency distribution of the values used in neighbouring countries and selected the median (see VANDE WALLE ET AL., 2005). The selected factors are shown in Table 3.6.

***Table 3.6**: Selected conversion and biomass expansion factors (VANDE WALLE ET AL., 2005)*

Species	Wood density ($t.m^{-3}$)				Carbon content ($t.t^{-1}$)			
	Min	Max	Med	#	Min	Max	Med	#
Spruce	0.34	0.45	0.38	15	0.40	0.51	0.50	5
Pine	0.39	0.60	0.48	13	0.40	0.55	0.50	9
Douglas fir	0.37	0.54	0.45	7	0.50	0.50	0.50	1
Larch	0.41	0.55	0.47	8	0.40	0.50	0.50	3
Other resinous	0.35	0.50	0.40	20	0.40	0.50	0.50	7
Beech	0.55	0.72	0.56	11	0.44	0.51	0.49	10
Oaks	0.50	0.72	0.60	9	0.45	0.50	0.50	3
« Nobles » species	0.52	0.69	0.59	9	0.50	0.50	0.50	1
Poplars	0.34	0.55	0.41	48	0.50	0.50	0.50	1
Other deciduous	0.38	0.77	0.55	34	0.45	0.50	0.50	6

Species	BEF 1 : Total aboveground biomass / Solid wood mass ($t.t^{-1}$)				BEF 2 : Total below ground biomass / total above ground biomass ($t.t^{-1}$)			
	Min	Max	Med	#	Min	Max	Med	#
Spruce	1.14	1.71	1.29	9	0.2	0.2	0.2	
Pine	1.14	1.40	1.32	5	0.16	0.16	0.16	1
Douglas fir	1.18	2.24	1.28	10	0.17	0.17	0.17	1
Larch	1.14	1.36	1.30	3	0.2	0.2	0.2	
Other resinous	1.14	1.71	1.33	5	0.18	0.25	0.20	3
Beech	1.16	2.04	1.34	9	0.23	0.25	0.24	2
Oaks	1.24	1.39	1.32	2	0.2	0.2	0.2	
« Nobles » species	1.29	1.29	1.29	1	0.2	0.2	0.2	
Poplars	1.4	1.4	1.4		0.2	0.2	0.2	
Other deciduous	1.24	1.40	1.32	2	0.20	0.22	0.21	2

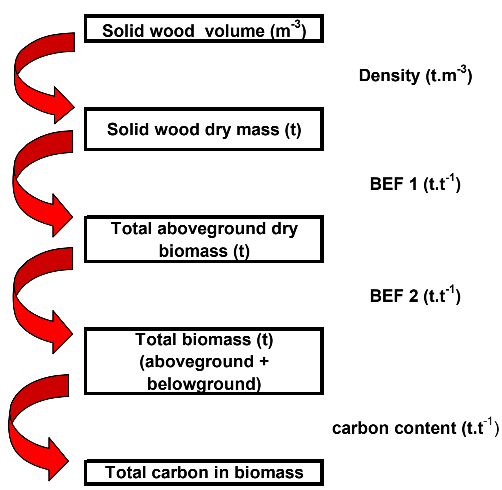


***Figure 3.6** Principle of the conversion of wood volume to carbon mass*

***Table 3.7** Principles of the EFOBEL model*

Age classes	2000 (reference Y0)		2001 (Y1=Y0+1)			...
	Areas (A in ha)	Solid wood volume (S in m^3)	Areas (A in ha)	Wood growth (W in $m^3.ha^{-1}.y^{-1}$)	Solid wood volume (S in m^3)	
0 - 1 y	$A1_{Y0}$	$S1_{Y0}$	$A1_{Y1}$ = areas of young trees		S1 Y1= young trees volumes	
1 - 2 y	$A2_{Y0}$	$S2_{Y0}$	$A2_{Y1} = A1_{Y0} + \Delta A1_{Y0}$	$W1_{Y1}$	$S2_{Y1}=S1_{Y0}+(A2_{Y1}*W1_{Y1})$	
2 - 3 y	$A3_{Y0}$	$S3_{Y0}$	$A3_{Y1} = A2_{Y0} + \Delta A2_{Y0}$	$W2_{Y1}$	$S3_{Y1}=S2_{Y0}+(A3_{Y1}*W2_{Y1})$	
⋮						

3.3.2 Modelling the evolution 2000-2012 of carbon stocks in the forest biomass: EFOBEL model

PERRIN ET AL. (2000) published a first model for the evolution of carbon stocks in the biomass of the forest trees. It was based on the Walloon forest inventories results of 1984 and 1999. The base of this first model was a linear interpolation of surfaces and total volumes between these two years of measurements. This work showed the existence of a forest carbon sink in Wallonia, with an annual sequestration of 1823 ktCO_2 per year in 1999 equivalent to 1.8 % of the total emissions of Belgium in 1990.

We then amended this first model in order to take into account the impacts of forest management practices on carbon stocks, to allow a calculation of uncertainties and to couple predictions of biomass carbon with estimates of soil carbon fluxes from YASSO. For this purpose, we took into account work of HÉBERT AND LAURENT (1995) and SHERIDAN (2002), simulating the dynamics of forest stands. We adapted and developed these algorithms in order to estimate the evolution of C stocks in the forest ecosystems in the short term. Both a single inventory and measurements of the forest growth increment by species between inventories were used to construct EFOBEL.

A full description of EFOBEL model can be found in LAITAT ET AL. (2004) and PERRIN (2005). The inputs of the model refer to every grid cells of the Belgian Forest Inventories as published for the year 2000: the solid wood volume and the area of the stands, by species and by age classes. The parameters are the annual growth increment for each species, the revolution, the period between harvest and replanting (also called latency), and the percentage replacement of one species by another according policy rules of the respective forest administrations. Table 3.7 shows the data processing from year to year for resinous and deciduous species. Productive areas (A) and solid wood volumes (S) are aggregated by species and age classes as established by the forest inventories for 2000 the first year of calculation (y_0). Surfaces and volumes of an age class "x" at the year "y_t" correspond to the surface of the age class "x - 1" at the year "y_t - 1" times a factor expressing reduction in forest area (ΔA) (in case of e.g. clear cut) and wood growth increment (W). Hence, surfaces and volumes "slip" from year to year from one class to the other until the harvest.

The deciduous species are classified according to their structure. Three structures are distinguished: "plantations" during ten years, "young stands" during the 30 consecutive years, and "mature stands" until the term of the harvest. At each year of simulation, 1/10th of the "plantations" slips into the class "young stands", and 1/30th of the "young stands" slips into the class "mature stands". In each annual simulation cycle, volumes evolve following the increase in volume and the harvest corresponding to the exploitation of the forests. The exploited percentage of surface is called the "removal rate". It depends on the specie or the age group or the structure. For the young stands, the rate of removal is 0 %. For the "coniferous trees" and "poplars", EFOBEL leaves the assumption that the harvest are spread out in time: the removal rates are fixed on the basis of distribution in Normal curve with a 20 years amplitude, i.e. a decadal interval on both sides of the revolution (PERRIN, 2005 for detailed methodology).

For the deciduous category, we considered the "annual availability". Within a plot, the assumption was a regular distribution of surfaces by age groups. The « annual availability » corresponds to the surface of the plot divided by the revolution. Each year, the availability is withdrawn mature stands HÉBERT AND LAURENT (1995). The assumptions of the model were described in PERRIN (2005). The outputs of this first step are the evolution of solid wood volume per sample plot, until 2012. For the 1990 – 1999 BOC evolution, we used a linear extrapolation. PERRIN ET AL. (2000) calculated the 1990 stock in the Walloon Region. For the Flemish forest, the linear extrapolation was based on the total forest area estimated for the years 1990 and 2000, and the knowledge of the harvested wood products during this period (VANDE WALLE ET AL., 2005).

3.3.3 Results and discussion

3.3.3.1 2000 BOC stock of Belgium based on forest inventory points

A first calculation of BOC was made by the conversion of the total solid wood volume to carbon for Belgium in 2000. Based on the use of median BEF parameter values for all species, the total carbon stock in Belgian forests amounted to 59.9 Mt C (LAITAT ET AL., 2004; VANDE WALLE ET AL., 2005). In total, 79.8 % of the Belgian living forest biomass carbon stock was located in Walloon forests, while Flemish forests contained 20 % of the forest carbon. The minimum parameter for each BEF resulted in a value of 42.8 Mt C for the total carbon stock in Belgium, while using the maximum BEFs s.l., BOC amounted to 83.5 Mt C.

The mean carbon stock in the biomass of Belgian forests was higher than the 59 ton C ha^{-1} in Dutch forests reported by NABUURS AND MOHREN (1993). CANNELL (1982) reports that the mean carbon stock in British broadleaved woodland is 62 ton C ha^{-1} on average and 21 ton C ha^{-1} in the much younger conifer stands. In Danish forests, BOC reaches 56.6 ton C ha^{-1} (VESTERDAL, 2000) and in France 59.0 ton C ha^{-1} (PIGNARD ET AL., 2000). All these figures are considerably lower than the stock in Belgium. However, German forests, with a stock of 103.4 ton C ha^{-1} contained a higher BOC stock per area than Belgian forests (BARITZ AND STRICH, 2000). Austrian forests appeared to have a carbon stock which was similar to the Belgian one at 82.1 ton C ha^{-1} WEISS ET AL. (2000) . All these results are of course dependent on the biomass expansion factors *s.l.* used for the calculations.

We proceeded to a Monte Carlo analysis to estimate the uncertainty linked to the carbon stock calculation. We combine the uncertainty linked to the forest inventories and the BEF selection. The forest inventories error is a combination of the error due to the measurement techniques (diameter, height, number of trees per plot) and the error linked to the surface and volume estimation for the whole region (error dependent on the sampling plot number per species; see RONDEUX (1999) for the methodology). For the BEF, a frequency distribution was established for the values used in neighbouring countries, assuming a normal distribution of the selected variable. The relative error could be derived from this frequency distribution. Consecutively a Monte Carlo analysis (10000 simulations) was applied to the calculation of the 2000 stock. According to our calculations, the relative confidence interval (CI95%) on the mean is 15.7% (PERRIN, 2005).

The prediction of BOC results from various scenarios. The most probable scenario for the period of 2008 to 2012 is the continuation of the current forestry practices in a scenario "Sylviculture as usual". The parameters relating to such a scenario are obtained either by the inventories themselves, or from the literature. For the "revolution " we adopted the following values:

- Spruce: the estimated ages of clear cut are indicated by the inventory. From the 1600 ha of clear cut plots, we calculated 72 years as an average revolution for public spruce stands and 56 years for private stands.
- Douglas: BOUDRU (1986) indicate a revolution from 70 to 80 years with an average of 75 years. The trees then reach a diameter from 60 to 80 cm and a total height from 35 to 40 m. We fix the average revolution at 75 years.
- Pines and "Other coniferous trees ": the average revolution is fixed at 60 years.
- Poplars: on the basis of forest inventories results, the average revolution is fixed at 40 years.
- Beeches, oaks and "Other deciduous ": the average revolution is fixed at 150 years. The average "latency" calculated starting from the data is three years.

The "conversion of the plantations of spruce" results from the compared analysis of the Walloon inventories from 1984 and 1999, which showed an annual reduction in the surface occupied by the spruce of 0,4 %, of which 0,3 % are converted into "non-productive" surfaces and 0,1 % in other coniferous or broadleaf stands. The same tendency is observed in Flemish forest for pines, converted mainly to broadleaf and other coniferous. "Sylviculture as usual" increases this stock by + 1,5% to 2,0% a year of the base year emissions (1990). Carbon stocks will than reach 101 t C ha^{-1} in 2012 (Figure 3.7).

In short-term scenarios, only longer revolution periods will significantly increase carbon stocks by + 6 to + 12% (for respectively Pine and Spruce) depending of the selected species. Changes in the latency or the specific composition of the forest do not affect the short-term carbon sequestration. The difference between the BAU scenario and the maximum scenario represents the maximum sequestration potential for article 3.4 of the Kyoto Protocol.

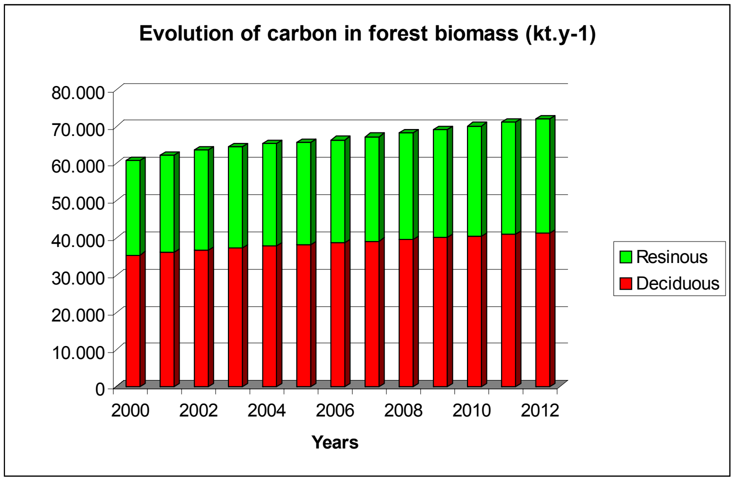


Figure 3.7 *Sylviculture as usual scenario of BOC evolution*

3.3.4 BOC stock change between 1990 and 2012 per landscape unit

BOC values apply only to forested LSUs. According to CLC-1990 and the Belgian Soil Association map, there are 159 forested LSUs in Belgium, with an average area of 39 km², divided over several polygons. They cover together 6,222 km², of which 2,000 km² under broadleaf forest, 1,407 km² under coniferous forest and 2,815 km² under mixed forest. The three forest types do not occur on all possible soil associations. In total 13 of the existing 65 associations do not contain any forest.

Since soil parameters are not measured in the Flemish inventory and the land use type mixed forest is not recognized in the Walloon inventory, classmatching of BOC data to an LSU is possible for part of the Belgian territory only. Through geomatching, the forest inventory point measurements are attributed to the LSU in which they are geographically situated. Compared to classmatching, this type of matching is less demanding in terms of ancillary information, since only geographic coordinates need to be available. The systematic sampling design of the forest inventories allows the assessment of within and between-LSU variability through geomatching. Consequently, this procedure is applied to the BOC datasets for 1990 (interpolated values), 2000 (measured values), 2012 (values modelled by EFOBEL). To those LSUs for which it fails to select any forest inventory points, the average BOC stock per forest type and region (Flanders or Wallonia) is attributed.

For 133 of the 159 studied LSUs, covering 6,204 km² or 99.7% of the total considered forest area, a BOC value could be determined using the geomatching procedure with the 2000 BOC dataset. The total number of observations included in the calculations (6,592) is smaller than the original number of observations (9,335) because with geomatching, only those plots situated within forested LSU as derived from CLC-1990 are used. The highest BOC stocks occur on the fertile deep loam or sand loam soils of central Belgium. The four associations with the highest BOC value are the associations 31 (162 t C ha^{-1}), 36 (155 t C ha^{-1}), 39 (136 t C ha^{-1}) and 37 (123 t C ha^{-1}) (data not shown). Low BOC stocks occur on dune soils (e.g. association 1 with 33 t C ha^{-1} and association 2 with 77 t C ha^{-1}), peat soils (e.g. association 54 with 56 t C ha^{-1} and 60 with 77 t C ha^{-1}) and sandy soils (e.g. association 25 with 70 t C ha^{-1}, or 15 with 73 t C ha^{-1}). BOC stocks per forest type in the year 2000 are as follows: broadleaf forest contains the largest stocks of OC (100 t C ha^{-1}) followed by coniferous forest (95 t C ha^{-1}) and mixed forest (87 t C ha^{-1}). BOC of the mixed forest type is significantly lower than the two other forest types, which do not differ significantly from each other. The geographic distribution of the BOC stock in Belgian forests in 2000 is shown in Figure 3.8. The high BOC stocks of the broadleaf forest on loam soils in central Belgium are visible, as well as some LSUs in the Walloon Ardennes, which are dominated by the soil associations 50, 51, 52 and 62 and the land use type coniferous or broadleaf forest. The low BOC stocks on the sand soils of Kempen, Sand Region and the coast region are visible as well.

