SPSD II

CARBON SEQUESTRATION POTENTIAL IN DIFFERENT BELGIAN TERRESTRIAL ECOSYSTEMS: QUANTIFICATION AND STRATEGIC EXPLORATION (CASTEC)

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PART 2

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

Part 2:
Global change, Ecosystems and Biodiversity

FINAL REPORT

Carbon Sequestration Potential in Different Belgian Terrestrial Ecosystems: Quantification and Strategic Exploration (CASTEC)

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CONTEXT AND GOALS

The aim of this project was to estimate the potential of important Belgian terrestrial ecosystems with respect to C sequestration and N_2O emissions. To elucidate this general outline, an extract from the original CASTEC proposal is taken.

Determining the C sequestration potential of the different Belgian terrestrial ecosystems will be performed via 3 vertical Task Forces (TF) (Fig. 1): TF1 covers cropland, TF2 covers grassland and urban land, and TF3 deals with forests and plantations. Each of these 3 TFs consists out of 3 subtasks. For TF1 this is: (I) a desk study summarising areas of cropland and the current knowledge on soil organic carbon (SOC) accumulation rates, (II) a validation study using historical data sets and establishment of the baseline (reference year for greenhouse gas reductions is 1990) SOC values of Belgian cropland, and (III) a scenario analysis giving the C sequestration potential for cropland and its interaction with the sources and sinks of greenhouse gases other than CO_2 (in co-operation with TF4a). For TF2 this is: (I) a desk study summarising the areas of the different types of grassland and urban land and their SOC accumulation rates, (II) gap filling research on SOC accumulation of permanent grassland and urban land, on impacts of land use change on SOC fluxes, and on interactions between nutrient cycles (synergy with TF4a), and (III) a scenario analysis giving the C sequestration potential of grassland and urban land. For TF3 this is: (I) to collect data on C pools and fluxes for a series of (model) forest ecosystems in Belgium, (II) to quantify the potential for C sequestration in short rotation bio-energy plantations using experimental data from model plantations, and (III) to co-operate with TF4a for the experimental assessment of CO_2 and CH_4 fluxes in model forests and plantations. These vertical TFs have important interaction with 2 cross cutting TFs. In TF4a potential interactions of soil management scenarios, which could increase C pools, with fluxes of non-CO_2 greenhouse gases (N_2O and CH_4) will be investigated. In forests/plantations CO_2 fluxes from soils will be included. In TF4b experiments using carbon-13 analysis will be set-up for the quantification of SOC accumulation and stabilization in cropland and grassland. The latter is needed to create a future database of soils, which are physically suitable for C sequestration. The proposal will finalise in a modelling TF (TF5). Herein, (I) an existing Dynamic Vegetation Model (DVM) will be optimized in order to predict C storage and C uptake in ecosystems of TF2 and TF3 in the short- and long-term and to establish the baseline (1990) stocks in these terrestrial ecosystems, and finally (II) a GIS based up-scaling procedure will be developed to assess (on a community level) the effect of land use changes and soil management changes (within TF1, TF2 and TF3) on C sequestration, relative to 1990.

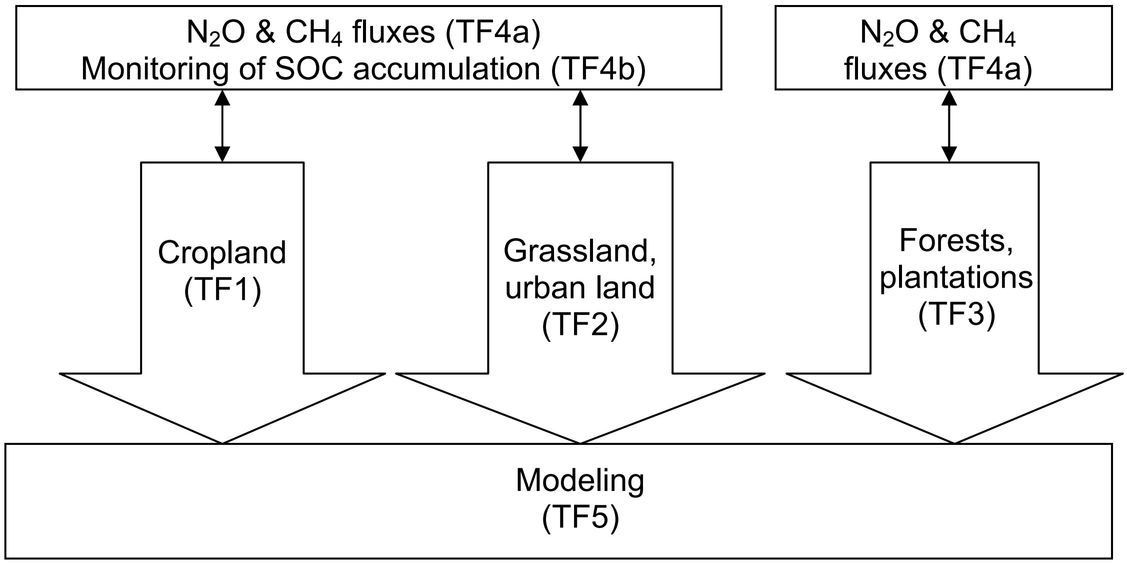


Fig. 1 *General overview of the CASTEC project*

ABSTRACT

In the context of climate change, terrestrial ecosystems are discussed both as a source as well as a possible sink of greenhouse gasses (CO_2, CH_4 and N_2O). The Kyoto Protocol states that carbon (C) sequestration through direct human-induced land use change and forestry activities, limited to afforestation, reforestation and deforestation (art. 3.3) and additional human-induced forest management, revegetation or agricultural activities, which have started after 1990 (art. 3.4), are allowed to be accounted for during the 2008-2012 commitment period. For Belgian terrestrial ecosystems, quantification of the net sequestration potential as well as the effect of possible management changes is therefore crucial. The study of these two items was divided into five different task forces (TF). The first three TF investigated the C sequestration potential in cropland and temporary grassland (TF1), in grassland (TF2) and in forest biomass (TF3). TF4 considered the effect of N_2O emissions from cropland and grassland soils on the total greenhouse gas budget from terrestrial ecosystems and some of the opportunities stable isotopes could bring along for carbon sequestration research. TF5, finally, adapted an existing dynamic vegetation model to elucidate forest C allocation.

The specific aims of TF1 were (1) to quantify soil organic carbon (SOC) stocks in Belgian cropland and their recent evolution, (2) to reduce uncertainties on the knowledge of factors that control SOC levels, and (3) to determine the net C sequestration potential of alternative cropland management. The SOC stocks in Flemish cropland were quantified to be 28162 kt OC (0-1 m) via analysis of a very large SOC dataset. A significant general decrease of these stocks of -0.87 t OC ha^{-1} yr^{-1} (0-1 m) on average was found for the 1990-1999 period. By means of a second SOC dataset, which was generated from a soil survey in the province of West-Flanders, this decreasing trend was confirmed. The impact of management on SOC stocks was evaluated indirectly by correlation analysis and by quantification of the labile fraction of soil organic matter (SOM) with physical fractionation. It was hypothesized that SOC levels in Flemish cropland soils are predominantly determined by management. Moreover, recent SOC stock losses are likely to be at least partly related to reduced application of manure since the establishment of the Manure Action Plan. A relatively high fraction of labile SOM was identified in West-Flanders’ cropland soils, which suggests a substantial part of current SOC stocks to be prone to future loss under current agricultural management. A long-term SOM model (DNDC) was used for simulation of future SOC stock evolutions. Under a business as usual scenario, the measured general baseline SOC loss in Flemish

cropland soils was confirmed to be prolonged, and to stabilize at -0.15 t OC ha^{-1} yr^{-1} (0-1 m) during the Kyoto commitment period. The net C sequestration potential of several management scenarios for that period was evaluated by comparing the simulated SOC stock evolutions under these scenarios with the simulated baseline. None of the scenarios investigated here, were able to counter the loss of SOC, as C sequestration per individual scenario was limited to less than 1% of the national CO_2 emission reduction to which Belgium has committed itself. In conclusion, the contribution of cropland and temporary grassland soils to achieve this aim will be modest.

SOC stocks were determined for grasslands under agriculture for Flanders and Wallonia. Both stocks declined with respectively 11% and 14% between the Kyoto reference year 1990 (respectively 38031 and 56036 kt OC) and 2000 (respectively 33695 and 48408 kt OC) (0-1 m). The decline can partly be explained by a decrease in grassland area but mainly by the decrease in SOC content per ha over these 10 years of 158 to 143 t OC in Flanders and of 150 to 135 t OC in Wallonia. The grassland area declined due to an increase in the maize production and a decrease in livestock. A possible explanation for the significant decrease in SOC content per hectare is the introduction of the Manure Action Plan. However such a plan was not introduced in Wallonia and also there a significant decrease in the SOC content was found. This means that another factor had more impact: the share of temporary grassland within the total grassland area (from 11 to 22% in Flanders, from 3 to 10% in Wallonia). Due to the significant lower SOC content of temporary compared to permanent grassland, the overall mean SOC content in grassland soils will also decline. A few regions in Flanders and Wallonia show a large potential to sequester C in their grassland soils. By a recovery in grassland area and by an increase in SOC, it would be possible to use this potential.

The total SOC stock for Flemish grasslands in nature reserves is estimated at 360 kt OC (0-1 m).

To determine the SOC stocks for grass-covered verges and grass-covered urban areas in Flanders, the area covered with grass had to be assessed first. For verges, the grass-covered area was estimated at 18027 ha, for urban areas at 11890 ha. The total SOC stocks (0-1 m) for grass-covered verges and urban areas come down to respectively 3520 and 1738 kt OC or to 10% and 6% of the total SOC stock of grassland under agriculture in Flanders. However, measures for additional C sequestration in these ecosystems seem to be rather limited.

The influence of management on the SOC content in the soil was investigated by intensive sampling and by setting up a field experiment on three soil types. SOC contents increased in the order: mowing, grazing + mowing, and grazing for all agro-pedological regions in Flanders. Permanent grassland has significant higher SOC content compared to temporary grassland. For the field experiment, the SOC concentration for all treatments on every soil type has increased compared to the starting SOC concentrations. However, no significant effects of the various management treatments introduced, were found. After 2-3 years of experimentation, it was difficult to conclude which management treatment is the best option. Based on the interaction of C sequestration with other greenhouse gases (especially N_2O and CH_4) and the positive effect of both treatments on the SOC content in the soils, using literature data grazed permanent grassland with low fertilisation could be the best combination between environment and agriculture. However, the Flemish implementation of the EU policy described in the Mid Term Review (EU-Directive 796/2004) will lead to disputable policy. On the one hand they are promoting that permanent grassland can be renewed after at least 5 years, whereas on the other hand the amount of organic matter in the soil should be maintained. These two measures lead to contradictory results.

The aim of TF 3 of the CASTEC project was to study the OC sequestration potential of Belgian forest ecosystems and bio-energy plantations. According to the Kyoto Protocol, C sequestered in managed forests is eligible for the calculation of emission reductions under article 3.4. As such, a first subtask under TF 3 was the calculation of the amount of C sequestered in Belgian forests. The total C stock in the living biomass of productive Belgian forests amounted to 60.9 Mt C in 2000 (12.3 Mt in Flanders and 48.6 Mt in the Walloon region), with a mean C-stock of 101.0 t C ha^{-1}. The EFOBEL model was used to predict the C sequestration in the Belgian forests during the first commitment period (2008-2012). The model results indicated a mean annual C uptake of 0.74 Mt C yr^{-1} for the Belgian forests under the 'business as usual scenario'. Starting from the available information for the year 2000, a linear back-calculation towards the year 1990 was performed. The conclusion was that Belgian forests acted as a C sink at a mean annual rate of 0.84 Mt C yr^{-1} during the period 1990-2000.

Afforestation of abandoned land falls within articles 3.3 and 3.4 of the Kyoto Protocol. A short-rotation tree plantation was established on former agricultural land (Zwijnaarde) to study the C sequestration in this type of ecosystem. In total, 24 plots of 400 m^2 were planted with poplar and willow (8 plots each), birch and maple (4 plots each). The mean actual aboveground biomass production after 4 years of tree growth amounted to 1.2, 2.6, 3.4 and 3.5 t DM ha^{-1} yr^{-1} for maple, birch, willow and poplar respectively. Based on the species-specific calorific value of the wood, the total energy content of the biomass was calculated. It appeared that only 1% of all households in Flanders could be provided by short rotation forestry (SRF) biomass energy if 10000 ha of SRF plantations would be established, which is only a minor contribution to the overall energy consumption. When biomass is used as fuel, CO_2 emission from the burning of fossil fuels can be prevented. However, because of the limited area available for SRF plantations in Flanders, the CO_2 emission reduction potential of SRF plantations in Flanders is only 0.09 % of the total CO_2 emissions. Finally, the C balance of the SRF plantation was calculated as the difference between the net C uptake by the trees and the CO_2 released from the soil as soil respiration. Measurements showed that the soil respiration still exceeded the net C uptake by the trees after the first rotation (4 years), resulting in a negative C balance. However, it is expected that the plantation will become a net C sink after 8 to 10 years.

Nitrous oxide is next to CO_2 an important greenhouse gas emitted by arable soils. One of the possibilities discussed in the literature to reduce net greenhouse gas emissions (by C sequestration) from soils is the use of zero or minimum tillage. A measuring campaign on both minimum and conventional tilled soils was established. N_2O losses from the minimum tilled soils (5.27 and 3.64 kg N_2O-N ha^{-1}) were considerably higher than from the conventional tilled soil (0.27 kg N_2O-N ha^{-1}) and this difference was high compared with literature data (2.4 kg N ha^{-1}) for the climatic region under consideration. The higher water content of the minimum tilled soils (slightly more clay and not drained) is presumably the major reason of these higher losses.
In this study also a model based approach (DNDC, FASSET, regression models) was used to obtain robust N_2O emission estimates. Using field measurements (22 year round measurements of N_2O) several regression models were tested. Although the lower emission rates (<10 kg N_2O-N ha^{-1}) were more or less well estimated, the higher emission rates (up to 30 kg N_2O-N ha^{-1}) were clearly underestimated. Therefore the possibilities of a process-based model (DNDC) were examined. In general, the DNDC simulations gave an overestimation of the measured N_2O losses, which decreased with increasing emissions. Croplands were consequently overestimated (7.4 kg N ha^{-1}) while, on the other hand, the similarity between simulated and measured emissions from grassland soils was low. Based on these results, up-scaling of the cropland soils to regional level was scheduled. Results will be discussed in the C sink cluster (OA/00/11). Based on the measured and simulated data (DNDC) an overall emission factor with a 95% confidence interval of 3.77 (-0.20; 7.73) and 7.34 (4.12; 10.56) was calculated, respectively. The FASSET model gave similar N_2O emission results as DNDC for emissions lower than 10 kg N_2O-N ha^{-1} but, in contrast to DNDC, underestimated the higher emissions. To elucidate the different behaviour between both models a sensitivity analysis will be performed (C

sink cluster OA/00/11). Another part of this study was to check the opportunity of stable isotopes as a powerful technique to monitor the evolution of C in the soil. An experiment was conducted but the results proved to be inconclusive due to interferences of N_2O on the $\delta^{13}C\text{-}CO_2$ measurements. To be able to repeat this experiment, the equipment had to be adjusted significantly. Two adjustments were tested (lowering the GC separation temperature and use of a packed instead of a capillar GC column) with samples containing an increasing N_2O/CO_2 ratio for which the latter one was retained.

The goal of TF5 was to simulate future scenarios for C fluxes and woody C stocks of deciduous forests (temperate beech forest). To reach this goal, six step were taken: (1) the development of a C allocation module (CAF), (2) a parameterisation of the model for a temperate beech forest, (3) an evaluation of FORUG on different sites and time scales, (4) a sensitivity analysis, (5) an uncertainty analysis, and (6) a scenario analysis to predict the impact of a global change scenario on the C balance of TF3 ecosystems. The comparison of FORUG simulations with EUROFLUX measurements has shown that the FORUG model is able to predict seasonal patterns of NEE (Net Ecosystem Exchange) for temperate forests at different latitudes in Europe. The new CAF module created the ability to simulate the evolution of the woody C stock of forests. The sensitivity analysis revealed the critical FORUG parameters. A better description of poorly described key processes (e.g. soil respiration) is recommended. The analysis of the output uncertainty resulted in a standard deviation of 0.88 t C ha^{-1} yr^{-1}, which was 24% of the mean value of NEE. The scenario analysis showed a major increase in GPP (gross primary production) and TER (total ecosystem respiration). But the increase of both GPP and TER will be of the same order of magnitude which involves only minor effects on the NEE. The scenario had also a major effect on the wood growth.

This report summarizes the most important findings from the CASTEC project. For more extensive information, reference can be made to the articles summed up in Annex 6 of the report or the different PhDs following from this project.

CHAPTER 1 CARBON SEQUESTRATION IN CROPLAND SOILS

1.1. INTRODUCTION

At present, and from a regional scale perspective, very little is known about the long-term Soil Organic Matter (SOM) balance of current cropland management. Recent interest in Soil Organic Carbon (SOC) sequestration since a couple of years, has led to a mushrooming of SOM research all over the world. Still, there are large uncertainties about the SOC sequestration potential of certain regions, management scenarios, their practical and economical feasibility, and the question even remains whether it is possible to actually measure and verify SOC sequestration. Next to SOC sequestration, maintenance and improvement of (SOM) levels is generally accepted as being the major prerequisite for sustainability of any agro- ecosystem. SOM research therefore has a second important benefit in that it focuses on the general question of sustainability of human management of (agro)ecosystems. The work presented here is aimed at improving our knowledge on the subject of SOM dynamics, particularly with respect to intensive cropland production.

The tasks of TF1 that were formulated in the research proposal are explicitly referenced to in the text as **Task I, II.1, II.2, II.3** and **III**. The outcome of the research of TF1 presented here can be subdivided into 4 main parts (**1.3.1, 1.3.2, 1.3.3, 1.3.4**), which cover the different tasks:

1. Measurement of SOC stock changes and analysis of long-term SOC stock evolution require current and past methodologies for SOC determination to be comparable. A comparison of different analysis methods was carried out (**1.3.1**).
2. A first main objective of the CASTEC research project was to actually quantify present and past SOC stocks in cropland soils and to look into recent SOC stock evolutions (**Task II.2 and II.3**). For right about every country sufficient SOC measurements are simply lacking. We did, however, dispose of a unique massive dataset of topsoil SOC measurements covering the whole of Flanders. The analysis of this dataset is discussed in **1.3.2.1**. This study has been discussed in detail by Sleutel et al. (2003a) and Sleutel et al. (2003b).
In order to verify the outcome of the study presented in **1.3.2.1**, a soil survey in the Province of West-Flanders was conducted (**Task II.1**), which proceeded on two past soil surveys dating back to the National Soil Survey in the 1950-ies, and to the beginning of the 1990-ies. The analysis of the data collected in this soil survey is discussed in **1.3.2.2**.
3. There is a general consensus in soil science that due to chemical and physical stabilization of a large part of the SOM, only the remaining more labile fraction of the SOM is likely to be affected by agricultural management at the time scale of years to

decades. The size of such a "decomposable" OM pool has been estimated to account for 10-40% of the whole SOM. We aimed at using physical fractionation to approximate this proportion of labile SOM in intensively managed cropland soils (**Task II.2**). First, a physical fractionation methodology was tested for its ability to isolate functional SOM pools using soils originating from two long-term field experiments (**1.3.3.1**). Second, this fractionation procedure was applied on a selection of soil samples from the survey of West-Flemish croplands (**1.3.3.2**).

4. The final part of this research was concerned with the prediction of future SOC stock changes in Flemish cropland soils. Localization of favorable regions where the effects of SOC sequestration measures are maximized is crucial in the development of successful agricultural policies for the Kyoto Protocol. Next to management, soil texture, climate and SOC content have the largest influence on SOC dynamics, and they exhibit a strong spatial variability. As a consequence, a modelling approach is required for detailed regional analysis of SOC dynamics. We used the DNDC C and N biogeochemical model (Li et al., 1994) which allowed for a spatial analysis of the results from the regional simulation of different management alternatives and provided a detailed insight into the spatial distribution of the net SOC-sequestration (**Task III**). First, the DNDC model was calibrated both at the site scale as well as at the regional scale. After calibration of crop and soil parameters and of the distribution of SOM in DNDC's OM pools, the model was used to predict SOC stock evolution in Flemish cropland soils.

1.2. MATERIALS AND METHODS

Basic information on the applied materials and methods is given in 3.1, 3.2, 3.3 and 3.4. A complete description and a full discussion of the results of this research are given in Sleutel (2005).

1.3. RESULTS AND DISCUSSION

1.3.1 Comparison of SOC determination methodologies

A large range of soil samples were measured according four different methodologies of SOC determination: the CNS-analyser and TOC-analyser dry combustion methods and the Springer & Klee and the Walkey & Black wet oxidation methods. It can be concluded from this study that the two modern combustion methods investigated closely matched the wet oxidation, and that the traditional assumptions of a 75% efficiency of the Walkey & Black method and a SOM/SOC mass ratio of 2.0 are generally valid for Flemish cropland soils.

1.3.2 SOC stocks in Flemish cropland soils

1.3.2.1 Analysis of a large SOC dataset

Evaluations of SOC stocks on a global scale are often based on assigning a carbon density to each one of a number of ecosystems or soil classes considered using data from soil profiles within these categories. The relative scarcity of SOC data hence always leads to the need of extrapolation when regional or national SOC surveys are to be made, which leads to increased uncertainties on the estimated stocks. Over 190 000 topsoil (0-24 cm) measurements have been made on Flemish cropland in the period 1989-2000 by the Belgian Pedological Service (Vanongeval et al., 2000). With this very large amount of data, covering the entire Flemish cropland area, at our disposal, the necessity to spatially extrapolate data, with its inherent additional errors, was omitted when calculating SOC stocks.

We used this unique large dataset to estimate SOC stocks for the Kyoto baseline year 1990 and to estimate changes in the SOC stock to a depth of 1m during the 1990-1999 period for Flanders. The SOC content data were available as means and standard deviation per community or per part of community (i.e. polygons). A weighted least squares difference regression (WLS regression), which specifically takes the standard deviation on the average SOC content per polygon into account, was used to detect trends in the SOC stock evolution with time. The SOC data of the communities was combined into 27 polygon groups having a normalized distribution, based on soil textural class (using a generalized version of the digital soil map of Flanders) and on spatial location in a GIS system (Arc-View 3.1) (Fig. 1.1). The estimated regression slopes (in %OC yr^{-1}) of each one of the 27 groups, are given in Fig. 1.2.

***Fig. 1.1** Map of 27 clustered polygon groups that were generated based on the dominant soil texture in each polygon and spatial location of the polygons*

Changes in SOC content were recalculated to SOC stock changes to a depth of 1 m by use of a depth distribution model (Sleutel et al., 2003a), by assigning soil bulk densities per textural class and using cropland surface data taken from the National Institute for Statistics. The sum of the SOC stocks of each group individually yielded a total Flemish 1990 stock of 28 162 kt OC. By analysis of this very large dataset we found that a significant decrease in the SOC stocks of Flemish cropland soils has occurred during the 1990-ies. The average loss of SOC was estimated to be -0.87 t OC ha^{-1} yr^{-1} (0-1 m depth). The outcome of this study corresponds well to previously predicted SOC losses from European arable lands by Vleeshouwers & Verhagen (2001). Regions which are relatively poor or rich in SOC stocks were identified, but except for the area surrounding Leuven, no clear reasons explaining these observations were found. The annual net CO_2 emission, which resulted from the estimated SOC decrease, amounted to 1275 kt CO_2 ha^{-1} yr^{-1} during the 1990-ies. Practically, restoration of the SOC lost since 1990 in Flemish cropland soils may well take about 40 years.

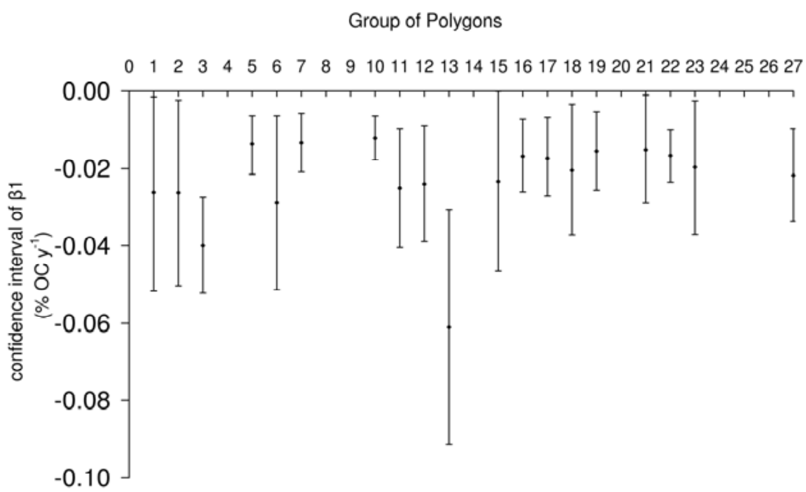


Fig. 1.2 *95% confidence intervals of the estimated regression slope for each polygon group considered*

1.3.2.2 Soil Survey of West-Flemish cropland soils

By revisiting locations that were sampled at the time of the national soil survey (1947-1962) and by Van Meirvenne et al. (1996) (1989-1994), paired observations were obtained of SOC measurements, plough layer depth and soil bulk density for a total of 116 cropland soils at three points in time (Fig. 1.3; Table 1.1).

