Carbon sequestration in short-rotation forestry plantations and in Belgian forest ecosystems

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Carbon sequestration in short-rotation forestry plantations and in Belgian forest ecosystems

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Koolstofvastlegging in plantages met korte omloopstijd en in Belgische bosecosystemen

Illustrations on the cover :
Front : birch trees at a short-rotation plantation (Zwijnaarde)
Back : Maskobossen (Jabbeke)

Illustrations between chapters :
Page 15 : short-rotation plantation (Zwijnaarde), washing of roots (greenhouse at the Faculty of Bioscience Engineering, Gent)
Page 115 : Aelmoeseneiebos (Gontrode), Maskobossen (Jabbeke), Merkenveld (Zedelgem), domain Cellen (Oostkamp)
Page 245 : Merkenveld (Zedelgem)

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Woord vooraf

In principe is het schrijven van een 'Woord vooraf' niet zo'n moeilijke opdracht. Er is immers geen statistiek voor nodig, er moet niets bewezen worden, en verwijzingen naar andere publicaties zijn ook al niet van toepassing. In de praktijk echter is het niet zo eenvoudig om in enkele woorden of zinnen alle mensen te bedanken die er toe bijgedragen hebben dat ik de afgelopen tien jaar met veel plezier wetenschappelijk onderzoek heb uitgevoerd, en dat ik er in geslaagd ben het boek dat nu voor u ligt af te werken.

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Zerkegem, maart 2007

Contents

PART A. SHORT-ROTATION FORESTRY PLANTATIONS

Chapter 5. Temporal variation and high resolution spatial heterogeneity in soil CO_2 efflux in a short-rotation tree plantation 67

Chapter 6. Carbon balance of a short-rotation tree plantation during the first five years of tree growth 89

PART B. BELGIAN FOREST ECOSYSTEMS

Chapter 8. Carbon sequestration in the Aelmoeseneie experimental forest from 1991 to 2005 131

Chapter 9. Growing stock-based assessment of the carbon stock in the Belgian forest biomass 153

Abbreviations and conversion factors

Abbreviations

α_c	quantum efficiency
ΔC_b	change in carbon stock in living biomass
ΔC_{dead}	change in carbon stock in dead biomass
ΔC_{soil}	change in soil carbon stock
A	photosynthetic rate
ABGC	above- and belowground carbon
$AC(pot)_{dep}$	potentially acidifying deposition
AG	aboveground
AGDM	aboveground dry mass
$AGDM_{mean}$	mean aboveground dry mass per tree
A_n	net photosynthetic rate
ARD	afforestation, reforestation and deforestation
AS	ash stand
BA	basal area
BB	budburst
BEF	biomass expansion factor *s.s.*
BEF *s.l.*	biomass expansion factor *s.l.* (collective expression for WD, BEF and CC)
BG	belowground
C	carbon
CAL	calorific value
CBH	circumference at breast height
CC	carbon content
$C_{dead\ LL}$	carbon stock in the litter layer
$C_{dead\ trees}$	carbon stock in lying dead trees
CE	conversion efficiency
CE_e	conversion efficiency for electricity production
CE_{th}	conversion efficiency for heat production
CFC	chlorofluorocarbon
CH_4	methane
C_i	internal CO_2 concentration
CO_2	carbon dioxide
COP	Conference of the Parties
CV	coefficient of variation
D	stem density of a plot
d_{30}	diameter at 30 cm height
D_{act}	actual stem density of a plot

DBH	diameter at breast height
D_{ini}	initial stem density of a plot
DM	dry mass
DOY	day of the year
DsBB	number of days since budburst
EC	eddy covariance
EE	energy efficiency
EM_c	amount of CO_2 emitted during a traditional electricity production process
EN_{conv}	energy input needed for conversion of biomass to usable energy
EN_{DM}	amount of energy stored in biomass
EN_e	amount of electrical energy
EN_{est}	energy input needed for the establishment of a SRF plantation
EN_{input}	amount of energy needed to produce and transport biomass, and to convert biomass into usable energy
EN_{output}	amount of usable energy produced
EN_{th}	amount of thermal energy
EN_{trans}	energy input needed to prepare and transport harvested wood
ER_c	CO_2 emission reduction potential
F	foliage production
FAWS	forests available for wood supply
FNAWS	forests not available for wood supply
GHG	greenhouse gas
GPP	gross primary productivity
GWP	global warming potential
H	tree height
Harv	harvest
HFC	hydrofluorocarbon
I	incident light level
I_c	light compensation point
IM	impact factor
Incr	increment
IPCC	Intergovernmental Panel on Climate Change
IR	infrared
J_{max}	potential electron transport rate at 25 °C
KP	Kyoto Protocol
LA	individual leaf area
LAI	leaf area index
LAI_{max}	maximum leaf area index
LDW	leaf dry weight
LFW	leaf fresh weight
LOI	loss-on-ignition
LWC	leaf water content
MAI	mean annual increment

MSE	mean squared error
N	nitrogen
N_2O	nitrous oxide
NBP	net biome productivity
NEE	net ecosystem exchange
NEP	net ecosystem productivity
NEP_{mass}	net ecosystem productivity determined by the mass-balance approach
NEP_{eco}	net ecosystem productivity determined by the ecological approach
NPP	net primary productivity
OB	oak-beech stand
P	phosphorus
PDM_{mean}	mean planted aboveground dry mass per tree
PFC	perfluorocarbon
PROD	biomass production of a plot
$PROD_{act}$	actual biomass production of a plot
$PROD_{pot}$	potential biomass production of a plot
R	root production
R:S	root-to-shoot ratio
R_a	autotrophic respiration
R_d	dark respiration
Resx	result of scenario x
R_f	foliage respiration
RFI	regional forest inventory
RGR	relative growth rate
R_h	heterotrophic respiration
RI	radiation integral
RMI	Royal Meteorological Institute of Belgium
RMSE	root mean squared error
R_r	root respiration
R_w	wood respiration
SF_6	sulphur hexafluoride
SLA	specific leaf area
SLA_{max}	maximum observed specific leaf area
SMC	volumetric soil moisture content
SOC	soil organic carbon
SOM	soil organic matter
SR	soil CO_2 efflux or soil respiration
SRF	short-rotation forestry
ST	soil temperature
SuR	survival rate
SVAR	semivariance
TA	total area of plantations
TABGC	total above- and belowground carbon

TBA	total basal area
T_{base}	base temperature
TER	total ecosystem respiration
TSB	total stem biomass
TSW	total solid wood, referring to the combination of stem and branches with a circumference exceeding 22 cm
TT	thermal time
TTSW	total total solid wood
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
V	stem volume
V_{cmax}	maximum rate of Rubisco activity at 25 °C
W	wood production
WD	wood density
WMO	World Meteorological Organization

Conversion factors

For the convenience of the reader, frequently used conversion factors are listed below :

$$12\ \mathrm{g\ C} \leftrightarrow 44\ \mathrm{g\ CO_2}$$

$$1\ \mathrm{\mu mol\ CO_2\ m^{-2}\ s^{-1}} \leftrightarrow 0.1584\ \mathrm{g\ CO_2\ m^{-2}\ h^{-1}}$$

$$6.31\ \mathrm{\mu mol\ CO_2\ m^{-2}\ s^{-1}} \leftrightarrow 1\ \mathrm{g\ CO_2\ m^{-2}\ h^{-1}}$$

$$100\ \mathrm{g\ C\ m^{-2}} \leftrightarrow 1\ \mathrm{t\ C\ ha^{-1}}$$

Chapter 1

General introduction

1.1. The enhanced greenhouse effect and the global carbon cycle

Analysis of air bubbles trapped in Antarctic ice revealed huge fluctuations in atmospheric carbon dioxide (CO_2) concentrations over the past half million years (Fig. 1.1). Periods of low CO_2 concentration correspond to ice ages, while higher carbon dioxide concentrations are linked to warmer eras. During this period, atmospheric CO_2 concentrations ranged between 180 ppm and 300 ppm (CDIAC 2006). The last ice age ended 14000 years ago, as atmospheric CO_2 levels rose from below 200 ppm to about 280 ppm (NASA 2006). Since then, atmospheric CO_2 levels stayed fairly constant at about 280 ppm (Barnola 1999).