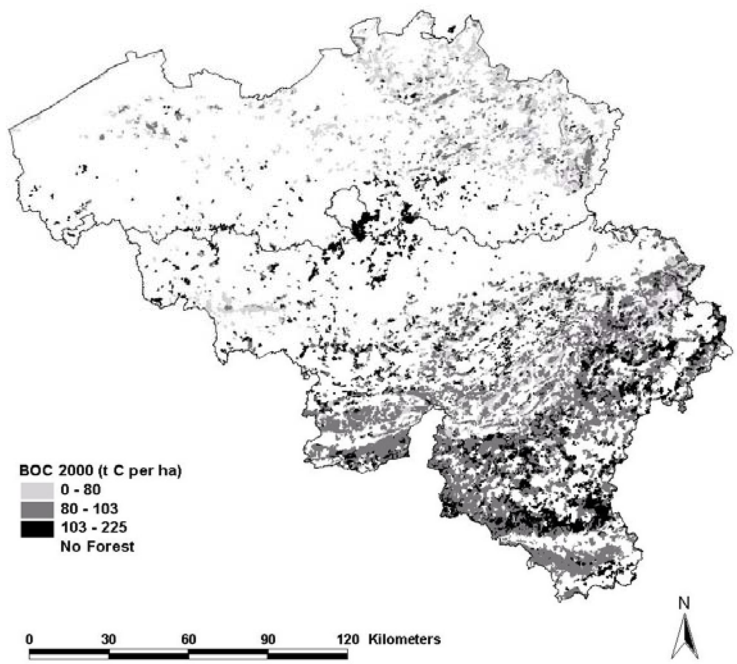


Figure 3.8 *Biomass organic carbon stock (t C ha^{-1}) in broadleaf, coniferous and mixed forest in 2000.*

The total BOC stock in 2000 in Belgian forests as delineated by CLC-1990 amounts to 57.8 Mt C in 6,222 km². Broadleaf forests store 19.9 Mt C (2,000 km²), coniferous forest 13.4 Mt C (1,407 km²) and mixed forest the remaining 24.5 Mt C (2,815 km²). Flemish forests store 9.4 Mt C (1,105 km²) and Walloon forests 48.4 Mt C (5,117 km²). This means that the average BOC stock in Flanders (85 t C ha^{-1}) is smaller than in Wallonia (95 t C ha^{-1}). Especially coniferous forests contain more BOC in Wallonia (103 t C ha^{-1}) than in Flanders (78 t C ha^{-1}). Flemish forests are on average younger than Walloon forests, with respectively

49% compared to 26% of the area covered by forest younger than 40 years. This could explain the lower BOC content of Flemish forests. Additionally, forests in Flanders occur mainly on poor sandy soils, with annual increments half as high as in Wallonia.

Applying the geomatching procedure to the modeling results per forest inventory point for the years 1990, 2000 and 2012, allows the computation of BOC fluxes between those years. The average fluxes per forest type are indicated in sections on GHG synthesis (3.5, 4.4.1 and 4.4.2).

3.4 NITROUS OXIDE FLUXES FROM AGRICULTURAL SOILS

3.4.1 Introduction

A methodology is proposed to take into account the spatial and temporal variability of the N_2O emissions from agriculture in national GHG inventories. Observed annual N_2O emission rates are used to establish statistical links between N_2O emissions, land use, seasonal climate, and nitrogen-fertilisation rate. This methodology is applied to Belgium over the period from 1990 to 2050. Nitrous oxide emissions are simulated taking into account different fertilisation rules to constrain the emissions. The evolution of the emissions during the period 1990 – 2050 is examined at the regional and national scale. Fertilisation data for this model are calculated using data at the municipality level and therefore need to be dis- and then re-aggregated to the level of the LSU. A similar methodology as described above for SOC content (3.2.3) is applied to achieve this.

N_2O is naturally produced in soils as a by-product of microbial processes. N_2O emissions from agricultural soils not only depend on the addition of nitrogen, but also on other factors, e.g. climate, soil and management practices (FIRESTONE AND DAVIDSON, 1989; GRANLI AND BOCKMAN, 1994; DAVIDSON AND VERCHOT, 2000). N_2O production varies both in time and in space due to the complex interactions between all these factors. Where soil moisture and mineral nitrogen availability are not limiting, seasonal and diurnal changes in temperature have been shown to be directly correlated with N_2O emissions for many soils in temperate climates (SKIBA AND SMITH, 2000; SKIBA ET AL., 2002). Land-use affects N_2O emission patterns at the field-scale. On an annual basis, croplands emit less N_2O than grasslands and differences appear among the different crop types (SKIBA ET AL., 1996; WAGNER-RIDDLE ET AL., 1997; HENAULT ET AL., 1998; KAISER ET AL., 1998a; SMITH ET AL., 1998a; DOBBIE ET AL., 1999). They show distinct emission patterns as a result of differences in management and nitrogen-fertilisation applied (Figure 3.9).

Different modelling approaches are used to predict N_2O emissions. Complex-mechanistic models consider all the proximal factors acting on nitrification and denitrification processes. Models of medium complexity have been developed with the objective of simulating terrestrial ecosystem carbon and nitrogen biogeochemistry (e.g. DAYCENT: PARTON ET AL. (2001); DNDC: LI ET AL. (1992); CASA: POTTER ET AL. (1996)). Application of these process-based models is hampered by the nature and the amount of input data required. At the simple end of the model-complexity spectrum are the highly empirical N_2O

flux models based on statistical analysis (EICHNER, 1990; BOUWMAN, 1996; KAISER AND RUSER, 2000; FREIBAUER, 2003) . Empirical models focus on the main driving variables and provide an opportunity to study system-responses rapidly and thus, assess uncertainty in the input variables. Most empirical N_2O models use the concept of the fertiliser-induced emission factor, which is defined as the emission from fertilised plots minus the emission from unfertilised control-plots expressed as a percentage of the N applied BOUWMAN ET AL. (2002a).

Simple regression models based on published N_2O soil-emission data have recently been included within a GIS framework to describe emissions in terms of broad environmental factors (SOZANSKA ET AL., 2002; BARETH AND ANGENENDT, 2003; LILLY ET AL., 2003). SOZANSKA ET AL. (2002) used a multilinear relationship to link N_2O emissions to soil moisture, soil temperature and soil carbon content. LILLY ET AL. (2003) did the same for soil nitrate content. BOUWMAN ET AL. (2002a) used a Residual Maximum Likelihood technique to link N_2O emissions to environmental factors, management related factors and factors related to the measurements. FREIBAUER ET AL. (2003) made the link with soil nitrogen content, soil drainage class and climatic region defined in terms of the probability of having seven consecutive freezing days.

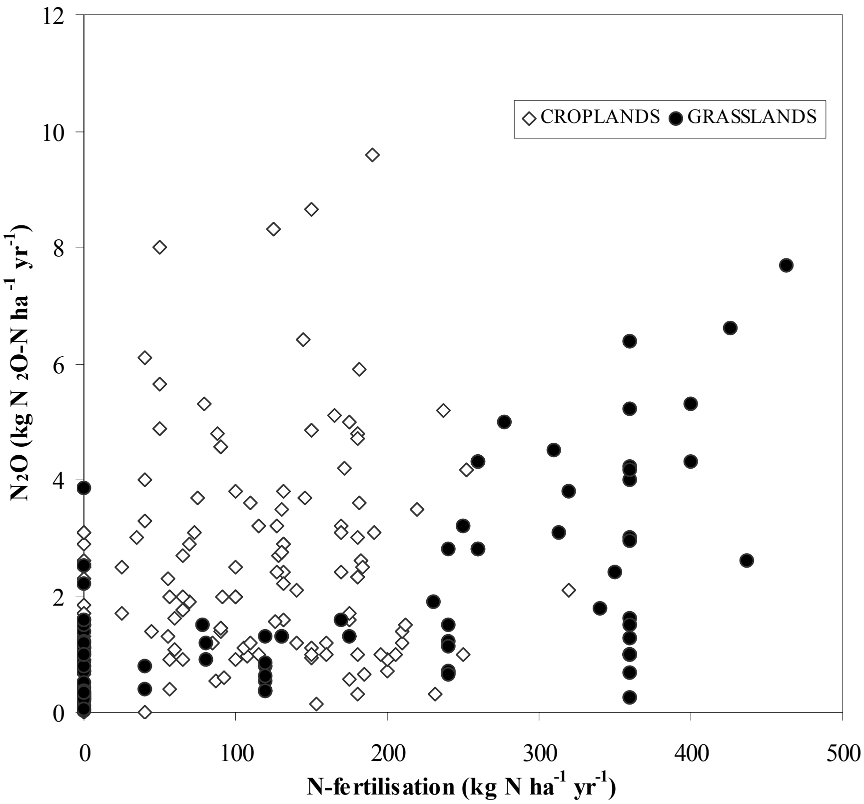


Figure 3.9 *Annual N_2O emissions (kg N_2O-N ha^{-1} yr^{-1}), from croplands and grasslands, as a function of nitrogen fertilisation (kg N ha^{-1} yr^{-1}).*

3.4.2 The N_2O emission models

The methods used to produce the models are described extensively in ROELANDT ET AL. (2005). A data base was constructed that included annual N_2O emissions reported in the literature completed with: soil texture (% clay, % silt, % sand), soil carbon and nitrogen content, nitrogen fertilisation rate, mean monthly air temperature and monthly precipitation. Emission data from croplands and grasslands were analysed separately. The statistical analysis was based on multi-linear regression and was carried out with the SAS software (SAS Institute Inc., Cary, NC, USA). A random sampling technique was used to create independent calibration and validation data sets.

3.4.2.1 Cropland model

The data set used for the model calibration included 80 observations. No correlation between N-fertilisation (N_{fert}) and N_2O emissions for croplands was found during the statistical analysis. A strong and highly significant correlation appeared between spring temperature (T_{spring}) and N_2O emission rate ($R^2 = 0.33$, $P < 0.0001$). A weaker, but significant, correlation occurred between summer precipitation (P_{summer}) and N_2O ($R^2 = 0.19$, $P < 0.0013$). By combining these variables a model sensitive to climate was obtained (equation 3.9):

$$N_2O = K + \beta_1 T_{spring} + \beta_2 P_{summer} \tag{3.9}$$

This model is further referred to as MCROPS with 'M' pointing to the METAGE project. For the calibration, MCROPS explained nearly 40% of the variance (adj.-$R^2 = 0.38$) of the annual N_2O emissions. The intercept and the parameters were highly significant. To reduce the influence of extreme data values, bias corrected parameters and confidence limits were estimated using a bootstrap technique. Corrected parameter values with their 95% confidence limits were:

$K = -8.1834$ ($\pm$ 5.2372) kg N_2O-N ha^{-1} yr^{-1};
$\beta_1 =$ 0.7511 ($\pm$ 0.3210) kg N_2O-N ha^{-1} yr^{-1} °C^{-1};
$\beta_2 =$ 0.0144 ($\pm$ 0.0180) kg N_2O-N ha^{-1} yr^{-1} mm^{-1}.

MCROPS was validated against observations from an independent data set (N = 56) (Figure 3.10). The model predicted the emissions with a R^2 equal to 0.35 (adj.-$R^2 = 0.34$) and a RMSE equal to 0.67 kg N_2O-N ha^{-1} yr^{-1}. The regression equation between observations and predictions was: Observation = -0.06 + 0.53 Prediction. The prediction term was significant, while the intercept was not. These results should be considered with caution: 1. the validation equation passes close to the origin; 2. the points predicted at 8 kg N_2O-N ha^{-1} yr^{-1} on their own strongly increase the validation R^2, but when these two points are removed the regression equation remains constant (results not shown). Compared to the BOUWMAN (1996) equation based on nitrogen fertilisation, MCROPS detects a trend through the N_2O emissions. The BOUWMAN (1996) equation does not show a significant relationship when applied to the validation set.

MCROPS is consistent with the general understanding that much more N_2O is emitted from soils that are warm and wet than those that are cold and dry (GRANLI AND BOCKMAN, 1994). Water availability and temperature control microbial activity and process kinetics. When nitrogen is not limiting, an exponential relationship between N_2O, water filled pore space and soil temperature is observed (SMITH ET AL., 1998b; DOBBIE ET AL., 1999; DOBBIE AND SMITH, 2003). During spring, when soils are mostly wet, temperature is the factor controlling microbial activity. During summer, water may be limiting and an increase in rainfall contributes positively to the annual emission rate (SMITH ET AL., 1998b; CLAYTON ET AL., 1997) .

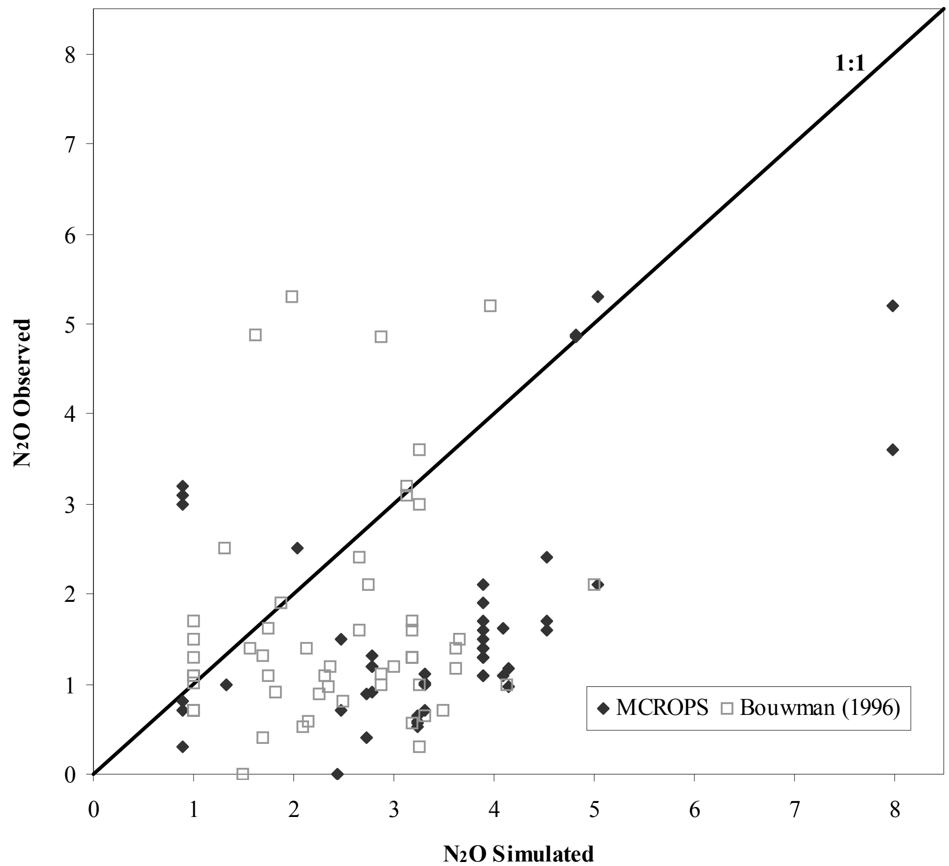


***Figure 3.10** MCROPS validation and comparison with* B*OUWMAN (1996) model.*

3.4.2.2 Grassland model

The modelling exercise focused on N_2O emissions characterised by nitrogen fertilisation rates lower than 500 kg N ha^{-1} yr^{-1}. Higher values were considered excessive compared to normal practice even in regions with very intensive agriculture. The effect of management options, such as grazing, mowing or stocking density were not investigated. The calibration data set comprised 59 observations.