The analysis of the dataset (see Sleutel et al., 2006 for details) confirmed the decreasing trend for Flanders with an average annual loss of -0.19 t OC ha^{-1} yr^{-1} between 1990 and 2003 for the cropland soils in the province of West Flanders. This result suggests that a clear trend break had occurred, ending the SOC accumulation that took place during several decades before the 1990-ies (Fig. 1.3). Land-use

Table 1.1 *Average soil bulk density and ploughing depth, average difference between the plough layer SOC content in 1990 and 2003-2004 (∆SOC), and average absolute change in SOC stock per soil textural class (Belgian classification)*

Soil Textural class	Soil bulk density (kg dm^{-3})	Ploughing Depth (cm)	∆SOC (%OC)	SOC stock change 2003-1990[a] (t OC ha^{-1} yr^{-1})
Z - sand	1.37	32.8	-0.083	-0.31
S - loamy sand	1.42	32.4	-0.061	-0.18
P - sandy loam	1.47	33.6	-0.078	-0.30
L – silt loam	1.50	32.0	-0.072	-0.18
A – silt/(silt loam)	1.43	29.8	-0.150	-0.49
E – (clay) loam	1.48	30.4	0.085	0.21
U - (sandy/silty) clay	1.20	30.4	0.086	0.20
Whole dataset	1.45	31.9[b]	-0.041	-0.19

[a] paired samples of the SOC stock in the plough layer

[b] average ploughing depth of 2003 assumed to be the one in 1990

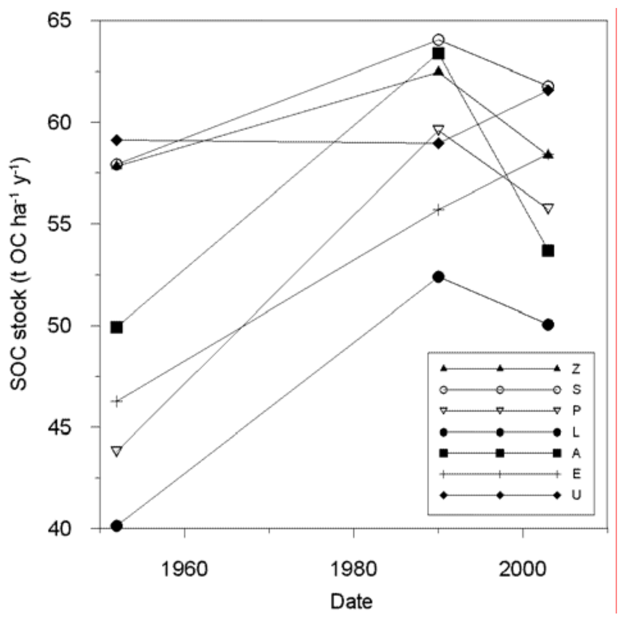


Fig. 1.3 *Average SOC stock in the plough layer of the 1952, 1990 and the 2003 surveys for each one of the 7 soil textural classes considered (Belgian classification: Z - sand; S - loamy sand; P - light sandy silt; L - sandy silt; A - silt; E - clay; U - heavy clay)*

changes, changes in ploughing depth, changes in N fertilization or climate change could not fully explain the recent decrease of SOC levels in these cropland soils. Logically, we then looked at changes in management of OM-input as a possible cause for the loss of SOC. The SOC balance of cropland soils in Europe may very well be predominantly dependent on management. However, no clear reasons could be found for the recently reported SOC stock decreases (Vleeshouwers & Verhagen, 2002; Sleutel et al., 2003). To investigate whether the OM input was indeed of major influence on the observed OC stock change, the amount of effective OC (OC_{eff}) input from incorporated plant residues and animal manure, was calculated. There was a significant positive relation between the changes in the average OC_{eff} input (ΔOC_{eff}) and ΔSOC: $\Delta SOC = 0.88\ \Delta OC_{eff} + 0.06$, which suggests that the observed SOC stock changes were primarily related to shifts in management (Fig. 1.4). This hypothesis is supported by the absence of a good relationship between SOC stocks or SOC stock changes with soil texture, which indicates that in these intensively managed cropland soils texture may have a limited role in SOC dynamics.

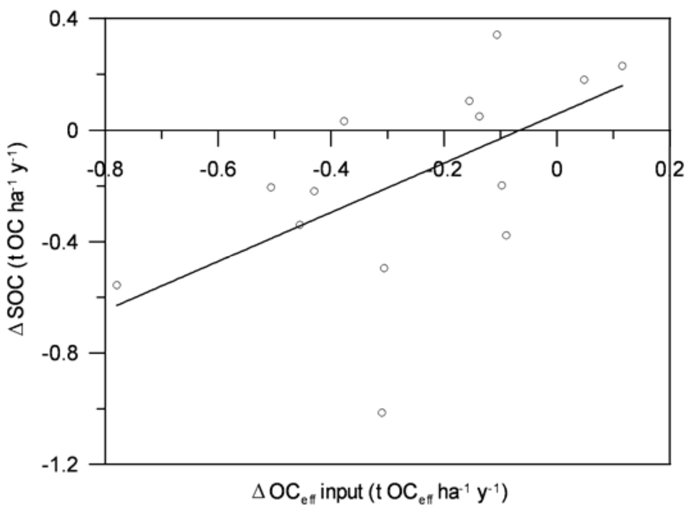


***Fig. 1.4** Relation between the difference in the average annual input of effective OC between 1990 and 2002 (ΔOC_{eff} input) per community and the observed average change in SOC level (ΔSOC) between 1990 and 2003 per community*

1.3.3 Physical Fractionation of SOM in conceptual OM pools

Firstly, a physical fractionation procedure based on the one proposed by Six et al. (2002) was tested for its ability to quantify shifts in SOM pools as a result of management after several decades. Samples from two long-term arable field experiments with each having a control and three manure and fertilizer input treatments were physically fractionated into two free Particulate Organic Matter (free POM), an occluded POM (iPOM) and the silt and clay associated OM fractions. In general, the relative distribution of OM over these fractions had shifted towards relatively more labile OM with increasing OM inputs (Fig. 1.5).

This result corroborates the hypothesis that newly formed OM is largely incorporated into relatively labile OM. An additional simple chemical fractionation of the silt and clay sized OM was inadequate in isolating a "passive" SOM pool. In conclusion, this study demonstrated that changes in the distribution of SOM over functional pools, resulting from management, can be measured after several decades.

The physical fractionation methodology, which was tested using soils from the above described field experiments, was then applied to characterize the SOM of ten of the cropland soils which were taken in the Soil Survey in West-Flanders (see 1.3.2.2). Relative large proportions of free POM (15-44% of the whole soil OM), which is assumed to be "unprotected OM", were found in these soils and a correlation existed between the amount of OC of this pool and the whole soil OC. In contrast to other research reported in the literature, no correlation was found between the silt and clay content and the whole soil OC level, nor was there a correlation between silt and clay content and the amount of silt and clay associated OC (Table 1.2). These observations corroborate the hypothesis of a predominant control of management over the SOC levels of these soils.

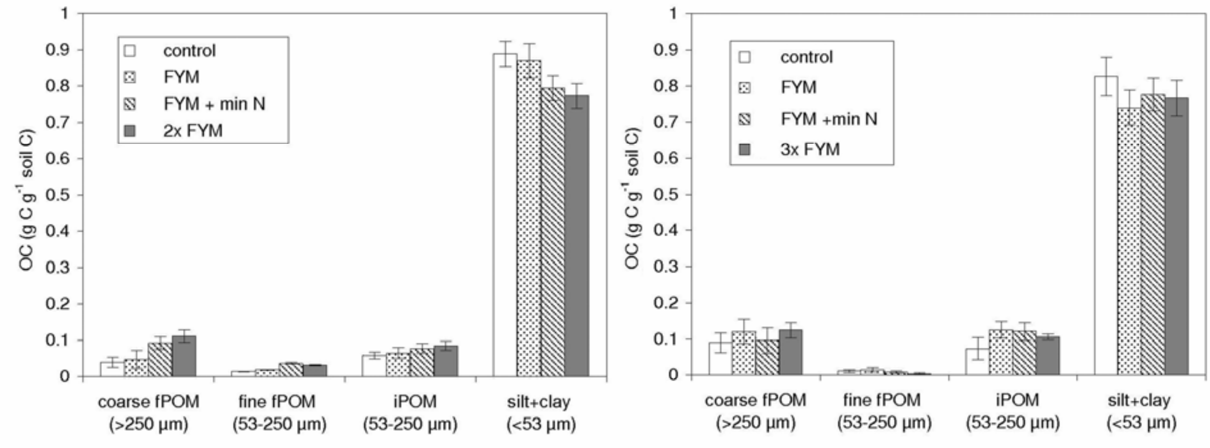


Fig. 1.5 *Relative distribution of the total OC over the coarse fPOM (>250 µm), the fine fPOM (53-250 µm), the iPOM (53-250 µm) and the silt+clay associated (<53 µm) SOC fractions for the different treatments at the Martonvásár (left) and Keszthely (right) sites. Columns represent average values of all repetitions; Y-error bars represent the corresponding standard deviation*

1.3.4 Regional simulation of long-term OC stock changes in Flemish cropland soils using the DNDC model

1.3.4.1 Large scale validation of the DNDC model

Detailed calculations taking into account variations in soil parameters, climate and management are needed to improve estimates of soil C sequestration. Because of the large spatial and temporal variability of Soil Organic Carbon (SOC) dynamics, a modeling approach is required in detailed regional analyses. In all previous regional

***Table 1.2** Pearson correlation coefficients (R) of OC in the size and density fractions considered (g C 100 g-1 soil), the whole soil OC content and the clay and silt content*

	Total OC	OC in free POM[a]	OC in iPOM	Silt + clay associated OC
Clay content	-0.334	-0.608*	-0.024	-0.116
Silt content	0.085	-0.404	0.720**	0.124
Silt+Clay	-0.263	-0.453	0.426	-0.251
Total OC		0.770**	0.187	0.924**
OC in free			-0.110	0.556*
OC in iPOM				0.007

* correlation is significant with P = 0.1 (2-tailed)
**correlation is significant with P = 0.05 (2-tailed)
[a] free POM = coarse POM (>250 µm) + fine inter microaggregate POM (53-250 µm)

or large scale modeling studies the adopted models were at best calibrated or validated to a very limited number of local field experiments when compared to the size of the area which was being studied (e.g. Hungary: Falloon et al., 1998; Canada: Smith et al., 2000). Due to the high spatial resolution of agricultural and soil information sources for this region a detailed modeling approach was possible for Flanders at the community scale. Moreover, contrary to these previous studies, since we disposed of the very large SOC measurements dataset (see 1.3.2.1) covering the whole study area (Flanders), a large scaled model validation could be carried out at the community scale.

First, the DNDC long-term SOC model was calibrated using data from two medium-term field experiments on silty soils (at Juprelle, 12 years and at Gembloux, 31 years) after adjustment of soil parameters to local conditions. The experiments involved mineral fertilizer, cereal straw, green manure and farmyard manure additions. The DNDC model was able to simulate the SOC level evolution of the field experiment at Juprelle very well (Fig. 1.6).

For the field experiment at Gembloux, simulated SOC levels were too high for two treatments involving cereal straw additions, which may imply a flaw in the ability of DNDC to simulate its decomposition under temperate conditions.

Secondly, a model validation was done on a large scale using high resolution, detailed spatial datasets of crop acreages, soil parameters, climatic measurements and a unique massive dataset of SOC stocks for the 1990-1999 period covering the entire study area. Based on a generalized soil map (Sleutel et al., 2003b) weighted averages for clay and silt content and soil bulk density were calculated for each of the 7 soil textural classes. Other soil physical parameters such as bulk density, saturated hydraulic conductivity and the volumetric soil moisture content at wilting point, which are required by DNDC, needed to be estimated using pedotransfer functions. Soil pH_{KCl} data from the Belgian Pedological Service were available as means and standard deviation at the municipality level for the 2000-2003 time period.

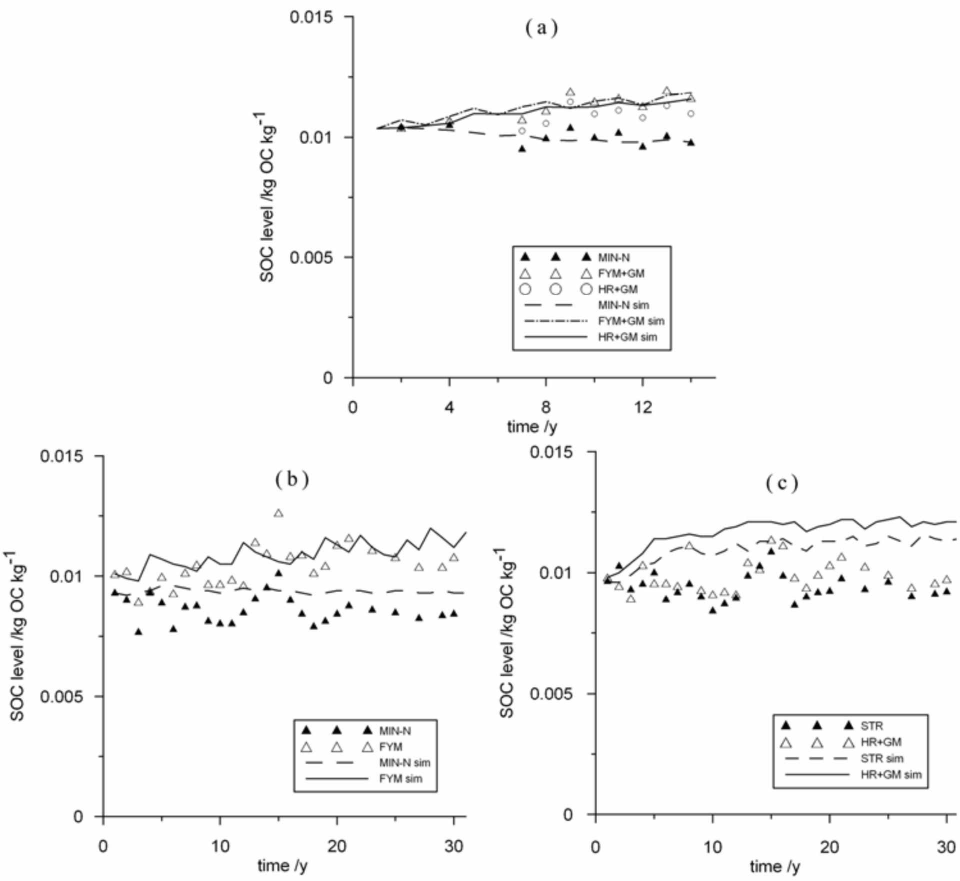


Fig. 1.6 *Evolution of the predicted and measured topsoil SOC content for the three OM treatments at the Juprelle (a) and Gembloux (b & c) field trials (see text). MIN-N: mineral fertilizer; FYM: farmyard manure; FYM+GM: farmyard manure and green manure; HR+GM: harvest residues and green manure; STR: straw*

The 9 predominant crops, constituting 92.3% of the total cropland area, were included for this study. An average cropping surface for the considered time period (1990-2000) for each one of the crops was calculated based on their acreages in 1990, 1993, 1996 and 1999 (NIS, 1990; 1993; 1996; 1999). Average crop yields were estimated from NIS statistics (NIS, 2000; 2001; 2002) and from crop variety field trials. Cropping dates, fertilizer application dates, plant N-uptake and plant C:N ratio's were based on different sources (see Sleutel et al., 2005). Daily measurements of precipitation and minimum and maximum air temperature for the 1990-2000 period from 18 weather stations of the Royal Meteorological Institute were used in the simulations. Every individual community was assigned to the closest climatological station. The version of DNDC used for this study (DNDC 8.1) comprises an interface for regional scale modeling. All spatially different data were linked together using a GIS platform (Arcview 3.1). Simulations were done for 304 different communities in total over an 11 year time period. Per year separate model runs were done for each one of the 9 crops for every community.

The regression line which was fitted to the actually measured SOC data (Sleutel et al., 2005) represented a decrease of 0.48 t SOC ha^{-1} yr^{-1} (0-30 cm). When considering the total 1990-2000 time period, the simulated average SOC stock per ha of all 304 communities decreased by -0.35 t SOC ha^{-1} yr^{-1}, i.e. about three quarters of the measured decrease. In this simulation total SOC was partitioned into different pools using the default DNDC repartitioning at the start of the simulation, i.e. 80% of the SOC in the "humus", 12% in the "humads" and 8% in the "litter" pool. Many authors have indeed conducted a preliminary model run which led to a steady state at the start of the simulation. Instead, in this study, as we disposed of the actually measured SOC stock evolution for the 1990ies, we fitted the simulated average SOC stock evolution for that period by adjusting the initial partitioning of DNDC's SOC pools towards a larger allocation into the labile pools. Although DNDC was able to simulate the SOC stock changes very well when regarding the study area as a whole, the simulated decrease of the SOC stocks was overestimated for communities predominantly having sandy textures and underestimated for communities with silt loam to silt textures. This study thereby pointed out the importance of the evaluation of dynamic SOM models before application on a larger scale.

1.3.4.2 Prediction of soc stock changes a under business as usual scenario and for alternative management options

After calibrating DNDC (1.3.4.1), simulations with DNDC of SOC stock changes during a period (2006-2012) were carried out at the community level for a Business As Usual (BAU) scenario and 7 alternative agricultural management options for SOC sequestration. The main objective was to simulate the net effect of different agricultural management options on topsoil SOC stocks during the Kyoto commitment period (2008-2012). This net effect is obtained as the difference in SOC stock evolution under a Business As Usual (BAU) scenario and under the particular alternative management options.

Climatic data for the 2001-2012 period were generated using the LARS-WG stochastic weather generator. A +0.32°C temperature change was taken into account, generating a future time series of climatic data. Possible future changes in precipitation were not regarded. Several agricultural management options were examined for their carbon sequestration potential in this modelling study (increased surfaces of green manuring, temporary pastures, organic farming and reduced tillage; expanded incorporation of cereal straw, vegetable fruit and garden compost and farmyard manure). All options were assumed to start in 2006. Details on the surfaces and OC input rates associated with the selected management scenario's are summarized in table 1.3.

Using the same crop surface and farming management data which was considered to be representative for cropland production in the study area during the 1990-2000 period a business as usual (BAU) scenario was simulated from 1990 till 2012. The evolution of the predicted average SOC stock (0-30 cm) for this BAU scenario showed that the slope of decreasing trend of the SOC stocks continues after 2000 but the decrease lowers to -0.15 t ha^{-1} yr^{-1} around 2002 and the slope of the decrease remains practically constant thereafter. The simulated baseline net SOC stock change (expressed in t OC ha^{-1} yr^{-1}) was negatively correlated with the initial SOC content (Pearson correlation coefficient = -0.49; P = 0.01), i.e. in communities with higher average SOC levels, the decrease of the SOC stock was larger.

Table 1.3 *Management options for carbon sequestration included in the scenario analysis, the associated surface on which they are practiced and the extra estimated OC input when compared to the baseline scenario*

Management option		Surface (ha)		Total surplus OC input (kt OC yr^{-1})	OC-input per ha (t OC ha^{-1} yr^{-1})	
		baseline	scenario		baseline	Scenario
Green manuring	x 2	23000	46000	46.4	0.128	0.256
	x 3		69000	92.8		0.384
Straw incorporation	50%	0	40175	60.1	0	0.166
Farmyard manure application	20%	47186	62915	30.7	0.292	0.389
	25%		78644	56.5		0.487
Compost application		x^a	x^a	10.1	0.007	0.028
Temporary pastures	x 1.5	47665	71498	-	-	-
	x 2		95330	-		-
Reduced tillage (Southern silty soils)		0	73860	-	-	-
Organic Farming (5% of total area)		0	18112	-	-	-

[a] Since no area figures were available, for the simulation compost was to be spread evenly before all crops except temporary pasture, winter barley and winter wheat (234188 ha)

Secondly, the SOC stock change was positively correlated with the clay content (Pearson correlation coefficient = 0.69; P = 0.01), i.e. the simulated SOC decrease was smaller in the heavier textured soils.

The average simulated SOC stock evolution between 2006 and 2012 for the individual selected management options has been compared to the BAU scenario in

Fig. 1.7. All simulated options resulted in a reduction of the SOC stock decrease compared to the BAU scenario but none of the options led to a stabilization of the SOC stocks. An average annual figure for the whole study area was obtained by dividing the total stock change by 5 (Table 1.4).

In general, this simulation study yielded lower C-sequestration rates than values cited in the literature. This seems to suggest that the total SOC sequestration potential for the study area presented here probably is a safe estimate and in reality better results may be achieved.

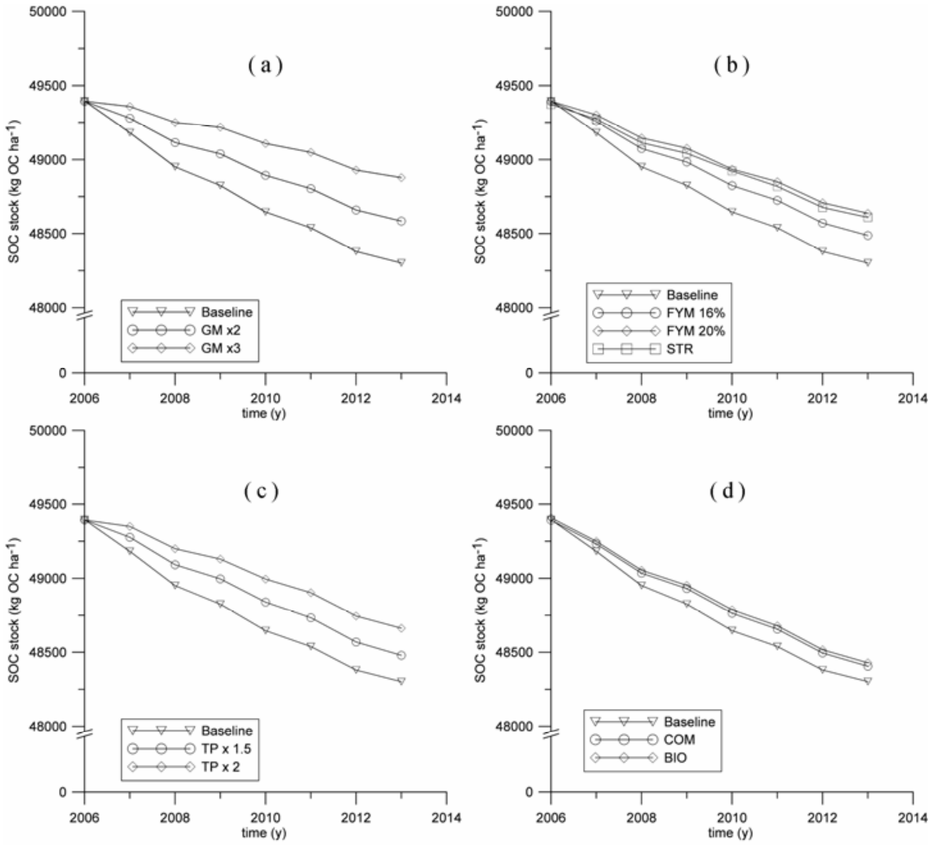


***Fig. 1.7** Evolution of the average simulated SOC stock (0-30 cm) for different management options starting in 2006: (a) Green Manuring (GM); (b) Farmyard Manure application (FYM) and cereal Straw incorporation (STR); (c) Expansion of the Temporary Pasture surface (TP); (d) application of Garden Fruit and Vegtable or Green Waste Compost (COM) and expansion of the Organic Farming area (OF)*

Table 1.4 *Increase of the SOC stock compared to the BAU scenario over the total study area predicted by DNDC for the Kyoto commitment period 2008-2012*

Management option		Total SOC sequestration (kt OC yr^{-1})
Green manuring surface	(current surface x 2)	11.5
	(current surface x 3)	23.1
Straw incorporation	(50% of all cereal straw produced)	13.3
Farmyard manure application	(20% of all N currently applied by animal manure)	7.5
	(25% of all N currently applied by animal manure)	13.1
Compost application	(5% of all N currently applied by animal manure)	4.4
Temporary pastures	(current surface x 1.5)	5.6
	(current surface x 2)	11.0
Reduced tillage	(on Southern silty soils)	0.5
Organic Farming	(expand to 5% of total cropland surface area)	4.3

Overall, the spatial variability of the SOC storage for the selected management options was strongly determined by the current spread of crops and management. The spatial distribution of the net SOC stock change resulting from adoption of the expanded temporary pasture and increased incorporation of cereal straw scenarios are given in Fig. 1.8 as examples.