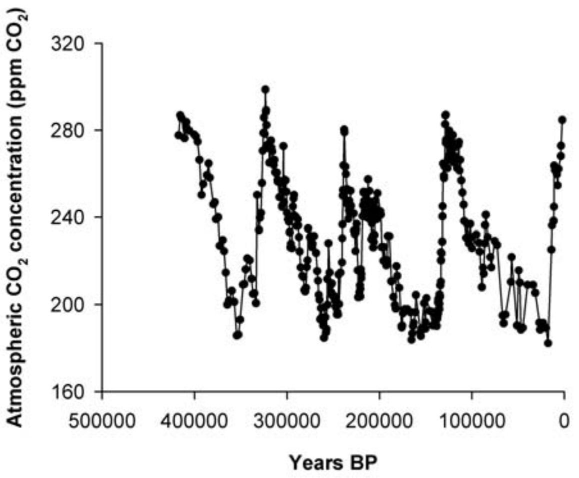

Figure 1.1. *Long-term record of atmospheric carbon dioxide concentrations obtained from Antarctic ice cores, taken at Vostok, Antarctica (graph based on data from CDIAC (2006))*

Intensified human activities, mainly fossil fuel burning and deforestation, increased carbon dioxide emissions since the dawn of the industrial revolution, about two centuries ago. This resulted in a steadily rising atmospheric CO_2 concentration (NASA 2006). As depicted in Fig. 1.2a, atmospheric CO_2 concentrations increased from 285 ppm in 1850 to 310 ppm in 1950. A further increase to 378 ppm CO_2 in 2004 was observed at the measuring station at Mauna Loa (Hawaii; Fig. 1.2b). As such, atmospheric CO_2 concentrations are higher today than they have been over the last half million years or longer (NASA 2006). Moreover, the rate of change in atmospheric CO_2 concentration, which was 1.3 ppm year^{-1} during the last 46 years on average, and 1.8 ppm year^{-1} in the period 1994 to 2004, is higher than has been observed ever before (Bolin and Sukumar 2000).

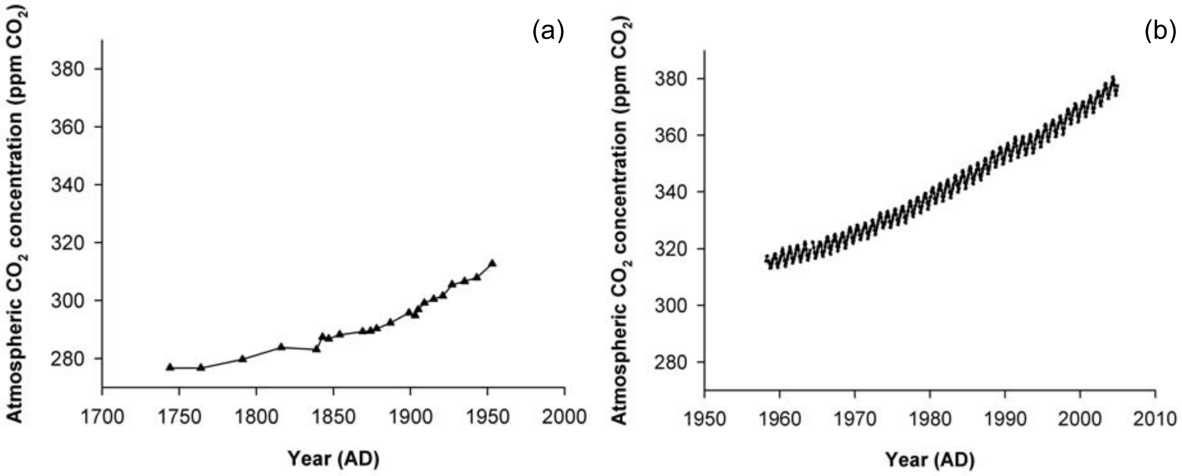

Figure 1.2. *Atmospheric carbon dioxide concentrations obtained from a) measurements on air occluded in Antarctic ice cores, taken at Siple Station and b) measurements on air samples taken at the measuring station at Mauna Loa, Hawaii (graphs based on data from CDIAC (2006)). The oscillations in atmospheric CO_2 concentrations depicted in b) are the result of the change in photosynthetic capacity of major terrestrial ecosystems over the year.*

The increasing amount of CO_2 in the atmosphere disturbs the natural greenhouse effect, of which a simplified scheme is given in Fig. 1.3. Most incoming (short-wave) solar radiation is absorbed by the earth's surface, and some is reflected back to space. On average, for the earth as a whole, incoming solar radiation is balanced by outgoing terrestrial radiation. Some of the (long-wave) infrared (IR) radiation emitted by the earth's surface passes relatively unimpeded through the atmosphere. The bulk of the IR radiation, however, is intercepted and absorbed by the atmosphere which in turn emits radiation both up- and downwards. The atmosphere consists mostly of nitrogen and oxygen (78 and 21 % on dry air, respectively), which are transparent to IR radiation. Other gases as water vapour, carbon dioxide and methane are present in much smaller quantities in the atmosphere, but they absorb and re-emit a large part of the thermal radiation leaving the earth's surface, and are therefore called greenhouse gases (GHGs) (Bengtsson 1994, Trenberth *et al.* 1995). Because of the heat-trapping characteristics of these gases, the lower part of the atmosphere and the earth's surface are warmed, and the average global surface air temperature, excluding Antarctica, is about 16 °C. Without this natural greenhouse effect, mean surface air temperature would be -18 °C, and life on earth would be impossible (Bengtsson 1994, AAS 2006).

The increasing release of greenhouse gases to the atmosphere by human activities results in an enhanced greenhouse effect, due to the increased heat-trapping capacity of the atmosphere (Walker *et al.* 1999, EPA 2006a). Consequently, global mean surface temperature is rising (Kasting 1998), and climates are changing. Measurements have shown that mean global surface temperature has increased by about 0.3 °C to 0.6 °C since the late 19th century, and by about 0.2 °C to 0.3 °C over the period from 1954 to 1994 (Houghton *et al.* 1995).

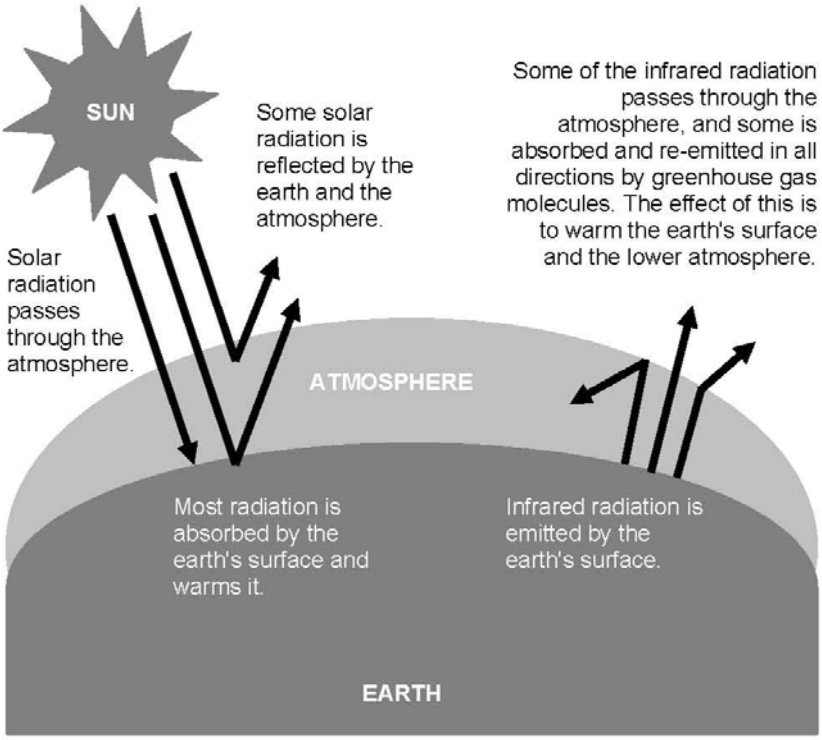

Figure 1.3. *Simplified overview of the greenhouse effect (after EPA 2006a)*

The effects of global warming are widespread and are expected to be disastrous (Grace 2004). Global warming could result in more frequent and more extreme weather events, such as droughts and floods. Evaporation will increase as climate warms, which will increase average global precipitation. Even relatively small rises in sea level, resulting from expansion of the oceans and retreating glaciers and ice-caps (Bengtsson 1994), will make densely settled coastal plains uninhabitable while an increased risk of certain diseases and pests can be expected due to shifting climatic zones (Houghton *et al.* 1995). Agricultural regions and natural ecosystems are also susceptible to climate changes that could result in increased insect populations and plant diseases. Degradation of natural ecosystems could lead to reduced biological diversity. Desertification will threaten some areas, while other regions could become colder because of direction changes in ocean currents (AAS 2006).

Besides CO_2, other greenhouse gases as methane (CH_4), nitrous oxide (N_2O) and chlorofluorocarbons (CFCs) are being emitted by human activities (Kasting 1998). The greenhouse potential of different gases is expressed as the Global Warming Potential (GWP) of the gas. Conventionally, GWP of CO_2 equals 1. GWP for methane is 21, while nitrous oxide has a GWP of 310 on a 100 years time horizon (Schimel *et al.* 1995, EPA 2006b). This means that methane and nitrous oxide have much stronger warming capacities than carbon dioxide. However, because of the enormous amounts of CO_2 being emitted to the atmosphere, this gas is the largest individual contributor to the enhanced greenhouse effect, accounting for about 64 % of the increase in heat trapping globally (Schimel *et al.* 1995).