On its own, N_{fert} explained 39% of the emissions variance. Seasonal climate during the growing season appeared significant, but the fraction of variance explained remains below 0.20. Winter temperature (T_{winter}) was significant and explained as much of the emissions variance as fertilisation. Combining N_{fert} and T_{winter}, the multivariate MGRASS model was obtained (equation 3.10):

$$N_2O = e^{(K+\beta_1 N_{fert}+\beta_2 T_{winter})} \qquad \textbf{3.10}$$

The parameters and the intercept were significant. The bias-corrected parameters were: K = –0.5095 (± 0.4083); β_1 = 0.0028 (± 0.0019); β_2 = 0.1245 (± 0.0929).. Validated against an independent data set (N = 36), MGRASS explained 48 % (adj.-R^2 = 0.47) of the observed emission variance. For the validation, MGRASS the RMSE was equal to 0.34. The MGRASS model predicts the observed N_2O emissions reasonably well except for a slight tendency to overestimate in the low emission range (Figure 3.11). The effects of nitrogen fertilisation rate on annual N_2O emissions from grasslands have been demonstrated in the literature (EICHNER, 1990; BOUWMAN, 1996; BOUWMAN ET AL., 2002b; KAISER ET AL., 1998b; KAMMANN ET AL., 1998) The strong link between ln(N_2O) and T_{winter} is the result of the limitation of microbial activity by temperature. The absence of soil-related predictors is partly due to the lack of data, but as suggested by FREIBAUER AND KALTSCHMITT (2003) N_2O emissions from grasslands do not correlate with soil information. MGRASS succeeded in passing all statistical tests and simulated values in the range of the observations during the validation process.

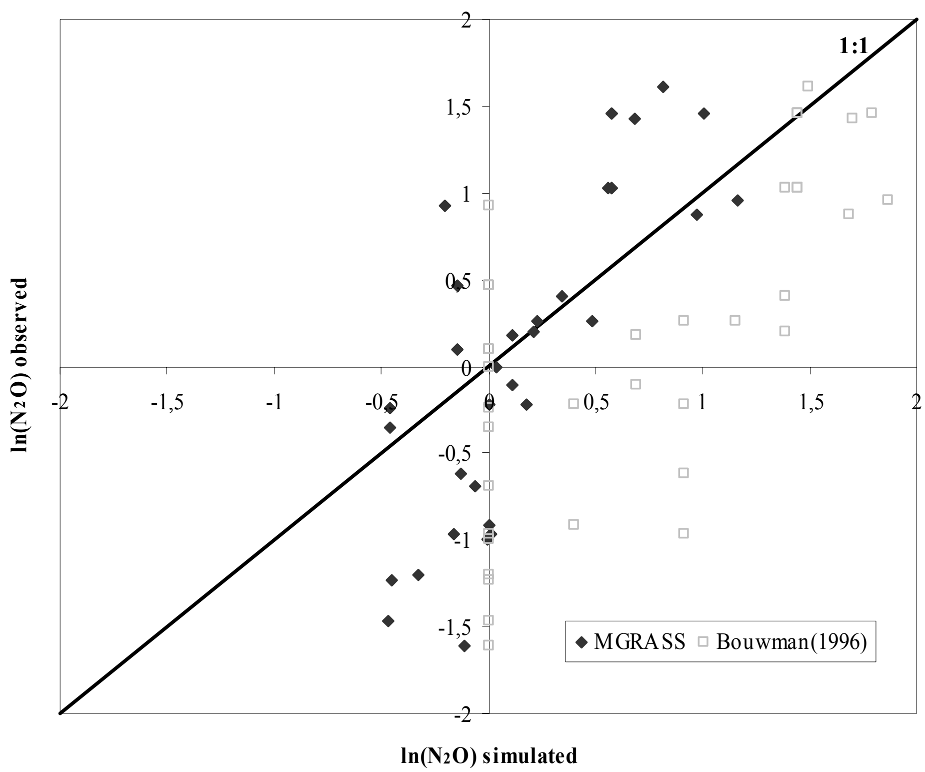


Figure 3.11 *MGRASS validation and comparison with BOUWMAN (1996) model.*

3.4.3 Spatial and temporal projection of the emissions

3.4.3.1 Data

The data used as input for the model are agricultural characteristics and climate. At the municipal level, the National Institute for Statistics (NIS) provided the following data: number of animals per type, crop type and surface, grassland surface, and animal housing (only 1996 is used in this study). Observed climate data for the 1990 to 2000 period were extracted (centroïd method) from the CRU TS 1.2 data-set MITCHELL ET AL. (2005) comprises 1200 monthly grids of observed climate, for the period 1901-2000, and covering the European land surface at a 10 minute spatial resolution. Future climate projections were used for the 2001 – 2050 period. The HadCM3 model (GORDON ET AL., 2000) produced the climate projections according to the IPCC - SRES A1F1 scenario (Nakicenovic et al., 2000).

3.4.3.2 Environmental regulations 1990 – 1996 – 2000 – 2004

The EU member states adopted the Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources the so-called" Nitrate Directive" in 1992. The objective of the directive is to reduce nitrate concentration in drinking water below 50 mg l^{-1}. The directive requires that member states designate vulnerable zones and implement action programmes and prescribe measures for reducing pollution by nitrates from agriculture in these areas.
In Belgium, the application of the directive is a regional responsibility. In Flanders, the directive was first transposed in 1991 with the "Mestdecreet" which was implemented as the "MestActiePlan 1" in 1996, subsequently revised as the "MestActiePlan 2" in 1999 and the "MestActiePlan 2bis" in 2000 and the "MestActiePlan 2ter" in 2003. The evolution of the fertilisation standards is summarised in Table 3.8.

In Wallonia, a first regulation was enacted to protect the quality of surface water ("Arrêté du Gouvernement wallon relatif à la protection des eaux contre la pollution par les nitrates à partir de sources agricoles (M.B. at 28/06/1994, p. 17372, abrogated (10/10/2002)). This regulation advised to limit the grassland fertilisation rate at 200kgNha^{-1} and that a maximum should be 350kgN ha^{-1}. In 2002, the Walloon Government transposed the Nitrate Directive in the Walloon law and initiated the "Sustainable Management Programme of Nitrogen in agriculture" (MB. 29/11/2002). The limits of the nitrogen fertilisation were more clearly established. Outside the vulnerable zones, only 120kgN of manure can be spread on croplands and not more than 210kgN of manure can be spread on grasslands.

Table 3.8 *Summary of the Flemish regulations concerning the application of nitrogen based fertilisers (Vlaamse Landmaatschappij, 2004).*

	Fertilisation limits outside vulnerable zone
Mestdecreet (23/01/1991)	Nihil
MestActiePlan 1 (01/01/1996)	Grasslands: 450 $kgNha^{-1}$ (maximum organic: 250 $kgNha^{-1}$; maximum mineral: 250 $kgNha^{-1}$). Maize: 325$kgNha^{-1}$ (maximum mineral: 200 $kgNha^{-1}$). Crops with low nitrogen needs: 170$kgNha^{-1}$ (maximum mineral: 125 $kgNha^{-1}$). Other crops: 325$kgNha^{-1}$ (maximum mineral: 225 $kgNha^{-1}$).
MestActiePlan 2 (11/05/1999) period: 01/01/1999 to 31/12/1999.	Grasslands: 444 $kgNha^{-1}$ (maximum organic: 444 $kgNha^{-1}$; maximum mineral: 250 $kgNha^{-1}$). Maize: 319$kgNha^{-1}$ (maximum organic: 319 $kgNha^{-1}$; maximum mineral: 194 $kgNha^{-1}$). Crops with low nitrogen needs: 164$kgNha^{-1}$ (maximum organic: 164 $kgNha^{-1}$; maximum mineral: 119 $kgNha^{-1}$). Other crops: 319$kgNha^{-1}$ (maximum organic: 319 $kgNha^{-1}$; maximum mineral: 219 $kgNha^{-1}$).
MestActiePlan 2bis (01/01/2000), period: 01/01/2000 to 31/12/2000.	Grasslands: 450 $kgNha^{-1}$ (maximum organic: 400 $kgNha^{-1}$; maximum mineral: 300 $kgNha^{-1}$) Maize: 300$kgNha^{-1}$ (maximum organic: 300 $kgNha^{-1}$; maximum mineral: 175 $kgNha^{-1}$). Crops with low nitrogen needs: 150$kgNha^{-1}$ (maximum organic: 150 $kgNha^{-1}$; maximum mineral: 100 $kgNha^{-1}$). Other crops: 300$kgNha^{-1}$ (maximum organic: 300 $kgNha^{-1}$; maximum mineral: 200 $kgNha^{-1}$).
MestActiePlan 2bis (01/01/2000), period: 01/01/2001 to 31/12/2002.	Grasslands: 450 $kgNha^{-1}$ (maximum organic: 325 $kgNha^{-1}$; maximum mineral: 350 $kgNha^{-1}$) Maize: 275$kgNha^{-1}$ (maximum organic: 275 $kgNha^{-1}$; maximum mineral: 150 $kgNha^{-1}$). Crops with low nitrogen needs: 125$kgNha^{-1}$ (maximum organic: 125 $kgNha^{-1}$; maximum mineral: 100 $kgNha^{-1}$). Other crops: 275$kgNha^{-1}$ (maximum organic: 225 $kgNha^{-1}$; maximum mineral: 200 $kgNha^{-1}$).
MestActiePlan 2ter (03/12/2003), period: 01/01/2003 to ….	Grasslands: 500 $kgNha^{-1}$ (maximum organic: 250 $kgNha^{-1}$; maximum mineral: 350 $kgNha^{-1}$). Maize: 275$kgNha^{-1}$ (maximum organic: 250 $kgNha^{-1}$; maximum mineral: 150 $kgNha^{-1}$). Crops with low nitrogen needs: 125$kgNha^{-1}$ (maximum organic: 125 $kgNha^{-1}$; maximum mineral: 100 $kgNha^{-1}$). Other crops: 275$kgNha^{-1}$ (maximum organic: 200 $kgNha^{-1}$; maximum mineral: 200 $kgNha^{-1}$).

3.4.3.3 Grassland fertilisation: model

The model is based on simplified rules. The animal waste production is estimated based on the livestock population and type (NIS data, VAN MOORTEL ET AL., 2000) and the amount of nitrogen excreted annually per type of animal. All the animal waste produced in a "municipality" is spread within the municipality limits. The cattle-demand for grass or hay has to be met by the grassland within the municipality. The daily mean consumption of hay per livestock unit (LU) is set to 6 kg. The hay production (A) of a given municipality needed to feed the local livestock (kg N yr^{-1}):

$$A = \sum_{i=1}^{n} I F_i P_i y \varepsilon \qquad 3.11$$

Where: F_i is the LU fraction of the livestock category "i"; P_i is the population of the livestock category "i" present in the municipality; y is the number of days during a year (365.6 days); ε is the conversion factor of hay dry matter into nitrogen (2.5 kg N per 100kg hay dry matter).

The animal waste produced in the municipality is spread first on the croplands according to crop type, N-demand and management type. The remainder manure is spread on the grasslands. To obtain the fertiliser-manure equivalent (N_{manure}) a conversion factor is applied on the different waste types (farmyard manure (0.2), slurry (0.45); SOLTNER, 1999). The N input from organic origin of the grasslands is computed as follows:

$$N_{organic} = N_{manure} + \alpha N_{grazing} + S_{past} N_{mineralisation} + S_{past} N_{leguminous} \quad 3.12$$

Where: $N_{grazing}$ is the nitrogen received by the pasture during the grazing period, the α coefficient (0.8) takes into account the losses on the way to the dairy (SOLTNER, 1999); $N_{mineralisation}$ is and average value of the nitrogen input due to the mineralisation of the soil organic matter (90kg N ha^{-1}; LAMBERT ET AL., 2003); $N_{leguminous}$ is the nitrogen added due to the leguminous plants nitrogen fixating activity (60 kg N ha^{-1}; LAMBERT ET AL., 2003); S_{past} is the surface (ha) covered by grassland in the municipality.

Fertilisation standards are then introduced which reduce the organic nitrogen input to the maximum allowed (Table 3.8). The mineral fertiliser input is deduced from the nitrogen balance of the pasture. In the case of a perfect balance, nitrogen input compensates for the nitrogen uptake by grass or hay production. The losses of efficiency of the fertiliser are taken into account by introducing a coefficient of nitrogen use efficiency (NUE = 0.7; SOLTNER, 1999). Mineral fertilisation rate is thus computed following equation 3.13:

$$N_{fertiliser} = \frac{(A - N_{organic})}{NUE} \quad 3.13$$

The N_2O emission data used to build the N_2O emission models did not take into account the nitrogen input due to the organic matter mineralisation nor the input due to N-fixation by the leguminous plants. In this exercise, the N_2O emissions are computed using the fertilisation rates balancing the N-exportations with mineral and organic fertilisers.

3.4.3.4 Results

Cropland

The amounts discussed below need to be taken with caution since the validation of MCROPS showed that it overestimates the emissions by a factor of c. 2 and that the uncertainty on the prediction is large (RMSE = 0.67 kg N_2O-N ha^{-1} yr^{-1}). Nitrous oxide emissions from cropland soils are highly sensitive to climate conditions and to the climate inter-annual variability (ROELANDT ET AL., 2005). For the year 1990, N_2O emissions from cropland soils are estimated to be equal to 3.5 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.3 kg N_2O-N ha^{-1} yr^{-1}) for both regions of Belgium (Fig. 3.12). A dry summer with a total precipitation of 119 mm is responsible for this low emission rate. In 1996, croplands N_2O emissions increased up to 4.5 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.4 kg N_2O-N ha^{-1} yr^{-1}). Summer 1996 experienced frequent thunderstorms and heavy rainfall that increased the precipitation up to 311 mm. The simulated N_2O emissions in

2000 show a strong increase compared to the preceding years. Spring 2000 was warmer than the other years and summer was moderately wet with a total precipitation of 206 mm. The combination of these two factors is responsible for the strong increase of the emissions up to 6.0 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.25 kg N_2O-N ha^{-1} yr^{-1}).

The emission rate simulated for 2004 shows a strong decrease. The emissions drop down to 3.3 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.88 kg N_2O-N ha^{-1} yr^{-1}) in Flanders and 3.2 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.43 kg N_2O-N ha^{-1} yr^{-1}) in the Walloon Region. The simulated spring temperature is normal (12.4°C) but the dry summer conditions (132.3 mm) strongly reduced the N_2O emissions to a level equivalent to the one reached in 1990.

Grassland

The average fertilisation rates are estimated based on the statistics for livestock (number, type and housing), land use between 1990 and 2004 and the regulation concerning the maximum nitrogen fertilisation rates allowed in both Flanders and Wallonia. From these estimations it results that the average fertilisation rate did not vary significantly during the last decade. The average fertilisation rate is 175 kg N ha^{-1} yr^{-1} for Wallonia and about 300 kg N ha^{-1} yr^{-1} for Flanders.

The nitrogen fertilisation being constant, the importance of the N_2O emissions from grassland depends on climate and more specifically on winter temperatures. The simulation results show that the emissions were high in 1990 due to a "warm" winter characterized by a mean temperature of 7.5°C (Figure 3.12). The grasslands emitted 3.3 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.3 kg N_2O-N ha^{-1} yr^{-1}) in Flanders and 2.1 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.61 kg N_2O-N ha^{-1} yr^{-1}) in Wallonia.