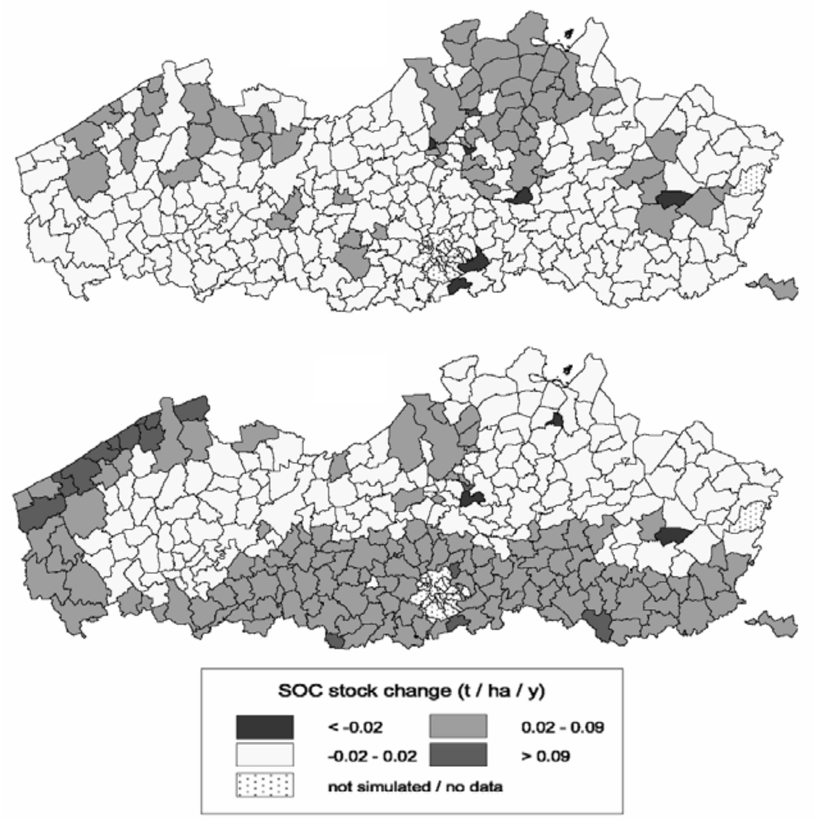


Fig. 1.8 *Spatial distribution of the simulated annual net SOC storage in the 2008-2012 Kyoto commitment Period expressed in t OC ha^{-1} yr^{-1} for different agricultural management options: Temporary Pasture surface x 1.5 (above) and Cereal straw Incorporation (below)*

1.4. CONCLUSIONS AND RECOMMENDATIONS

The most striking outcome of this work is probably that, starting from a research subject aimed at estimating the SOC sequestration potential of cropland soils, we found a general trend of recent SOC losses in Flanders. Further loss of SOC from these soils may have harmful effects such as increased susceptibility to soil erosion, diminished soil fertility and degradation of soil structure. Further research looking into the role of changing management in the process of SOM loss is certainly needed. Such research will be crucial for the development of an agricultural policy for sustainable soil management. A first step would be the establishment of long-term field experiments, which are lacking in Belgium, although they are the single best means for the study of long-term SOM dynamics. Secondly, there is at present a very strong demand for intensive monitoring of SOC stock changes at a large scale, and this will undoubtedly continue in the future. A functional soil monitoring network is one of the priorities in the European Soil Thematic Strategy. The development of such a monitoring network for soil quality in Flanders is a crucial step in this respect.

In conclusion, SOC sequestration in Flemish cropland soils has a limited potential. Up to 1% of the national CO_2 emission reduction, to which Belgium has committed itself by ratification of the Kyoto Protocol, may be achieved. Taking into account a possible increase of N_2O and CH_4 emissions, the net greenhouse gas emission balance may even become negative for some management options. Furthermore some efforts towards SOC storage may have undesired environmental side effects such as increased NO_3^--N leaching, NH_3 emissions or phosphorous accumulation. Given the large scaled recent losses of SOC, the preservation of the SOC stocks still present should be the priority of agricultural policy. The implementation of management practices such as ley-farming, green manuring, incorporation of cereal straw, compost application and organic farming should be further promoted. Preservation of SOM is crucial for the sustainability of agricultural production and for soil quality as a whole.

1.5. ACKNOWLEDGEMENTS

We wish to thank the Federal Science Policy for financing the CASTEC project and for their particular additional support in obtaining the SOC dataset from the Soil Service of Belgium. Many thanks to Mathieu Schatteman, Sofie Schepens and Luc Deboosere for their skillful technical assistance. Thanks to Maria Rosario Prat Roibas, Maria Isabel Moyano Campanario, Ana de la Paz Haza and Benoit Singier for their contribution to this research in preparing their thesis.

CHAPTER 2 CARBON SEQUESTRATION IN GRASSLAND SOILS

2.1 INTRODUCTION

Besides efforts to reduce the emission of their greenhouse gases, countries can also reach their Kyoto targets (during the first commitment period 2008-2012) by enhancing carbon (C) sequestration in soils. Articles 3.3 and 3.4 of the Kyoto Protocol allow C sequestration in terrestrial ecosystems. Especially article 3.4 is of importance for grasslands and allows net changes in greenhouse gas emissions through additional human induced activities in agricultural soils, forest management and revegetation.

As such, C sequestration in agricultural soils becomes a potentially suitable mechanism to ensure compliance with the EU's obligations to reduce their greenhouse gas emissions (ECCP, 2003). The European Climate Change Programme (2003) mentioned an overall potential for greenhouse gas mitigation by agricultural soils via C sequestration up to 60-70 Mt CO_2 yr^{-1} during the first commitment period. This corresponds to 1.5-1.7% of the EU's anthropogenic CO_2 emissions and to 19-21% of the total reduction of 337 Mt CO_2 yr^{-1} to which the EU is committed during that period. Overall, soils of agro-ecosystems are C depleted and represent a potential CO_2 sink. According to Paustian et al. (1997), temperate industrialized countries have the largest capacity to pursue C sequestration strategies in their terrestrial ecosystems. In general, grasslands have more SOC belowground and are far more important for C sequestration in soil (Hougthon et al., 1999; Römkens et al., 1999; Vleeshouwer & Verhagen, 2002) and vegetation (roots, stubble and standing vegetation below mowing/grazing level) than cropland. The longer vegetation period of grasses leads to an additional C allocation belowground (2200 kg C ha^{-1} yr^{-1} for grasses in comparison to 1500 kg C ha^{-1} yr^{-1} for cereals) (Kuzyakov et al., 2000 & 2001). Moreover, grasslands are not ploughed every year and the vegetation is perennial. By this, losses in C through mineralization are limited and there is a continuous input of plant material into the soil.

To determine the possibilities of article 3.4 and to assess the impact of land use and land use change activities, it is necessary to have accurate soil organic C (SOC) stocks for the Kyoto reference year 1990. Establishing a baseline C stock for 1990 is crucial, determining whether agricultural land is a sink or a source for C. In Belgium, this kind of information for grassland agro-ecosystems is absent. Even at European level, there is little information available for grasslands with regard to the total C sequestration potential of measures for increasing SOC stocks in agricultural soils and of the limiting factors (ECCP, 2003).

2.2 MATERIAL AND METHODS

2.2.1 Determination of Soil Organic Carbon stocks

2.2.1.1 Grassland under agricultural management (Flanders + Wallonia)

Three databases were used to estimate SOC stocks in Flanders: the Aardewerk database (Van Orshoven et al., 1988), the database of the Soil Service of Belgium (Hendrickx et al., 1992; Vanongeval et al., 2000) and the database collected by the Department of Crop Husbandry and Ecophysiology (Agricultural Research Centre, Ministry of the Flemish Government, CLO-DFE) during 2001-2003. For the last mentioned database (n = 1500), soil samples were taken at different depths (0-10, 10-30 and 30-60 cm) in each agro-pedological region. Samples were treated and analysed according to Mestdagh et al. (2004a). For the Walloon region, SOC data to a depth of 15 cm (per community) from two laboratories (Réquasud and Michamps) for 1990 and 2000 and the Aardewerk database (Van Orshoven et al., 1988) were used.
Grassland areas, permanent grassland as well as temporary grassland (grazing and mowing plus grazing) are published annually by the National Institute of Statistics (NIS, 1991 and 2001). The temporary grasslands which are only mown are included in cropland calculations. The grassland areas for the different agro-pedological regions are shown in Table 2.1.

Bulk density was measured on the fields sampled in the database collected by CLO-DFE for Flanders as described by Mestdagh et al. (2004b). For the Dunes, bulk density was not measured. Therefore, the bulk density of the Campines was used because both regions have a similar sandy texture. For the Walloon region, bulk density measured in the Aardewerk database (Van Orshoven et al., 1988) per agro-pedological region was used. For calculation of total SOC stocks, SOC data, bulk density and areas are required. First, a model (Mestdagh et al., 2004a) was used to extrapolate all data to the same depth (1 m). The data were compensated for bulk density and multiplied with the respective area data depending on the region. The SOC data from a limited depth (6 cm for Flanders, 15 cm for Wallonia) to 1 m SOC contents could in this way be extrapolated.

2.2.1.2 Grassland in nature reserves (Flanders)

Soil samples were taken in Flemish grasslands in nature reserves because no SOC dataset was available which could be used to extrapolate the SOC data to 1 m. The samples were analysed with the method of Walkley and Black as described in Mestdagh et al. (2004a). For the Dunes, the same SOC content as for the Campines was taken, assuming that both regions have a similar soil texture. For the Silt region, the mean SOC content from Sandy loam region was used to calculate a total stock.

Table 2.1 *Area data (in ha) for grasslands under agriculture for the different agro-pedological regions in Belgium (NIS, 1991 and 2001)*

Agro-pedological region	Grassland area 1990	Grassland area 2000
Campines	53788	52935
Silt region (Flanders)	11771	11412
Polders	21679	20981
Sandy loam region (Flanders)	72451	68724
Sandy region	73458	71795
Dunes	2698	2488
Pasture area of Liege (Flanders)	2327	1864
Pasture area of Liege (Wallonia)	55430	50285
Ardennes	93574	89112
Condroz	51617	49076
Famenne	38062	37441
Campines of Hainault	368	341
High Ardennes	28139	24965
Jura region	22673	23558
Silt region (Wallonia)	58713	57298
Pasture area of the Fagne	12600	11959
Sandy Loam region (Wallonia)	12863	13448

In this way, the stocks for the not sampled regions could be estimated.

The total grassland area in nature reserves in Flanders was 2113 ha in 2002 (Nature report, 2003). Detailed numbers on the area of natural grassland in the different agro-pedological regions are not available (the area is not included within the grassland area of the NIS). An approach of the area of grassland in nature reserves was calculated for a specific agro-pedological region.
Bulk density was measured on the sampled fields. For the Dunes and for the Silt region, the bulk density values of respectively the Campines and the Sandy loam region were used. For those two regions, no bulk density was measured.
For the calculation of the SOC stock in natural grassland, it was not necessary to extrapolate the data. SOC data are available to a depth of 1 m and therefore, the stock can be calculated by adding the SOC contents of the different layers (0-10, 10-30, 30-60 and 60-100 cm). The SOC contents are calculated by the following formula:

Total SOC (0-100 cm) (t C ha^{-1}) = $[\sum (\%SOC/100) * BD_{xi} * x_i)] * 100$ (1)

Where %SOC is the percentage soil organic carbon, BD the bulk density and x_i represents the distance over which was sampled (in this case 10, 20, 30 and 40 at respectively 0-10, 10-30, 30-60 and 60-100 cm). To calculate the total stock, the SOC contents need to be multiplied by the area considered.

2.2.1.3 Verges and urban areas (Flanders)

Soil samples were taken in different types of verges for the different agro-pedological regions and analyzed as described in Mestdagh et al. (2005a). For the Dunes, no samples were taken and the SOC data from the Campines were used, as both regions have a very similar sandy soil texture.

To calculate the SOC stocks in gardens and parks as well as in recreation areas, the mean SOC contents of temporary agricultural grasslands of each agro-pedological region were used (see later) because we assume that the grass in gardens, parks, recreation areas and sport fields is often resown. No soil samples were taken from the Dunes and in the Pasture area of Liege (Mestdagh et al., 2005a).

The length and widths of roads, railways and waterways was determined for every community according to Mestdagh et al. (2005a). To determine the total area of the verges, the length and the width needs to be multiplied. Area data for the categories 'gardens and parks' and 'recreation areas' (camping sites, sport fields, playing grounds, swimming pools and racecourses) are reported by the National Institute of Statistics. A reduction factor of 0.50 was used because not the total surface is covered with grass (Mestdagh et al., 2005a). The bulk densities, measured during the sampling of agricultural grasslands were used to calculate the SOC content for the verges and urban land (Mestdagh et al., 2004b).

Soil samples could only be taken to a depth of 60 cm due to the presence of public facilities (gas, water, electricity). However, for comparison reasons with SOC stocks of other terrestrial ecosystems, calculation to a depth of 1 m is necessary. The SOC content to 60 cm was calculated using equation 1. X_i represents here 10, 20 and 30, the depth over which was sampled at respectively 0-10, 10-30 and 30-60 cm. However, because no data were available for the layer 60-100 cm, it is still necessary to extrapolate data for the layer 60-100 cm as described in Mestdagh et al. (2005a). To calculate the total SOC stock to 1 m depth in a specific region, the SOC content to 1 m (in t OC ha^{-1}) was multiplied by the area (ha) considered.

2.2.2 Management and manageability of SOC stocks

2.2.2.1 Soil sampling in the different agro-pedological regions (Flanders)

The following management practices were considered: grazing (4 Livestock Units, LU ha^{-1}), mowing (number of cuts depending on the amount of N fertilizer used and the weather conditions), mowing + grazing (in function of the growth rate of the grass, the stocking rate and supplementary feeding), permanent (grasslands which are at least 5 years old (EU-Directive 796/2004) and temporary (grasslands of 4 years old or younger, or fields which are resown every or second year). In the different agro-pedological regions in Flanders, grasslands were sampled and the soil samples were analyzed with the method of Walkley and Black for OC concentration (Mestdagh et al., 2004a). Bulk density was measured according to Mestdagh et al. (2004a). SOC contents to a depth of 60 cm were calculated using equation 1, where in this case x_i is 10, 20 and 30 at respectively 0-10, 10-30 and 30-60 cm.

2.2.2.2 Field experiments

The experimental fields were located on a sandy loam soil (Merelbeke), clay (Watervliet) and sand soil (Geel) in Flanders. All fields on the three locations have since decades been used as cropland. On the sandy loam soil, the experiment was started in spring 2001, for the clay and sand soil in spring 2002.
On the grassland for agricultural use, four factors were applied: the level of fertilisation, the cutting regime, the grass mixtures and the removal of the hay (Mestdagh et al., 2005b). On the verges, three different factors were applied. The factors were the number of cuts, the grass mixture and the removal of the hay (Mestdagh et al., 2005b).
Before the experiment was set up, soil samples were taken at the different locations at 0-10 cm. In October 2004 on the three locations, the different plots were sampled again. The samples were analysed according to Mestdagh et al. (2004a).

2.3. RESULTS AND DISCUSSION

2.3.1 Grassland in agriculture (Flanders + Wallonia)

The mean SOC contents for the different agro-pedological regions declined between 1990 and 2000 (Table 2.2), except for the Polders and the Dunes where the SOC content did not change. The mean SOC content varied with soil type, with the lowest content for the Silt region and the highest for the Polders for Flanders and for Wallonia the Silt and Sandy Loam region (lowest) and the Pasture Area of Liege (highest). For 1990 and 2000 the SOC stock was estimated at respectively 38031 and 33695 kt OC for Flanders and at respectively 56036 and 48408 kt OC for Wallonia. There is an overall decline in SOC stock in Flanders between 1990 and

2000, but especially in the north and the east of Brussels and in the eastern Campines. The northeastern and western parts of Flanders have the largest SOC stocks. These regions are characterized by a high grassland density and also have a history of relative intensive farming. Also for the Walloon region, there is an overall decline in SOC stocks. Especially the Pasture Area of Liege and the Silt region have decreased the most. The largest stocks are for the Ardennes and for the Pasture Area of Liege, despite its large decrease. The total SOC stocks for the different agro-pedological regions are shown in Table 2.2.

The observed decrease in SOC stocks in grasslands between 1990 and 2000 can partly be explained by a decrease of 3% and 4% in total grassland area for respectively Flanders and Wallonia (NIS, 1991 & 2001). The grassland area for Flanders in 1990 was 238173 ha while in 2000 it was only 231450 ha. For the Walloon region, the grassland area declined from 374083 ha in 1990 to 357483 ha in 2000. An important reason for the decrease in grassland area was the enormous increase in maize cultivation. The area of maize production in Flanders and Wallonia increased with respectively 37% and 32% between 1990 and 2000 (NIS, 1991 & 2001). This increase was due to the fact that the cultivation of maize was/is frequently used for cattle feeding, the cultivation is fully mechanized (low cost) and the government supports and subsidizes maize production. In addition there was a decrease in livestock between 1990 and 2000 (NIS, 1991 & 2001). The decrease in total SOC stock can, however, mainly be explained by a decrease in SOC content per hectare of 158 t OC ha^{-1} in 1990 to 143 t OC ha^{-1} in 2000 in Flanders and of 150 t OC ha^{-1} in 1990 to 135 t OC ha^{-1} in 2000 in Wallonia. One of the possible explanations for the significant decrease in SOC content per hectare is the approval of the Manure Action Plan (MAP) in Flanders, by which farmers became strongly limited in the application of N via chemical fertilizer and manure (before 1990 the amount was unlimited). As such, the MAP may have contributed indirectly to a decrease in organic matter (OM) because there was no longer a large supply of OM to the soil. As for Wallonia, such an action plan was not introduced but also here we found a decline in SOC content. This means that other factors had more impact on the mean SOC content in Flanders and Wallonia than the decreased supply of OM to the soil. Both in Flanders and Wallonia, the share of temporary grassland within the total grassland area has increased; from 11% to 22% for Flanders and from 3% to 10% for Wallonia between 1990 and 2000. This event is likely to have a larger impact on the decrease in SOC content per agro-pedological region. More temporary grassland contributes to the total grassland area, to the mean SOC content and the total SOC stock in 2000 so it is expected that, because of their significant lower SOC content (see 3.4), the mean SOC content per agro-pedological region will also decline. Besides, we want to remark that the database does not exists of paired data (fields sampled in 1990 are not necessarily the ones in 2000). Consequently, the changes calculated give a general trend and magnitude of change.

A few regions in Flanders show a large potential to sequester C in their grassland soils. Especially for the regions north and east of Brussels and the eastern Campines, there is a potential to restore the SOC stock. By a recovery in grassland area and by an increase in SOC content, it would be possible to use this potential. The regions in the north and east of Brussels have a sandy loam, sand and silt soil texture and are lying in the agro-pedological regions which knew a large decline in mean SOC content over 10 years (Table 2.2). But overall, the large increase in temporary grassland within the total grassland area has led to an overall decline in the total SOC stock (to 1 m depth). For Flanders most grassland was lost in the Sandy Loam and Sandy region whereas, for Wallonia nearly all regions lost a lot of grassland area. In Wallonia, the Silt region, Sandy Loam region and the Pasture Area of Liege had the largest decreases in SOC content and because also there the loss in area occurred, there is a large potential for these regions to increase their SOC stocks by a recovery of grassland area but mainly by an increase in SOC content. For the regions with lower mean SOC contents, the increase needs management practices which build up soil C such as grazing and permanent grass. These practices should lead to a higher input of organic matter into the soil and (or) a decrease of the decomposition rate of soil organic matter. The capacity of soils to store C is finite and sequestration will go on to saturation at a new equilibrium which in most cases is under the original C level. When the required management is no longer maintained, the new soil C levels will rapidly drop again. C sequestration has, next to its potential to mitigate the CO_2 concentration, also an improving effect on soils. Overall, with higher SOC levels, there is an increasing capacity to bind

Table 2.2 Mean SOC contents (t OC ha^{-1}), standard errors (se) and total SOC stocks (kt OC) for the different agro-pedological regions for 1990 and 2000 (based on the data from the Soil Service of Belgium, Réquasud and Michamps). The results between parentheses are based on a limited number of samples with a large error and must be treated with caution.

Agro-pedological regions	1990		2000		1990	2000
	content	se	content	se	Stock	Stock
Campines	163	3	142	3	8801	7579
Silt region (Flanders)	126	4	111	4	1505	1309
Polders	211	11	213	13	4637	4327
Sandy Loam region (Flanders)	150	3	132	3	11003	9599
Sandy region	148	3	134	3	11034	10006
Dunes	(249)	(25)	(253)	(30)	(609)	(568)

Pasture area of Liege (Flanders)	181	—	160	—	442	307
Pasture area of Liege (Wallonia)	203	5	168	3	11592	9025
Ardennes	151	4	145	3	13832	13128
Condroz	148	2	143	4	7566	7022
Famenne	135	3	131	3	5167	4939
Campines of Hainault	(148)	—	(130)	(55)	(54)	(59)
High Ardennes	182	9	167	4	5225	4157
Jura region	140	9	130	7	3118	2916
Silt region (Wallonia)	104	1	82	2	6102	4519
Pasture area of the Fagne	160	2	144	6	2041	1689
Sandy Loam region (Wallonia)	104	—	71	4	1339	956

nutrients, a better soil structure and aeration, larger ability to store water and a higher protection against erosion. All factors are also influencing crop/grass growth positively (win-win situation).

2.3.2 Flemish grasslands in nature reserves

The mean SOC content (1 m) for natural grassland and the total stocks for the different agro-pedological regions are shown in Table 2.3. The total stock for Flemish natural grassland is estimated at 360 kt OC. The value of the stock for the silt region and the Dunes is put in parentheses because this value is an estimation calculated on basis of data from respectively the Sandy loam region and the Campines.
The Polders have the highest mean SOC content. This is due to the fact that nearly all fields sampled in the Polders were located in the flooding area of a river or stream and were flooded for a large part of the year. Because of the anaerobic conditions for long periods of time, the OM is decomposed at a slower rate. There is a trend for higher mean soil organic carbon contents in natural grasslands which can partly be explained by their permanent character.

2.3.3 Flemish grass covered verges and urban land

The total area for grass covered verges along roads comes down to 7413 ha, along waterways to 9542 ha and along railways to 1072 ha. This gives a total area of verges of 18027 ha. The mean SOC contents for grassy verges to 1 m depth are shown in Table 2.4. The total SOC stock to 1 m for grass covered verges along roads is to 1438 kt SOC, for waterways at 1873 kt SOC and for railways at 209 kt SOC. The total SOC stock for the three types of grass covered verges was estimated at 3520 kt SOC. The calculated total grassy area of Flemish verges in this study accounts for 8% of the total grassland area in Flanders. This represents 10% in comparison to the total SOC stock in grasslands. The potential for additional C sequestration and mitigating atmospheric CO_2 for verges along roads is likely to be small due to i) the increasing pressure of traffic on the verges; ii) road maintenance and maintenance for public facilities (gas, water and electricity) and iii) the limited possibilities in changing their management. Along railways, the possibilities for a changing management are also very limited because half of the total surface consists of woody vegetation and the grassy parts are mown as often as necessary to assure a clear view for train drivers. However, the above-mentioned problems do not often occur in verges along waterways and therefore, any potential for C sequestration should be focused on these verges. A possible measure to enhance C sequestration in these verges is to increase the frequency of grazing. Only a very small percentage of the verges along waterways are grazed, the remaining part is mown. In general, grazing enhances the SOC stock (Hassink & Neeteson, 1991; Schuman et al., 1999). Planting trees in all three types of verges could also lead to an additional C sequestration mainly in the aboveground biomass but less in the soil.

Table 2.3 *Mean SOC contents (t OC ha^{-1}), standard errors and total SOC stocks (kt OC) for natural grassland to 1 m for the different agro-pedological regions*

Agro-pedological region	mean SOC content	se	SOC stock
Campines	192	18	92
Silt region	—	—	(15)
Polders	253	13	49
Sandy loam region	142	8	92
Sandy region	165	9	107
Dunes	—	—	(5)

The total area covered with grass for the category 'gardens and parks' was estimated at 9530 ha and for the category 'recreation areas' at 2360 ha. The mean SOC contents to 1 m, used to calculate the total SOC stocks, are shown in Table 2.4. For the grassy parts of the category 'gardens and parks', a total SOC stock of 1392 kt SOC was calculated and of 346 kt SOC for the category 'recreation areas'. For the grassy urban area, the total area accounts for 5% of the total grassland area and the total SOC stock accounts for almost 6% of the total SOC stock for Flemish grasslands. Management measures to enhance C sequestration in the urban areas are quite difficult because i) gardens are private property and cannot easily be subjected to laws and ii) parks and recreation fields keep their grasses very short. However, the IPCC (2001a) mentions a management strategy for urban and peri-urban land. Tree planting, waste management and wood production could lead to a net annual rate of change in C stocks of 0.3 t C ha^{-1} yr^{-1}.