Carbon dioxide is cycling naturally between the atmosphere, oceans and terrestrial biosphere, as illustrated in Fig. 1.4. During photosynthesis, plants utilize solar energy to combine CO_2 from the atmosphere with water to form organic matter and to release oxygen to the air. This photosynthesis is balanced, on average, by plant and animal respiration, and by decomposition of dead organic material. In a similar way, CO_2 is rapidly exchanged between the atmosphere and the surface ocean, and between the surface ocean and marine biota. Until the early 19th century, carbon (C) fluxes to and from the atmosphere were approximately balanced (Kasting 1998).

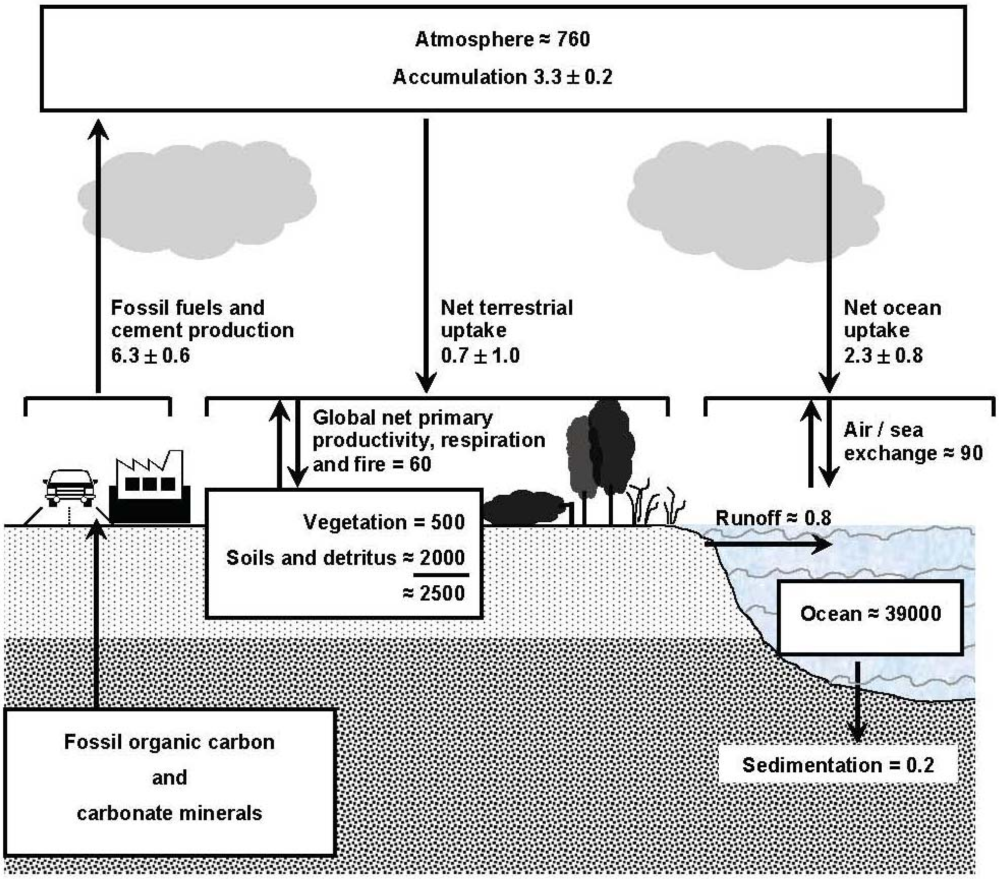

Figure 1.4. *The global carbon cycle, showing the carbon stocks in pools (boxes, in Gt C =* 10^{15} *g C) and carbon fluxes (arrows, in Gt C year*$^{-1}$*) relevant to the anthropogenic perturbation as annual averages over the decade from 1989 to 1998. The emission of carbon due to land-use changes is not depicted, but is assessed at 1.6 ± 0.8 Gt C year*$^{-1}$ *for this period (after Bolin and Sukumar 2000).*

The major (natural) fluxes in the global carbon cycle are more than ten times as large as the man-made injection of CO_2 to the atmosphere. However, human activities, mainly fossil fuel burning and deforestation, strongly disturbed the dynamic equilibrium between the natural carbon fluxes during the last two centuries (Goudriaan 1987, Trenberth *et al.* 1995, Schulze *et al.* 2002). About half of the amount of CO_2 emitted by human activities accumulates in the atmosphere, while one third is taken up by the ocean (Fig. 1.4). The remaining part is often referred to as the 'missing carbon sink' (Scholes *et al.* 1999). This part of the global carbon budget is the least understood, but it is evident that this amount of carbon is being removed from the atmosphere by increased carbon storage in the terrestrial biosphere (Kasting 1998). Several scenarios could cause the land to take up more carbon dioxide than is released each year. Regrowth of forests since the massive deforestation in the northern hemisphere over the last century could account for the missing carbon, while changing climate could also contribute by stimulating plant growth (Taylor and Lloyd 1992, Schimel *et al.* 1995, Bolin and Sukumar 2000, Houghton 2005). In order to predict how atmospheric CO_2 levels and climate may change in the future, it is critically important to understand the processes controlling the sources and sinks of carbon (Taylor and Lloyd 1992, Kasting 1998, NASA 2006).

1.2. Importance of forests in the global carbon cycle

Terrestrial ecosystems can influence the climate system through exchanges of carbon dioxide, influencing as such atmospheric CO_2 concentrations. As can be seen in Table 1.1, the total amount of carbon stored in vegetation of terrestrial ecosystems is assessed at 466 Gt C. On a global scale, soil carbon stocks largely exceed carbon stocks in vegetation, and amount to 2011 Gt C. The ratio soil-to-vegetation C stock ranges from about 1 in tropical forests to 5 in boreal forests, and much larger factors in grasslands and wetlands. Changes in soil carbon stocks are therefore at least as important for carbon budgets as changes in vegetation carbon stocks (Bolin and Sukumar 2000).

Forest ecosystems contain more than three fourths of the carbon stored in terrestrial vegetation, as carbon is stored in stems and branches, foliage and roots of trees (Bolin and Sukumar 2000). Moreover, while only 28 % of the total area is covered with forests, forest soils contain 39 % of all carbon stored in soils (Table 1.1). Conversion of forests to agricultural land releases carbon, mostly from trees, to the atmosphere through burning and decay. Depending on the agricultural practices applied, there may be an accompanying decline in the quantity of carbon stored in the soil (Scholes *et al.* 1999, Schlesinger and Andrews 2000). Conversely, regrowth of forests on abandoned lands withdraws carbon from the atmosphere and stores it again in trees and soils (Melillo *et al.* 1995). Because of the high carbon storage capacity of forests, and the long residence time of carbon in forests, more and more attention was dedicated in recent years to the mitigating role that forest ecosystems can play in reducing the build-up of CO_2 in the atmosphere (Winjum and Schroeder 1997).

Table 1.1. *Global carbon stocks in vegetation and top 1 m of soils of terrestrial ecosystems (Bolin and Sukumar 2000)*

	Area (10^6 km^2)	**Carbon stocks (Gt C)**		
Biome		**Vegetation**	**Soils**	**Total**
Tropical forests	17.6	212	216	428
Temperate forests	10.4	59	100	159
Boreal forests	13.7	88	471	559
Tropical savannas	22.5	66	264	330
Temperate grasslands	12.5	9	295	304
Deserts and semideserts	45.5	8	191	199
Tundra	9.5	6	121	127
Wetlands	3.5	15	225	240
Croplands	16.0	3	128	131
Total	151.2	466	2011	2477

1.3. Carbon cycle in a forest ecosystem

The uptake of carbon from the atmosphere by plants is called gross primary productivity (GPP). Plant respiration releases CO_2 back to the atmosphere, and reduces GPP to net primary productivity (NPP) (Fig. 1.5). In an ecosystem, further CO_2 losses occur because of decomposition of dead organic matter (Schulze 2000). NPP minus heterotrophic respiration results in net ecosystem productivity (NEP). A positive NEP indicates that the ecosystem has accumulated carbon during the considered time period, while a negative NEP denotes a loss of carbon from the ecosystem to the atmosphere (Melillo *et al.* 1995). On a time scale of years, most forests accumulate carbon through tree growth and an increase in soil carbon, until the next disturbance occurs (Bolin and Sukumar 2000). On the longer term (decades to centuries), and at a regional level (including a range of ecosystems, called a biome), additional C losses are caused by disturbances such as fire, wind-throw, drought, pests and human activities (*e.g.*, wood harvest). The resulting net imbalance can be interpreted as the net biome productivity (NBP) (Mooney *et al.* 1999, Bolin and Sukumar 2000, Grace 2004).