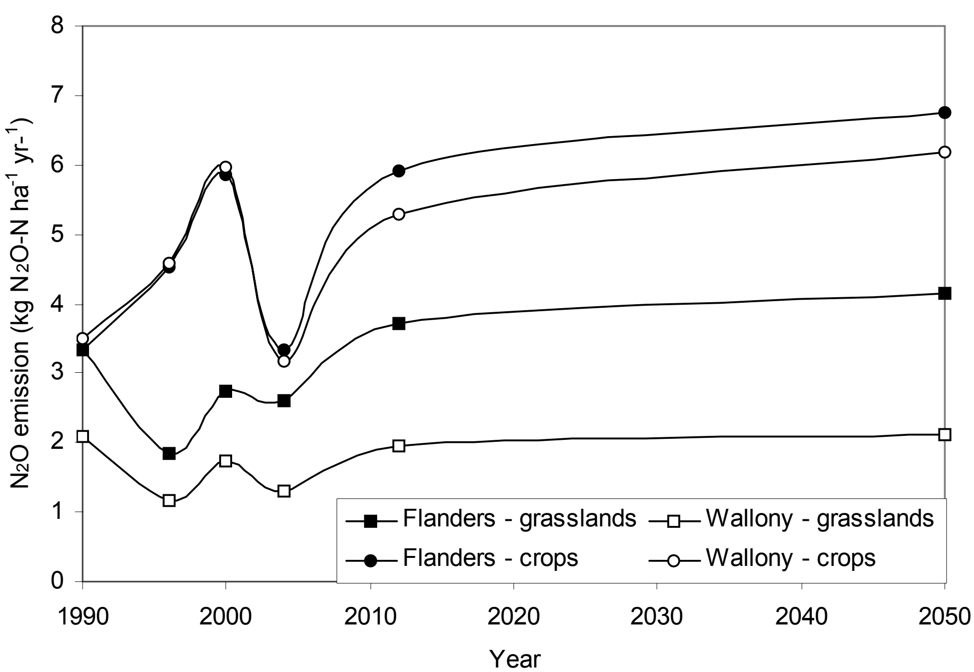


Figure 3.12 *Evolution of the mean N_2O emission rate (kg N2O-N ha^{-1} yr^{-1}) from agricultural soils on a regional basis*

In contrast, the winter of 1996 was much colder with an average temperature of 3°C. This justifies the low emission rate for grasslands in both regions with 1.8 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.15 kg N_2O-N ha^{-1} yr^{-1}) in Flanders and 1.2 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.34 kg N_2O-N ha^{-1} yr^{-1}) in Wallonia. The year 2000 experienced less extreme winter temperatures with an average of 6.3°C. Grasslands emissions followed the same pattern as the winter temperature with a regional difference due to nitrogen fertilisation rate. The mean N_2O emission for Flanders is equal to 2.7 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.24 kg N_2O-N ha^{-1} yr^{-1}). The mean emission in Wallonia is equal to 1.7 kg N_2O-N ha^{-1} yr^{-1} (SD = 0.49 kg N_2O-N ha^{-1} yr^{-1}).

N emissions for landscape units

Average N emissions are calculated for the average climate and the average fertilization rate per LSU and then applying the two models. Climate variables are derived for the association centroids (see 3.4.3.1). The NIS data on fertilization are available per municipality. These values are dis- and then re-aggregated to the level of the LSU according to the principles described earlier for attributing per-municipality SOC values to the LSU (under 3.2.3.2 and in Figure 3.2b). For future years, the amount of fertilizer is considered constant and equal to the 2004 quantities. The average N-N2O flux for the period 1990-2000 and 2000-2012 is shown in Figure 3.13. For 1990-2000, these results are based on the average fertilization rate and the average climate over this period. For 2000-2012, the fertilization rate of the year 2004 is used in combination again with the average climate.

3.4.3.5 Discussion

A previous inventory of N_2O emissions of agricultural soils in Belgium used the IPCC default model. VAN MOORTEL ET AL. (2000) estimated a direct N_2O emission rate of 7.56 10^6 kg N_2O-N yr^{-1} for Belgium during the year 1996. This corresponds to an average of 5.6 kg N_2O-N ha^{-1} yr^{-1} for croplands and grasslands. This is in agreement with our results. VAN MOORTEL ET AL. (2000) underlined also the difference between the regional emission rates due to the intensive pig and poultry farming in Flanders. This difference appears only for the emissions from grasslands in this study. A spatial and sectorial disaggregation of N_2O emissions from Belgian agriculture was proposed by BOECKX ET AL. (2001) where N_2O emissions have been differentiated per agro-pedological region and calculated per farm type. Differences appeared between the northern and the southern part of the country. For Flanders, the average N_2O emission rate is equal to 13.6 kg N2O-N ha^{-1} yr^{-1} while for Wallonia agricultural soils emit 6.7 kg N2O-N ha^{-1} yr^{-1}. These emission rates are in the same range as the ones observed in this study although emissions in Flanders are considerably higher than the values of the present study.

The two studies cited above both used the IPCC default methodology. Compared to the present study they did not take into account the influence of land use or climate. The N_2O emission models used in the Metage project are based on a larger data set than the IPCC model and are sensitive to seasonal climate. However, large uncertainties still exist due to the empirical nature of the models, the simplified approach used to simulate the nitrogen fertilization rate on the grasslands and the coarse resolution of the climate data. These uncertainties are not quantified yet but some consistent conclusions can be drawn. The N_2O emissions from agricultural soils differ between croplands and grasslands in amplitude and in their response to climate. Emission rates from cropland depend on crop type, management, fertilization rate, soil characteristics and on climate. Such a large amount of cross-scale interacting factors are difficult to manage with a single model. This study puts forward the link between the emission rate and the seasonal climate and shows that a warm moist world will increase the N_2O emissions from cropland. Grassland emission rates depend through an exponential relationship on winter temperature and on nitrogen fertilization rate. The latter factor is the only one on which human activity can act directly.

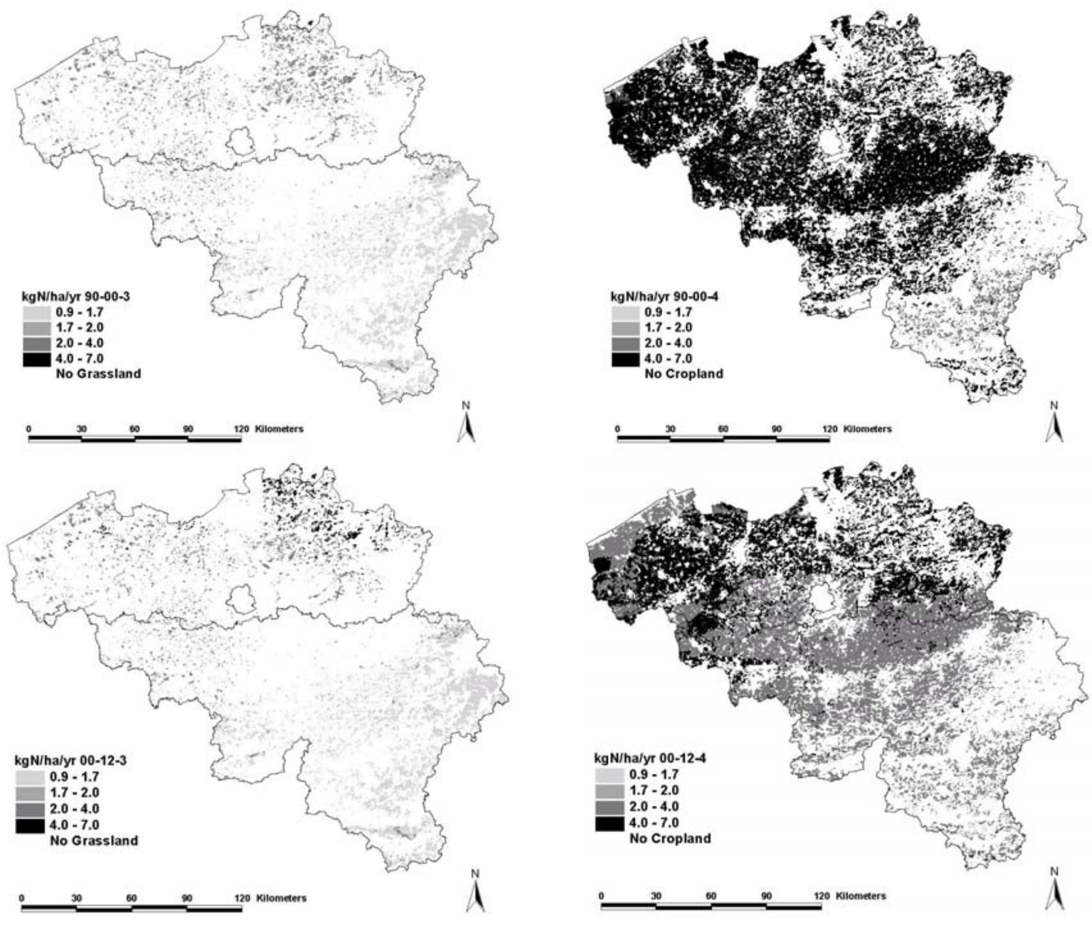


Figure 3.13 *Spatial distribution of N_2O-N fluxes for the periods 1990-2000 (upper graphs) and 2000-2012 (lower graphs) for grassland (left hand graphs) and cropland (right hand graphs)*

3.5 SYNTHESIS OF GREENHOUSE GAS FLUXES EXPRESSED IN GLOBAL WARMING POTENTIAL

Since all the fluxes (carbon from biomass and soils and N_2O from soils) are estimated per LSU, it is possible to combine them into a greenhouse gas synthesis, expressed in CO_2 equivalents. CO_2 equivalents reflect the global warming potential of all greenhouse gases compared to the potential of CO_2. The generally accepted global warming potential of N_2O is 310, meaning that 1 kg N_2O is 310 times as powerful as 1 kg CO_2 as a greenhouse gas. Consequently, 1 kg N_2O is equal to 310 kg CO_2-eq (IPCC, 1997).

Table 3.9 *Mean annual GHG fluxes in CO_2eq per land use and source for the period 1990 – 2000*

	Flux per area (t $CO_2 ha^{-1} y^{-1}$)			Fluxes for Belgium (kt CO_2 y^{-1})			
	SOC	BOC	N_2O	SOC	BOC	N_2O	sum
Broadleaf forest	2.76[1]	5.09	-	552.0	1018.2	-	1570
Coniferous forest	1.72	5.78	-	242.3	812.6	-	1054.9
Mixed forest	2.70	4.70	-	761.2	1323.5	-	2084.7
Grassland	-1.85	-	-0.94	-681.4	-	-595.7	-1277.1
Cropland	-0.83	-	-2.27	-1174.4	-	-1596.2	-2770.7
						total	*662.0*

[1] *Positive fluxes refer to sequestration of GHGs (increase in terrestrial C stocks), while negative fluxes represent GHG fluxes towards the atmosphere*

Figure 3.13 shows the total GHG flux between 1990 and 2000, expressed in t CO_2-eq ha^{-1}. A positive value refers to an increase in SOC stock, and thus a sequestration of greenhouse gases. The map shows important GHG losses in Flanders. Table 3.9 shows that the losses are mainly due to a decrease in SOC stocks in grassland and cropland as well as the emission of N_2O from these ecosystems. Overall a limited net sink of 0.66 Mt CO_2 y^{-1} was registered from 1990 to 2000 (Table 3.9). As was shown in the previous section N_2O emissions under grassland are generally lower than under cropland (Figure 3.13). Forest biomass and soils under forest sequester C, leading to a net sink under forest. Therefore, clear regional differences in GHG emissions occur with CO_2 losses occurring in the northern and middle part of the country dominated by intensive agriculture and C sequestration restricted to the areas dominated by forest and grassland i.e. the Ardennes and Jura, but also in the Kempen, Famenne, High Ardennes and Fagne (Figure 3.13).

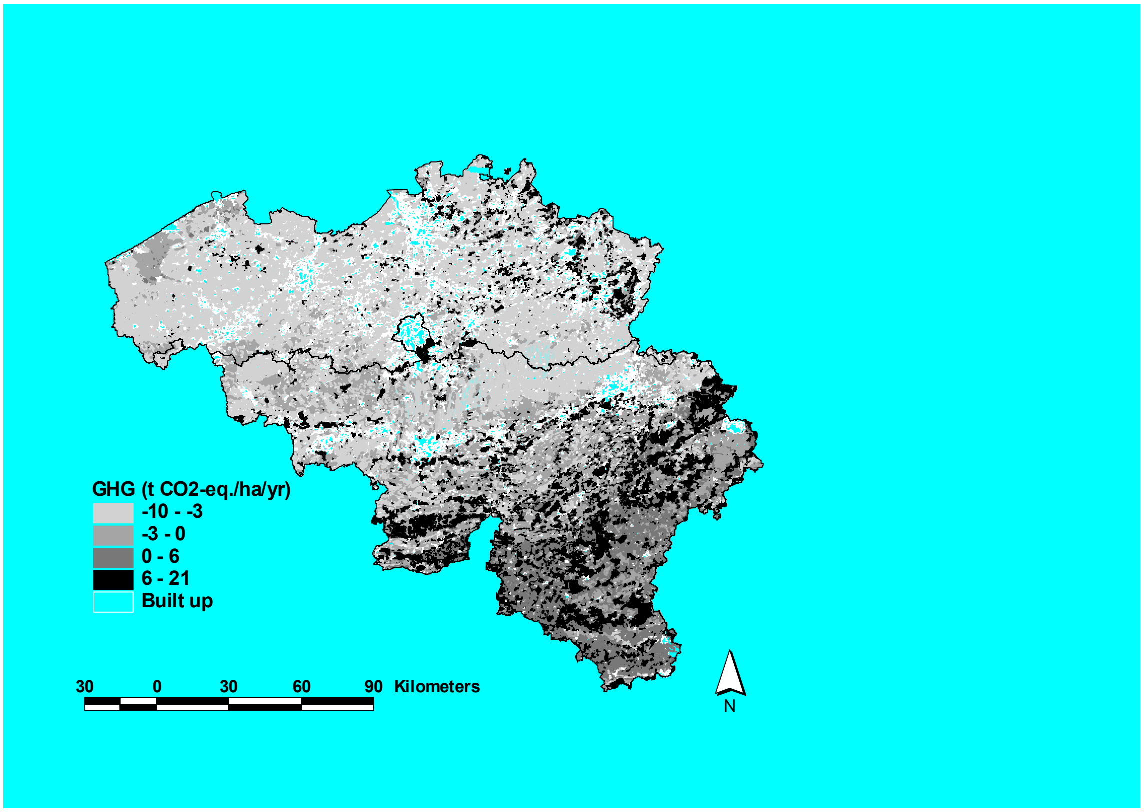


Figure 3.14 *Total GHG flux between 1990 and 2000 (t CO_2-eq $ha^{-1}yr^{-1}$).*

4. GREENHOUSE GAS EMISSIONS UNDER GLOBAL CHANGE SCENARIOS

4.1 Introduction

One of the objectives of the METAGE project was to determine the fluxes of greenhouse gases from terrestrial ecosystems under global change scenarios. So far, the LSUs formed the basic units for which greenhouse gas fluxes of the past were calculated. In our approach changes in the size or distribution of these LSUs were not considered, although it was argued in section 2.1 that land use change will have occurred from 1960 until 2000 and therefore will have had a limited impact on the size and the distribution of LSUs and hence on GHG fluxes. In this chapter, we will demonstrate an approach, which allows the effect of global change on GHG fluxes from terrestrial ecosystems to be taken into account. This requires the estimation of GHG fluxes from landscape units that are affected simultaneously by climate and land use change. Since the fluxes of CO_2 from soils and biomass were estimated by the stock change approach, this implies that the full land use and climate history for each unit should be used in the carbon stock modelling. As a first approximation, we have opted for a simpler approach proposed by KIRSCHBAUM ET AL. (2001) This approach takes land use change into account by calculating the difference in the C stocks under the original and the new land use. Evidently, both C stocks have to be calculated considering the evolution of the climate over the modelling period. Estimates of N_2O emissions are based on direct flux measurements. The empirical N_2O emission models (section 3.4.2) were developed separately for cropland and grassland and take seasonal climate into account. These models can therefore be applied to estimate annual emissions under conditions of different climate and land use.

Climate and land use scenarios are based on the story lines developed by the IPCC in their Special Report on Emission Scenarios (NAKICENOVIC ET AL., 2000). The four SRES marker scenarios differ according their socio-economic development pathways, which reflect alternatives in the importance of economic or environmental considerations at either the regional or global scales. The EU project ATEAM (http://www.pik-potsdam.de/ateam/) has produced monthly precipitation and temperature data for the period 1900 to 2100 in 10' grid cells covering the EU15 (MITCHELL ET AL., 2004). Until 2000 these data are derived from interpolation of measured climate data, and from 2000 until 2100, predictions of the HADCM3 global circulation model under the different scenarios are available (Nakicenovic et al., 2000). Land use maps with the same 10' grid cells have been produced for the years 2020, 2050 and 2080 according to the methodology of EWERT ET AL. (2005) and ROUNSEVELL ET AL. (2005). This approach uses a simple supply/demand model (with global economic inputs derived from the IMAGE 2.2 model) of agricultural area quantities at the European scale and the disaggregation of these quantities using scenario-specific, spatial allocation rules. Here we use two scenarios to evaluate the effects of climate change on SOC stocks and N_2O fluxes: the A1FI scenario, which is a globally and economically oriented scenario also referred to as 'world markets-fossil fuel intensive', and the B1 scenario which has the smallest climate changes and is referred to as 'global sustainability'. Only the A1FI scenario was used to calculate a spatially distributed land use map for 2050 under this scenario. This scenario assumes that agricultural production becomes concentrated in optimal locations with widespread abandonment in marginal agricultural areas. The areas of non-optimal production were considered to be the Less Favoured Areas receiving subsidies within the framework of the CAP. The remainder of the EU was marked as productive areas.