Table 2.4 *Mean SOC contents and standard errors (t C ha^{-1}) for temporary grasslands and grass covered verges in Flanders to 1 m depth for the different agro-pedological regions*

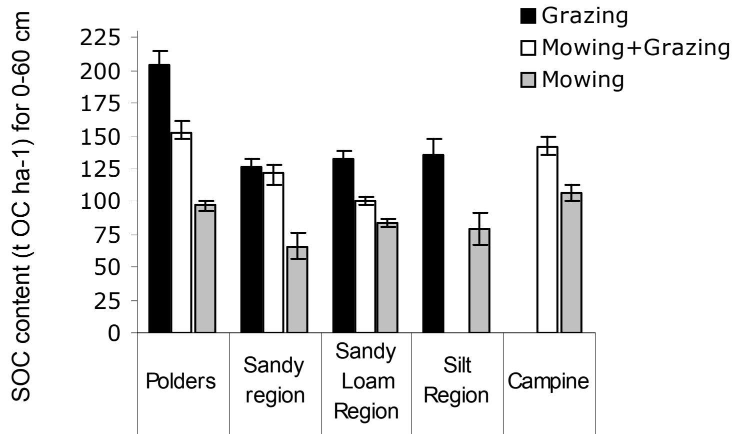
Agro-pedological region	Temporary grassland		Grass covered verges	
	Mean SOC	se	Mean SOC	se
Campines	176	10	190	25
Silt region	106	15	163	26
Polders	135	5	183	22
Sandy region	162	14	215	13
Sandy loam region	129	5	186	26
Pasture Area of Liege	—	—	262	17

2.3.4 Management and manageability of SOC

The intensive sampling in Flanders made it possible to assess mean SOC concentrations for different management options, grassland age and for different soil types. Management options for building up C stocks in the soil are those which either increase the input of OM in the soil and/or decrease the rates of decomposition of the organic material.

The SOC content was highest for grazing and lowest for mowing alone (Fig. 2.1). This trend is found for all agro-pedological regions. Higher amounts of SOC in the soil under grazing have also been found by other authors (Hassink & Neeteson, 1991; Van den Pol Van Dasselaar & Lantinga, 1995; Schuman et al., 1999). The higher SOC concentrations under grazed grasslands can be explained by a higher return of organic matter to the soil under grazing. This is partly due to higher losses of plant material when compared to cutting, leaving behind of faeces (Hassink & Neeteson, 1991); redistribution of C within the plant/soil system and animal trampling which enhances physical breakdown, soil incorporation and the rate of decomposition of litter (Schuman et al., 1999). Under mowing, a large part of the primary production is exported from the field as hay or silage and a smaller part of the assimilated C is translocated to the roots and stubble (Van den Pol Van Dasselaar & Lantinga 1995). Kuikman (1996) claims that, if frequent cutting results in translocation of C from the roots to the shoots, there will be less C present belowground with mowing. There are more aboveground losses of C, more C is invested in the shoots and the C in reserve in the roots is mobilized to compensate for the aboveground losses. Without cutting the grass, the distribution of assimilates would depend on the physiological stage of the growing plant. Wolf & Janssen (1991) found that 40% of the total dry matter was distributed to the roots and stubble residues for mown grasslands, whereas for grazed grasslands, 47% of the total dry matter was distributed to roots and stubble residues. However, according to Schuman et al. (1999), heavy grazing could lead to more soil C than light grazing.

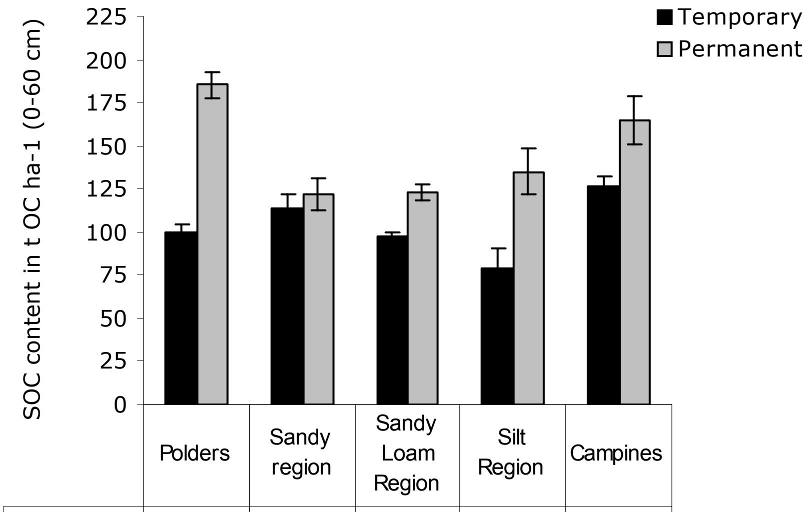
But they noticed that increased grazing pressure could be expected to reduce soil carbon. Grasses will respond to defoliation by increasing C allocation to new leaves while decreasing allocation to roots. So frequent grazing could result in lower C inputs into the soil from the roots. These results are confirmed by the IPCC (2001a) and by the simulation models of Parton et al. (1987). The influence of mowing + grazing has, to our knowledge, not been investigated. Nevertheless, as could be expected, intermediate values between grazing and mowing for the mean SOC content were found for the mowing + grazing treatment.



	Polders	Sandy region	Sandy Loam Region	Silt Region	Campine
Grazing	204	127	132	135	
Mow ing+Grazing	152	122	100		142
Mow ing	97	66	84	79	107

Fig. 2.1 *SOC contents (t OC ha^{-1}) and standard errors to 60 cm for three management treatments in different agro-pedological regions in Flanders*

Figure 2.2 clearly shows that permanent grasslands contain significantly more SOC than temporary ones. Ploughing and resowing of the soil increases soil organic matter mineralization, released as CO_2 in the atmosphere.



	Polders	Sandy region	Sandy Loam Region	Silt Region	Campines
Temporary	100	114	97	79	126
Permanent	185	122	123	135	165

Fig. 2.2 *Mean SOC contents (t OC ha^{-1}) with standard error (se) to 60 cm for temporary and permanent grassland in Flemish agro-pedological regions*

Management measures which can lead to an increase in soil organic matter (such as grazing or higher N fertiliser application) can nevertheless also lead to higher emissions of CO_2 equivalents (as N_2O and CH_4). Moreover, these interactions can lead to possible negative impacts on the environment (e.g. eutrophication of drinking water). Grazing for example can store more C belowground than cutting but has also higher emissions of N_2O and CH_4. Due to their higher 'Global Warming Potential' (1 kg N_2O = 310 kg CO_2, 1 kg CH_4 = 21 kg CO_2) a considerable amount of CO_2, which was extra sequestered by grazing, can be lost as CO_2 equivalents in the atmosphere (Mestdagh et al., 2003).

2.3.5 Field experiments

Overall, SOC concentration has increased for all treatments for every soil type compared to the starting SOC concentrations but there was no significant effect of the soil type. On the light clay soil (Watervliet), there was a significant increase in SOC concentration for nearly all treatments compared to the starting values (35% higher for agricultural grassland and 15% for verges). On the fine sandy soil (Geel), the increase was much smaller (2% and 4% for respectively agricultural grassland and verges). Nevertheless, SOC is likely to increase when cultivated soils are turn within permanent grassland soils or when there is a management for high grass productivity (Post & Kwon, 2000). After 3 years of experiment on the sandy loam soil (Merelbeke), agricultural grassland management systems gave an overall increase of 14% compared to original concentrations and the verges showed a significant C storage explained by the high belowground biomass production compared with grassland for agricultural use (Iantcheva et al., 2004). For all locations, no significant differences between and within the treatments were found which is in agreement with Smith (2004). He found that a detectable change in SOC takes 3-15 years (or longer) depending on the increase in C input and the detection power of background SOC. However, after 2-3 years of experiment, it is difficult to conclude which management treatment is the best option. Therefore, the observation period must be increased and further monitoring is essential.

2.4. CONCLUSIONS AND RECOMMENDATIONS

SOC stocks for both Flanders and Wallonia have decreased between 1990 and 2000 with respectively 11% and 14%. This was partly due to a decline in grassland area but mainly to a decrease in SOC content (t OC ha^{-1}). The total SOC stocks could be increased or maintained by a recovery of the grassland area (lost as cropland or urban land), by an improved management of the verges and by an increase in the SOC content of grassland soils by encouraging management practices which enhance C sequestration such as grazing and permanent grassland. Permanent grassland is described in the EU-Directive 796/2004 as grassland that is not incorporated in crop rotation for at least 5 years. The Flemish Government follows this directive and postulates in the cross compliance of the Mid Term Review: the farmer can only cultivate permanent grassland if it is compensated by an equivalent area of permanent grassland which is maintained for at least five years. However, because the storage of OM in the soil is a very slow process (see also field experiments), enormous amounts of SOC are lost every time the grass is ploughed. OC in the soil with ploughing is lost faster and easier than the reverse process of C sequestration. It is impossible to recover the amount of C lost by maintaining grassland 5 years. Besides, in their good agricultural and environmental practices, the Flemish Government mentions that the OM in the soil must be maintained by suitable practices. However, as the cross compliance is now described, more OM will be lost. Contradictions are found within the Flemish implementation of the Mid Term Review and one should be aware that this could lead to disputable policy. Finally, experiments which investigate the impact of management on the change in SOC concentration should last longer than the project period. It is therefore recommended to continue this experiment as it is expected to find detectable changes in SOC concentrations within 10 years.

The atmosphere, soil, vegetation and the management applied (e.g. grazing, mowing, fertilizer use) all interact with each other. Therefore, management measures can not be judged on their potential to store C without taking the fluxes of other greenhouse gases (especially N_2O and CH_4) into account. Grazing should enhance the C sequestration but it also has a high emission of N_2O and CH_4, a higher NO_3^- leaching potential which is also limited to preserve drinking water and to resist the eutrophication of water and soil compared with mowing. Another contrasting factor is the application of fertilizers. Different studies have investigated the influence of fertilizer use on the sequestration of C in the soil but they are not unanimous. Fertilizers give a lot of advantages for the farmers but they have a strengthening effect on the emission of N_2O, the leaching of NO_3^-, production of CH_4 (more N, more production so more animals on the grassland) but it also leads to a decrease in the biodiversity of ecosystems and its role for C sequestration is not clear. On the other hand, permanent grassland has fewer emissions than temporary grassland. Therefore, grazed permanent grassland with low fertilisation could be the best combination between environment and agriculture.

As for the future, grassland areas should be recovered and permanent grassland should be promoted. Besides, organic carbon contents should be increased by improved management, verges along rivers and canals could be grazed and trees should be planted for additional C storage. Also one needs to be careful when formulating measures which enhance C storage as some of them can offset the benefit. Finally, if we continue with the business as usual scenario, further losses in SOC will occur. The implementation of the cross compliance policy of the Mid Term Review for permanent grassland needs better relation with research results.

2.5. ACKNOWLEDGEMENT

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CHAPTER 3 CARBON SEQUESTRATION IN FOREST ECOSYSTEMS

3.1. INTRODUCTION

In accordance with the aim of Task Force 3 of the CASTEC project, this chapter describes the carbon (C) sequestration potential of Belgian forest ecosystems and bio-energy plantations.

3.2. BELGIAN FOREST ECOSYSTEMS

According to the Kyoto Protocol (KP) and the following Conference Of the Parties (COP) agreements, carbon sequestered in managed forests is eligible for the calculation of emission reductions under the KP Article 3.4. As such, the first subtask of Task Force 3 concerned the study of the carbon sequestration in Belgian forest ecosystems. Reporting to the United Nations Framework Convention on Climate Change (UNFCCC) however requires clear definitions on forests. Therefore, a shortlist is given of necessary information for correct interpretation of the results:

- minimum area for forest (Flanders : 0.5 ha, Walloon region : 0.3 ha,), canopy closure (Flanders : 20%, Walloon region : 10%), dendrometric measurements (minimum diameter at breast height of 7 cm) and volume calculations were taken from the regional forest inventories (FI) (FGAD 2001, Lecomte & Rondeux 1994);
- only productive forests were included in the calculations (Table 3.1). Forest area was derived from the updated forest map for Flanders and from the regional forest inventory for the Walloon region. For Flanders, the area of productive forests was calculated as the total forest area minus the forest reserves;
- Afforestation, Reforestation and Deforestation (ARD) activities were derived from consecutive forest mapping or inventorying;
- forest management : business as usual (BAU) scenario;
- information on harvested wood was derived from the IVANHO database (see Appendix 1) and the forest inventory for Flanders and the Walloon region respectively;
- annual growth increment Flanders : see Appendix 2;
- rotation length : see Appendix 3 for Flanders, Perrin et al. 2000 for the Walloon region;
- area per species was derived based on total forest area and the percentage of each species basal area in the total forest basal area (FGAD 2001) (Appendix 4).

Table 3.1 *Area of productive forests used in the carbon calculations (2000)*

Forest type	Flanders (ha)	Walloon (ha)
D (deciduous)	74750	233600
C (coniferous)	54540	225100
MD (mixed deciduous)	6800	
MC (mixed coniferous)	8610	
Total	**144700**	**458700**

3.2.1. Carbon stock in the above- and belowground Belgian forest biomass, in the year 2000 (Van Camp et al., 2004; Vande Walle, et al. 2005)

Qualitative analysis of different biomass assessments led us to the conviction that the carbon stock in the Belgian forests can be reasonably approximated by a multiplication of the standing stock with median values of biomass expansion and conversion factors (BEFs), critically selected from European literature (Appendix 5) as advised by the IPCC (2003). As such, the mean total C-stock in the living biomass of productive Belgian forests amounted to 60.9 Mt C in the year 2000 (12.3 Mt in Flanders and 48.6 Mt in the Walloon region). The overall mean C-stock amounted to 101.0 t C ha^{-1}. This value was in the higher range of values reported for the neighbouring countries, mainly due to a high mean growing stock in the Belgian forests (261.9 m^3 ha^{-1}). When calculations were based on the minimum BEF values, a total C-stock of 42.8 Mt C was found, while the maximum BEF values gave a total C-stock of 83.5 Mt C in the living Belgian forest biomass. Carbon stock in the standing dead wood of the Flemish forests amounts to only 1.4% of the total carbon stock (no information for the Walloon region).

Wood density values reported in literature appeared to introduce the largest variability in the assessment of the carbon stock in the Belgian forest biomass. Additional measurements of wood densities in Belgian forests could help to reduce the uncertainty related to this factor. Because of the time-consuming and destructive character, the establishment of new allometric equations does not have the highest priority for improving the carbon stock assessment in the Belgian forests. Accurate volume estimations however are critical for good biomass predictions with the available BEFs. Therefore, it is advised that regional forest inventories should include a well-defined methodology and accuracy assessment on their volume calculations. As the median C-content value for all species except beech was equal to the default IPCC-value of 50% carbon in dry matter, it seems appropriate to use this value for future calculations.

A Monte Carlo based error propagation revealed an uncertainty (GPG, 2003) of 15% on the total carbon stock of 60.9 Mt C. The uncertainty was larger for coniferous (39% and 28%) than for deciduous forests (18% and 20%) for the Flemish and the Walloon region respectively. Table 3.2 summarizes the different input uncertainties.

***Table 3.2** Uncertainties associated with each input variable in the carbon accounting (D: deciduous; C: coniferous; M: mixed)*

Variable	Forest type	Uncertainty (%) F - W
Volume (m^3)	D including MD	2.6 - 1.4
	C including MC	3.3 - 2.0
Area (ha)	D including MD	5.0 - 1.0
	C including MC	5.0 - 0.8
BEF3	D including MD	10.0 - 15.0
	C including MC	28.0 - 33.0
Carbon content	D including MD	9.8 - 14.0
	C including MC	10.0 - 29.4

3.2.2. Carbon sequestration in the Belgian forest biomass during the first commitment period (2008-2012)

The EFOBEL model (Perrin et al., 2000) was used to predict the future C-sequestration in the Belgian forests. Information on wood volume and forest area, divided over species and age classes, for every grid cell of the Belgian forest inventories served as input for the model. Parameters in the model are annual growth increment revolution, latency and forest conversion. The EFOBEL model gives solid wood volume (m^3) for the projection year as output. The business as usual scenario implies no forest conversion, or forestation. For the Walloon region, model runs were also applied for forest conversion from spruce to Douglas fir and for lengthening/shortening the rotation of spruce with 10 years.

The 'business as usual' scenario resulted in a total C-stock in the productive forests of 67.5 Mt C in the year 2008 and 71.2 Mt C in the year 2012. This means that the Belgian forests will act as a sink for carbon during the First Commitment period, with a mean carbon uptake of 0.74 Mt C yr^{-1}. The obliged CAP on this value (only 15% of this value may be taken into account) results in an actual carbon sequestration of 0.11 Mt C yr^{-1}. This value largely exceeds the CAP of 0.03 Mt C yr^{-1} set for forest management options under the article 3.4 of the Kyoto Protocol.

3.2.3. Carbon stock in the Belgian forest biomass in the reference year 1990

The year 1990 is the reference in relation to discussions concerning the Kyoto Protocol. Carbon stock calculations for the living biomass in this reference year were based on the median BEF values used for the calculations of the C-stock in 2000. The availability of information related to growth increment, wood harvest and ARD activities was very limited. Consequently, no statistics were considered acceptable for precise calculations.

Starting from the available information for the year 2000, a linear back-calculation towards the year 1990 was performed. In the period 1990 to 2000, Belgian forests acted as a carbon sink at a rate of 1.24 Mt CO_2 eq. yr^{-1} and 1.84 Mt CO_2 eq. yr^{-1} for Flemish and Walloon forests respectively. This implies that the mean annual carbon uptake by the Belgian forests amounted to 0.84 Mt C yr^{-1}, which is slightly higher than the 0.74 Mt C yr^{-1} predicted for the First Commitment period (2008-2012). As stated above, the calculation of the 1990 carbon stock is based on a number of assumptions concerning forest area, wood harvest and annual growth increment, and should therefore be considered with some caution. More precise calculations will be possible when the second Flemish forest inventory will be completed, as this will provide more detailed (species-specific) growth increment information. Future calculations will be more precise when more exact information on both harvest and ARD activities will be available.

3.2.4. Carbon allocation to the forest compartments (Flemish case study)

Although it was concluded by Johnson & Curtis (2001) and Heath et al. (2002) that forest harvesting does not significantly influence the carbon stock in forest floor litter (FF) and soil organic carbon (SOC) over the entire profile, management activities involving ARD activities or forest conversion do affect those forest compartments.

Flemish forests allocate 12.6 Mt C to their living biomass, 3.9 Mt to the litter layer (L, F and H layer) (FF) and 40.1 Mt to the SOC to a depth of 1.2 m. SOC (kt C ha^{-1}) in deciduous forests is higher than in coniferous forests and this for all provinces (Fig. 3.1). This can be partly attributed to the fact that deciduous trees incorporate more fine roots into the deeper soil layers but this is also caused by the soil type. Soils planted with deciduous trees generally have a more loamy texture and especially a higher moisture content, which favours a higher carbon stock.

Forest litter layer and SOC data were made available from the forest inventory by INBO. For detailed information on the same results for the Walloon region, reference is made to the final report of the METAGE project.

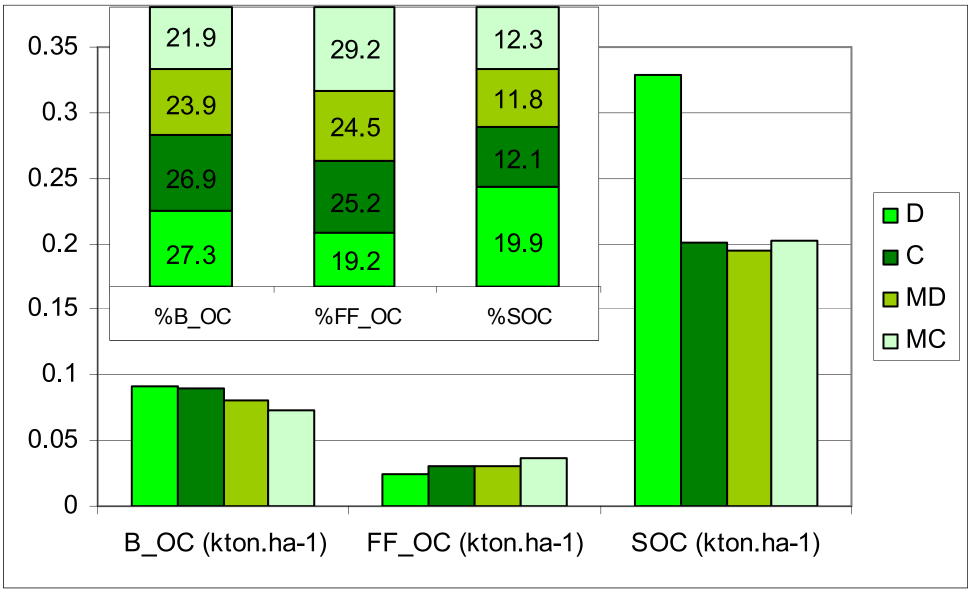

Fig. 3.1 *Allocation of total Flemish forest carbon (12.6 Mt C) to the forest types and pools; D: deciduous, C: coniferous, MD: mixed deciduous, MC: mixed coniferous; B_OC: above- and belowground biomass, FF_OC: forest floor litter, SOC: soil organic carbon*

3.2.5. Conclusion

A good approximation can be made of actual and projected carbon stock and sequestration in Belgian forests based on regional forest inventories and general reported BEFs. It is emphasized though that more precise carbon accounting can be achieved by establishing species-specific wood density values, by including dead wood, species- and age-specific annual growth increment, transparent volume calculations with corresponding accuracy assessments and more centralised information on wood harvesting (by expanding an IVANHO-like database to all forests) and on ARD activities.

Overall, deciduous forests appeared to have a higher carbon stock in the different forest compartments than coniferous forests. Especially SOC in deciduous forests has a large part in the total carbon stock. The general trend in Flanders of shifting towards more mixed and deciduous forests seems promising in view of carbon uptake. More research is needed on similar soil types (texture and moisture content) for deciduous and coniferous forests with regard to their carbon allocation and stock.

Belgian forests have acted throughout the period 1990-2000 as a carbon sink, amounting to 3.08 Mt CO_2 eq. yr^{-1}. They will remain acting as a carbon sink (BAU scenario) in the first commitment period (2008-2012) at a predicted rate of 2.71 Mt CO_2 eq. yr^{-1}. Deforestation in Flanders during the period 1990-2000 generated a carbon loss to the atmosphere of 0.16 Mt CO_2 eq. yr^{-1}.

3.3. SHORT-ROTATION TREE PLANTATIONS

3.3.1. Introduction

Especially in Europe, Short Rotation Forestry (SRF) is rapidly expanding because of the reduced dependence on foreign sources of energy, the availability of abandoned land under the European set-aside policy and because of the CO_2 mitigation capacities of SRF plantations. Afforestation of abandoned agricultural land also falls within the articles 3.3 and 3.4 of the Kyoto Protocol on Land Use, Land Use Change and Forestry (LULUCF) activities. In this paragraph, results linked to task II and III of Task Force 3 are presented.

3.3.2. Short-rotation plantation

In March and April 2001, a short-rotation plantation was established on shallowly tilled former agricultural land at Zwijnaarde (Campus Ardoyen of Ghent University; (51°02' N, 3°43' E). At the start of the experiment, the sandy soil of the plantation had a homogeneous upper layer of 30 cm depth, with a mean organic carbon concentration of 1.0%, a nitrogen content of 1000 ppm and a pH_{KCl} of 4.5. The total area of the plantation is 9600 m^2, and is composed of 24 plots of 400 m^2 each (25 m * 16 m). Cuttings of poplar (*Populus trichocarpa x deltoides* - Hoogvorst) and willow (*Salix viminalis* - Orm) and two-year old saplings of birch (*Betula pendula Roth*) and maple (*Acer pseudoplatanus L.* - Tintigny) were planted with an initial density of 20000 stems ha^{-1} (poplar and willow) and 6667 stems ha^{-1} (birch and maple) respectively. No weed control, fertilisation or irrigation was performed in the plantation.

3.3.3. Aboveground biomass production

Site- and species-specific allometric relationships of the form $AGDM = a . D_{0.3}^b$ were established based on destructive measurements throughout one rotation period (4 years). In this equation, AGDM is aboveground dry matter (g), $D_{0.3}$ is diameter at 0.3 m height (mm) and a and b are regression parameters. Age and tree height did not appear to be a significant predictor variable for this first rotation. The regression coefficients for the applied power regressions are given in Table 3.3.

Both potential and actual (taken mortality into account) aboveground biomass production were calculated after 4 growing seasons (Table 3.4).

The high mortality of birch can be explained by a severe plant shock (drying of root tissue), while the applied poplar clone appeared to have no resistance to cancers and bacterial infections. The large variability in biomass production between subplots can be explained by the physical soil characteristics (texture, acidity, matrix potential, root penetrability and bulk density) up to 1 m depth. However, their predictive capacity appeared to be rather low (R^2_{adj} of 0.21) (Van De Casteele, 2004).