In Fig. 1.6, a more detailed overview of the main carbon pools and fluxes in forest ecosystems is given. Exact values for the C stocks contained in the pools and for the fluxes are not given here, as they vary strongly between forest ecosystems. As was already illustrated in Table 1.1, large amounts of carbon are stored in the above- and belowground biomass components, in the litter layer, and in the mineral soil. Other carbon pools that are not indicated on the graph comprise the shrub and herb layer, and the dead wood lying on top of the forest floor. The main carbon flux in a forest ecosystem is the gross photosynthesis (GPP). Part of the carbon taken up by the plants is released by respiration of the foliage (R_f), the woody biomass components (R_w) and the roots (R_r). These respiration fluxes add up to the total autotrophic respiration R_a. Decomposition of organic matter by heterotrophic respiration (R_h) also releases carbon to the atmosphere. R_a together with R_h defines the total ecosystem respiration or TER. The balance between GPP and TER, the net ecosystem productivity (NEP), determines whether the forest acts as a source or a sink for carbon (Schulze *et al.* 2002). NEP values of forests may reach values of 7 t C ha^{-1} year^{-1} (Bolin and Sukumar 2000), and are described to depend on climatic conditions, soil fertility, stand age, stand structure and species composition. A change in the balance between photosynthesis and respiration will change the carbon stock in forests, and also has potential to alter the CO_2 content of the atmosphere (Melillo *et al.* 1995).

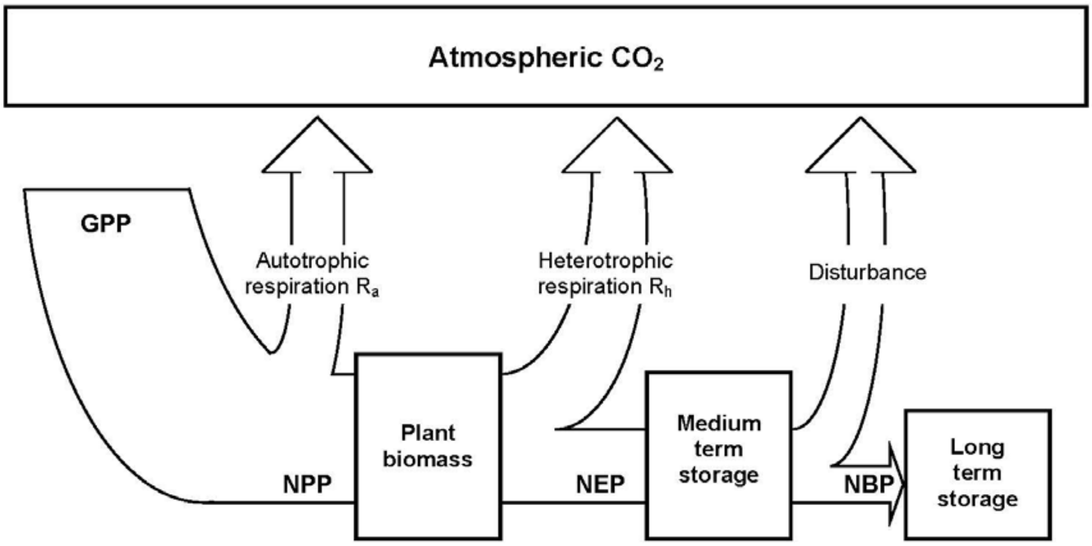

Figure 1.5. *Terms used to define various components of ecosystem productivity; GPP : gross primary productivity, NPP : net primary productivity, NEP : net ecosystem productivity, NBP : net biome productivity (after Scholes* et al. *1999)*

The ancient hypothesis of ecological equilibrium assumed that forest ecosystems tend towards a stage where assimilation and respiration are balanced. However, even old-growth forests have been shown to be carbon sinks (Carey *et al.* 2001, Schulze *et al.* 2002, Poulton *et al.* 2003). As such, three main phases can be considered in the development of a forest stand. During the early phase of stand development, which is expected to last 10 to 20 years, a forest is likely to be a source of carbon, as trees are small, and as such, photosynthesis cannot compensate for the carbon loss from soil and detritus. In a second phase, biomass production in the young forest stand is high, while the carbon content of the soil layer is fairly constant. In this phase, forests are a strong carbon sink. This phase can last for a century or even longer. In a third phase, the amount of carbon in the living biomass reaches a more or less constant level. Due to tree mortality, root and foliage decay, the soil carbon content will enhance. Consequently, the total carbon stock in the forest ecosystem can still increase, even while the vegetation carbon stock slightly diminishes (Bolin and Sukumar 2000). Therefore, replacing mature forests by actively growing plantations in order to capture more carbon from the atmosphere can be counterproductive, as exploitation of old forests results in large carbon losses, due to the harvest of a very large C stock and to soil disturbance, which favours carbon losses through soil respiration processes (Schulze *et al.* 2002).

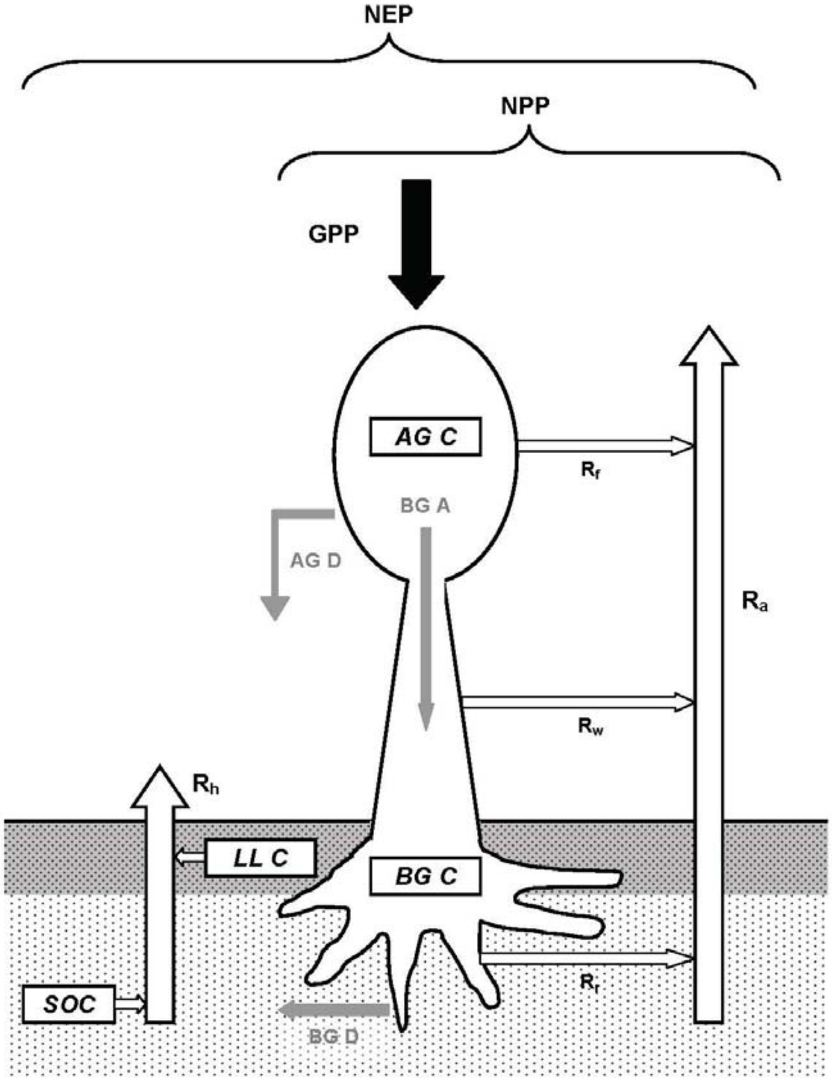

Figure 1.6. *Main carbon pools (boxes) and fluxes (arrows) in a forest ecosystem; AG C : aboveground biomass carbon stock, BG C : belowground biomass C stock, LL C : C stock in the litter layer, SOC : C stock in the mineral soil layer; GPP : gross primary productivity; R_f : foliage respiration, R_w : wood respiration, R_r : root respiration, R_a : autotrophic respiration; R_h : heterotrophic respiration; AG D : aboveground detritus (litter fall and mortality), BG D : belowground detritus (fine root turnover, exudation and root mortality), BG A : belowground allocation; NPP : net primary productivity; NEP : net ecosystem productivity (after Bolin and Sukumar 2000, Williams* et al. *2005)*

1.4. International agreements and obligations

In response to the growing awareness about global change, the World Meteorological Organization (WMO) and the United Nations Environment Programme (UNEP) established the Intergovernmental Panel on Climate Change (IPCC), in 1988. Four years later, in 1992, 162 countries adopted a treaty, known as the United Nations Framework Convention on Climate Change (UNFCCC) at the first Earth Summit held in Rio de Janeiro, Brazil (AAS 2006). The stated objective of the Framework Convention is "to achieve stabilization of the greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Such a level should be achieved within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner". In this treaty, developed countries were asked to reduce their greenhouse gas emissions to 1990 levels by the year 2000, as to avert extreme climate change. It was also stated that on the longer term (2100), the global greenhouse gas emissions have to decrease with 40 to 50 % in comparison to 1990. Countries that ratified the UNFCCC were committed to a number of obligations, including the reporting of their national greenhouse gas emissions, the development of strategies and measures to reduce these emissions, among which enhancing energy use efficiency and searching for alternative and renewable energy sources, and the protection of greenhouse gas sinks such as forests (Schimel *et al.* 1995).