4.2 AN EXAMPLE OF A SPATIAL EXPLICIT GLOBAL CHANGE SCENARIO FOR BELGIUM

The spatially distributed land use map of 2050 for the A1FI scenario produced at the European level (10' or 18*18 km grid cells at the latitude of Belgium) is too coarse to be compared to the Corine land use map of 1990 with a 250*250 m resolution. Therefore, this land use map was downscaled to the 250 m resolution using the methodology of Dendoncker et al. (subm.). First, logistic relationships were established between the current land use and a range of environmental, spatial and economic variables. Then, a probability surface for the occurrence of each land use type was established. Finally, according to the percentage of each land use in the 18 * 18 km grid cells, the 250 m grid cells were attributed to a land use class starting with the ones with the highest probability and continuing until the target land use distribution was reached. A hierarchy in the attribution of land use to the grid cells was maintained in the following order: urban > cropland > grassland > biofuels > forest. The result of the downscaling is a land use map with the following distribution of land use types: (i) 12,969 km² of crops, covering 42% of the national territory, (ii) 2,295 km² of grassland (8%), (iii) 6,245 km² of forest (20%), (iv) 677 km² of crops for liquid biofuels (2%), (v) 961 km² of crops for solid biofuels (3%), (vi) 6,711km² of built up land and surplus land (22%), (vii) 729 km² of other types of land use (2%) which have not changed from 1990 to 2050, such as estuaries and moorland. The average area of a LSU-polygon is 41 ha; all polygons of an LSU sum up to 10,435 ha on average. Note that in this chapter the LSU refers to the new land use map intersected by soil associations, in contrast to the previous chapters where the term LSU referred to an intersection of the Corine land cover dataset and the soil association map.

4.3 CALIBRATION OF A CARBON DYNAMIC MODEL USING DATA FOR THE LANDSCAPE UNITS

4.3.1 Description of the RothC-26.3 model

The RothC-26.3 model was developed to simulate the turnover of organic carbon in non-water logged agricultural soils on a time scale of years to centuries. RothC-26.3 has been tested against long-term experiments in a range of soils and climate conditions in Western Europe (COLEMAN ET AL., 1997; SMITH ET AL., 1997). This model was selected for the regional scale modelling exercise in Belgium, because of its simplicity and the availability of data at the LSU level to run the model. The RothC-26.3 model requires three types of data:

- Monthly weather data (rainfall (mm), evapotranspiration (mm), mean air temperature (°C);
- Soil data (clay content (%), inert organic carbon (IOM), initial soil organic carbon (SOC) stock (t C ha^{-1}), depth of the soil layer considered (cm);

- Land use and management data (soil cover, monthly input of plant residues (t C ha^{-1}), monthly input of farmyard manure (FYM) (t C ha^{-1}), residue quality factor (DPM/RPM ratio).

The RothC-26.3 model splits SOC into four active compartments and a small amount of inert organic matter. The active compartments are Decomposable Plant Material (DPM), Resistant Plant Material (RPM), Microbial Biomass (BIO) and Humified Organic Matter (HUM). The IOM compartment is resistant to decomposition. Incoming plant carbon is split between DPM and RPM, depending on the DPM/RPM ratio of the particular incoming material. For most agricultural crops and improved grassland a DPM/RPM ratio of 1.44 is used (COLEMAN ET AL., 1997). These pools in turn decompose exponentially to form CO_2 and BIO+HUM. The clay content determines the proportion between the CO_2 and BIO+HUM produced. The decomposition rate is modified as a function of temperature, soil moisture deficit and the presence of a plant cover. The internal structure and internal parameters of the model were unaltered. Further details of the RothC model can be obtained from the GCTE SOMNET website[1].

4.3.2 Calibration of plant input

RothC-26.3 was used to simulate the evolution of SOC stock for long-term experiments in order to obtain correct carbon plant input for arable soils in Belgium,. Long-term experiments representing the dominant crop rotation in Belgium have been conducted by the Centre de Recherche Agronomique de Gembloux since 1959 (FRANKINET ET AL., 1993). Observed SOC stocks from the control experiment (export of crop residues and no addition of animal manure) were compared with model simulations from 1959 until 1994. Since the long term experiments were used to calibrate the model for later application on the LSUs, it was preferred to use the climate data which is also available for the LSUs: monthly precipitation (mm) and average monthly mean air temperature (°C) from the spatial climate database (NEW ET AL., 2002), potential evapotranspiration using the empirical formula of Thornthwaite (SHAW, 1994). IOM was estimated from the initial SOC stock using the equation proposed by FALLOON ET AL. (1998). The depth of the modelled soil layer corresponds to the plough depth of 22 cm. RothC was run to equilibrium using average monthly climate data for the 1900-1959 period and iteratively fitting carbon inputs to match the initial SOC stock and thus the distribution in four active compartments; DPM, RPM, BIO, HUM (COLEMAN ET AL., 1996). The model was then run for the period 1959-1994 with monthly climate data for each year. The plant C input to the soil was estimated by optimising the total SOC stock predicted by the model to the measured data of the control experiment. The performance of the model was evaluated on the root mean square error (RMSE) used by SMITH ET AL. (1997). The RMSE indicates the mean difference between observed and predicted SOC stocks expressed as a percentage of the mean. The best fit between modelling and experimental SOC stock (RMSE: 0.8 %) was obtained by using a carbon plant input of 2.3 t C ha^{-1} y^{-1}.

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http://www.rothamsted.bbsrc.ac.uk/aen/somnet/.[1]

4.3.3 Parameterisation of the RothC model for the landscape units

After calibration of the plant input, the RothC model was applied for each LSU under cropland and grassland starting in 1960. Climate input variables; monthly rainfall (mm) and average monthly mean air temperature (°C) were extracted from an interpolated climate database (MITCHELL ET AL., 2004) for 'centroids' of the polygons forming a soil association. The parameterisation of the model was similar to the one used for the Gembloux experiments, apart from the soil depth that was increased to 30 cm, the most frequently used soil depth in SOC change studies. Hence, the plant C input obtained during the calibration had to be multiplied by a factor 30/22 in order to reflect the increase in soil thickness. A single plant C input was used for all arable soils in Belgium. Because of the lack of long term experiments for grasslands in Belgium, carbon plant input for grassland was retrieved from the Rothamstead experiment and represents: 3.9 C t/ha for 30 cm soil depth (COLEMAN ET AL., 1997).

In a first attempt, only plant residues were considered as carbon input for the RothC model, then also carbon input from manure was used. The latter were calculated using a methodology based on livestock numbers, types, production of excrements per unit of livestock and type of manure produced (FYM or slurry) in each agricultural region (Table 4.1). Details of these calculations can be found in DENDONCKER ET AL. (2004). While modelling the SOC dynamics for the cropland and grassland LSUs using carbon input from both plant residues and manure, it was observed that RothC predicted unrealistically high SOC stocks. As a compromise, the difference between the manure and slurry produced and the baseline of 1960 were used in the model. An average manure input was thus determined for each decade. This means that manure and slurry were used as input for RothC in the Polders, Sand, Campine and Sand loam region and no extra input was used for the other regions, since manure production in these regions remained more or less stable or declined from 1960 to 2000 (Table 4.1).

Table 4.1 *Annual manure and slurry production, converted to carbon content, in each agricultural region.*

	FYM and slurry (t C ha^{-1} yr^{-1})				
Agricultural region	**1958**	**1970**	**1980**	**1990**	**2002**
Dunes-Polders	1.38	1.82	1.77	2.29	2.38
Sand	1.78	2.80	3.48	4.37	3.85
Campine	1.51	2.78	3.41	4.62	4.07
Sand-loam	1.33	1.96	2.26	2.77	2.53
Loam	1.05	1.19	0.99	1.05	1.07
Condroz	1.07	1.15	0.92	0.96	0.99
Herbagère Liégeoise	2.44	2.39	1.99	1.80	1.67
Campine Hennuyère	1.25	1.48	1.15	1.13	1.06
High Ardenne	1.52	1.81	1.71	1.63	1.48
Herbagère Famenne	1.62	1.67	1.37	1.37	1.40
Famenne	1.15	1.29	1.14	1.28	1.39
Ardenne	1.25	1.47	1.36	1.72	1.79
Jura	0.99	1.15	1.36	1.28	1.31

4.3.4 Comparison of modelled and observed SOC stock (period 1960-2000)

For each LSU, SOC stocks predicted by the RothC model were compared to the SOC data for 1990 and 2000 provided by LETTENS ET AL. (2005). When only carbon from crop residues was used in RothC, it becomes clear that the model underestimated SOC stocks. This is particularly true for grasslands (Figure 4.1a +c). Using both growing plants and animal excrements as carbon input to RothC, outlined in the previous section, a better fit between observed and modelled data was obtained (Figure 4.1b+d). Only the latter approach will be used from now on. When the modelled SOC stocks are plotted against the observed values a good agreement can be demonstrated (Figure 4.2). The RMSE calculated separately for both years is quite low and ranges from 1.45 % for 1990 to 1.62 % for 2000. This implies that the SOC stocks of each LSU can be modelled with an accuracy of 1.02 t C ha^{-1} in 1990 and 1.10 t C ha^{-1} in 2000.

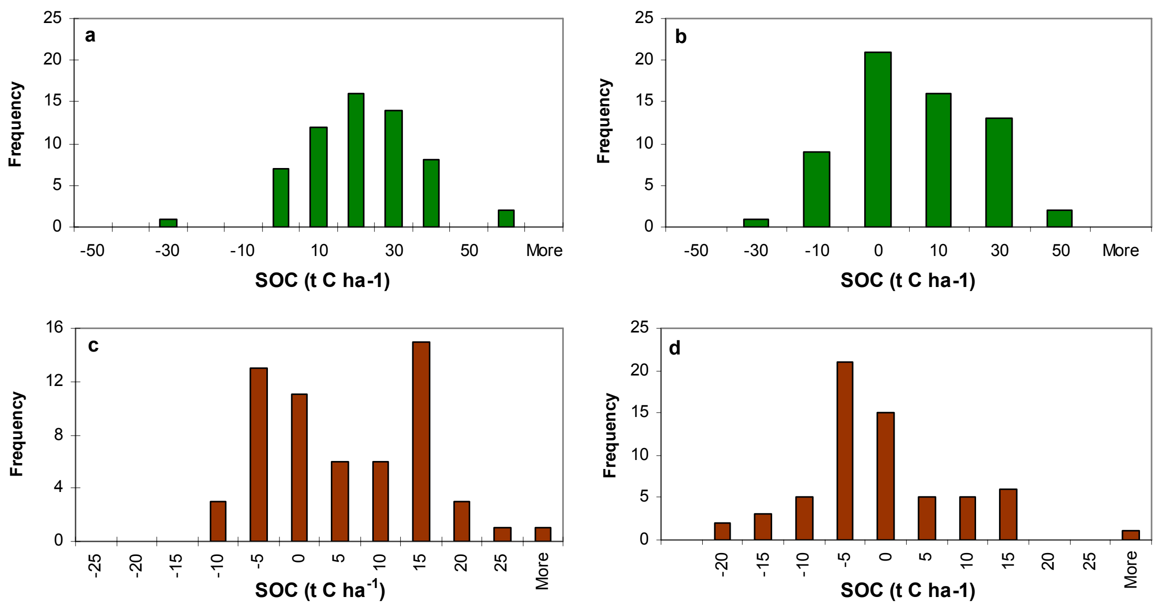

Figure 4.1 *Frequency distribution of differences between observed and modelled SOC stock in 2000 for each soil association. Different approaches to carbon input were tested:*
a) grasslands, C input exclusively from plants;
b) grasslands, C input from plants and manure;
c) croplands, C input exclusively from plants;
d) croplands, C input from plants and manure.

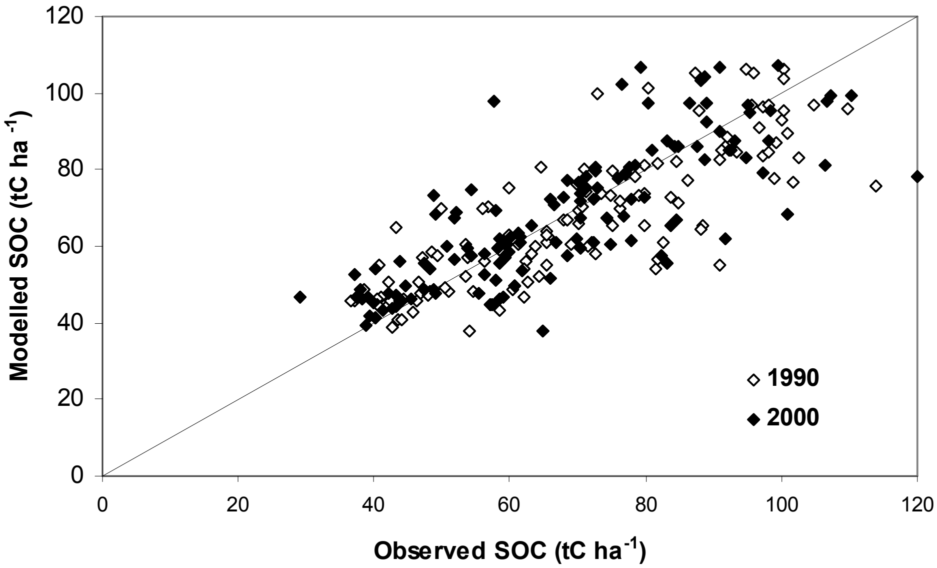

Figure 4.2 *Modelled against observed SOC stocks in 1990 and 2000 for the LSUs under cropland and grassland (t C ha^{-1}). The line of perfect agreement is indicated.*

The evolution of the average SOC stock in all grassland and cropland LSU was calculated using a weighting by the surface of each LSU (Figure 4.3). The observed SOC stocks in 1990 and 2000 with their 95% confidence limits are given as well. Overall, the confidence limits are narrow due to the large amount of data (see section 3.2.4). The RothC model predicts the general increase in SOC stock for grassland relatively well, whereas the model predicts a slight increase in SOC for croplands against an observed decrease. However, the deviation between modelled and observed data is restricted to 6 ton C ha^{-1}, which corresponds to a rate of 0.15 t C ha^{-1} y^{-1}. The decrease in SOC stocks observed between 1990 and 2000 (Figure 3.3) was not well represented by the RothC model. This decrease occurs mainly in northern and central Belgium and probably corresponds to a slight decrease in the production of FYM and slurry in the Sand region and the Campine (Table 4.1).