Table 3.3 *Number of sample trees (n), regression coefficients a and b (with standard error), and the adjusted coefficient of determination for the species-specific allometric relationship* $AGDM = a \cdot D_{0.3}^{b}$

Species	n	a	b	Adjusted R^2
Birch	18	0.292 (0.078)	2.242 (0.078)	0.980
Maple	49	0.067 (0.088)	2.662 (0.050)	0.983
Poplar	18	0.295 (0.076)	2.223 (0.077)	0.980
Willow	34	0.135 (0.022)	2.553 (0.059)	0.983

Table 3.4 *Mean survival rate (%) and mean potential and actual aboveground biomass production (t DM ha^{-1} yr^{-1}) of birch, maple, poplar and willow after four years of tree growth; different letters indicate significant differences (p = 0.05)*

Species	Mean survival rate	Potential biomass production (t DM ha^{-1} yr^{-1})		Actual biomass production (t DM ha^{-1} yr^{-1})	
	(%)	Mean (s.d.)	Range	Mean (s.d.)	Range
Birch	75.8	3.3 $(0.7)^b$	2.7 - 4.4	2.6 $(0.7)^{a,b}$	2.1 - 3.7
Maple	96.8	1.2 $(0.5)^a$	0.7 - 1.7	1.2 $(0.5)^a$	0.6 - 1.7
Poplar	86.3	4.2 $(0.9)^b$	2.8 - 5.4	3.5 $(0.7)^b$	2.6 - 5.0
Willow	97.6	3.5 $(1.5)^b$	1.4 - 5.9	3.4 $(1.5)^b$	1.4 - 5.8

Although biomass production in the plantation at Zwijnaarde is rather low compared to European studies on nutrient rich soils (10 to 20 t DM ha^{-1} yr^{-1}), it is comparable to other studies on marginal land (2 to 3 t DM ha^{-1} yr^{-1}). A fertilization experiment for the tree species studied revealed that the physical soil properties were the growth limiting factor (Vanlerberghe, 2004). As such, fertilizing did not increase the production. In most other studies, weed control, fertilization or irrigation are applied to enhance growth and biomass production.

3.3.4. Belowground biomass production

In 2001, roots were fully excavated for several trees per species, while in 2004, roots were excavated per unit area and root borer samples for willow. More than 50% of total biomass in SRF is situated belowground with a root:shoot ratio exceeding 1 (Table 3.5). More than 70% of all willow roots can be found in the upper 30 cm of the soil.
More information on the vertical root distribution for willow and accompanied SOC input to the soil layers can be found in De Vos (in prep.).

Table 3.5 *Percentage of root biomass (g DM) in the total biomass (g DM) and root:shoot ratio (R:S) for the four experimental species*

Species	% (2001)	R:S (2001)	% (2004)	R:S (2004)
willow	54.70	1.21	52.60	1.11
poplar	47.65	0.91		
birch	54.59	1.28		
maple	70.13	1.54		

3.3.5. Energy production from biomass

The calorific value of wood indicates its energy content. This calorific value (kJ g^{-1} DM) was determined for the four species with an oxygen bomb calorimeter. The mean calorific values found in the plantation (Table 3.6) are comparable to calorific values of different bio-energy crops mentioned in literature, which ranged from 13.5 to 24.0 kJ g^{-1} DM. Combining information of aboveground biomass (t DM ha^{-1}) after 4 years of tree growth and the calorific value of the wood, gave the amount of energy stored in the biomass. Table 6 shows that although birch was planted at a lower density (6667 trees ha^{-1}), resulting in a lower mean biomass production (Table 3.4), the total amount of energy stored in the biomass was slightly higher for birch than for willow, because of the higher calorific value of birch wood.

It is planned that in Flanders a maximum of 10000 ha will become available for short-rotation forestry in the near future. When co-burning (conversion efficiency (CE) of 37%) is chosen to produce electricity from biomass grown at these plantations, the total number of households (mean need of 3300 kWh_e yr^{-1}) that can be provided with electricity from these 10000 ha SRF plantations is 22095 per year (Table 3.6). This is only 1% of the total number of 2.2 million of households living in Flanders. This implies that the possible contribution of electricity from SRF biomass to the overall electricity consumption in Flanders is very limited, though not unimportant.

3.3.6. CO_2 emission reduction potential by aboveground biomass

Burning of biomass from SRF plantations can be considered as a CO_2 neutral process. As such, the carbon emission reduction resulting from the substitution of fossil fuels by SRF electrical energy was calculated by multiplying the maximum amount of electrical energy that could be produced by 10000 ha of SRF plantations of birch, maple, poplar and willow (at marginal soils and tested densities) with the amount of CO_2 released during 'traditional' electricity production processes (Fig. 3.2). The oldest coal plant in Belgium emits 263.9 kg CO_2 per GJ electricity produced, while the emission was estimated at 136.1 kg CO_2 GJ_e^{-1} for the most modern gas turbine. Besides co-burning, two other conversion processes were considered: burning of biomass (CE of 16%) and gasification (CE of 27%).

***Table 3.6** Calorific value (kJ g^{-1} DM), energy stored in the biomass after 4 years of tree growth (MWh ha^{-1}) and number of households that can be provided by electricity produced from this biomass*

Species	Cal. value kJ g^{-1} DM	Biomass energy (MWh ha^{-1})	Households (# yr^{-1})
Birch	21.3	77.4	21723
Maple	19.4	25.8	7234
Poplar	19.9	78.8	22095
Willow	19.6	74.8	20966

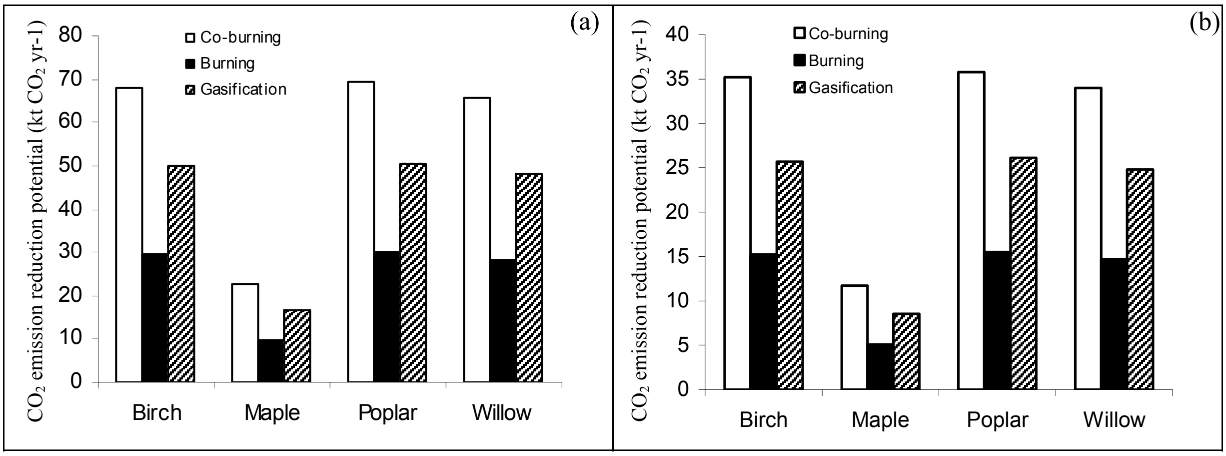

***Fig. 3.2** CO_2 emission reduction potential (kt CO_2 yr^{-1}) of 10000 ha of birch, maple, poplar and willow SRF plantations compared to (a) the oldest coal plant of Belgium and (b) the most modern gas turbine*

Emission reductions were highest when bio-energy production systems were compared to the oldest Belgian coal plant. Depending on the conversion process chosen (co-burning, burning or gasification), using electricity produced by biomass instead of fossil fuels can reduce CO_2 emissions with 9.8 to 69.3 kt CO_2 yr^{-1} (Fig. 3.2a). Reductions compared to a modern gas turbine ranged from 5.1 to 35.8 kt CO_2

yr^{-1} (Fig. 3.2b). The total CO_2 emission in Flanders in the year 2000 amounted to 76264 kt CO_2. This means that the CO_2 emission reduction potential of this kind of SRF plantations in Flanders is only 0.09% of the total CO_2 emissions and thus of minor significance in the view of reaching the Kyoto Protocol targets.

3.3.7. Soil respiration

An automatic closed chamber system (EGM-1, PP-systems) was used to measure temporal and spatial variation of the soil respiration (g CO_2 m^{-2} h^{-1}). Simultaneously, soil temperature (°C) was measured at a depth of 5 cm next to the PVC soil respiration ring (diameter 18.8 cm). In one birch plot, 18 rings were installed to study the variation in soil respiration at a short distance. In the plots where the carbon balance was studied (see next paragraph), three rings were installed. A detailed study of the influence of soil moisture on soil respiration was performed in three willow plots, where 4 rings per plot were available.

Soil temperature explained between 60 and 95% of the temporal variation of the soil respiration measured at individual ring level, or at plot level (Looman, 2005). Two models were established to describe the short-distance spatial variation in soil respiration within the studied birch plot:

$$SR = (0.037 \cdot C_{wb.0-30} + 0.041 \cdot w_{02.15-30}) \cdot e^{(0.186 \cdot C_{wb.0-30}) \cdot ST} \tag{1}$$

and

$$SR = \left[\frac{(0.424 \cdot C_{wb.0-30})}{0.000607 \cdot N_{5-15} + (0.299 \cdot tree_3)^{-((ST-10)/10)}} \right] \tag{2}$$

with SR : soil respiration (g CO_2 m^{-2} h^{-1}); $C_{wb.0-30}$: carbon content (%) in the mineral soil layer from 0 to 30 cm, determined by the Walkley and Black method; $w_{02.15-30}$: the amount (g DM) of fine roots (diameter < 2 mm) in the mineral soil layer from 15 to 30 cm; N_{5-15} : nitrogen content (ppm) of the mineral soil layer 5-15 cm; $tree_3$: distance from the soil respiration ring to the third nearest tree (cm), and ST : soil temperature at 5 cm depth (°C).

On a larger scale, the quality of the leaf litter was an important factor in explaining the differences in soil respiration between the different plots. This was translated in the following soil respiration model:

$$SR = (0.00267 \cdot N_{litter}) \cdot e^{(0.102 \cdot bd_{0-5} \cdot ST)} \tag{3}$$

with SR : soil respiration (g CO_2 m^{-2} h^{-1}); N_{litter} : N-content of the leaf litter (g kg^{-1}); bd_{0-5} : bulk density of the upper mineral soil layer (0-5 cm); ST : soil temperature at 5 cm depth (°C). Fig. 3.3 shows the relation between the measured soil respiration values and the modelled results for the year 2004.

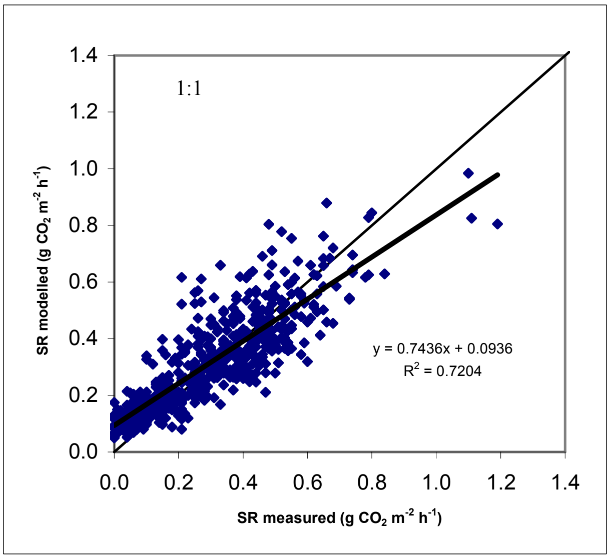

Fig. 3.3 *Comparison between measured and modelled soil respiration rates (g CO_2 m^{-2} h^{-1}) for the year 2004; measurements for all plots of the short-rotation plantation at Zwijnaarde*

Including soil moisture in the temporal soil respiration model increased the R^2 with 4.3, 19.1 and 3.6% for the three willow plots (Looman, 2005). This indicates that at some places in the field, soil moisture can be very important in determining the soil respiration rates. However, the difference in soil moisture in these three plots did not explain the spatial variation in the total soil respiration between these 3 plots. These spatial differences were mainly determined by the C- and N-content of the upper soil layer (0-5 cm), and the bulk density of this layer. It should be noted that for the rings in these plots, no information on roots was available so far.

3.3.8. Carbon balance

In this study, the carbon balance is considered as the net amount of carbon taken up by the trees minus the carbon released from the soil by soil respiration. The carbon balance was studied on six plots of the plantation in Zwijnaarde, and on two control plots, which were not planted with trees. If more carbon is taken up by the trees than is released by soil respiration, the site acts as a sink for carbon. In the opposite case, the site is a source of C.

To determine the carbon uptake, monthly measurements of Leaf Area Index and Specific Leaf Area (m^2 g^{-1}) were performed. Every month, the diameter at 30 cm, d_{30} (mm), of 20 trees per plot was measured, and transformed in aboveground biomass AGDM by the allometric relations mentioned above. As the root-to-shoot ratio was also determined in Zwijnaarde, the amount of belowground biomass was calculated based on the AGDM of the tree. Fifty percent of the dry matter was supposed to be carbon. The difference between the total above- and belowground carbon stock in the biomass between the 1st January and the 31st December was considered as the annual carbon uptake by the trees.

As described in the previous paragraph, soil respiration (g CO_2 m^{-2} h^{-1}) and soil temperature (°C) were measured fortnightly with an automatic closed chamber system (EGM-1, PP-systems). Integration of soil respiration measurements over time gave the total soil respiration during a specific period or year.

In 2002, the amount of carbon taken up by the trees was lower than the total soil respiration for all plots (Fig. 3.4a). However, compared to two plots which were not planted with trees, the overall carbon balance for the planted plots was less negative, which means that these plots were a smaller source for carbon than when the area would not be planted with trees.

In the year 2003, the willow plot appeared to be a sink for carbon (Fig. 3.4b), while the other plots were still losing C. In 2004, the carbon uptake in the two poplar plots was larger than the soil respiration. The maple and birch plots, and the non-planted plots, were still a source of C after 4 years of tree growth. For these plots, the balance was more negative in 2004 than in 2003.

The amount of carbon taken up by the trees was increasing from year to year (Fig. 3.5a), and did therefore not explain the more negative C-balance in 2004 compared to 2003. The soil respiration on the other hand was lower in 2003 than in 2002 and 2004 (Fig. 3.5b). This lower value found in 2003 was mainly caused by lower temperatures in this year, in combination with a dry summer period.

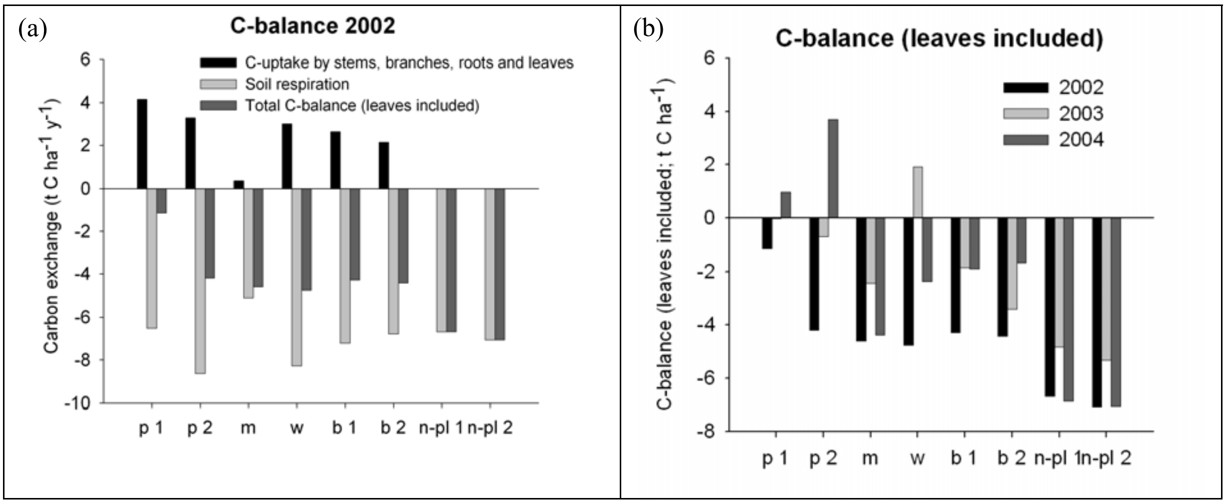

Fig. 3.4 *(a) C-uptake by the trees, soil respiration and C-balance for the year 2002; (b) C-balance for the years 2002, 2003 and 2004; p = poplar, m = maple, w = willow, b = birch, n-pl = non-planted (control) plots*

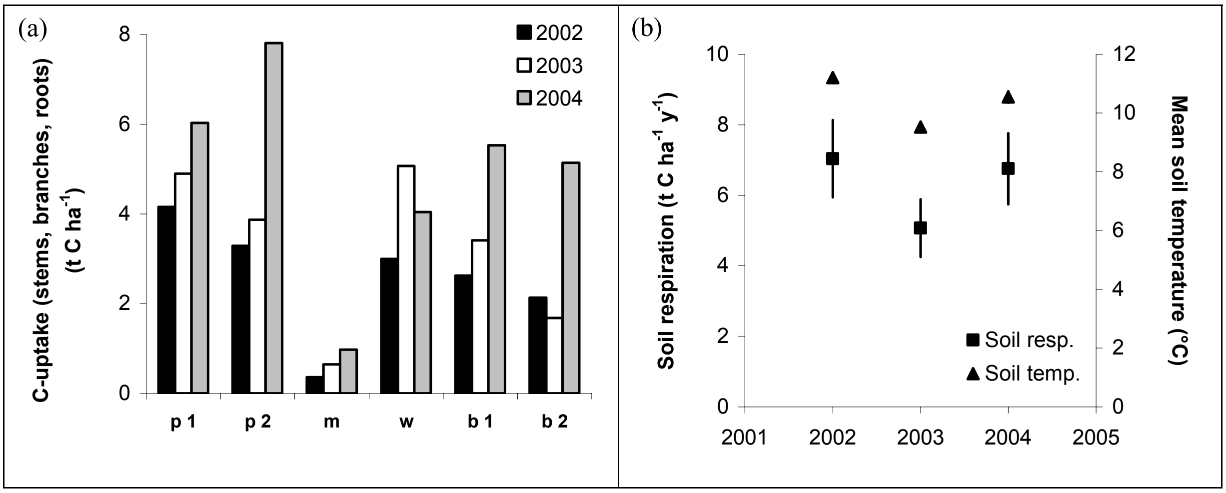

Fig. 3.5 *(a) C-uptake by the trees in 2002, 2003 and 2004; (b) mean soil respiration (squares) and mean soil temperature (triangles) in 2002, 2003 and 2004; error bars indicate 1 s.d.; p = poplar, m = maple, w = willow, b = birch*

3.3.9. Insect population

In 2003, an indication for the biological value of the plantation was sought through inventorying and characterising 4 terrestrial arthropods (Araneae, Isopoda, Chilopoda and Coccinellidae). Although the plantation shows more connection with the road sides than the forests, some species were found that have forests as their preferred though not unique habitat (Lepthyphantes zimmermanni and Oenopia conglobata). The fact that these species occur confirms the value of such biomass plantations for biodiversity. Since SRF will never reach the climax situation of a mature forest, the value of biomass plantations lies in a repeated temporary stepping stone function to bridge the gaps between surrounding forests.

3.3.10. Conclusion

With the implementation in due course of a clear judicial statute for biomass plantations in Flanders and the continuous need for renewable energy, it is most likely that the poorer agricultural soils and marginal soils will be primarily used for woody biomass production in Flanders. Tree species selection on such marginal sites should be done precautiously, with a preference for birch, especially on drier soils. Birch should be seeded in situ to prevent a severe plant shock. Several examples in Scandinavia already apply seeded birch for biomass or pulp production.

The possible contribution of electricity or heat from SRF biomass to the overall electricity or heat consumption in Flanders is very limited, but has potential at small scale. Although carbon is sequestred by SRF, their respiration after the first rotation still exceeds the carbon uptake, resulting in a negative carbon balance. It is expected that the SRF will become a net sink after 8 to 10 years (Hansen, 1993).

However, besides energy production and carbon sequestration, SRF plantations can have many other benefits as there are : the improvement of soil physical properties, prevention of soil erosion, capturing of airborne particulates, visual and acoustic buffer function or (temporary) habitat for plants and animals.

CHAPTER 4 NON CO_2 GREENHOUSE GAS EMISSIONS FROM SOILS

4.1. INTRODUCTION

The research considered thus far concerned in essence opportunities of C sequestration in soils based on art. 3.3 and 3.4 of the Kyoto Protocol. The Kyoto Protocol, however, establishes that between 2008 and 2012 a general reduction of 7.5% of a basket of greenhouse gases has to be achieved compared to the reference year 1990. Not only CO_2 but also N_2O and CH_4 belong to that basket so they should be considered, as well. Because of their high global warming potentials (296 and 23, respectively) the total greenhouse gas budget of any ecosystem is more sensitive to changes in the latter two gas fluxes than to changes in soil CO_2 emissions (IPCC, 2001b). As C and N cycles are tightly linked, changes in C accumulation could influence N_2O and/or CH_4 emissions. Therefore the benefits of possible increased soil C accrual in relation to greenhouse gas emissions could be offset to an extent that depends on the magnitude of the two other gases (Robertson et al., 2000; Six et al., 2004) (TF4a).

Nitrous oxide emissions from soils are predominantly formed during the biological processes of nitrification and denitrification and are determined by several different factors (Beauchamps, 1997). As these factors can change fast both in time and space, emission measurements are heavily constrained by large measuring errors (Helgason et al., 2005) caused by this temporal and spatial variability (Velthof et al., 1996; Choudhary et al., 2002). Methane, having an equal variability, is both produced and consumed, respectively by methanogens and methanotrophs in the soil. Production occurs only under anaerobic, highly reducing conditions in the absence of nitrate, sulphate or ferric iron. Thus, anaerobic soils, such as rice paddies and wetlands, that are rich in organic matter, can be significant methane sources (Topp and Pattey, 1997). Consumption occurs in contrast, only under aerobic circumstances and is performed by two distinguishable groups of methanotrophs. Addition of N fertilization was proven to inhibit this oxidation process.

Net emissions of non-CO_2 greenhouse gases are higher from arable than from forest soils (Suwanwaree and Robertson, 2005; Boeckx, 1999; Borken and Beese, 2005; Goossens et al., 2001; Venterea et al., 2005). Therefore, our primary concern was to estimate emissions from arable soils. As CH_4 fluxes from arable soils are only minor to the N_2O emissions focus was laid on the latter greenhouse gas. The high temporal and spatial variability of N_2O emissions together with a multitude of factors affecting these emissions in a complex way limit the practical feasibility to measure N_2O emissions under all possible, different conditions and make predictions of the losses based on simple regression equations (Plant, 2000) difficult. Both available and new measurements were used to validate the reliability of existing models (DNDC, FASSET) to estimate N_2O emissions from arable land. The regional integration tool of DNDC permitted to up-scale these simulations and estimate the N_2O losses from cropland soils in the Flemish region, which will give additional information to the C sequestration simulation results for croplands (1.3.4).

4.2. MATERIALS AND METHODS

Short descriptions of the experiments are mentioned in the text. A more extended version can be found in Beheydt et al. (submitted a), Beheydt et al. (submitted b), Beheydt et al. (in preparation a) and Beheydt et al. (in preparation b).

4.3. RESULTS AND DISCUSSION

4.3.1 N_2O emissions of minimum tilled vs. tilled soils for Belgium (Beheydt et al., submitted b)

Research stipulated the important potential of C sequestration in soils by the adoption of no tillage (West and Post, 2002; Freibauer et al., 2004). However, taking all greenhouse gases into account, as dictated by the Kyoto Protocol (CO_2, N_2O and CH_4) the conversion from conventional to no tillage provided more contrasting results. Both net emissions and net sequestration (Six et al., 2004) were found, with N_2O emissions determining the directions of the overall greenhouse gas (GHG) flux. Knowing whether or not extra N_2O emissions remain smaller than the C sequestration potential after conversion, becomes primordial to assess the overall effect. According to Venterea et al. (2005) and Helgason et al. (2005) there is at this moment little consensus, whether reduced tillage leads to increased or decreased N_2O emissions and what are the most important factors regulating the magnitude or direction of the effect. Performing a meta-analysis Six et al. (2004) concluded that long-term tillage effects on total GHG emissions depend both on climatic regime and duration of the adoption period. It may be clear, however, that next to these two factors, also other factors like fertilizer management (Venterea et al., 2005), residue management (Baggs et al., 2003), etc. will influence the obtained results in a more or lesser extent. As there are no measurements available for Belgium, there is even no clue whether this conversion will result in a net emission or net sequestration. Therefore a measuring campaign was started, measuring two minimum tilled soils cropped with maize (MTM) and summer oath (MTO) and a tilled field, cropped with maize (CTM), all under a business as usual.

Total N_2O losses over the measuring period (6/3/2004-8/2/2005) were markedly higher from the minimum tilled soils (MTO: 3.64 kg N_2O-N ha^{-1}; MTM: 5.27 kg N_2O-N ha^{-1}) than from the tilled soil (0.27 kg N_2O-N ha^{-1}). Peak emissions of N_2O are strongly represented during spring and summer time and were often associated with the combination of precipitation (or high soil water contents) and management events (killing off of the cover crop, N fertilization…). On the basis of an acetylene inhibition experiment, N_2O losses were assigned predominantly to denitrification (data not shown). Large differences, about twice as high as the values mentioned by Six et al. (2004), were found between minimum and conventionally tilled soils. The major factor responsible for the large difference seems to be the lower water content of the tilled soil (Fig. 4.1), which could be caused by the drainage, the higher carbon content and/or the higher clay content of the minimum tilled soils (Table 4.1) (Vereecken et al., 1989).