At the third Conference of the Parties (COP), which was held in December 1997 in Kyoto (Japan), the UNFCCC was extended with the Kyoto Protocol (KP). This protocol can be considered as a first modest step to stabilize atmospheric greenhouse gas concentrations. The KP sets the collective global target of reducing greenhouse gas emissions by 5.2 % compared to 1990 levels in the industrialized countries (so-called Annex I countries), and this during the first commitment period (2008-2012) (Schulze *et al.* 2002, AAS 2006). Under the KP, not only carbon dioxide is considered, but a basket of six (groups of) greenhouse gases is taken into account : CO_2, methane, nitrous oxide, sulphur hexafluoride (SF_6), hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs). These two latter groups of gases are replacing chlorofluorocarbons, of which the use was seriously restricted because of the damage they caused to the ozone layer in the atmosphere. To enable comparison of the greenhouse strength of the different gases, their Global Warming Potential is used to express the greenhouse strenght in CO_2 equivalents.

At 16th February 2005, the Kyoto Protocol entered into force, following ratification by Russia on 18th November 2004. As such, 163 countries had ratified the protocol, and they were representing 61.6 % of the total CO_2 emissions of all Annex I countries. This was far more than the 55 countries and 55 % of the total of CO_2 emissions for

1990 of the Annex I countries, which were the prerequisites for the protocol to become legally binding.

Under the Kyoto Protocol, the overall reduction target for the 15 then-members of the European Union was set at 8 % of 1990 emission levels (Schulze *et al.* 2002). Within the European Union, differential reduction rates were applied. For Belgium, the emission reduction commitment was fixed at 7.5 % compared to 1990. In March 2004, it was agreed that Flanders should reduce its GHG emissions with 5.2 % compared to 1990 levels and Wallonia with 7.5 %, during the first commitment period. The remaining GHG emission reduction needed to reach the Belgian commitment will be compensated by the federal government, which will purchase emission rights at the international market (Wittoeck 2006). In 2003, total greenhouse gas emissions in Belgium amounted to 147719 kt CO_2 equivalents, an increase with 1.4 % compared to 1990 (EMIS 2006). CO_2 contributed 85.5 % to the total Belgian GHG emissions. Moreover, CO_2 emissions increased with 6.2 % during the period 1990 to 2003, while emissions of methane (6.8 % of GHG emissions) and nitrous oxide (8.5 % of GHG emissions) were reduced with 20.9 and 7.7 %, respectively.

In the Kyoto Protocol, direct reference is made to the carbon sequestration capacity of forest ecosystems. Art. 3.3 states that afforestation, reforestation and deforestation (ARD) activities since 1990 can be used to meet the emission reduction commitments. Afforestation refers to the planting of trees on former arable land or grassland, while reforestation can be described as the planting or natural regeneration of trees on arable land or grassland, which had been forest before cultivation by humans (Thuille and Schulze 2006). Under Art. 3.4, additional human-induced activities related to changes in greenhouse gas emissions by sources and removals by sinks in the agricultural and the land-use change and forestry categories can be taken into account for the calculations of GHG budgets, provided that these activities have taken place since 1990. In this article, it is also stated that each country that wants to use these activities to meet the commitments under the Kyoto Protocol should establish its level of carbon stocks in 1990, and should estimate the changes in carbon stocks in subsequent years (KP 1997).

1.5. Objectives of the thesis

The ongoing scientific and political discussions on the role of terrestrial ecosystems in general, and forests in particular, in the global carbon cycle reveal the need for more in-depth studies of the carbon cycle in forest ecosystems (Schulze 2000, Thuille and Schulze 2006). As improved insight in carbon sequestration in forests can help to find measures to mitigate global warming, the overall aim of this PhD was to study carbon sequestration in Belgian forest ecosystems. In Art. 3.3 and 3.4 of the Kyoto Protocol, short-rotation forestry (SRF) plantations as well as mature forest ecosystems are mentioned as possible means to sequester carbon from the atmosphere. Therefore, both types of ecosystems were investigated in the framework of this PhD study. The specific objectives of this research project were :

1) to study the establishment and biomass production of SRF plantations in Flanders,
2) to determine biomass carbon stocks in different (Belgian) forest ecosystems,
3) to investigate methodologies to assess the carbon balance of a forest ecosystem,
4) to examine the carbon balance of different forest ecosystems and
5) to assess the role of forest ecosystems in reaching the Kyoto Protocol target for Flanders and Belgium.

1.6. Thesis outline

Two main parts can be distinguished in this thesis. In Part A, several aspects of carbon sequestration in short-rotation forestry plantations are described. Part B of the thesis is focussed on the carbon sequestration in mature Belgian forest ecosystems.

1.6.1. Short-rotation forestry plantations

Part A comprises Chapter 2 to Chapter 6. In these five chapters, different aspects of carbon sequestration in short-rotation forestry ecosystems are discussed. Measurements were performed on the SRF plantation established at Zwijnaarde (51°02' N, 3°43' E), which is situated 10 km south of the centre of Ghent. In spite of the general procedure of mentioning only objectives in the general introduction, an extensive description of the study site is provided here, in order to prevent needless repetition of site description throughout Chapters 2 to 6.

During March and April 2001, the plantation was established on former agricultural land by hand planting. The site is characterized by a temperate maritime climate, with moderate temperature variation, prevailing westerly winds, a heavy cloud cover and regular rainfall. Long-term (1961-1990) mean values for the mean air temperature and the annual amount of rainfall are 9.8 °C and 821 mm, respectively (RMI 2006). At the start of the experiment, the upper 30 cm layer of the sandy soil (< 2 µm : 3.0 %, 2 - 50 µm : 7.5 %, 50 µm - 2 mm : 89.5 %) had a mean organic carbon concentration of 1.0 % and a pH_{KCl} of 4.5. The original area of the plantation was 9600 m^2, and was composed of 24 plots of 400 m^2 each (25 m x 16 m). Birch (*Betula pendula* Roth) and maple (*Acer pseudoplatanus* L. - Tintigny) were planted as 2 year old saplings, with a density of 6667 stems ha^{-1} (spacing distance : 1.0 m x 1.5 m) on four plots each. Twenty cm unrooted cuttings of poplar (*Populus trichocarpa* x *deltoides* - Hoogvorst) and willow (*Salix viminalis* - Orm) were planted on eight plots each, with an initial density of 20000 stems ha^{-1} (spacing distance : 1.0 m x 0.5 m). The plantation activity was preceded by tillage of the upper 20 cm of the soil. Afterwards, no specific management activities such as weed control, fertilization or irrigation were performed on the plantation. In Fig. 1.7, a schematic overview of the plantation is given. It should be noted that due to building activities in the neighbourhood of the plantation, plots W2, W23 and M24 were destroyed in October 2002. As such, it was no longer possible to perform measurements on these three plots.

The most important carbon flux in forest ecosystems is photosynthesis, the fixation of CO_2 by trees (Fig. 1.6). The photosynthetic capacity of tree species is mainly determined by the leaf photosynthetic characteristics, while specific leaf area (SLA) is an important leaf parameter in the upscaling of photosynthesis from leaf to stand level. These leaf traits were determined for birch and poplar as described in Chapter 2. This chapter focuses on seasonal variation of these leaf characteristics, and on the difference between the two tree species, both growing under SRF conditions. The information provided in this chapter can serve as input for modelling the carbon cycle of short-rotation forestry plantations.

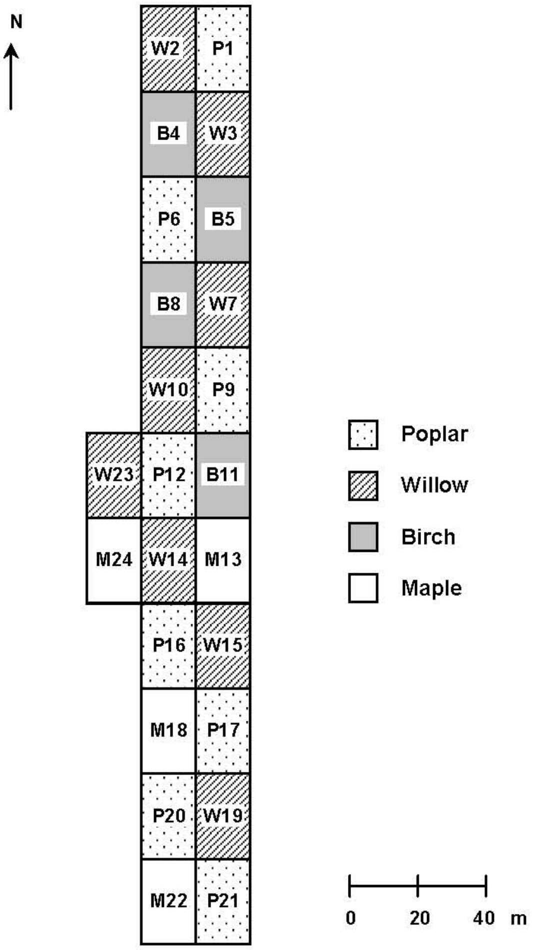

Figure 1.7. *Schematic overview of the short-rotation plantation at Zwijnaarde, which was established on former agricultural land; P : poplar, W : willow, B : birch, M : maple; plots W2, W23 and M24 were destroyed in October 2002.*

In Chapter 3, the aboveground biomass production after 4 growing seasons (2001 to 2004) is determined for all plots of the plantation. An extensive literature review is provided, as to compare results obtained at Zwijnaarde with the production observed at other short-rotation plantations, established under comparable (climatic) conditions. The specific aim of this chapter was to compare the production capacities of the different tree species under study, as to give advice on species choice for future establishment of SRF plantations in Flanders.