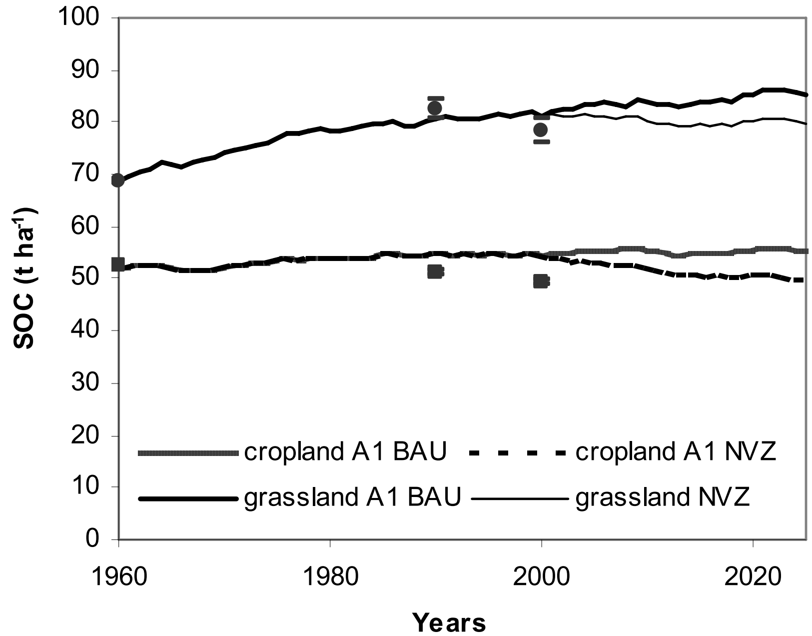

Figure 4.3 *Mean SOC stock for cropland and grassland obtained from RothC model runs on all LSUs. From 2000 onwards two scenarios of climate and land management (A1 BAU, A1 NVZ) were used. The mean SOC stocks for grassland (diamonds) and cropland (squares) are given with their confidence limits (Lettens et al., 2005).*

4.4 GREENHOUSE GAS EMISSIONS UNDER SRES SCENARIOS FOR THE PERIOD 2000-2050

After parameterisation of the GHG emission models for the LSUs (SOC in agricultural land, SOC and BOC in forests) and a quantitative assessment of the fit between predicted and modelled values, we will illustrate how these models can be used to predict greenhouse gas emissions under global change scenarios. Finally, the empirical N_2O emission models, developed in section 3.4, will be used to estimate N_2O fluxes in 2012 and 2050. Although the selection of only two years does not allow an assessment of the inter annual variability in N_2O fluxes, a synthesis of both CO_2 and N_2O fluxes will be attempted.

For this demonstration the monthly climate under the A1FI and B1 scenarios were used as well as the land use in 2050 under the A1FI scenario. Two scenarios of land management were combined with the climate scenarios:

- A business as usual approach (BAU) refers to a sustained input of farmyard manure and slurry at the levels of 2002 (Table 4.1).
- An alternative scenario reflects the input that soils received in the 1960's. For the Walloon region this alternative scenario does not differ from the BAU scenario since FYM and slurry production have not increased over time (Table 4.1). For Flanders,

FYM production in 1960 ranged from 1.33 to 1.78 t C ha^{-1}. This corresponds to an organic nitrogen input of 120 to 160 kg N ha^{-1}, assuming equal amounts of slurry and FYM with CN ratios of 14 for FYM and 8 for slurry. This manure input corresponds with the limits of organic N for Nitrate Vulnerable Zones (NVZ) in Flanders (VLM, 2004), and hence the scenario is referred to as 'NVZ'.

Since the scenarios induce a range of changes to the main driving forces, their effects on greenhouse gas emissions will be discussed in steps. First the effects of climate and land management change on SOC stocks will be discussed and later the combined effect of climate and land use change on total GHG emissions from terrestrial ecosystems expressed in CO_2 equivalents will be addressed.

4.4.1 Effect of climate change and land management on SOC and BOC stocks

Cropland and grassland

The SOC stocks in cropland will remain more or less constant under the BAU scenario, whereas the SOC stocks of grasslands will continue to increase (Figure 4.3 and Table 4.2). These overall stocks, however, do not allow detecting the trends as a result of differences in land management, which exist between the Flemish and the Walloon Region. Flemish croplands will increase their SOC content within the next 25 years with the strongest increase to be expected towards the beginning of this period (Table 4.2). In contrast, SOC contents in Walloon croplands will decrease. Grasslands show an increase in both regions, with the strongest increase in Flanders. The different evolution is not surprising under a BAU scenario. The input of FYM and slurry in the Flemish region will eventually lead to a higher SOC stock (Table 4.1). Since the input remains constant the increase in SOC content will gradually level off. The SOC stocks in both cropland and grassland will decrease under the NVZ scenario (Fig. 4.3). The evolution in the Walloon region does not differ compared to the BAU scenario and was already discussed above. The NVZ scenario induces important decreases in both cropland and grassland in Flanders. Furthermore, the decreasing trend corresponds to the observed data, and implies that the reduction of manure input started already in the 1990's as can be seen in Table 4.1. However, the observed decrease in the Flemish agricultural regions was not strong enough to result in a decreasing trend in the model output. The B1 scenario showed more or less the same results as the A1FI scenario (not shown). Smith et al. (2005) demonstrated a divergence in average carbon stocks in Europe between RothC model runs with different scenarios. These differences increase until at least 2080, but they are still small in 2025.

Table 4.2 *Regional SOC stocks in the topsoil (0-30 cm) for cropland and grassland. Results of Roth-C model runs starting in 1960 onwards. Future stocks represent the situation under the A1FI-BAU scenario.*

	SOC stocks (Mt C)		
	2000	2012	2050
Flanders all agricultural soils	54.0	55.3	56.4
Flanders grassland	11.6	12.0	12.6
Flanders cropland	42.5	43.3	43.8
Walloon all agricultural soils	52.8	52.6	52.1
Waloon grassland	18.3	18.6	19.0
Walloon cropland	34.5	34.1	33.1

SOC and BOC stocks in forests

The long-term evolution of forest BOC and SOC is subject to large uncertainties. There is a long list of factors, which may influence the forest carbon stock (e.g. forest fires, pests or change in timber prices). Forest carbon models like EFOBEL do not take all these factors into account. We used expert judgment to estimate the C stock in 2050. The main stakeholders were consulted to define an "optimal forest" in 2050. The main hypothesis were:

- The total Belgian forest area will remain constant (in accordance with the actual policies and the protection of natural areas);
- The area occupied by Norway Spruce and pine will be significantly reduced due to climate change and the necessity to optimise the forest specific composition with regard to the edaphic conditions;
- We assume a significant increase of Douglas fir and a decrease of pine and larch (for commercial reasons) during the next 30 years;
- The actual increase of deciduous areas will be lower (for the moment, only a volume corresponding to a half of the annual biological growth is harvested).

Table 4.3 *Forest structure of the Belgian forest in 2050 estimated by expert judgment.*

	Spruce	Douglas fir	Larch	Pine	Other coniferous	Beech	Oak	Mixed deciduous *	Other deciduous	Poplars	Coppice only	Total
Productive areas (ha)												
Wallonia	12 000	35000	4000	10000	15000	60000	60000	60000	60000	10000	15000	**449 000**
Flanders	2248	1016	2408	50144	800	9228	16968	12132	25896	22572	0	**143 412**
Belgium	122248	36016	6408	60144	15800	69228	76968	72132	85896	32572	15000	**592 412**
Solid wood volume ($m^3.10^3$)												
Wallonia	32238	7719	958	2005	3609	14458	14458	14458	14458	2619	0	**106979**
Flanders	604	224	577	10053	192	2224	4089	2923	6240	5912	0	**33038**
Belgium	32842	7943	1535	12058	3801	16681	18546	17381	20698	8531	0	**140017**

The quantitative estimation takes into account the actual age structure and the harvest potential during the next 30 years for each species. The result is shown in the table 4.3. Overall compared to 2000, the solid wood volume decreases with c. 15% in Wallonia and increase with 2.5 % in Flanders (Tables 3.5 and 4.3). Decreases in BOC over the same period are concentrated in the Ardennes and in the Campine, while the forests around Brussels retain a high BOC and some increases occur in the Jura (Figures 3.8 and 4.4).

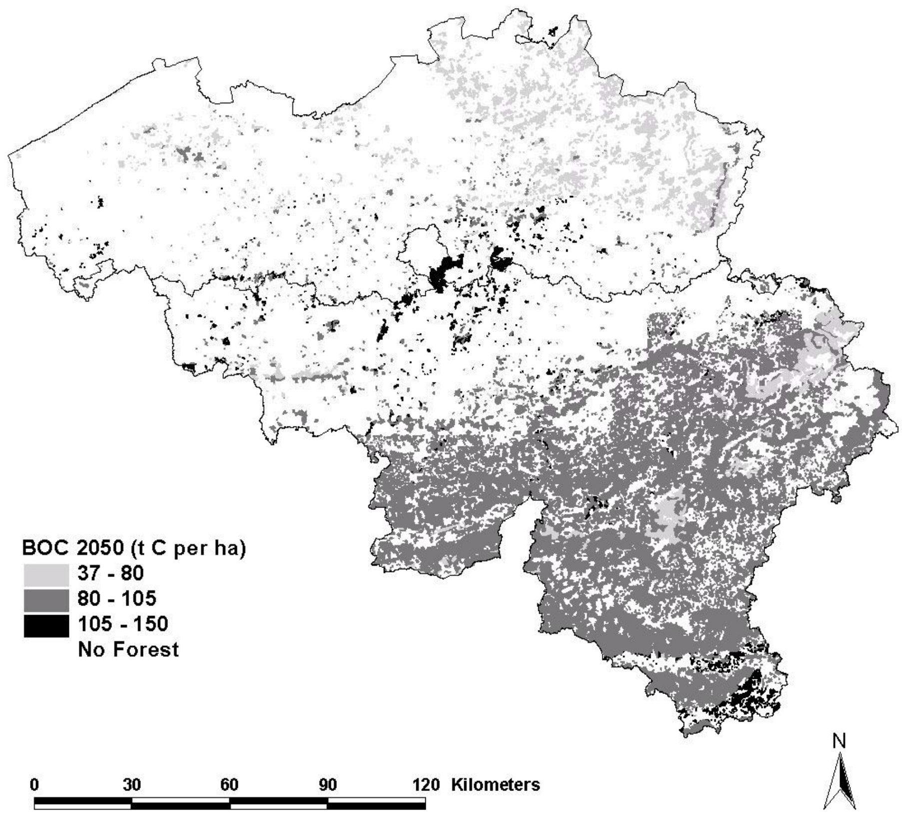

Figure 4.4 *Biomass organic carbon stock (t C ha^{-1}) in LSUs under forest in 2050*

4.4.2 Long term prediction of Greenhouse gas fluxes including the effects of simultaneous land use and climate change (AIFI-BAU scenario)

So far the effects of changes in climate and land management were evaluated for the existing LSUs. In this section we will explore an example of land use change occurring under a future climate and model the effects of this scenario on GHG emissions. Since a continuous evolution of the land use over time is lacking, we have to make some assumptions about the timing of the land use change. We assume that land use change occurs between 2012 and

2050 and that this leaves enough time for a LSU subjected to land use change to reach equilibrium SOC and BOC under the new land use and climate.

The land use map for 2050 under the A1FI scenario was compared to the Corine 1990 land use map. Meanwhile, C stocks and GHG emissions under the A1FI climate scenario were modelled for forest, grassland and cropland under a BAU management scenario (see previous section). The effects of land use change were calculated by comparing the C stocks for the total area under consideration for the areas of different land use types in 1990 and 2050 (Table 4.4). Care was taken to keep the total area under consideration constant in order to avoid 'leakage' of CO_2. This approach does not allow considering the temporal dynamics of land use change, since all types of land use are characterised by C stocks obtained over a long period. Overall, we observe a decrease in cropland and grassland at the expense of forest, biofuels and built up areas and surplus land (Table 4.4).

The future land use distinguishes 'liquid' biofuels from 'solid' biofuels. The former represent annual crops such as rape seed cultivated to produce diesel fuel, whereas the latter represent perennial plantations of short coppice rotations or Miscanthus grass. Surplus land is a category of land, which no longer has an identifiable economic use, and is considered, therefore, to be abandoned. Since no SOC stocks were modelled for biofuel and the increase in urban areas, some assumptions on the dynamics of SOC denities under these new types of land use had to be made. Densities from arable lands were used for liquid biofuels and stocks for grassland were used for solid biofuels and the increased area of built up areas and surplus agricultural land. The attribution of grassland SOC densities to these classes is somewhat arbitrary, but it can be argued that solid biofuels are perennials like grasses with a relatively important belowground biomass. Literature on SOC in built up areas is scarce, but Pouyat et al. (2002) point to relatively high stocks. BOC from biofuels and agricultural crops was not considered for the GHG balance. However, fossil fuel replacement by biomass can be discussed separately at a rate of 4.2 t C ha^{-1} yr^{-1} (SMITH ET AL., 2000).

Table 4.4 *Organic Carbon (OC) stocks per ha and per LU type in 2012 and 2050. SOC stocks refer to the 0-30 cm topsoil.*

	2012			**2050**		
Land use	**Area 1990** (km²)	**OC stock** (t C ha^{-1})	(Mt C)	**Area 2050** (km²)	**OC stock** (t C ha^{-1})	(Mt C)
Forest	6201	243.5	151,0	6245	250.1	156,2
Grassland	3678	83.1	30,6	2295	86,6	19,9
Cropland	14125	54.8	77,4	12969	53,1	68,9
Liquid biofuels				677	53,1	3,6
Solid biofuels				961	86,6	8,3
Increase urban + surplus				859	86,6	7,4
Total	*24004*		*258,9*	*24007*		*264,3*

Overall, BOC and SOC stocks are predicted to increase from 2012 to 2050 (Table 4.4). The increase in the areas under forest, solid biofuels and built up areas, characterised by high SOC and BOC densities, at the expense of cropland with a low SOC density compensates the decrease in grassland, which is also characterised by a high SOC density. The net increase in SOC stock results in a sequestration of 0.52 Mt CO_2 yr^{-1}. This does not include an additional sequestration of c. 2.52 Mt CO_2 yr^{-1} as a result of fossil fuel substitution by the burning of biomass. The C-sequestration potential for the period 1990-2000 was estimated at 2.85 Mt CO_2 yr^{-1} (Table 3.9). The decline in C-sequestration after 2012 can largely be attributed to the slowing down of the growth rates of the forests (Table 4.3).

Table 4.5 *Mean N_2O-fluxes per agricultural land use type in 2000, 2012 and 2050*

Land use	Area[1]	1990-2000 N$_2$O flux		2000-2012 N$_2$O flux		Area[2]	2012-2050 N$_2$O flux	
	(km²)	(t CO_2 eq $ha^{-1}yr^{-1}$)	(Mt CO_2-eq.yr^{-1})	(t CO_2-eq $ha^{-1}yr^{-1}$)	(Mt CO_2 eq.yr^{-1})	(km²)	(t CO_2eq $ha^{-1}yr^{-1}$)	(Mt CO_2eq yr^{-1})
Grassland	6289	0.94	0.59	0.94	0.59	3925	1.09	0.43
Cropland	7062	2.27	1.60	1.86	1.31	6484	2.38	1.54
liquid biofuels						677	2.38	0.16
solid biofuels						961	1.09	0.10
Total N_2O flux			*2.19*		*1.90*			*2.24*

[1] *N_2O fluxes are calculated from the net grassland and cropland areas, which are fertilised. These are calculated from the areas declared by the farmers (INS).*
[2] *Predicted areas under cropland and grassland are corrected to represent the net areas.*

The implications of the climate and land use change on N_2O emissions from soils can sometimes be more important than the potential for CO_2 sequestration. Although the uncertainties on the N_2O modelling are quite large and the interannual variability can be more important than the effects of a gradual trend in the climate, N_2O emissions are indeed large and in the same order of magnitude as C- sequestration (1.9-2.24 Mt CO_2eq. yr^{-1}, Table 4.5). However, no significant increase in emissions between 2000 and 2050 were observed. Although terrestrial ecosystems were a small net sink from 1990 to 2000 (0.66 Mt CO_2eq. yr^{-1}; Table 3.9), under the A1FI-BAU scenario a net source of –1.72 Mt CO_2eq. yr^{-1} is predicted after 2012 (Tables 4.4 and 4.5).

This section demonstrates the potential of a spatially explicit GHG modelling approach driven by spatial data sets on climate, soils, management and land use. However, the results are derived from a single scenario (A1FI) assuming a business as usual land management. This scenario merely represents a possible future. Refining the spatially explicit approach and applying other more detailed scenarios, in particular for land management will be the next step forward. Even under extreme climate and land use scenarios important decrease of C-stocks in terrestrial ecosystems have not been found. This is in contradiction with the European wide modelling work by Smith et al. (2005). This contrast can perhaps be explained by the use of a BAU management scenario with high organic inputs from manure and the small climate differences in Belgium.