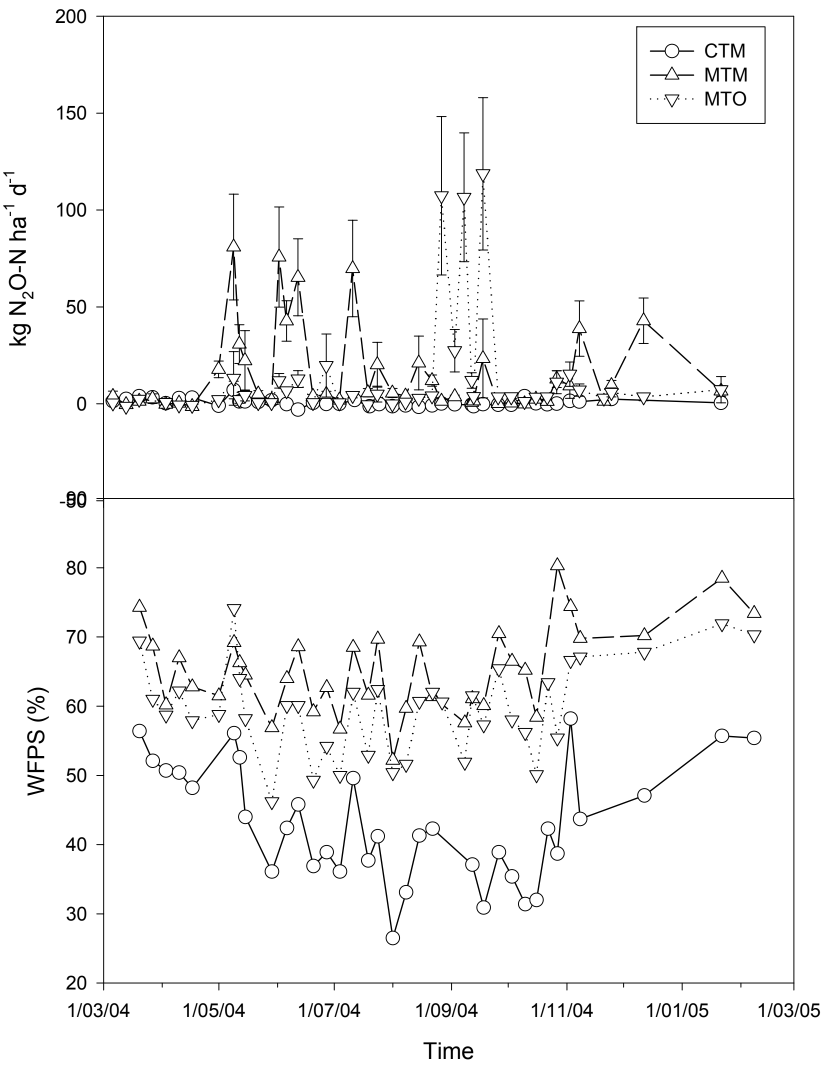

Fig. 4.1 *N_2O emissions and water filled pore space (WFPS) for the minimum tilled and tilled fields (error bars for N_2O represent standard errors)*

Crop choice had an influence on the N_2O emissions from the minimum tilled soils (summer oath emitted less than maize ($p < 0.05$)). Sowing and extra fertilization of the cover crop after summer oath, however, nullified this picture (1/09/2004, Fig. 4.2). Emission losses expressed per kg N applied were also quite high from the minimum tilled fields (up to 0.04 kg N_2O-N/kg N applied for the MTM without background correction) if compared to the default value of 0.0125 suggested by the IPCC (2001c). The conventional tilled field, on the other hand, reached only a value of 0.001.

Table 4.1 *Soil properties of the three fields for 0-20 cm*

	OC	Clay	Silt	Sand	$CaCO_3$	pH_{KCl}	BD
			(%)				(g cm^{-3})
MTO	1.31	30.0	61.8	8.2	0.95	5.8	1.16
MTM	1.66	28.2	62.5	9.3	0.68	6.5	1.25
CTM	0.95	12.0	74.4	12.6	0.54	5.8	1.22

4.3.2 Validation of DNDC for N_2O emissions from Belgian agricultural soils (Beheydt et al., submitted a)

To cope with the up-scaling of direct N_2O emissions from agricultural soils, different models, ranging from simple regressions to completely process-based models have been developed. Data requirement is often an extra limitation. Therefore regression models have been quite popular (Bouwman, 1996; Freibauer and Kaltschmitt, 2003; Roelandt et al., 2005). As these regression models neglect several variables (because their sensitivity was below the cut off value or because the dataset used for making the model did not distinguish for these variables), they can not always be used to test different management or mitigation scenarios, in contrast to the more complicated models (Li et al., 2005). Moreover, with regard to N_2O emissions, the complexity of interactions between controlling factors (Beauchamps, 1997) is the very reason for Plant (2000) suggesting detailed, mechanistic simulation models, rather than empirical summary models for extrapolation purposes. In addition, by using more complex models, N_2O emissions from agriculture can be reported according to the tier 3 methodology in the IPCC guidelines (IPCC, 2001c). In the literature, several mechanistic simulation models for estimating N_2O emissions at field scale are described. The aim of this part of the study was to validate DNDC (Denitrification-Decomposition; Li et al., 1992), for the simulation of N_2O emissions from arable land and grassland using region specific (Belgium) N_2O flux data. The DNDC model was chosen as it has been widely applied throughout the world under a broad range of conditions with promising results to estimate direct N_2O losses from the soil (Li et al., 2005).

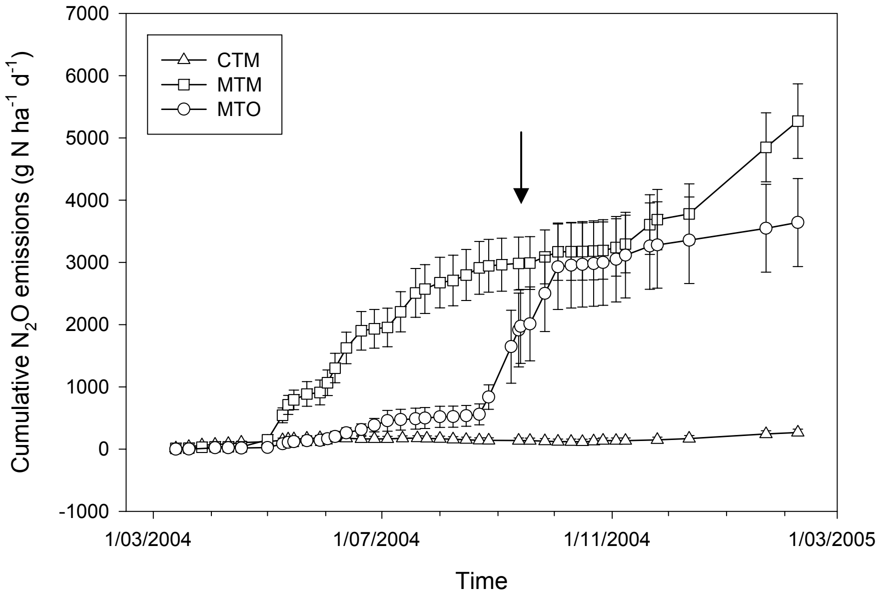

Fig. 4.2 *Cumulative N_2O emissions (error bars represent standard errors) for the different crops (the arrow indicates the timing of the cover crop fertilization on the MTO field)*

Although the main concern is to predict reliable total N_2O emissions, comparing estimated and measured time patterns of the emissions can be equally important to verify whether the model captures the most important processes. The agreement between the temporal pattern of the simulated N_2O emissions and the measurements varied between good and poor using DNDC default parameters for field capacity (FC) and wilting point (WP). Therefore pedotransfer functions (PTF's) (Vereecken et al., 1989) assessed for the local conditions, were used to calculate the water filled pore space (WFPS) at FC and at WP, which clearly improved the simulations of the WFPS. According to Frolking et al. (1998) accurate simulation of soil moisture appears to be a key requirement for reliable simulation of N_2O emissions. Further adoptions with changed crop and soil parameters, based on regional data, or changes to the default distribution of C in humus, humads and litter pools, as was suggested by Sleutel (2005) for C simulations, did not improve the N_2O simulations and only slight changes were observed (data are not shown). Therefore it was decided to use the crop and soil parameters provided by DNDC 8.3P except replacing the original values of FC and WP with the results calculated based on the PTF's of Vereecken et al. (1989).

Immediately after each fertilizer or manure application DNDC simulates important N_2O emission peaks, having a major impact on the total simulated N_2O emissions. These peaks, however, were not always observed in the measurements. In general, simulations gave higher and more frequent N_2O peaks compared to actual measurements in the field. Only for a few grassland sites the simulated peaks were lower (data not shown). Contrasting results for the temporal variation of N_2O emissions have also been found by other authors (Smith et al., 2002; Brown et al., 2002; Cai et al., 2003). Li et al. (2001) argued that DNDC is able to capture general patterns and magnitudes of N_2O emissions observed in the field although discrepancies exist.

Performing a regression analysis for temperate Western European sites, Freibauer and Kaltschmitt (2003) distinguished their data into three groups. Both cropland (with the exception of alpine and extensively snow-covered pre-alpine regions, which are not under discussion here) and grassland sites were characterized by annual mean N_2O emissions generally below 3 kg N_2O-N ha^{-1} yr^{-1} and maximum emission rates below 10 kg N_2O-N ha^{-1} yr^{-1}. Although most of the fields in our data set fitted into this emission range, some croplands and grasslands had emissions which were two- or three-fold higher than the suggested maximum (Fig. 4.3). The simulated N_2O emissions varied also over a broad range (0.5 to 42.6 kg N_2O-N ha^{-1}). Just as the measured emissions, several simulated fields showed values considerably higher than the maximum value suggested by Freibauer and Kaltschmitt (2003) (Fig. 4.3). It has been assumed that a high N fertilization rate (Boeckx et al., 2001) may be one of the major driving forces of these high emissions. According to Li et al. (1996) and Li et al. (2001) the soil organic C (SOC) content is the most sensitive factor for N_2O emissions. To obtain a measure for the uncertainty on the simulations (error bars in Fig. 4.3) this factor was varied (±15% C).

Although DNDC tends to overestimate the total N_2O emissions (solid line in Fig. 4.3) it can be seen that individual results (Fig. 4.3) showed both (large) under- and overestimations. The magnitude of the overestimation, however, decreases with increasing N_2O emissions. The regression for the cropland soils only (Sim.= 1.08*Meas.+7.4 R^2=0.85) is considerably better than for the grasslands (Sim.= 0.26*Meas.+9.0 R^2=0.05) although the former overestimates all measurements, while the latter is both over- and underestimating the measured data. The problematic behaviour of DNDC for grassland soils was also seen for several fields in The Netherlands (H. Kros, personal communication).

Using the available management data, additional simulations for the measured tilled and minimum tilled soils (see 4.1) were run. The simulated total N_2O emissions resulted for the two minimum tilled soils in lower mean values, while the conventional tilled soil was predicted to emit more N_2O than measured. The total N_2O emissions simulated for the minimum tilled soils lay within the 95% confidence interval of the measured N_2O emissions. The simulation for the conventional tilled soil, in contrast, was about 10 times as high as the upper value of the 95% confidence interval, stressing the overestimation by the model. Addition of these three results to Fig. 4.3 did not affect the conclusions derived above.

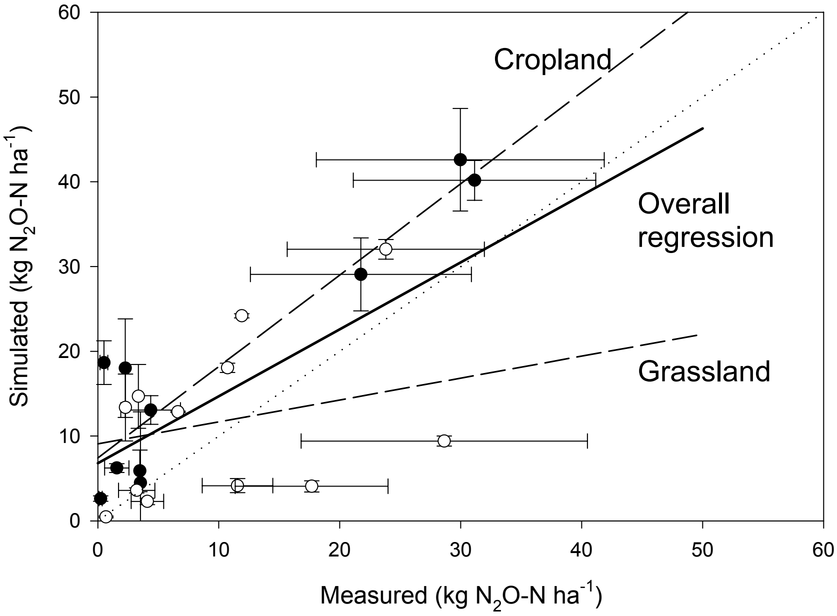

Fig. 4.3 *Overview of the total measured and simulated (DNDC) N_2O emissions for the different croplands (full bullet) and grasslands (empty bullet) (error bars for the measurements are the standard deviations; error bars for the simulations are changes in total N_2O emissions for ±15% changes in SOC)*

With the exception of fertilization events, DNDC did not show any significant N_2O peaks after other management events (destruction of cover crop, harvest…) which were measured from the minimum tilled soils. The WFPS data indicated also another problem with the model. It is known that the composition (physical and biological) of the upper few centimetres in a minimum tilled soil are quite different from the below centimetres, influencing the occurring biological processes. The inability of DNDC to capture the correct WFPS at the different depths could possibly hamper the correct predictions of the N_2O emissions as measurements showed that the peak emissions were associated with high WFPS. The simulated emission pattern of N_2O from the tilled soil agreed with the measurements in the fact that no striking peak emissions were seen. However, the continuous overestimation of the background emission significantly influenced the total losses.

4.3.3 Comparing DNDC with FASSET (Beheydt et al., in preparation a)

Several of the fields were also simulated with another process-based model: FASSET (Jacobsen et al., 1998). FASSET stands for 'farm assessment tool' and was originally developed in Denmark to enable farmers to perform decisions on their farm concerning management, land use… based on the economical consequences (e.g. environmental consequences leading to taxes) of these decisions. Greenhouse gas emissions are part of the environmental impact of a farm and a sub-model (Chatskikh et al., 2005), simulating these emissions was used to obtain a second estimate of the direct N_2O losses from the measured Belgian fields.

Comparing the total N_2O simulations with the measurements (for measured values lower than 10 kg N_2O-N ha^{-1}) relatively similar results were obtained by both models (Fig. 4.4). For some simulations FASSET predicted better results and for other simulations it was the other way around. For the higher N_2O emissions measured, however, FASSET simulations were lower than the DNDC. In contrast to DNDC, which both under- and overestimated the measurements, FASSET underestimated all measurements. In general, cropland measurements were best and most consequently estimated by the DNDC model (although overestimated) while also for grasslands the DNDC tended to predict better results. However, as was already mentioned before, these latter simulations with DNDC were not really satisfying.

Concerning the time pattern of the emissions, FASSET showed generally a more fanciful behaviour than DNDC and predicted lower background emissions than DNDC. However, due to the large errors associated with the measurements and the different behaviour of both models for all simulations, no further conclusions can be drawn. To elucidate the differences between DNDC and FASSET better, a sensitivity analysis will be performed (study is part of C-sink cluster OA/00/11).

4.3.4 Up-scaling DNDC (Beheydt et al., in preparation b)

The ultimate aim of the simulations with DNDC is performing an up-scaling exercise for Flanders (and or Belgium) in analogy with Sleutel et al. (2005) to determine the most interesting regions for N_2O emission reduction. As simulations are still running, no more information can be added, yet. A full overview of the results will be given in Beheydt et al. (in preparation b) as part of the C-sink cluster (OA/00/11).

4.3.5 Use of regression methods and the calculation of an emission factor based on the dataset (Beheydt et al., submitted a)

To verify our simulation results with DNDC, several regression models (Bouwman, 1996; Freibauer and Kaltschmitt, 2003; Roelandt et al., 2005), which may be used in a temperate climate system, were applied on the available dataset. For the regression model of Bouwman (1996) grazed grasslands and N_2 fixing crops are omitted from the measured data. Calculations were made for total emissions so no corrections for background emissions were applied (Fig. 4.5a).

The two other regression models (Fig. 4.5b and c), which were designed specifically for temperate climate systems, seem to face the same problem as the model of Bouwman (1996). Although all the regression models predict the lower emissions well (up to ±10 kg N_2O-N ha^{-1}), none of these models was able to capture the higher measured losses. As exactly these high emission fluxes have a higher-weighed contribution to the total seasonal N_2O losses, it is inherently important to have these high fluxes well estimated for regional inventories.

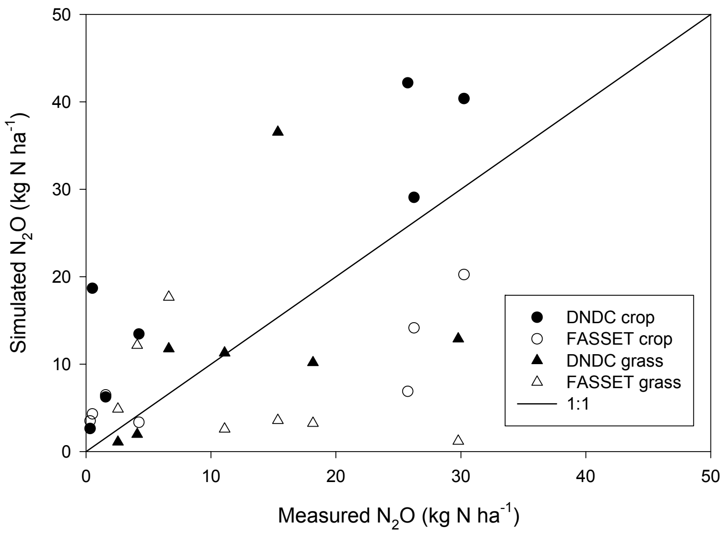

Fig. 4.4 *Simulated N_2O emissions using DNDC (filled symbols) and FASSET (open symbols) compared to the measured N_2O emissions*

Shortage of input data (land management, soil data…), or a limited number of N_2O measurements throughout a year could explain part of this discrepancy. In our validation study all N_2O measurements were carried out at irregular intervals and/or were event directed (e.g. after fertilization or rainfall). Large errors can occur because of the temporal integration and linear interpolation (Freibauer and Kaltschmitt, 2003). The large inherent uncertainty of N_2O emission data (Bouwman, 1996) has to be kept in mind as well. Using these regression-based models for up-scaling will severely underestimate the total N_2O emissions.

Although the DNDC model both under- or overestimated N_2O emissions for specific sites (Fig. 4.4), the general agreement between simulated and measured total N_2O losses was better than for the regression models. Improving the DNDC simulations for N_2O in grasslands will make this model an interesting tool to test different mitigation scenarios on a regional scale and enhance the quality of the reporting to the IPCC to a Tier 3 methodology.

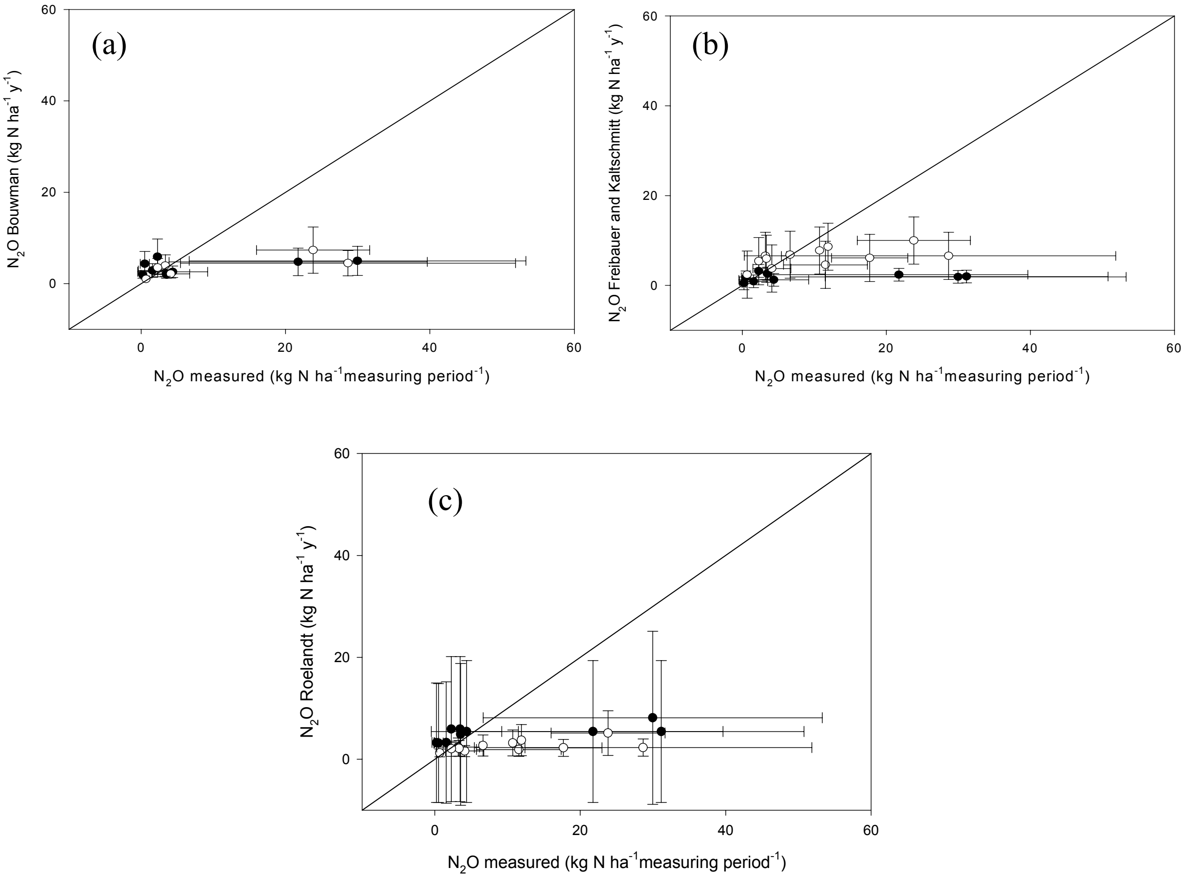

***Fig. 4.5** Comparison of the measured and calculated N_2O emissions using the regression methods of Bouwman (1996) (a), Freibauer and Kaltschmitt (2003) (b) and Roelandt et al. (2005) (c) (error bars are the 95% confidence intervals except for Bouwman (absolute uncertainty)) for croplands (full bullet) and grassland (empty bullet)*

Calculating an emission factor based on this data set and applying the same omissions as Bouwman (1996) (without grazed grassland and N_2-fixing crops), an EF of 3.77 with a 95% confidence interval of (-0.20; 7.73) for the region under consideration was found. Making a distinction between cropland and grassland an EF of, respectively, 4.27 (-0.7.17; 9.25) and 3.36 (-1.05; 7.77) was found. DNDC estimates for the considered fields resulted in a general EF of 7.36 (4.17; 10.56) or 8.81 (4.04; 13.57) and 6.41 (5.03; 7.80) for cropland and grassland, respectively.

4.4 CONCLUSIONS AND RECOMMENDATIONS

N_2O and CH_4 are important natural greenhouse gases emitted from soils, determining in many cases whether a soil will act as a net sink or a source of greenhouse gases. Concerning N_2O, which is in most cases the most important greenhouse gas emitted from the soil, only a limited amount of measurements are available for Belgium. This is partially because of the large effort needed for a measuring campaign (financial and practical), but also because of the multitude of

factors affecting these emissions. Therefore models should be used to extrapolate the emission measurements. Still, high quality measuring data for validation is needed.
Simple regression models can be used to make first estimates, but all three tested models lacked to predict the higher N_2O emission rates well. Another drawback of these models is that they do not allow testing different management options. Process-based models (DNDC) in contrast are not hampered by those constraints. Up-scaling the emissions to a regional scale using these models should permit to identify the areas having the most important losses.
Several management options are discussed in the literature to decrease greenhouse gas emissions from arable soils. One of these options, reducing tillage, was shown to increase N_2O emissions in this study. As several factors affecting these emissions were not taken into account (fertilizer management, residue management), extra measurements are necessary to strengthen this conclusion.

CHAPTER 5 QUANTIFICATION AND VERIFICATION OF ANNUAL NEW SOIL CARBON ACCUMULATION

5.1 INTRODUCTION

Classic SOC measurements are hampered by the slow and small changes of SOC after management changes compared to the large background concentrations. Therefore C sequestration experiments have to last several years (Smith et al., 2004) to obtain measurable differences. The major aim within TF4b was to test whether other options, like the use of stable isotopes, could be used to increase the resolution of the SOC determination. Thus far one experiment was performed but the results were inconclusive at our laboratory. This first experiment, however, also proved that the used equipment had a shortcoming for the type of samples which had to be measured. Adaptations were made and tested (Beheydt et al., 2005).

5.2. MATERIALS AND METHODS

Short descriptions of the experiments are mentioned in the text. A more extended version can be found in Beheydt et al. (2005).