CO_2 emissions from fossil fuel burning can be prevented when biomass produced at SRF plantations is used as a substitute. Therefore, the total CO_2 emission reduction potential of short-rotation forestry plantations in Flanders is assessed in Chapter 4, based on results presented in Chapter 3. Biomass production is translated into energy production capacity, by taking into account the calorific value of wood. In this chapter, the capacity of SRF in Flanders to deliver electricity is also described.

Soil CO_2 efflux, the combination of root respiration and heterotrophic respiration (Fig. 1.6), is the second largest carbon flux in forest ecosystems. In Chapter 5, soil CO_2 efflux is studied in detail in three willow plots (W3, W15 and W19, see Fig. 1.7). On the one hand, the temporal variation of soil CO_2 efflux is extensively described. On the other hand, high resolution spatial variation of this carbon flux is discussed. In this chapter, soil CO_2 efflux models are established, as to describe the relationship between soil CO_2 efflux and the main factors driving temporal and spatial variation of this carbon flux.

The last chapter of part A, Chapter 6, assembles all information on carbon fluxes in the SRF plantation at Zwijnaarde. In this chapter, the overall carbon balance of the plantation is calculated, and this for the first five years of tree growth (2001 to 2005). Two different approaches were selected to assess the net ecosystem producitivity (NEP), which can be considered as the carbon balance of the plantation (Fig. 1.6). The first research question to be answered in this chapter is if the plantation acted as a sink or source for carbon during the first five years of tree growth. Moreover, the NEP values calculated according to the two approaches are compared to assess the comparability of the results. Finally, the interannual variation in NEP, and the possible causes of this variability, are discussed.

1.6.2. Belgian forest ecosystems

Part B begins with the determination of the total carbon stocks in the experimental zone of the Aelmoeseneie forest, which is situated at Gontrode (50°58' N, 3°49' E). This experimental zone comprises two different forest stands, an oak-beech stand and an ash stand. More detailed information on species composition, leaf area index, humus and soil type of the two stands can be found in Chapter 7. Carbon stocks in above- and belowground vegetation, in the litter layer and in the soil are determined and compared for the oak-beech and the ash stand in this chapter too.

In Chapter 8, carbon sequestration in the experminental zone of the Aelmoeseneie forest is assessed. In December 1990, December 1997 as well as in January 2006, a full inventory of both the oak-beech and the ash stand was performed. Allometric relationships are applied to translate dendrometric information into biomass increment. Based on the carbon stock change method, mean net ecosystem productivity (NEP) for the periods 1991 to 1997 and 1998 to 2005 is assessed for both stands. Uncertainty on NEP results was also assessed.

After the case study of the Aelmoeseneie experimental forest, the total carbon stock in the above- and belowground biomass of all Belgian forests was calculated for the year 2000, as described in Chapter 9. Besides the determination of the total standing carbon stock, a detailed analysis was performed as to identify the biomass expansion factor category that has the biggest impact on the C stock assessment. This information can be of use for future forest inventories in Belgium.

As 1990 is the reference year in relation to the Kyoto Protocol, a linear back-calculation is performed in Chapter 10, as to assess the carbon sink or source strength of the Belgian forests in the period 1990 to 2000. Additionnally, the amount of carbon stored in or released from the forests was compared to the cap which was fixed under the Kyoto Protocol. This information can help the government to decide on including activities under Art. 3.4 in the reporting to the Kyoto Protocol.

A comprehensive overview of the main findings can be found in Chapter 11. In this chapter, results of all previous chapters are integrated, as to come to a final conclusion with regard to the research topics listed in § 1.5. Besides a synthesis of the results, recommendations for future investigation are also given in Chapter 11.

Chapter 2 to Chapter 10 are all adapted from submitted or published manuscripts. References from all chapters are brought together in Chapter 12. Finally, a summary, both in English and in Dutch, and a curriculum vitae are added to this book.

PART A.

SHORT-ROTATION FORESTRY PLANTATIONS

Chapter 2

Seasonal dependence of specific leaf area and leaf photosynthetic characteristics of birch and poplar growing in a short-rotation forestry plantation

Adapted from : Vande Walle, I., R. Samson, N. Van Camp, K. Verheyen and R. Lemeur. Seasonal dependence of specific leaf area and leaf photosynthetic characteristics of birch and poplar growing in a short-rotation forestry plantation. *Submitted to Biomass & Bioenergy.*

Abstract

Specific leaf area (SLA) and leaf photosynthetic characteristics are important parameters in the upscaling of carbon fluxes from leaf to canopy level. In this study, the seasonal dependence and height variation of SLA and photosynthetic characteristics were studied for young birch (*Betula pendula* Roth) and poplar (*Populus trichocarpa* x *deltoides* – Hoogvorst) trees. The study site was a short-rotation plantation situated on a sandy soil, located at Zwijnaarde, near Ghent (Belgium).

Considering birch and poplar together, SLA values ranged from 9.2 to 22.8 m^2 kg^{-1} dry mass, maximal photosynthetic rate A_{max} ranged from 7.1 to 21.2 µmol CO_2 m^{-2} s^{-1}, while values from 22.4 to 94.3 µmol CO_2 m^{-2} s^{-1} and from 42.6 to 143.9 µmol e^- m^{-2} s^{-1} were found for the maximum rate of the Rubisco activity at 25 °C, V_{cmax}, and the potential electron transport rate at 25 °C, J_{max}, respectively. These values corresponded well with those reported in literature.

SLA of both species decreased during the growing season. This seasonal trend is important if SLA is used to determine leaf area index of a forest stand. For the leaf photosynthetic characteristics, such a seasonal trend was less obvious. For both species, leaves of the upper canopy layer appeared to have higher A_{max}, V_{cmax} and J_{max} values than leaves growing in the lower part of the canopy, while no clear difference in SLA was found for both layers. The observed vertical gradient in leaf photosynthetic characteristics should be taken into account in the upscaling of photosynthesis from leaf to canopy level.

Empirical SLA models were established for birch and poplar separately. These models link SLA to the number of days since budburst (DsBB). The adjusted R^2 value was 0.49 for birch and 0.78 for poplar. When the maximum observed SLA was included in the poplar model, the adjusted R^2 was 0.89. Comparison of modelled with measured values showed a good performance of these empirical models for the plantation at Zwijnaarde.

2.1. Introduction

Because of their large carbon sequestration capacity, temperate forests play an important role in the global carbon cycle (Schimel 1995). Therefore, a lot of effort has been spent during the last decades in developing models to simulate the carbon cycle and to calculate the carbon balance of forest ecosystems (Valentini *et al.* 2000). One of the major challenges of applying mechanistic models for simulating growth processes is to scale CO_2 fluxes from leaf to canopy (Kull and Jarvis 1995, De Pury and Farquhar 1997, Walcroft *et al*. 1997, Rasse *et al.* 2001). Often, photosynthesis parameters are measured at leaf level. Afterwards, upscaling techniques are applied to integrate the fluxes determined at leaf level and to assess as such the total carbon balance of a whole canopy or region (Hoffmann 1995, Kull and Jarvis 1995, Leuning *et al*. 1995, De Pury and Farquhar 1997, Deraedt and Ceulemans 1998, Wang and Leuning 1998, Dungan *et al.* 2004, Le Goff *et al*. 2004).

Specific leaf area (SLA, the ratio of leaf surface area to leaf dry mass) and leaf photosynthetic characteristics are of particular importance for modelling the carbon cycle of an ecosystem. SLA is often used to convert leaf dry weight into leaf area (Gower 1991, Hoffmann 1995, Maguire and Bennett 1996, Cermák *et al*. 1998, Verwijst and Telenius 1999, Lafarge and Hammer 2002, Bouriaud *et al.* 2003). Leaf area index (LAI) is on its turn a crucial factor to scale up photosynthesis from leaf to canopy level (Baldocchi and Harley 1995, Ceulemans *et al.* 1996, De Pury and Farquhar 1997, Wang and Leuning 1998, Schieving and Poorter 1999, Lebaube *et al.* 2000, Stenberg *et al.* 2001, Wilson *et al.* 2001, Le Goff *et al*. 2004). The use of an over- or underestimated SLA value might therefore lead to an erroneous LAI estimation, and thus to an erroneous estimation of the canopy carbon exchange. This can diminish the accuracy of the ecosystem simulations (De Pury and Farquhar 1997, Wilson *et al.* 2001). As such, better knowledge of horizontal variation, vertical gradients and seasonal dynamics of specific leaf area within tree crowns contributes to a better understanding and quantification of light dynamics and, consequently, of the photosynthesis and respiration processes of the whole canopy (Temesgen 2003).