5 CONCLUSION

The Metage project focused on the main fluxes of greenhouse gases from terrestrial ecosystems: CO_2 fluxes from the soil or woody biomass in forests, as well as direct N_2O emissions from agricultural soils. Right from the start an explicitly spatial approach was chosen covering the entire non-built up area of Belgium. Hence, GHG fluxes and C stocks are calculated for the so-called landscape units (LSUs), which are characterised by homogeneous soil, climate, land use and land management characteristics. It is assumed that the per-LSU and total CO_2 flux is equal to the observed change in soil organic carbon (SOC) or biomass organic carbon (BOC) stock in CO_2 equivalents over a certain time span and that the per-LSU-fluxes can be aggregated to yield total fluxes at regional or national levels. An empirical approach was followed to relate N_2O fluxes to soil and climate data that is available at the LSU level. This spatial framework has three main advantages: i) spatial patterns of GHG fluxes can be represented, ii) the spatial units cover the entire non-urban surface, thus avoiding inconsistencies, such as leakage or double counting, iii) fluxes can be calculated at various levels from the LSU to the land use type or for all terrestrial ecosystems in Belgium. Although the most important spatial and temporal trends in GHG fluxes are discussed in this report, it became clear during the project that there are often requests from end-users (MIRA-T reports, Tableau de Bord de l'Environnement wallon, National GHG reporting) for GHG fluxes at different levels of aggregation than the ones chosen for this report. A spatial data set containing all information on GHG fluxes and C stocks was organised in a user-friendly GIS system. This system is available through a link on the Metage website (www.geo.ucl.ac.be), and allows the user to calculate stocks and fluxes according to their own criteria.

Heterogeneous data sources with large spatial and temporal variability were used to calculate SOC changes for the period 1960 until 2000, including the IPCC baseline year 1990. This implies that significant changes in SOC stock and hence fluxes of CO_2 could not always be detected at the LSU level. The large amount of data ensured, however, that regional CO_2 fluxes could be derived for each type of land use. Overall the SOC stock of terrestrial ecosystems in Belgium increased from 1960 to 2000. Grassland and forest sequestered CO_2 between 1960 and 2000. Croplands emitted CO_2 in Wallonia and sequestered CO_2 in Flanders resulting in a slight emission at the national level. A majority of agricultural LSUs contained more SOC in 1990 than in 2000. This observation could not be verified for forested LSUs due to lack of data but the expectation is that forested LSUs continued to store more carbon due to more ecological forest exploitation practices. With regard to agriculture, changes in application practices of animal manure, as driven by economic factors and policy, are found to be the major explanatory factor for the observed changes. Although carbon dynamics simulation models are reported to produce reliable SOC estimates when applied to long-term experiments, their application at the regional scale is more problematic. The management and input from organic amendments proves difficult to reconstruct for longer periods. A soil carbon simulation model (RothC) was run for the LSUs from 1960 until 2000. The deviation between modelled and observed data (RMSE) in 1990 and 2000 is restricted to 1.5-1.6 % of the SOC stock in an LSU. However, the slight decrease in SOC stocks observed between

1990 and 2000 was not well represented by the RothC model. All those findings must be put in the context of a possibly important source of error in SOC inventories: the applied OC measurement technique and its associated correction factor.

The forest inventories were used to estimate the BOC stocks. Two critical factors had to be addressed before BOC stocks for the baseline year (1990), 2000 and 2012 could be calculated. First, biomass expansion factors had to be decided upon and then a model had to be developed to simulate the growth of the overall young forest stands (EFOBEL). It is not surprising that the choice of the biomass expansion factors is the main source of error. The highest stocks per ha reside in broadleaf forest and the lowest in mixed forest. Forest in Flanders contains on average less BOC than in Wallonia and this may be explained by the lower average forest age in Flanders. Based on comparison with other BOC inventories it is concluded that the geomatching methodology delivers satisfying results, considering its limited data requirements and the advantage that it reflects the spatial heterogeneity of the LSU as regards BOC.

Multivariate linear relationships were developed that link **N_2O** emissions to climate, land use and N-fertilisation. Two empirical models were presented for cropland (MCROPS) and grassland (MGRASS) emission patterns. The analysis demonstrated that seasonal climate has an impact on the annual N_2O emissions. Emissions from crops were found to be sensitive to spring temperature and summer precipitation. The grassland-emission pattern is, however, driven by nitrogen-fertilisation and winter temperature. Compared against independent data sets, the empirical models were able to explain 35% of the variance in annual N_2O emissions for croplands and 48% of the variance for grasslands. Based on a larger data set, MCROPS and MGRASS improve the statistical reliability compared to the IPCC default methodology. These models are innovative by linking seasonal climate to annual N_2O emissions in an empirical approach. MCROPS demonstrates the relationship between annual N_2O emissions and seasonal climate patterns independently of any management information. MGRASS improves the IPCC methodology by adding climatic information to nitrogen fertilisation information. When these empirical models were applied to the LSUs clear spatial patterns occurred with higher emissions in Flanders.

The implications of one of the IPCC **climate and land use scenarios** (A1 FI) on GHG emissions were tested. The A1FI scenario is a globally and economically oriented scenario also referred to as 'world markets-fossil fuel intensive'. Land use change was taken into account from 2012 onwards by means of new LSU's based on a predicted land use map for 2050. Considering the fact that N_2O emissions remain important under global change scenario and are in the same order of magnitude as C- sequestration (1.9-2.24 Mt CO_2eq. yr^{-1}), a synthesis of the fluxes of GHG from terrestrial ecosystems can only be made by considering all GHG fluxes and expressing them in CO_2eq according to their global warming potential. This synthesis reveals that, under the A1FI-BAU scenario, terrestrial ecosystems gradually become a net source of GHG's: from a sink of 0.66 Mt CO_2eq. yr^{-1} (1990-2000) to a source of –1.72 Mt CO_2eq. yr^{-1} (2012-2050). The effects of land management prevail over those of land use change under a BAU scenario. This can largely be attributed to the slowing down of the growth rates of the forests.

The main aim of the METAGE project was to develop a spatially explicit approach to estimate greenhouse gases (GHGs) from terrestrial ecosystems under global change scenarios. These predictions are based on models using baseline emissions as a starting point and current trends to evaluate their accuracy. Since current GHG fluxes were not known at the regional scale a large part of the project dealt with GHG fluxes for the 1960-2000 period, and the prediction of GHG fluxes under global change scenarios received less attention. The importance of land management on CO_2 fluxes was underlined by the regional differences in the period 1960-2000: intensification of agriculture in large parts of Flanders and extensification in the southern part of Wallonia with the dominance of fast growing forests. N_2O fluxes were very sensitive to seasonal climate. Predictions of GHG fluxes under global change scenarios are only possible, when land management aspects are taken into account along side land use and climate. It proved difficult to reconstruct land management data (FYM and slurry as well as rotations) specific enough to reconstruct the spatial patterns in SOC dynamics based on statistical data for agricultural regions (14 regions cover the entire country). Therefore, it is recommended to improve the regional scale SOC modelling starting with pilot zones for which more detailed input data can be retrieved.

During the project the new data and guidelines became available. These were, unfortunately, not taken into account in the development of the spatial framework. A new Corine land cover data set is now available for 2000, and the IPCC have issued more detailed Guidance on accounting LULUCF. From a confrontation of SOC data with historical land use change data from the National Institute for Statistics, it is suggested that recent land use changes do not unacceptably distort results. However, the availability of the new Corine land cover data set would allow confirming this suggestion. The reporting of land areas within the context of Articles 3.3 and 3.4 must include information on the geographical limits of the portions of territories subjected to these activities. The Good Practice Guidance gives two reporting methods: i) the broad representation of geographical units including/understanding of multiple legal, administrative or ecological units subjected to the human activities; ii) the complete identification of each unit. Although the geographical limits of the LSUs are defined and they are contiguous covering the entire non-built up territory, the spatial variability of their SOC contents is in many cases too large to detect significant changes between two time slices. Therefore CO_2 fluxes can often not be detected from the individual LSUs and a regional approach remains necessary. The availability of the new Corine data set will not resolve this problem, and the only way forward is the re-sampling LSUs instead of relying on existing data.

The reliability of the empirical N_2O models would be improved by more long-term data with a finer resolution for the climate and a better geographical spread and temporal distribution. This is especially true for the MCROPS model. Furthermore, differences amongst crop and grassland types need to be investigated together with the effects of land management. Incorporating management effects within these types of models would improve their capacity to support mitigation policy.

6 ACKNOWLEDGEMENTS

The research was carried out under the second Science for Sustainable development programme of the Belgian Science Policy (contract numbers, EV/10/14A, EV/16/14B and EV/01/14C), and the support is gratefully acknowledged. The authors wish to express their thanks to the agencies that provided data on soil carbon, Belgian Soil Service, Requasud, Centres d'Information Agricole des provinces de Luxembourg, Liège et Hainaut, the Institute for Forestry and Game management and the Division des Forêts et de la Nature du Ministère de la Région walonne, and on forest inventories, Afdeling Bos en Groen (AMINAL, Vlaamse Gemeenschap) and Direction générale des Ressources Naturelles et de l'Environnement (Ministère de la Région walonne).

7 REFERENCES

Afdeling Bos & Groen, 2000, De bosinventarisatie van het Vlaamse Gewest. Resultaten van de eerste inventarisatie 1997-2000. Brussels: Ministry of the Flemish Community.

Arrouays, D., Deslais, W. and Badeau, V. , 2001. The carbon content of topsoil and its geographical distribution in France. Soil Use and Management 17, 7-11.

Bareth, G. and Angenendt, E., 2003. Economic-ecological modelling of climate-relevant emissions from agriculture at regional level. Berichte uber Landwirtschaft 81, 29-56.

Baritz, R. and Strich, S., 2000. Forests and the National Greenhouse Gas Inventory of Germany. Biotechnology, Agronomy, Society and Environment 4, 267-271.

Batjes, N. H., 1996. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science 47, 151-163.

Boeckx, P., Van Moortel, E. and Van Cleemput, O., 2001. Spatial and sectorial disaggregation of N_2O emissions from agriculture in Belgium. Nutrient Cycling in Agroecosystems 60, 197-208.

Boon, W., 1984. Onderzoek naar het verband tussen het koolstofpercentage en de volumedichtheid van de grond. Bodemkundige Dienst van België. Heverlee, Belgium.

Boudru, M., 1986. Forêt et Sylviculture: sylviculture appliquée. Presses Agronomiques, Gembloux, Belgium.

Bouwman, A. F., 1996. Direct emission of nitrous oxide from agricultural soils. Nutrient Cycling in Agroecosystems 46, 53-70.

Bouwman, A. F., Boumans, L. J. M. and Batjes, N. H., 2002a. Modeling global annual N2O and NO emissions from fertilized fields. Global Biogeochemical Cycles 16, art. No. 1080.

Bouwman, A. F., Van Vuuren, D. P., Derwent, R. G. and Posch, M., 2002b. A global analysis of acidification and eutrophication of terrestrial ecosystems. Water Air and Soil Pollution 141, 349-382.

Brown, S., 2002. Measuring carbon in forest: current status and future challenges. Environmental Pollution 116, 363-372.

Brye, K. R. and Slaton, N. A., 2003. Carbon and nitrogen storage in a typic Albaqualf as affected by assessment method. Communications in Soil Science and Plant Analysis 34, 1637-1655.

Cannell, M., 1982. World forest biomass and primary production data. Academic Press, London, 1982.

Chan, K. Y. and Heenan, D. P., 1999. Lime-induced loss of soil organic carbon and effect on aggregate stability. Soil Science Society of America Journal 63,1841-1844.

Clayton, H., McTaggart, I. P., Parker, J., Swan, L. and Smith, K. A., 1997. Nitrous oxide emissions from fertilised grassland: A 2-year study of the effects of N fertiliser form and environmental conditions. Biology and Fertility of Soils 25, 252-260.

Coleman, K. and Jenkinson, D.S., 1996. RothC--26.3 - A model for the turnover of carbon of carbon in soil. in D.S. Powlson, P. Smith and J.U. Smith, eds., Evaluation of soil organic matter using existing long-term datasets. NATO ASI Series, vol.38. Heidelberg: Springer-Verlag, pp. 237 - 246.

Coleman, K., Jenkinson, D. S., Crocker, G. J., Grace, P. R., Klir, J., Korschens, M., Poulton, P. R. and Richter, D. D., 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma 81, 29-44.

Davidson, E. A. and Verchot, L. V., 2000. Testing the hole-in-the-pipe model of nitric and nitrous oxide emissions from soils using the TRAGNET database. Global Biogeochemical Cycles 14, 1035-1043.

Davidson, E. A., Savage, K., Bolstad, P., Clark, D. A., Curtis, P. S. et al., 2002. Belowground carbon allocation in forests estimated from litterfall and IRGA-based soil respiration measurements. Agricultural And Forest Meteorology 113, 39-51.

De Vos, B., Vandecasteele, B., Deckers, S. and Muys, B. (subm.) Capability of loss-on-ignition to predict total organic carbon in noncalcareous forest soils. Communications in Soil Science and Plant Analysis.

Dendoncker, N., Bogaert, P. and Rounsevell, M., (subm) A statistical method to downscale aggregated land use data.

Dendoncker, N., Van Wesemael, B., Rounsevell, M. D. A., Roelandt, C. and Lettens, S., 2004. Belgium's CO2 mitigation potential under improved cropland management. Agriculture Ecosystems & Environment 103, 101-116.

Diaz-Zorita, M., 1999. Soil organic carbon recovery by the Walkley-Black method in a Typic Hapludoll. Communications in Soil Science and Plant Analysis 30, 739-745.

Dieter, M. and Elsasser, P., 2002. Carbon Stocks and Carbon Stock Changes in the Tree Biomass of Germany's Forests. Forstwissenschaftliches Centralblatt 121, 195-210.

Dobbie, K. E. and Smith, K. A., 2003. Impact of different forms of N fertilizer on N_2O emissions from intensive grassland. Nutrient Cycling in Agroecosystems 67, 37-46.

Dobbie, K. E., McTaggart, I. P. and Smith, K. A., 1999. Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. Journal of Geophysical Research-Atmospheres 104, 26891-26899.

Drover, D. P. and Manner, H., 1975. Comparison between Walkley-Black and a dry combustion method for determining organic carbon in a humic brown soil, Papua-New-Guinea. Communications in Soil Science and Plant Analysis 6, 495 - 500.

Eichner, M. J., 1990. Nitrous oxide emissions from fertilized soils: Summary of available data. Journal of Environmental Quality 19, 272 - 280.

Ewert, F., Rounsevell, M.D.A., Reginster, I., Metzger, M. and Leemans, R., 2005. Future scenarios of European agricultural land use. I Changes in crop productivity. Agriculture, Ecosystems and Environment 107, 101-116.

Falloon, P. D., Smith, P., Smith, J. U., Szabo, J., Coleman, K. and Marshall, S., 1998. Regional estimates of carbon sequestration potential: linking the Rothamsted Carbon Model to GIS databases. Biology and Fertility of Soils 27, 236-241.

Firestone, M. K. and Davidson, E. A., 1989. Exchange of trace gases between terrestrial ecosystems and the atmosphere. in M.O. Andrea and D.S. Schimel, eds. Trace Gases between Terrestrial Ecosystems and the Atmosphere, John Wiley & Sons Ltd, New York. pp. 7 - 21.

Frankinet, M., Raimond, Y., Destain, J.P., Roisin, C. and Grevy, L., 1993. Organic matter management and calcic amendmends in order to maintain or improve soil fertility. in P. Paoletti, W. Foissner and D. Coleman, eds., Soil biota, nutrient cycling and farming systems. Lewis Publishers, Boca Raton, pp. 27-39.

Freibauer, A., 2003. Regionalised inventory of biogenic greenhouse gas emissions from European agriculture. European Journal of Agronomy 19, 135-160.

Freibauer, A. and Kaltschmitt, M., 2003. Controls and models for estimating direct nitrous oxide emissions from temperate and sub-boreal agricultural mineral soils in Europe. Biogeochemistry 63, 93-115.

Freibauer, A., Rounsevell, M. D. A., Smith, P. and Verhagen, J., 2004. Carbon sequestration in the agricultural soils of Europe. Geoderma 122, 1-23.

Garten, C. T. and Wullschleger, S. D., 1999. Soil carbon inventories under a bioenergy crop (switchgrass): Measurement limitations. Journal of Environmental Quality 28, 1359-1365.