5.3 RESULTS AND DISCUSSION (Beheydt et al., 2005)

Stable isotope techniques are described in the literature as sensitive methods to determine the different sources and sinks (and thus the increase or decrease) of greenhouse gases emitted from soil. A first experiment to determine the behaviour of C in soil after manure addition according to Högberg and Ekblad (1996) resulted in un-interpretable results. Among other reasons it proved that the $\delta^{13}C$ and $\delta^{18}O$ measurements of CO_2, however, can be obscured if N_2O is present in the sample (Craig and Keeling, 1963). This interference can happen because a common procedure for extracting CO_2 from gas samples, originally introduced by Craig (Craig, 1953), uses a cryogenic separation whereby N_2O is condensed along with CO_2. The capillary column present in our equipment (setup C) which should separate N_2O and CO_2 did not (Fig. 5.1). Theoretical corrections can be used to obtain an unbiased $\delta^{13}C$ value for mixed air samples. However, these assumptions are based on air samples and are not necessarily valid for the gaseous mixture found in direct soil emissions. Even more, certain factors necessary for theoretical correction are difficult to measure. It was shown these theoretical corrections could not remove the effect of N_2O on $\delta^{13}C$ and $\delta^{18}O$ measurements for the necessary measuring range (N_2O/CO_2: 0-100) (Fig. 5.1). Therefore the capillary column was replaced by a packed column (setup P) allowing an online separation of CO_2 and N_2O in the IRMS samples (Fig. 5.1). A new experiment is scheduled to use this changed setup.

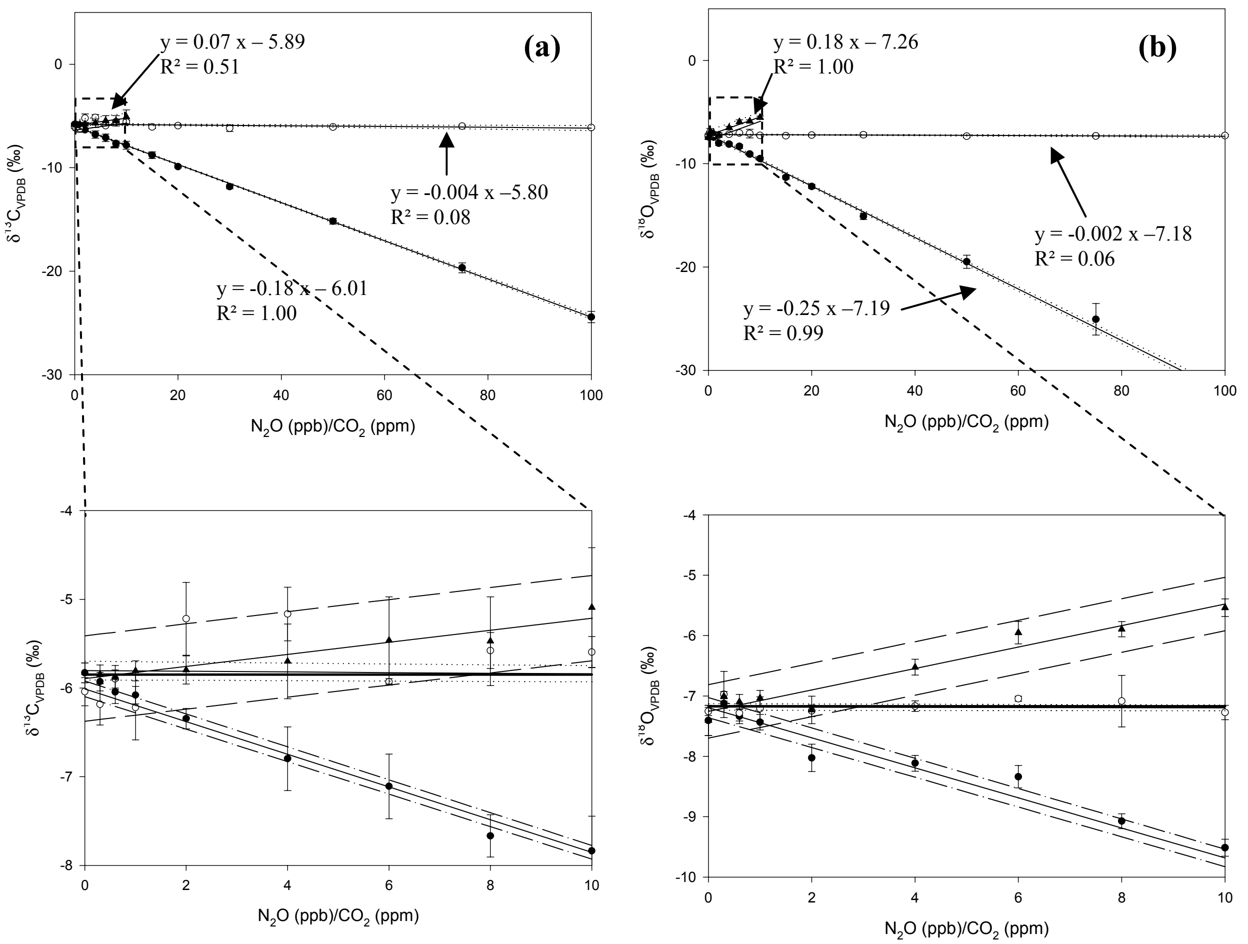

Fig. 5.1 *Influence of the N_2O/CO_2 (ppb/ppm) ratio on the $\delta^{13}C$ (a) and $\delta^{18}O$ (b) measurement of CO_2 (● : setup C; ○: setup P; ▲: theoretically calculated values) with indication of the 95% confidence interval on the mean of the regressions and the expected value (bold line) ($\delta^{13}C$ = -5.85‰; $\delta^{18}O$ = -7.17‰) (n = 5; error bars show the standard deviations)*

5.4 CONCLUSIONS AND RECOMMENDATIONS

The ability to use stable isotopes to elucidate the fate of carbon in the soil, both in the short and in the longer term, is described in literature. Performing experiments on this matter allowed improving the measurement technique and removing an interfering factor. New experiments based on this improved knowledge are scheduled.

5.5 ACKNOWLEDGEMENT

We wish to thank the Federal Science Policy for financing the CASTEC project enabling us to write chapter 4 and 5 of this document. Additional financial support of the FWO allowed a short-term mission and collaboration with the Danish Institute of Agricultural Sciences (Tjele) more particularly I wish to thank J.E. Olesen, D. Chatskikh and especially J. Berntsen for their help. Thanks also to T.J. Clough and

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CHAPTER 6 OPTIMIZATION OF AN EXISTING DYNAMIC VEGETATION MODEL

6.1 INTRODUCTION

This section is corresponding to TF5 (modelling) of the CASTEC project. The original goal of TF5 of the CASTEC project was quite ambitious: "The aim of the model is to simulate future scenarios of above- and belowground carbon sequestration of grassland, forests and plantations and to recalculate the 1990 C-stocks for TP2 and TP3 ecosystems". Time constraints and some typical characteristics of the original FORUG model (e.g. extensive parameterisation, small time step (half hour), vegetation model) led to a more feasible goal and more realistic view on the FORUG model in the CASTEC project. This goal was to simulate future scenarios for carbon fluxes and woody carbon stocks of deciduous forests (temperate beech forest).

To reach this goal, six steps can be distinguished: (1) the development of a carbon allocation module to make the link between fluxes and stocks, (2) a parameterisation of the model for a temperate beech forest, (3) an evaluation of FORUG on different sites and time scales to give an idea of the validity of the model, (4) a sensitivity analysis to determine the key parameters, (5) an uncertainty analysis to test the reliability of FORUG and (6) a scenario analysis to predict the impact of a global change scenario on the carbon balance of TF3 ecosystems. This chapter will focus on the results, detailed descriptions of the model and the methods can be found in the references.

6.2 MATERIALS AND METHODS

6.2.1 Model description

6.2.1.1 The flux model

The FORUG model is a multi-layer process-based model that simulates CO_2- and H_2O-exchange between forest stands and the atmosphere. The main model outputs are the Net Ecosystem Exchange (NEE), the Total Ecosystem Respiration (TER), the Gross Primary Production (GPP) and the evapotranspiration. In this text, we focus on the NEE as output, because the net exchange of carbon between forests and the atmosphere determines the role of forests in the global carbon cycle (Law et al., 2001).

One understory and three upperstory canopy layers are considered. A radiation module calculates the available direct and diffuse Photosynthetic Active Radiation (PAR) in each vegetation layer (Spitters, 1986; Spitters et al., 1986). In each layer, the intercepted PAR is calculated for the sunlit and shaded leaf fraction (Lemeur, 1973). This intercepted PAR drives the photosynthesis submodel. Photosynthesis and stomatal conductance are calculated according to Farquhar et al. (1980) and Ball et al. (1987). Photosynthesis and leaf respiration parameters are temperature dependent as described in Medlyn et al. (2002) and de Pury and Farquhar (1997). Soil respiration is calculated using a simple exponential function of soil temperature as described by Granier et al. (2002). Woody biomass respiration is calculated using a temperature function according to Ceschia et al. (2002). To simulate the NEE, the FORUG model uses 54 parameters. For a detailed model description see Samson (2001), Boonen et al. (2002), Verbeeck (2002) and Verbeeck et al. (2006).

6.2.1.2 The carbon allocation module

A carbon allocation module (CAF) was implemented in the FORUG model. At tree level, partitioning of carbon, among different organs and functions determines the relative growth rate of the various plant components. Furthermore, elucidating the partitioning of carbon between short- and long-term storage pools leads to a more detailed understanding of ecosystem carbon cycling. These issues are of primary relevance in evaluating the interactions and feedback effects between terrestrial ecosystems and greenhouse gases in the atmosphere.

The module was developed to simulate short- (e.g. 1 year) and long-term (e.g. 50 years) carbon allocation in a deciduous forest under different climate change scenarios. The computational formulation of the carbon allocation module was based on existing mechanistic models (TREEDYN3 Bossel, 1996; FAGUS Hoffman, 1995). The partitioning strategy of the module is based on source-sink relationships and characterised by a hierarchical structure. The different carbon sinks, namely maintenance and growth respiration of fruits, leaves, fine roots and woody biomass (subdivided in branches, stem and coarse roots) were ranked according to a priority level order. The sink with the highest priority (e.g. maintenance respiration) was computed to be supplied first; then, if some assimilates were left after its demand was met, the sink with the second priority (e.g. wood growth) was computed to be supplied and so on.

Two main improvements were made to this classical hierarchical organisation. First, an active role was given to the carbon reserve of the tree (non-structural carbohydrates), considered to be of crucial importance in determining the carbon allocation pattern and more generally to the survival of the tree. Secondly, the hierarchical organisation of carbon supply was made dynamic and flexible by allowing changes in sink priority over the time of the year and switching out some of the sinks according to the phenological phase (e.g. dormant season, bud-burst, full autotrophy of the leaves and senescence). Moreover, the module includes explicit formulation of all the relevant ecophysiological processes linked to carbon allocation, namely: vegetative respiration, biomass growth and losses, carbon relocation before abscission, crown competition for light, tree mortality and self thinning. For a detailed description of the CAF module see Campioli (2004) and Campioli et al. (2004).

6.2.2 Data

The reliability of a model can only be tested with accurate experimental data. The eddy covariance data collected by the EUROFLUX project (Moncrieff et al., 1997; Valentini, 1999; Aubinet et al., 2000) cover a wide range of forest types, stand age and locations throughout Europe. Moreover, the accessibility makes this dataset very convenient for the validation of a vegetation model.

The data used for the parameterisation and evaluation of the FORUG model are the flux and meteorological data for three beech (*Fagus sylvatica* L.) sites. The missing data are filled by means of look-up tables (Falge et al., 2001). The selected sites are listed in Table 6.1. Half-hourly meteorological data were used as model input.

Table 6.1 Selected EUROFLUX sites with beech as dominant species

Site Name	*Country*	*Latitude (N)*	*Longitude (E)*	*Elevation (m)*	*Age (y)*
Collelongo	***Italy***	***41°50'***	***13°35'***	***1550***	***100-105***
Hesse	***France***	***48°40'***	***07°05'***	***300***	***35***
Sorø	***Denmark***	***55°29'***	***11°38'***	***40***	***80***

The data used for the sensitivity and uncertainty analysis are the meteorological and flux data of the beech site in Hesse. The Hesse forest is 30 to 35 years old and well described by Granier et al. (2000, 2002).

6.2.3 Parameterisation

The flux model was parameterised mainly based on literature data for the three respective sites. When literature data were missing, parameterisation was done by minimising the Root Mean Square Error (RMSE) between simulated and measured GPP or TER. For this parameterisation, only daily values of GPP and TER were used, excluding interpolated values as these values are in fact also modelled and not measured. For a more detailed description of the parameterisation method see Samson et al. (2004). Verbeeck et al. (2006) present a list of the parameter values used for the flux model. For the CAF module the prior parameterisation was done only for Hesse forest. A detailed list of the parameter values of the CAF module used for the simulations for Hesse forest is given by Campioli (2004).

6.2.4 Model evaluation

The goodness-of-fit of the parameterised model was first evaluated by a visual comparison of the daily values of modelled and measured NEE, and by comparing the modelled annual fluxes of GPP and ecosystem respiration to observed values, and secondly by calculating the R^2, MSE_s and MSE_u for the carbon fluxes, as was explained by Kramer et al. (2002). The mean squared error (MSE) is defined as:

$$MSE = \frac{\sum (y_o - y_p)^2}{N}$$

where y_o and y_p are the observed and predicted values of the dependent variable, and N is the total number of observations. The use of MSE makes it possible to discriminate between systematic (MSE_s) and unsystematic errors (MSE_u). Kramer et al. (2002) describe in detail how MSE_s and MSE_u are calculated.

6.2.5 Sensitivity analysis

Process-based carbon flux models usually only predict discrete model outputs without mentioning the output uncertainty. To what extent can these model outputs be trusted? This question is important when predictions for the future are concerned. Uncertainty should be taken into account to answer it.

The Monte Carlo technique is a very useful and robust tool to propagate input uncertainty through a model in order to determine the output uncertainty. However, to carry out Monte Carlo simulations, the uncertainty distributions or the probability density functions (PDF) of the model inputs should be known. PDF's can be estimated based on experimental data. For more complex ecophysiological flux models however, the estimation of PDF's is one of the bottlenecks to perform uncertainty analyses.

When dealing with a large amount of parameters (the FORUG model has 54 parameters) the question rises whether it is necessary to take into account all parameters in the uncertainty analysis. To determine the parameters contributing most to the output uncertainty, a sensitivity analysis has to be conducted to rank the parameters. It is sufficient to account only for these most sensitive parameters in the uncertainty analysis. A tool to rank parameters for uncertainty is the use of the Monte Carlo technique in combination with a multiple linear regression, called a least square linearization (Lei and Schilling, 1996), which splits up output uncertainty into its sources.

Simulations were run in which inputs are assigned probability distributions and the effect of variance in the inputs on the output distribution is assessed. This sensitivity analysis is used to rank the parameters of the FORUG model for uncertainty. This method is very helpful in screening the most important inputs.

6.2.6 Uncertainty analysis

An uncertainty analysis is carried out to test the reliability of the FORUG model. The Monte Carlo technique is a numerical technique to calculate the output uncertainty of a model. It is robust and relatively easy to implement.

In Monte Carlo simulations, the PDF of the inputs is required. If enough data are available, input PDF's can be determined based on knowledge and measurements (e.g. found in literature). But unfortunately, information about distributions of parameters is often not available. As a result, input distributions are in many cases estimated based on "expert knowledge".

6.2.7 Scenario analysis

For the scenario analysis, a climate scenario created in the framework of the BELFOR project was used (Vande Walle and Lemeur, 2001). In the BELFOR project, site-specific synthetic weather data sets were developed on a sub-hourly basis for the 20th and 21st centuries from: (1) two to four years of measured meteorological data, depending on the site, and (2) outputs of a general circulation model (GCM), called CGCM, for the 1900-2100 period.

Climate change scenarios are tightly coupled to the predicted increase in atmospheric CO_2 due to human activities. Therefore the FORUG predictions for the 21st century are based on both climate change and atmospheric CO_2 rise scenarios. The site-specific synthetic weather data sets were generated for three of the BELFOR sites : Vielsalm, Brasschaat and Chimay.

CGCM computes the atmospheric general circulation in response to an increase in the concentration of atmospheric CO_2 over the 20th and 21st centuries based on the IS92a ("business as usual") IPCC scenario.

For the BELFOR project, monthly averages of GCM-simulated weather data were extracted to half hourly values for the climate parameters, for the two grid cells corresponding to the Belgian territory. The scenario for Brasschaat was taken here as the "Flemish" scenario used for the FORUG simulations. The IS92a CO_2 scenario of the IPCC was added to this scenario for Brasschaat.

According to CGCM simulations, temperature appears to be the only climatic variable which will be substantially modified in the course of the 21st century. Predicted temperature increase for Belgium from 2000 to 2100 approximates 3 ºC. Predicted changes for the other five climatic variables appear minimal for the 21st century.

6.3 RESULTS

6.3.1 Model evaluation

The goodness-of-fit test (Table 6.2) shows a good correspondence between measured and simulated daily NEE. Values found for R^2, MSE_s and MSE are comparable with values found by Kramer et al. (2002) for six process-based models.

Table 6.2 Results of the FORUG evaluation for beech

Site	R^2	MSE_s	MSE
Collelongo	0.84	0.39	2.39
Hesse	0.83	0.02	1.33
Sorø	0.79	0.10	1.03

6.3.2 Seasonal pattern of NEE

Figure 6.1 to 6.3 show the measured and modelled NEE for the beech site in Hesse for the years 1997 to 1999. The model gives an accurate prediction of the seasonal pattern.

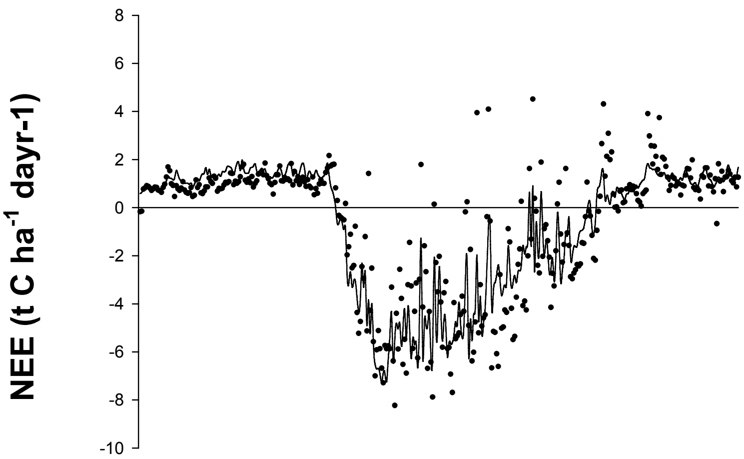

***Fig. 6.1** Seasonal pattern of the Net Ecosystem Exchange (NEE) (t C ha^{-1} $dayr^{-1}$) for beech in Hesse (1st January to 31st December 1997). FORUG simulations (line) compared with EUROFLUX measurements (dots).*

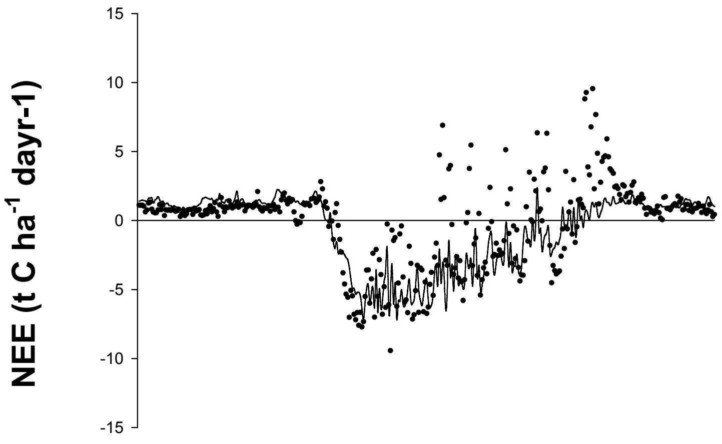

Fig. 6.2 *Seasonal pattern of the Net Ecosystem Exchange (NEE) (t C ha^{-1} dayr^{-1}) for beech in Hesse (1st January to 31st December 1998). FORUG simulations (line) compared with EUROFLUX measurements (dots)*

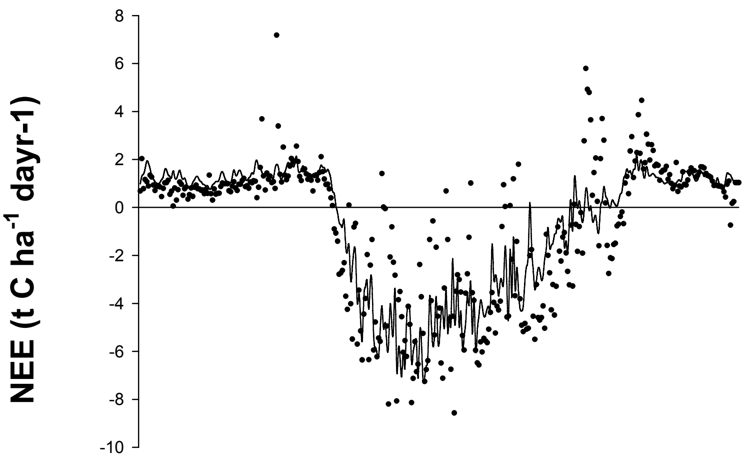

Fig. 6.3 *Seasonal pattern of the Net Ecosystem Exchange (NEE) (t C ha^{-1} dayr^{-1}) for beech in Hesse (1st January to 31st December 1999). FORUG simulations (line) compared with EUROFLUX measurements (dots)*

6.3.3 Seasonal pattern of allocation

Figure 6.4 gives the simulated seasonal pattern of the carbon stocks in the wood, the leaves and the reserve pool. The simulations of the CAF module are given for the years 1997 and 1998 for the beech forest in Hesse.

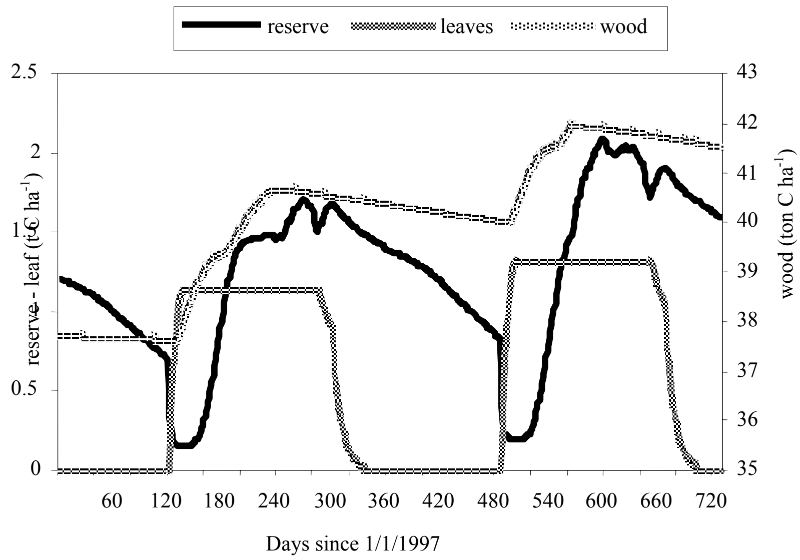

Fig. 6.4 *Seasonal pattern of the carbon stock in the wood, the leaves and the reserve pool for the Hesse beech forest for the years 1997 and 1998*

6.3.4 Sensitivity analysis

The sensitivity analysis resulted in a ranking of the FORUG parameters based on their contribution to the uncertainty on the NEE. For the 10 most important parameters, the contribution expressed as percentage can be found in Table 6.3. This table shows that more than 97% of the output uncertainty is caused by the uncertainty of these 10 parameters. This means that the other 44 parameters determine only 4% of the output uncertainty and are less important.
More than 63% of the overall output uncertainty is determined by the two coefficients (a_{soil} and b_{soil}) of the soil respiration model. The soil respiration is calculated using a simple exponential equation:

$$R_{soil} = a_{soil}\exp(b_{soil}T_{soil}) \qquad \text{(Granier et al., 2002)}$$

Table 6.3 also shows that the light extinction coefficient for diffuse radiation (k_d) plays an important role. Other sensitive parameters appearing in the list are connected to the photosynthesis process: the initial quantum yield (α_F), the maximum carboxylation rate (V_{cmax}), the activation energy of the temperature dependence of V_{cmax} (E_{av}) and the Michaelis-Menten constant for carboxylation (K_c). The rest of the list contains less sensitive parameters: the Michaelis Menten constant for oxygenation (K_o), the CO_2 compensation point (Γ^*) and the de-activation energy of the temperature dependence of J_{max} (H). The other 44 parameters don't appear in the list and have a contribution that is even smaller than 0.55%.