Photosynthesis models at leaf level include descriptions of light response curves or A-C_i curves. Light response curves link net photosynthetic rate A_n (µmol CO_2 m^{-2} s^{-1}) to incident light level (I, µmol m^{-2} s^{-1}), while A-C_i curves describe the relationship between photosynthetic rate A (µmol CO_2 m^{-2} s^{-1}) and internal CO_2 concentration (C_i)

of the leaf. This latter type of photosynthesis curves is often used to assess the impact of a rising atmospheric CO_2 level on the carbon cycling in forest ecosystems (Medlyn *et al.* 1999, Urban and Marek 1999). Key photosynthetic parameters, determining the photosynthetic capacity of a plant (species), comprise A_{max} (maximal photosynthetic rate) for light response curves, and V_{cmax} (maximum rate of Rubisco activity at 25 °C) and J_{max} (potential electron transport rate at 25 °C) for A-Ci curves (Medlyn *et al.* 1999, Gonzalez-Real and Baille 2000).

Since 1970, there has been an increasing interest in short-rotation forestry (SRF). At the beginning, the main reason for this interest was the capacity of the biomass produced by SRF to substitute fossil fuels. Nowadays, research is focussed on the carbon sequestration potential of this type of forestry. An example of this sequestration is the enhancement of the carbon stock of the soil when SRF plantations are established on previous agricultural land. Poplar and willow are generally considered as the most convenient tree species to be used for short-rotation forestry (Barigah *et al.* 1994, Hervé and Ceulemans 1996, Scarascia-Mugnozza *et al.* 1997, Ceulemans and Deraedt 1999, Fang *et al.* 1999, Tahvanainen *et al.* 1999, Verwijst and Telenius 1999, Kopp *et al.* 2001, Proe *et al.* 2002, Nordh and Verwijst 2004, Laureysens *et al.* 2005). However, birch can also be an interesting species to be grown under SRF, especially on marginal (*e.g.,* dry) soils (Ferm 1993, Jõgiste *et al.* 2003, Vande Walle *et al.* 2007a and 2007b). Morphological (*i.e.,* area and dry mass) and physiological leaf parameters (*i.e.,* photosynthetic capacity) differ between tree species, and vary with height in the canopy and with leaf age (Schieving and Poorter 1999, Casella and Ceulemans 2002, Proe *et al.* 2002, Marshall and Monserud 2003). Information on these parameters and their temporal and spatial variability is needed to model the carbon balance of short-rotation forestry ecosystems.

The aims of the present study were i) to determine the seasonal variation of the specific leaf area (SLA) of birch and poplar growing in a short-rotation forestry system, ii) to establish an empirical SLA model, and iii) to quantify both the temporal and vertical variation of leaf photosynthetic characteristics for these two species. The results of this study will contribute to the adaptation of the mechanistic dynamic forest growth model FORUG (Boonen *et al.* 2002, Verbeeck *et al.* 2006) for simulating the carbon cycle of short-rotation forestry plantations.

2.2. Material and methods

2.2.1. Short-rotation plantation

The study site is extensively described in § 1.6.1. For the study presented here, two birch plots (B5 and B8, Fig. 1.7) and two poplar plots (P12 and P17, Fig. 1.7) were used.

2.2.2. Specific leaf area (SLA) and leaf water content (LWC)

The average specific leaf area (SLA, m^2 kg^{-1} dry mass or m^2 kg^{-1} DM), the average individual leaf area (LA, cm^2) and the average leaf water content (LWC, %) were determined at plot level at two, four and eight moments in the growing seasons of 2001, 2002 and 2003, respectively. In total, 20 leaf samples per plot were collected randomly at each sampling date. Leaf samples were put in bags, and transported immediately to the laboratory, where the leaves were dried with tissue paper to remove any surface water when necessary. After removing petioles (Garnier *et al.* 2001, Le Goff *et al.* 2004, Niinemets *et al.* 2004), the individual leaf area LA was measured with the LI-3000 Portable Area Meter, coupled with the LI-3050A Transparent Belt Conveyer. Leaf fresh weight (LFW, g DM) was noted too. After leaves were dried at 50 °C until constant weight, leaf dry weight (LDW, g DM) was determined. SLA was calculated as the individual leaf area (LA) divided by the corresponding leaf dry weight (LDW). The percentage leaf water content (LWC) was determined from the leaf fresh weight and the leaf dry weight as LWC = 100 . (LFW - LDW) / LDW (Al Afas *et al.* 2005). As sampling time can have an effect on the SLA value (Tardieu *et al.* 1999), all samples were taken between 9 and 12 a.m. local time. No rehydration procedure was applied, as it was the aim to detect field values rather than potential values for the leaf traits studied (Garnier *et al.* 2001).

2.2.3. Meteorological information

One of the objectives of this study was to establish an empirical model which links SLA to climatic variables. There was no possibility to collect meteorological information directly on the plantation. Therefore, data registered at official meteorological stations of the Royal Meteorological Institute of Belgium (RMI) were used. Global (direct and indirect) solar radiation data for Ukkel (50°49' N, 4°19' E) and mean air temperature values for the measuring station at Kruishoutem (50°55' N, 3°31' E) were used for this study, as these were the stations nearest to the plantation with this kind of information available (distance to the plantation was 51 and 20 km for Ukkel and Kruishoutem, respectively). Mean air temperature is defined as the mean value of the maximum and the minimum air temperature over 24 hours (RMI 2001, 2002 and 2003).

2.2.4. SLA model

In addition to the day of the year (DOY), the number of days since budburst (DsBB) was considered as a possible variable to build an empirical SLA model. Budburst (BB) was defined as the day on which the first leaf appeared on at least 3 trees in a plot (unpublished data). In addition to these two variables, both radiation integral and thermal time were considered in this study as possible model variables. Summation of daily global solar radiation data resulted in the radiation integral (RI) for a specific date, expressed in MJ m^{-2}. Thermal time TT_{date} (degree-days or °C d) was calculated as given by eq. 2.1 :

$$TT_{date} = \sum_{i=1}^{n} (T_i - T_{base}) \cdot \Delta t_i \qquad \textbf{(2.1)}$$

where TT_{date} is thermal time (°C d), calculated for a specific date, n is number of time steps since the considered starting date, T_i is mean air temperature for time step i (°C), T_{base} is base temperature for the process under consideration (°C) and Δt_i is duration of time step i (days).

The time step in this study equalled one day. A day for which the daily mean air temperature was below the base temperature was counted as zero. T_{base} was set equal to 5 °C, in accordance to the value often reported as base temperature in literature (Cannell and Smith 1986, Hoffmann 1995, Tardieu *et al.* 1999, Kopp *et al.* 2001). In a subsequent exercise, base temperatures of 0 °C and 10 °C were also tested. However, the Pearson's correlation matrix revealed that the highest correlation between SLA and the independent variable TT was found for a base temperature of 5 °C. Both the radiation integral and the thermal time were calculated starting from six dates : 1st and 16th January, 1st and 16th February, 1st and 16th March. Pearson's correlation coefficients between SLA and RI or TT differed less than 1 % for these six dates. Therefore, only one of these dates (1st March) was used to establish the final SLA model. This starting date was at least 26 days before budburst (unpublished data). Concluding, the following four variables were selected to build the SLA model : day of the year (DOY), number of days since budburst (DsBB), radiation integral (RI) from 1st March onwards and thermal time (TT), calculated with the mean air temperature and a base temperature of 5 °C, since 1st March.

2.2.5. Leaf photosynthetic characteristics

Measurements of photosynthetic characteristics were performed on fully developed leaves during three periods in the growing season of 2004 : 14th to 24th June, 15th to 26th July and 1st to 7th September. In the first period (June), leaves from one birch (B8) and one poplar plot (P12) were used. In the second period (July), two birch (B5 and B8) and two poplar plots (P12 and P17) were investigated. In September, two birch plots and only one poplar plot (P17) were studied, as all leaves of the other poplar plot (P12) already died by that time. Mean tree height in May 2004 was 275 cm, 289 cm, 393 cm and 350 cm for the plots B5, B8, P12 and P17, respectively. In each measuring campaign and for each plot, 6 leaves from the upper part (> 2 m), and 6 leaves of the lower part (< 2 m) of the canopy were selected for the determination of photosynthetic characteristics. In the field, randomly chosen branches were cut off, submerged immediately in a bottle with water and transported to the laboratory within 30 minutes after harvest. There, measurements of photosynthetic characteristics took place, after the branches had been recut under water, to ensure an undisturbed water transport towards the leaves during the measurements.