Gillman, G. P., Sinclair, D. F. and Beech, T. A. , 1986. Recovery of organic-carbon by the Walkley and Black procedure in highly weathered soils. Communications in Soil Science and Plant Analysis 17, 885 - 892.

Goodman, L. A., 1960. On the exact variance of products. Journal of the American Statistical Association 55, 708 - 713.

Gordon, C., Cooper, C., Senior, C. A., Banks, H., Gregory, J. M. et al., 2000. The simulation of SST, sea ice extents and ocean heat transports in a version of the Hadley Centre coupled model without flux adjustments. Climate Dynamics 16, 147-168.

Gracia, C., Sabaté, S., Vaydera, J. and Ibànez, J., 2002. Aboveground biomass expansion factors and biomass equations of forests in Catalonia. COST E21, WG1-biomass Workshop.

Granli, T. and Bockman, O. C. 1994. Nitrous oxide in agriculture. Norwegian Journal of Agricultural Sciences 12, 7 - 128.

Guo, L. B. and Gifford, R. M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8, 345-360.

Henault, C., Devis, X., Lucas, J. L. and Germon, J. C., 1998. Influence of different agricultural practices (type of crop, form of N-fertilizer) on soil nitrous oxide emissions." Biology and Fertility of Soils 27, 299-306.

Howard, P. J. A., Loveland, P. J., Bradley, R. I., Dry, F. T., Howard, D. M. and Howard, D. C., 1995. The Carbon Content Of Soil And Its Geographical-Distribution In Great-Britain. Soil Use And Management 11, 9-15.

Hébert, J. and Laurent, C., 1995. Estimation de la disponibilité de la ressource forestière. Cas d'une essence traitée en futaie équienne monospécifique. Revue Forestière Française, 47, 572-580.

IPCC, 1997. Revised 1996 IPCC guidelines for national greenhouse gas inventories. Intergovernmental Panel on Climate Change. Genève:

IPCC, 2005. Good Practice Guidance for land use, land-use change and forestry.. : Intergovernmental Panel on Climate Change. Genève

Janzen, H. H., 2005. Carbon cycling in earth systems - a soil science perspective. Agriculture, Ecosystems and Environment 104, 399 - 417.

Johnson, D. W. and Curtis, P. S., 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology And Management 140, 227-238.

Kaiser, E. A. and Ruser, R., 2000. Nitrous oxide emissions from arable soils in Germany - An evaluation of six long-term field experiments. Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 163, 249-259.

Kaiser, E. A., Kohrs, K., Kucke, M., Schnug, E., Heinemeyer, O. and Munch, J. C., 1998a. Nitrous oxide release from arable soil: Importance of N- fertilization, crops and temporal variation. Soil Biology & Biochemistry 30, 1553-1563.

Kaiser, E. A., Kohrs, K., Kucke, M., Schnug, E., Munch, J. C. and Heinemeyer, O., 1998b. Nitrous oxide release from arable soil: importance of perennial forage crops. Biology and Fertility of Soils 28, 36-43.

Kammann, C., Grunhage, L., Muller, C., Jacobi, S. and Jager, H. J., 1998. Seasonal variability and mitigation options for N_2O emissions from differently managed grasslands. Environmental Pollution 102, 179-186.

Kirschbaum, M.U.F., Schlamadinger, B., Cannell, M.G.R., Hamburg, S.P., Karjalainen, T. et al., 2001. A generalised approach of accounting for biospheric carbon stock changes under the Kyoto Protocol. Environmental Science & Policy 4, 73-85.

Laitat, E., Karjalainen, T., Loustau, D. and Lindner, M., 2000. COST E21 Workshop: Contribution of forests and forestry to mitigate greenhoue effects. Biotechnology Agronomy Society and Environment 4, 241-251.

Laitat, E., Perrin, D., Sheridan, M., Lebègue, C. and Pissart, G., 2004. EFOBEL un modèle de calcul de la sequestration du carbone par les forêts, selon les termes des Accords de Marrakech et les engagements de rapportage de la Belgique au Protocole de Kyoto. Biotechnology Agronomy Society and Environment 8, 27-40.

Lambert, R., Peeters, A. and Toussaint, B., 2003. Proposition d'une méthode de raisonnement de la fertilisation azotée en prairie. Centre de recherche agronomique de Gembloux.

Lecomte, H. and Rondeux, J., 1994. L'inventaire forestier régional wallon: brève présentation méthodologique. Silva Belgica, 101, 9-16.

Leroy, I., Van Meirvenne, M., Depuydt, S. and Hofman, G., 2000. Digitalisatie en verwerking van historische bosbodemprofielgegevens: Eindverslag. Vakgroep Bodembeheer en Bodemhygiëne, Universiteit Gent.

Lettens, S., Van Orshoven, J., van Wesemael, B., De Vos, B. and Muys, B., 2005. SOC content and SOC content change of landscape units in Belgium derived from heterogeneous datasets for 1990 and 2000. Geoderma 127, 11-23.

Li, C. S., Frolking, S. and Frolking, T. A., 1992. A Model of Nitrous-Oxide Evolution from Soil Driven by Rainfall Events.1. Model Structure and Sensitivity. Journal of Geophysical Research-Atmospheres 97, 9759-9776.

Lilly, A., Ball, B. C., McTaggart, I. P. and Horne, P. L., 2003. Spatial and temporal scaling of nitrous oxide emissions from the field to the regional scale in Scotland. Nutrient Cycling in Agroecosystems 66, 241-257.

Liski, J., Perruchoud, D. and Karjalainen, T., 2002. Increasing carbon stocks in the forest soils of western Europe. Forest Ecology And Management 169, 159-175.

Milne, R. and Brown, T. A., 1997. Carbon in the vegetation and soils of Great Britain." Journal Of Environmental Management 49, 413-433.

Mitchell TD, Carter TC, Jones P et al. (2004) A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed record (1901–2000) and 16 scenarios (2001–2100). Tyndall Centre Working Paper, 55, Tyndall Centre for Climate Change Research, University of East Anglia, Norwich, 30pp

Nabuurs, G. and Mohren, G., 1993. Carbon in Dutch forest ecosystems. Netherlands Journal of Agricultural Sciences 41, 309-326.

Nakićenović, N., J. Alcamo, G. Davis, B. de Vries, J. Fenhann, S. Gaffin, K. et al., 2000. Special report on emissions scenarios. Cambridge University Press, Cambridge

Neter, J., Kutner, M. H., Nachtsheim, C. J. and Wasserman, W., 1996. Applied linear statistical models. McGraw-Hill Companies, Inc., New York.

NIS, 1990. " Recensement agricole et horticole au 15 mai 1990". Institut National de Statistiques, Ministère des Affaires Economiques, Bruxelles.

NIS, 2000. " Recensement agricole et horticole au 15 mai 2000". Institut National de Statistiques, Ministère des Affaires Economiques, Bruxelles.

Parton, W. J., Holland, E. A., Del Grosso, S. J., Hartman, M. D., Martin, R. E., Mosier, A. R., Ojima, D. S. and Schimel, D. S., 2001. Generalized model for NOx and N_2O emissions from soils. Journal of Geophysical Research-Atmospheres 106, 17403-17419.

Perrin, D. 2005. Modélisation des flux de carbone de sols forestiers à différentes échelles spatiale et temporelle. Unpublished PhD thesis, Faculté Universitaire des Sciences Agronomiques de Gembloux.

Perrin, D., Temmerman, M. and Laitat, E., 2000. Calculation on the impacts of forestation, afforestation and reforestation on the C-sequestration potential in Belgian forests ecosystems." Biotechnology, Agronomy, Society and Environment 4, 259-262.

Pignard, G., Dupouey, J.-L., Arrouays, D. and Loustau, D., 2000. Carbon stocks estimates for French forests. Biotechnology, Agronomy, Society & Environment 4, 285-289.

Post, W. M. and Kwon, K. C., 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology 6, 317-327.

Potter, C. S., Matson, P. A., Vitousek, P. M. and Davidson, E. A., 1996. Process modeling of controls on nitrogen trace gas emissions from soils worldwide. Journal of Geophysical Research-Atmospheres 101, 1361-1377.

Pouyat, R., Groffman, P., Yesilonis, I. and Hernandez, L., 2002. Soil carbon pools and fluxes in urban ecosystems. Environmental Pollution 116, S107-S118.

Rawls, W. J., 1983. Estimating soil bulk-density from particle-size analysis and organic-matter content. Soil Science 135, 123 - 125.

Rice, J.A., 1995. Mathematical statistics and data analysis. : Duxbury Press.

Roelandt, C., van Wesemael, B. and Rounsevel, M.A.D., 2005. Estimating annual N_2O emissions from agricultural soils in temperate climates. Global Change Biology 11, 1701-1711.

Rogasik, J., Schroetter, S., Funder, U., Schnug, E. and Kurtinecz, P., 2004. Long-term fertilizer experiments as a data base for calculating the carbon sink potential of arable soils. Archives of Agronomy and Soil Science 50, 11 - 19.

Rondeux, J., 1999. La mesure des arbres et des peuplements forestiers. Gembloux: Les Presses Agronomiques de Gembloux.

Rounsevell, M.D.A.; Ewert, F.; Reginster, I.; Leemans, R.; Carter, T.R., 2005. Future scenarios of European agricultural land use. II. Projecting changes in cropland and grassland. Agriculture, Ecosystems and Environment 107, 117-135.

Shaw, E., 1994. Hydrology in practice. Chapman & Hall, London.

Sheridan, M., 2002. Essai de quantification et de prévision de l'évolution du stock de C de la forêt wallonne. Gembloux.

Skiba, U. and Smith, K. A., 2000. The control of nitrous oxide emissions from agricultural and natural soils. Chemosphere - Global Change Science 2, 379-386.

Skiba, U., Hargreaves, K. J., Beverland, I. J., O neill, D. H., Fowler, D. and Moncrieff, J. B., 1996. Measurement of field scale N_2O emission fluxes from a wheat crop using micrometeorological techniques. Plant and Soil 181, 139-144.

Skiba, U., van Dijk, S. and Ball, B. C., 2002. The influence of tillage on NO and N_2O fluxes under spring and winter barley. Soil Use and Management 18, 340-345.

Skjemstad, J. O., Spouncer, L. R. and Beech, A., 2000. Carbon conversion factors for historical soil carbon data: CSIRO, Land and Water, Adelaide.

Sleutel, S., De Neve, S. and Hofman, G., 2003. Estimates of carbon stock changes in Belgian cropland. Soil Use And Management 19, 166-171.

Smith, K. A., McTaggart, I. P., Dobbie, K. E. and Conen, F., 1998a. Emissions of N_2O from Scottish agricultural soils, as a function of fertilizer N. Nutrient Cycling in Agroecosystems 52, 123-130.

Smith, K. A., Thomson, P. E., Clayton, H., McTaggart, I. P. and Conen, F., 1998b. Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. Atmospheric Environment 32, 3301-3309.

Smith, P., Milne, R., Powlson, D. S., Smith, J. U., Falloon, P. and Coleman, K., 2000. Revised estimates of the carbon mitigation potential of UK agricultural land. Soil Use And Management 16, 293-295.

Smith, P., Powlson, D.S., Smith, J.U., Falloon, P. and Coleman, K., 2000. Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. Global Change Biology 6, 525-539.

Smith, J., Smith, P., Wattenbach, M., Zaehle, S., Hiederer, R., Jones, R.J.A., Montanarella, L., Rounsevell, M., Reginster, I. and Ewert, F. 2005. Projected changes in mineral soil carbon of European croplands and grasslands, 1990-2080. Global Change Biology 11, 1-12.

Smith, P., Smith, J. U., Powlson, D. S., McGill, W. B., Arah, J. R. M. et al., 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. Geoderma 81, 153-225.

Soltner, D., 1999. Les grandes productions végétales. Sciences et Techniques agricoles, Sainte-Gemmes-Sur-Loire.

Sozanska, M., Skiba, U. and Metcalfe, S., 2002. Developing an inventory of N_2O emissions from British soils. Atmospheric Environment 36, 987-998.

Tate, K. R., Scott, N. A., Saggar, S., Giltrap, D. J., Baisden, W. T., Newsome, P. F., Trotter, C. M. and Wilde, R. H., 2003. Land-use change alters New Zealand's terrestrial carbon budget: Uncertainties associated with estimates of soil carbon change between 1990-2000. Tellus Series B-Chemical And Physical Meteorology 55, 364-377.

Tavernier, R. and Maréchal, R., 1972. Bodemassociatiekaart 1:500,000. Atlas van Belgïe, bladen 11A en 11B.

UNECE/FAO, 2000. Forest resources of Europe, CIS, North America, Australia, Japan and New Zealand; UNECE/FAO contribution to the Global Forest Resource Assessment 2000. Main Report (ECE/TIM/SP/17).

Vande Walle, I., Van Camp, N., Perrin, D., Lemeur, R., Verheyen, K., van Wesemael, B. and Laitat, E. (in press). Growing stock-based assessment of the carbon stock in the Belgian forest biomass. Annals of Forest Science.

VanMeirvenne, M., Pannier, J., Hofman, G. and Louwagie, G., 1996. Regional characterization of the long-term change in soil organic carbon under intensive agriculture. Soil Use And Management 12, 86-94.

Van Moortel, E., Boeckx, P. and Van Cleemput, O., 2000. Inventory of nitrous oxide emissions from agriculture in Belgium - calculations according to the revised 1996 Intergovernmental Panel on Climate Change guidelines. Biology and Fertility of Soils 30, 500-509.

Van Muysen, W., Govers, G., Van Oost, K. and Van Rompaey, A., 2000. The effect of tillage depth, tillage speed, and soil condition on chisel tillage erosivity. Journal Of Soil And Water Conservation 55, 355-364.

Van Oost, K., Govers, G. and Desmet, P., 2000. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. Landscape Ecology 15, 577-589.

Van Orshoven, J., Deckers, J. A., Vandenbroucke, D. and Feyen, J., 1993. The completed database of Belgian soil profile data and its applicability in planning and management of rural land. Bulletin des Recherches Agronomiques de Gembloux 28, 197 - 222.

Van Orshoven, J., Maes, J., Vereecken, H. and Feyen, J., 1991. A procedure for the statistical characterization of the units of the Belgian soil map. Pedologie, 1991, XLIpp. 193 - 212.

Van Wesemael, B., Lettens, S., Roelandt, C. and Van Orshoven, J., 2005. Modelling the evolution of regional carbon stocks in Belgian cropland soils. Canadian Journal of Soil Science 85, 511-521.

Vejre, H., Callesen, I., Vesterdal, L. and Raulund-Rasmussen, K., 2003. Carbon and nitrogen in Danish forest soils - Contents and distribution determined by soil order. Soil Science Society Of America Journal 67, 335-343.

Vesterdal, L., 2000. Carbon sequestration in Danish forests. Biotechnology, Agronomy, Society & Environment 4, 272-274.

Vlaamse Landmaatschappij, 2004. Mestbank: manure registration at the municipality level (in Dutch). [On line] Available at: http://www.vlm.be/mestbank/startpagina.htm (accessed 2-11-2005)

Wagner-Riddle, C., Thurtell, G. W., Kidd, G. K., Beauchamp, E. G. and Sweetman, R., 1997. Estimates of nitrous oxide emissions from agricultural fields over 28 months. Canadian Journal of Soil Science 77, 135-144.

Walkley, A. and Black, I. A., 1934. An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. Soil Science 37, 29 - 38.

Wang, S. Q., Tian, H. Q., Liu, J. Y. and Pan, S. F., 2003. Pattern and change of soil organic carbon storage in China: 1960s-1980s. Tellus Series B-Chemical And Physical Meteorology 55, 416-427.

Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J. and Dokken, D.J. (eds.), 2000. Land use, land use change and forestry, A special report of the IPCC. Cambridge University Press.

Weiss, P., Schieler, K., Schadauer, K., Radunsky, K. and Englisch, M., 2000. Die Kohlenstoffbilanz des Österreichischen Waldes und Betrachtungen zum Kyoto-Protokoll." Umweltbundesamt – Monographien Band 106.