Table 6.3 *Results of the ranking for uncertainty. The contribution (%) of the uncertain parameters to the overall uncertainty on the FORUG model output: Net Ecosystem Exchange (NEE). These results are based on 2000 Monte Carlo simulations for the year 1997 for the Hesse forest in France*

Parameter	**Description**	**Process**	**% contribution to the overall uncertainty**
b_{soil}	Coefficient	Soil respiration	49.67
k_d	Extinction coefficient for diffuse radiation	Light extinction	14.91
a_{soil}	Coefficient	Soil respiration	13.66
α_F	Initial quantum yield	Photosynthesis	5.24
V_{cmax}	Maximum carboxylation rate	Photosynthesis	4.51
K_c	Michaelis-Menten constant for the carboxylation	Photosynthesis	3.17
E_{av}	Activation energy of temperature dependence of $V_{c\max}$	Photosynthesis	2.80
Γ^*	CO_2 compensation point	Photosynthesis	1.84
H	De-activation energy of temperature dependence of $J_{\max}$	Photosynthesis	1.02
K_o	Michaelis-Menten constant for oxygenation	Photosynthesis	0.55

6.3.5 Uncertainty analysis

The distribution of the simulated NEE for the year 1997 for the Hesse forest is shown in Figure 6.5. This distribution is based on 2000 Monte Carlo simulations. The uncertainty analysis only accounted for the 10 parameters appearing in Table 3. The mean simulated output value is -3.63 t C ha^{-1} yr^{-1} (a negative value of NEE indicates a net carbon uptake by the ecosystem). The standard deviation of the output distribution is 0.88 t C ha^{-1} yr^{-1}. The variance is 0.77. In the Euroflux project, a total NEE value of -2.83 t C ha^{-1} yr^{-1} was measured for the year 1997. The measured value of the NEE differs from the mean simulated value, but falls within the range of one standard deviation of the output distribution.

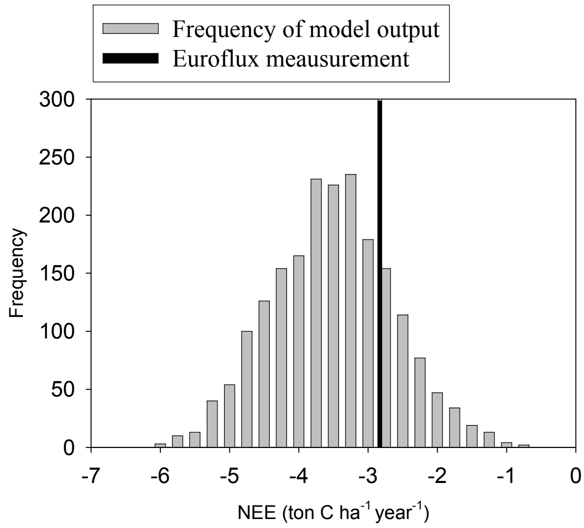

Fig. 6.5 *Distribution of the total Net Ecosystem Exchange (NEE, t C ha^{-1} yr^{-1}) in 1997 for the Hesse forest in France. The calculation of this distribution is based on 2000 Monte Carlo simulations. Only the uncertainty of 10 key parameters was taken into account. The black bar shows the measured (Euroflux) NEE value and does not represent a frequency*

The uncertainty on the daily NEE is illustrated in Figure 6.6 by an example of a 50 days period in the summer of 1997 for the Collelongo beech forest in Italy. In this example, we test if the measurements fall within the model uncertainty. The black dots represent the daily sums of the measured NEE. Based on 5000 Monte Carlo simulations, the output distribution of the NEE was calculated. The simulated distributions are shown as the 5% and 95% confidence boundaries around the mean modelled NEE. So this is not a simulation of a discrete time series, but a band which contains 90% of all Monte Carlo simulations.
Figure 6.6 shows that most of the measured points fall within the confidence intervals of the model predictions. Comparable results were found for the rest of the growing season (data not shown).

6.3.6 Scenario analysis

The IS92a scenario was applied on a virtual Flemish beech forest planted in 1967 (forest A). In this scenario, the GPP rises from 14.6 t C ha^{-1} yr^{-1} in 1997 up to 21.5 t C ha^{-1} yr^{-1} in 2100 (Figure 6.7). This is an increase of 7 t C ha^{-1} yr^{-1}. The TER rises from 10.1 t C ha^{-1} yr^{-1} to almost 18.9 t C ha^{-1} yr^{-1} (Figure 6.8). This is an increase of 9 t C ha^{-1} yr^{-1}. The NEE will fluctuate in this scenario between -2 t C ha^{-1} yr^{-1} and -4 t C ha^{-1} yr^{-1} (Figure 6.9).

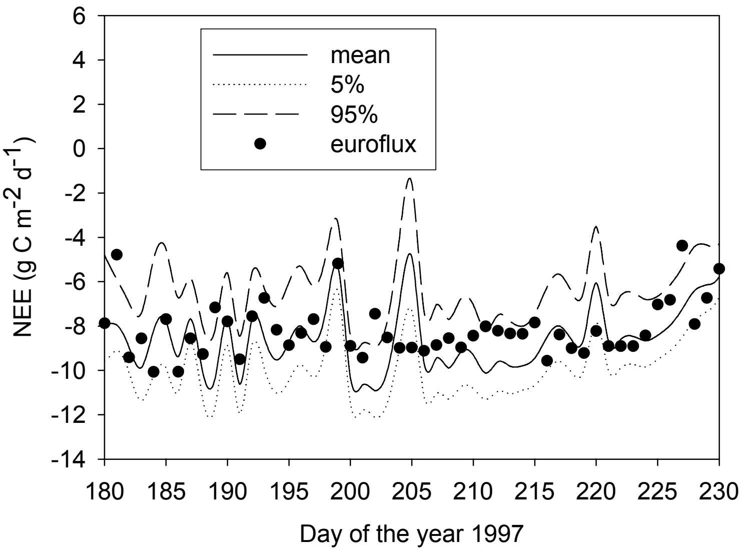

Fig. 6.6 *Measured (dots) Net Ecosystem Exchange (NEE) and mean predicted (full line) NEE with confidence intervals (dashed and dotted lines) for a 50 days period in the 1997 growing season for the Collelongo site (Italy). Predicted confidence intervals are based on 5000 Monte Carlo simulations*

For the evolution of the C stock in woody biomass, two virtual forests were considered. The first forest (A) is the forest planted in 1967. This forest is 30 years old in 1997, which is comparable to the Hesse forest. The second virtual forest (B) is planted in 2005. This forest will be 30 years old in 2035. The simulations were started in 1997 and 2035 respectively, because parameters of an adult beech forest were used. Both forests started with a woody biomass of 37.5 t C ha^{-1}. Forest (A) ends in 2100 with a woody biomass of 105 t C ha^{-1}. Forest (B) ends in 2100 with a woody biomass of 104 t C ha^{-1} (Figure 6.10). Forest (B) is growing faster because of the higher temperatures and the higher atmospheric CO_2 concentrations. From 1997 to 2007 forest (A) has a mean wood increment of 2.0 t C ha^{-1} yr^{-1}. Forest (B) has a mean wood increment of 2.6 t C ha^{-1} yr-1 from 2035 to 2045. We can conclude that forest (B) will reach the commercial timber age relatively faster than forest (A). However, we can not conclude that forest (B) has sequestered more C in a shorter period, because the most important long-term carbon stock, the soil C stock, is not considered here.

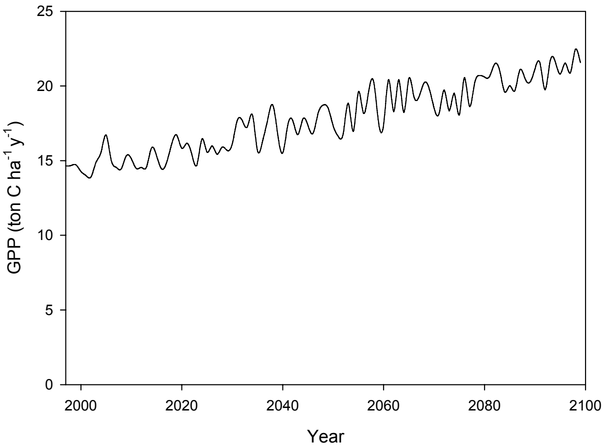

Fig. 6.7 *GPP (ton C ha^{-1} yr-1) scenario (1997-2100) for beech in Flanders*

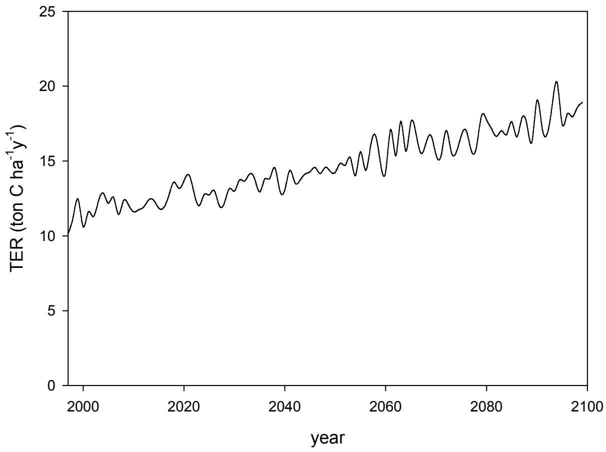

Fig. 6.8 *TER (ton C ha^{-1} yr-1) scenario (1997-2100) for beech in Flanders*

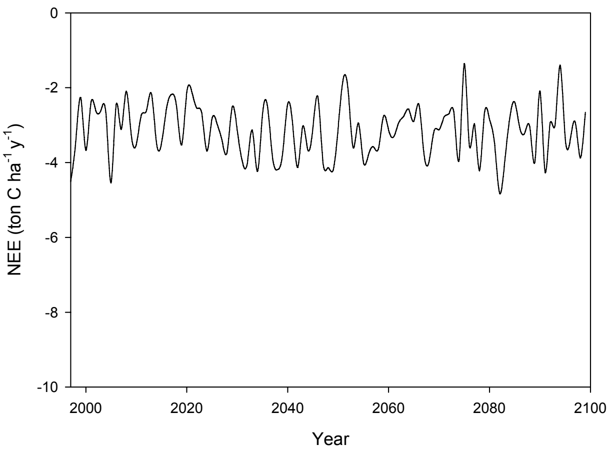

Fig. 6.9 *NEE (ton C ha^{-1} yr-1) scenario (1997-2100) for beech in Flanders*

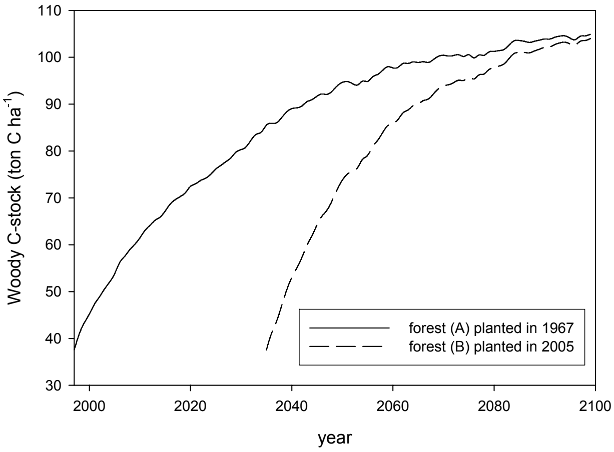

Fig. 6.10 *C stock in the woody biomass (ton C ha^{-1}) scenario's (1997/2035-2100) for beech in Flanders*

In Table 6.4 the results of the scenario study are compared with the results of the scenario study of the BELFOR project conducted with the ASPECTS model (Vande Walle and Lemeur 2001). The results for GPP are comparable. The increase in mean annual GPP at the end of the 21st century is in both cases 43% compared to the mean annual GPP of the 20th century. But for the NEE, there is a major difference between the FORUG prediction and the ASPECTS results. The ASPECTS model simulates an increase of 26% of the NEE, while the FORUG model gives a decrease of 4.4%. This small decrease in NEE is due to the fact that the increase of GPP is coupled with an equivalent increase in TER in the FORUG scenario analysis. The ASPECTS simulations resulted in a smaller increase in TER, which can be partly due to the scenario of Vielsalm.

Table 6.4 *Results of the scenario study compared with the scenario study of the BELFOR project conducted with the ASPECTS model. Mean values of GPP and NEE (t C ha^{-1} yr^{-1})*

	FORUG (beech, virtual Forest A)		ASPECTS (beech, Vielsalm)	
	GPP	NEE	GPP	NEE
1990-1999	14.8	-3.12	12.5	-2.41
2090-2099	21.3	-2.98	17.9	-3.04
% increase	43.4%	-4.4%	43.2%	26.1%

6.4 CONCLUSION

The comparison of FORUG simulations with EUROFLUX measurements has shown that the FORUG model is able to predict seasonal patterns of NEE for temperate forests at different latitudes in Europe. The new CAF module created the ability to simulate the evolution of the woody carbon stock of forests.

The sensitivity analysis revealed the critical FORUG parameters. Future research should focus on these critical parameters and their corresponding processes. A better description of poorly described key processes is recommended. In particular the exponential soil respiration model needs revision. A more complex model taking into account soil water content, will probably temper the high sensitivity for the soil respiration parameters.

The analysis of the output uncertainty resulted in a standard deviation of 0.88 t C ha^{-1} yr^{-1}, which is 24% of the mean value of NEE. Hirsch et al. (2004) found a higher uncertainty of 35% of the mean value of the net carbon flux simulated with the CARLUC (3PG) model for the Brazilian Amazon.

The scenario analysis showed that the IS92a scenario will cause a major increase in GPP and TER. But the increase will be of the same order of magnitude which involves only minor effects on the NEE. The scenario had also a major effect on the wood growth. The wood increment of a 30 year old forest is higher in 2035 than in 1997. Unfortunately, we can not predict the effect of this global change scenario on the long term carbon stock in the soil.

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APPENDICES

Appendix 1 Volume (m³) harvested wood products for the Flemish (non-private) forests, derived from the IVANHO database (FGAD)

Species	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Spruce												61	1075	1131	4219	3646
Douglas fir												60	900	1119	4232	4217
Larch												206	2856	4146	7775	6490
Pine												7558	73458	76478	125993	105095
Other coniferous												262	6890	6314	5371	2568
Sub-total coniferous												8147	85178	89188	147589	122017
Beech												718	13466	16440	13202	23159
Oak												486	3277	3445	3972	4686
"Noble" deciduous												835	11298	12984	17286	20027
Other deciduous												585	1444	4567	3431	5271
Poplars												4290	10562	24985	24913	24534
Sub-total deciduous												6914	40046	62421	62804	77677
Total	**45600**	**104210**	**34475**	**47047**	**38613**	**72447**	**58607**	**51256**	**73514**	**83114**	**88496**	**15061**	**125225**	**151609**	**210393**	**199693**

Appendix 2 Annual growth increment for Flemish forests

Experimental

Level II site	**plant year**	**species**	**annual growth ($m^3ha^{-1}y^{-1}$)**
11 Wijnendale	1923	beech	**10,12**
12 Meerdaal		Q robur	**9,99**
		Q petrea	**9,12**
13 Hallerbos	1944	beech	**17,15**
14 Ravels	1930	Pinus nigra	**14,88**
15 Brasschaat	1929	Pinus Sylvestris	**7,14**
17 Buggenhout	1845	beech	**12,43**
		Q robur	**3,39**
18 Houthulst	1922	Q robur	**9,71**
		beech	**7,08**
		birch	**3,46**
		douglas fir	**5,89**
19 Pijnven	1930	Pinus Sylvestris	**13,37**
20 Zutendaal	1943	Pinus Sylvestris	**9,17**
21 Zoniën	1906	beech	**17,11**
		larix E	**13,57**

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Analyse van de luchtverontreiniging en de gevolgen daarvan op het bosecosysteem. SEB, Labo voor Bosbouw, 169p. (In Dutch)

Literature review (Löwe et al. 2000)

species	**mean annual growth ($m^3ha^{-1}j^{-1}$)**
Pine	**7,0**
Douglas fir	**16,0**
Larch	**8,5**
Spruce	**10,5**
Other conifers	**10,0**
Beech	**5,2**
Oak	**6,0**
Mixed noble deciduous	**5,9**
Other deciduous	**5,9**
Poplars	**13,0**

Remark:
On average the mean annual growth for all forests in Flanders is considered 5 $m^3ha^{-1}y^{-1.}$
Thus, in order to comply with the aims of sustainable forest management, the maximum allowable yearly harvest amounts to 4 m³ha-1y-1 (FGAD).

Appendix 3 Information on the applied rotation length, according to the guidelines for forest management plans (FGAD)

Minimum criteria for harvest (final cut) :

	minimum age (years)	mean CBH (cm)
coniferous	**60**	
poplar	**15**	
Beech		**150**
Quercus rubra		**150**
Quercus robur		**150**
Quercus petraea		**150**
Maple		**120**
Prunus avium		**120**
Ash		**90**

Depending on the site index, the age of harvest differs.
The general policy is to vary the rotation lengths and 'the older the better'.

Criteria for thinnings :

	Age (years)	Rotation length (years)
Coniferous	**< 40**	**3**
	> 40	**6**
	70 - 90	**9 - 12**
deciduous	**< 70 – 80**	**4 – 6**
	> 70 – 80	**8 - 12**

Appendix 4 Used in the uncertainty assessment – prediction towards first commitment period (EFOBEL)

	FL-volume (m3)	%uncertainty	SE		RW - volume (m3)	%uncertainty	SE
Spruce	545066	20,06%	54670	Spruce	52502800	2,20%	589317
Douglas fir	358678	31,31%	56154	Douglas fir	2387200	13,20%	160771
Larch	775204	17,67%	68488	Larch	2081200	9,50%	100874
Pine	12837128	3,42%	219239	Pine	3743300	6,80%	129870
Other resinous	197602	44,33%	43799	Other resinous	4955300	7,50%	189616
Sub-total resinous	**14713679**	**3,29%**	**246611**	**Sub-total resinous**	**65669800**	**1,97%**	**660407**
Beech	2470699	12,53%	154745	Beech	12277965	4,70%	294421
Oak	3683966	8,48%	156225	Oak	20372371	3,00%	311822
"Noble" deciduous	2325889	10,87%	126444	"Noble" deciduous	15041416	4,20%	322316
Other resinous	3742446	8,07%	150981	Other resinous	9661648	5,20%	256329
Poplars	5217264	7,28%	189865	Poplars	2703900	17,10%	235901
Sub-total deciduous	**17440264**	**3,94%**	**350976**	**Sub-total deciduous**	**60057300**	**2,09%**	**639655**
TOTAL	**32153943**	**2,61%**	**428953**	**TOTAL**	**125727100**	**1,43%**	**919400**

	FL-area (ha)	%uncertainty	SE		RW - area (ha)	%uncertainty	SE
Spruce	2930	5,00%	75	Spruce	172400	0,90%	792
Douglas fir	1280	5,00%	33	Douglas fir	10800	4,50%	248
Larch	3060	5,00%	78	Larch	8300	5,20%	220
Pine	63600	5,00%	1622	Pine	15300	3,70%	289
Other resinous	1010	5,00%	26	Other resinous	20700	3,20%	338
Sub-total resinous	**71880**			**Sub-total resinous**	**227500**	**0,83%**	**967**
Beech	7710	5,00%	197	Beech	42300	2,10%	453
Oak	14270	5,00%	364	Oak	82100	1,50%	628
"Noble" deciduous	10170	5,00%	259	"Noble" deciduous	57300	1,90%	555
Other resinous	21650	5,00%	552	Other resinous	43400	2,00%	443
Poplars	19020	5,00%	485	Poplars	9900	4,80%	242
Coppice only	0	5,00%	0	Coppice only	15300	10,00%	781
Sub-total deciduous	**72820**			**Sub-total deciduous**	**250300**	**0,98%**	**1252**
TOTAL	**144700**			**TOTAL**	**477800**	**0,65%**	**1582**

Area calculations are based on the share of each species in the total forest basal area conform the methodology suggested by FGAD.
Volume calculations and volume uncertainty assessment is based on bootstrap resampling techniques.
Error propagation is achieved through Monte Carlo simulation in the EFOBEL model (Perrin et al. 2000).

Appendix 5 Biomass expansion and conversion factors applied for the carbon accounting in Belgian forests (CASTEC, 2005)

Parameter	*Unit*		*Pine*	*Douglas fir*	*Larch*	*Spruce*	*Beech*	*Oak*	*Mixed 'noble'*	*Poplars*
Density	ton DM m^{-3}	min.	0,390	0,370	0,410	0,340	0,550	0,500	0,520	0,340
		max.	0,600	0,540	0,550	0,450	0,720	0,720	0,690	0,550
		median	0,480	0,450	0,470	0,380	0,560	0,600	0,590	0,410
BEF 1 : aboveground biomass / total solid wood	ton DM ton^{-1} DM	min.	1,140	1,180	1,140	1,140	1,160	1,240	1,290	n.a.
		max.	1,397	2,240	1,356	1,710	2,040	1,390	1,290	n.a.
		median	1,318	1,280	1,296	1,290	1,340	1,315	1,290	n.a.
BEF 2 : belowground biomass / aboveground biomass	ton DM ton^{-1} DM	min.	0,160	0,170	n.a.	n.a.	0,230	n.a.	n.a.	n.a.
		max.	0,160	0,170	n.a.	n.a.	0,250	n.a.	n.a.	n.a.
		median	0,160	0,170	n.a.	n.a.	0,240	n.a.	n.a.	n.a.
BEF 3 : total AG + BG biomass / total solid wood	ton DM ton^{-1} DM	min.	1,430	1,500	1,500	1,500	1,500	1,500	1,500	1,500
		max.	2,000	2,000	2,000	2,000	1,750	1,500	1,500	1,500
		median	1,500	1,710	1,750	1,750	1,670	1,500	1,500	1,500
Carbon content	ton C ton^{-1} DM	min.	0,400	0,500	0,400	0,400	0,437	0,450	0,500	0,500
		max.	0,554	0,500	0,500	0,512	0,507	0,500	0,500	0,500
		median	0,500	0,500	0,500	0,500	0,490	0,500	0,500	0,500
			0,36	0,38475	0,41125	0,3325	0,458248	0,45	0,4425	0,3075

Parameter	*Unit*		*Coniferous*	*Deciduous*	*Coppice*
Density	ton DM m^{-3}	min.	0,350	0,380	0,00
		max.	0,500	0,770	0,00
		median	0,400	0,550	0,00
BEF 1 : aboveground biomass / total solid wood	ton DM ton^{-1} DM	min.	1,140	1,240	0,00
		max.	1,712	1,400	0,00
		median	1,333	1,320	0,00
BEF 2 : belowground biomass / aboveground biomass	ton DM ton^{-1} DM	min.	0,180	0,200	0,00
		max.	0,250	0,220	0,00
		median	0,200	0,210	0,00
BEF 3 : total AG + BG biomass / total solid wood	ton DM ton^{-1} DM	min.	1,500	1,500	0,00
		max.	2,000	1,500	0,00
		median	1,750	1,500	0,00
Carbon content	ton C ton^{-1} DM	min.	0,400	0,450	0,00
		max.	0,500	0,500	0,00
		median	0,500	0,500	0,00

Appendix 6 Peer reviewed articles published or in preparation based on research results of the CASTEC project

Beheydt D, Boeckx P, Clough TJ, Vermeulen J, Sherlock RR, Van Cleemput O (2005). Methods to adjust for the interference of N_2O on $\delta^{13}C$ and $\delta^{18}O$ measurements of CO_2 from soil mineralization. Rapid Commun. Mass Sp. 19:1365-1372.

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Beheydt D, Berntsen J, Boeckx P, Olesen JE, Van Cleemput O (in preparation). Comparison and sensitivity of direct N_2O emission estimates from arable soils using the DNDC and FASSET model.

Beheydt D, Boeckx P, Sleutel S, De Neve S, Hofman G, Li C, Van Cleemput O (in preparation). Regional direct N_2O emissions from an intensive agricultural system using the DNDC model.

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Carlier L, De Vliegher A, Rotar I (2005). Importance and functions of European grasslands. Buletin USAMV-CN 61:17-26.

Iantcheva A, Mestdagh I, Lootens P, Carlier L (2004). Assessment of seasonal variability of belowground biomass in Belgian grasslands influenced by management treatments. Bulgarian J. Agricul. Sci. 10:15-23.

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Mestdagh I, Sleutel S, Lootens P, Van Cleemput O, Carlier L (2005a). Soil organic carbon stocks in verges and urban areas of Flanders, Belgium. Grass Forage Sci. 60:151-156.

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Sleutel S, De Neve S, Beheydt D, Li C, Hofman G (submitted). Regional simulation of long-term organic carbon stock changes in cropland soils using the DNDC model: 1. Large scale model validation to an extensive database. Soil Use Manage.

Sleutel S, De Neve S, Beheydt D, Li C, Hofman G (submitted). Regional simulation of long-term organic carbon stock changes in cropland soils using the DNDC model: 2. Scenario analysis of management options. Soil Use Manage.

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Vande Walle I, Van Camp N, Van de Casteele L, Verheyen K, Lemeur R (submitted). Short-rotation forestry of birch, maple, poplar and willow in Flanders (Belgium) I. Biomass production after 4 years of tree growth. Biomass Bioenerg.

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Van Waes C, Mestdagh I, Lootens P, Carlier L (2005). Possibilities of near infrared reflectance spectroscopy for the prediction of organic carbon concentrations in grassland soils. J. Agricul. Sci. 143:487-492.

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SPSD II (2000-2005)

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EV-12 CARBON SEQUESTRATION POTENTIAL IN DIFFERENT BELGIAN TERRESTRIAL ECOSYSTEMS: QUANTIFICATION AND STRATEGIC EXPLORATION (CASTEC) SPSD II