Light response curves and A-C_i curves were measured by the automatic dynamic chamber system LI-6400. The temperature in the cuvette was fixed at 25 °C, which is an optimal temperature for the photosynthesis of C3 plants (Niinemets 1999). The light response curve was measured at a CO_2 concentration of 400 ppm in the cuvette, which is near to the ambient concentration in the field. Light intensities were decreased from 1500 or 1800 µmol PAR m^{-2} s^{-1} (for lower and upper leaves, respectively) to darkness in 10 steps, emphasizing the lower light levels. Afterwards, the light intensity was kept constant at the saturating value of 1500 or 1800 µmol PAR m^{-2} s^{-1}, and the A-C_i curve was measured by changing the external CO_2 concentration in the cuvette from 50 to 1500 ppm in 11 steps.

SPSS 11.5 was used to determine the photosynthetic characteristics given by eq. 2.2, the mathematical expression of the light response curve :

$$A_n = A_{max} \cdot \left(1 - exp\left(\frac{-\alpha_c \cdot I}{A_{max}} \right) \right) - R_d \qquad \textbf{(2.2)}$$

where A_n is net photosynthetic rate (µmol CO_2 m^{-2} s^{-1}), A_{max} is maximal photosynthetic rate (µmol CO_2 m^{-2} s^{-1}), α_c is quantum efficiency at the light compensation point (µmol CO_2 $µmol^{-1}$ PAR), I is incident photosynthetic active radiation (µmol PAR m^{-2} s^{-1}) and R_d is dark respiration rate (µmol CO_2 m^{-2} s^{-1}). An additional parameter, the light compensation point I_c (µmol PAR m^{-2} s^{-1}), was calculated as the light intensity I at which the net photosynthetic rate A_n equals zero.

The response curves of photosynthesis to intercellular CO_2 concentration (C_i) and to incident photosynthetic active radiation (I) were used to calculate the key parameters of the A-C_i model as proposed by Farquhar and von Caemmerer (1982), a model widely used in ecophysiological studies (Baldocchi and Harley 1995, De Pury and Farquhar 1997, Niinemets and Tenhunen 1997, Walcroft *et al.* 1997, Deraedt and Ceulemans 1998, Garcia *et al.* 1998, Niinemets *et al*. 1998, Evans and Poorter 2001, Rasse *et al.* 2001, Warren and Adams 2001, Casella and Ceulemans 2002). An Excel spreadsheet was used which was developed for determination of these parameters in the framework of the ECOCRAFT network (Medlyn *et al.* 1999). In the model of Farquhar and von Caemmerer (1982), the net photosynthetic rate A_n is determined as the minimum of the Rubisco limited photosynthetic rate and the photosynthetic rate which is limited by the rate of the regeneration of the substrate ribulose-1,5-biphosphate, which is on its turn driven by the electron transport reactions. The main characteristics of the A-C_i curve are V_{cmax}, the maximum rate of the Rubisco activity at 25 °C (µmol CO_2 m^{-2} s^{-1}), and J_{max}, the potential electron transport rate at 25 °C (µmol e^- m^{-2} s^{-1}). V_{cmax} can be determined from the measurements at low internal CO_2 concentrations, while J_{max} can be calculated based on the slope of the light response curve. An extensive description of all equations used to calculate the parameters V_{cmax} and J_{max} can be found in Medlyn *et al.* (1999). For all leaves on which photosynthetic parameters were determined, individual leaf area (LA), specific leaf area (SLA) and leaf water content (LWC) were determined as described in § 2.2.2.

2.2.6. Statistical analysis

Statistical analysis was performed with the programme SPSS 11.5. Normality of data was tested with the Shapiro-Wilk test for samples with 50 or fewer observations, and with the Kolmogorov-Smirnov test with a Lilliefors significance level for testing normality of data sets with more than 50 samples. When data were not normally distributed, non-parametric tests (Mann-Whitney U-test and Kruskal-Wallis test) were used to determine significant differences between mean values. When data were normally distributed, ANOVA analysis was applied. In the case of homogeneous variances, significant differences were separated by Duncan's multiple range test; when variances were not homogeneous, Tamhane's T2-test was applied for this separation. Tests were performed at the 95 % confidence level.

2.3. Results

2.3.1. Specific leaf area, individual leaf area and leaf water content

In Fig. 2.1, the temporal evolution of the SLA values is presented. As can be seen, there was a significant decreasing trend of SLA during the growing season, especially clear for the year 2003, when a lot of measurements were made. A t-test (for normally distributed data) or a Mann-Whitney U-test (for data that were not normally distributed) was used to check the difference in SLA between the two plots of a single tree species on a specific date. When the two birch plots were compared, almost no significant differences between the plots could be detected, while the SLA value found in the poplar plot P12 was significantly higher than the value in P17 on 8 of the 11 measuring dates.

In Table 2.1, the minimum and maximum values for the mean LA, SLA and LWC are presented per plot. These are values from 2003, as this was the only year where measurements were done during the complete growing season. The lower LA values observed for birch illustrated that the leaves of birch were obviously smaller than the poplar leaves, while there was an overlapping range in SLA and LWC. The range of individual leaf area, calculated as maximum LA divided by minimum LA and expressed in terms of percentage, was 656 % for birch and 516 % for poplar, which is much larger than the range of SLA, which was 185 % and 224 % for birch and poplar, respectively.

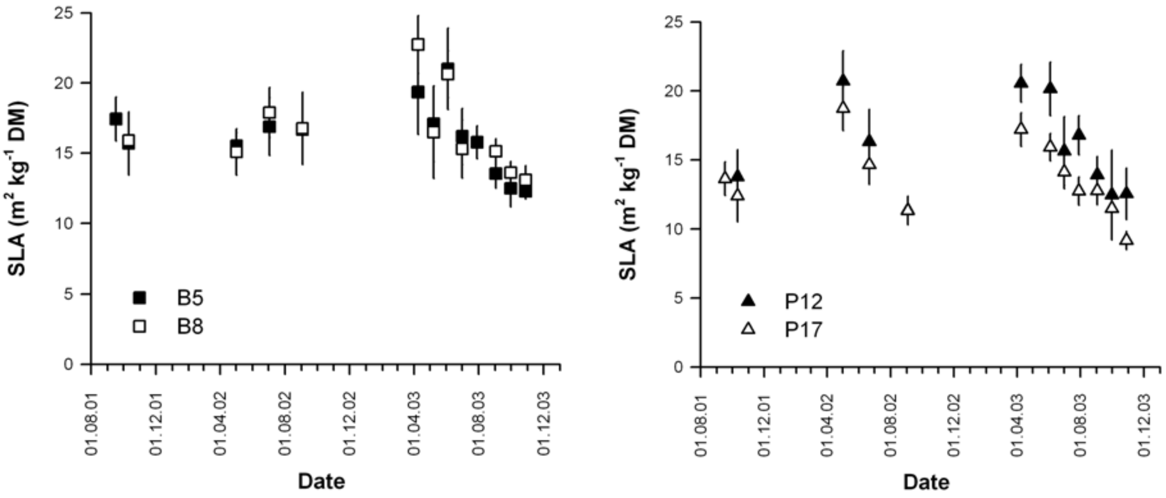

Figure 2.1. *Temporal evolution of the specific leaf area SLA measured in two birch plots (left pane, B5 and B8) and two poplar plots (right pane, P12 and P17) of the plantation at Zwijnaarde (n = 20); error bars represent ± 1 standard deviation.*

Table 2.1. *Minimum (Min.) and maximum (Max.) values of mean individual leaf area (LA, cm^2), mean specific leaf area (SLA, m^2 kg^{-1} DM) and mean leaf water content (LWC, %) for two birch and two poplar plots measured during the growing season of 2003; n = 8 for B5, n = 7 for B8, P12 and P17; standard deviation is given between brackets.*

		Birch		**Poplar**	
		B5	**B8**	**P12**	**P17**
LA	Min.	1.8 (0.8)	2.8 (1.0)	19.4 (6.0)	15.9 (10.1)
	Max.	11.8 (4.1)	9.8 (3.3)	58.1 (57.1)	82.0 (58.3)
SLA	Min.	12.3 (0.6)	13.1 (1.0)	12.5 (3.3)	9.2 (0.7)
	Max.	21.0 (2.9)	22.8 (2.1)	20.6 (1.4)	17.2 (1.2)
LWC	Min.	76.4 (12.8)	96.1 (10.9)	9.2 (1.0)	16.0 (7.2)
	Max.	220.2 (25.0)	215.9 (18.3)	279.6 (20.6)	237.8 (26.2)

The mutual relationship between the measured leaf traits is depicted in Fig. 2.2. As can be seen, there was a positive relationship between LDW and LA, and between SLA and LWC for both species, and a negative relationship between SLA and LA. No significant correlation was found between LWC and LA.

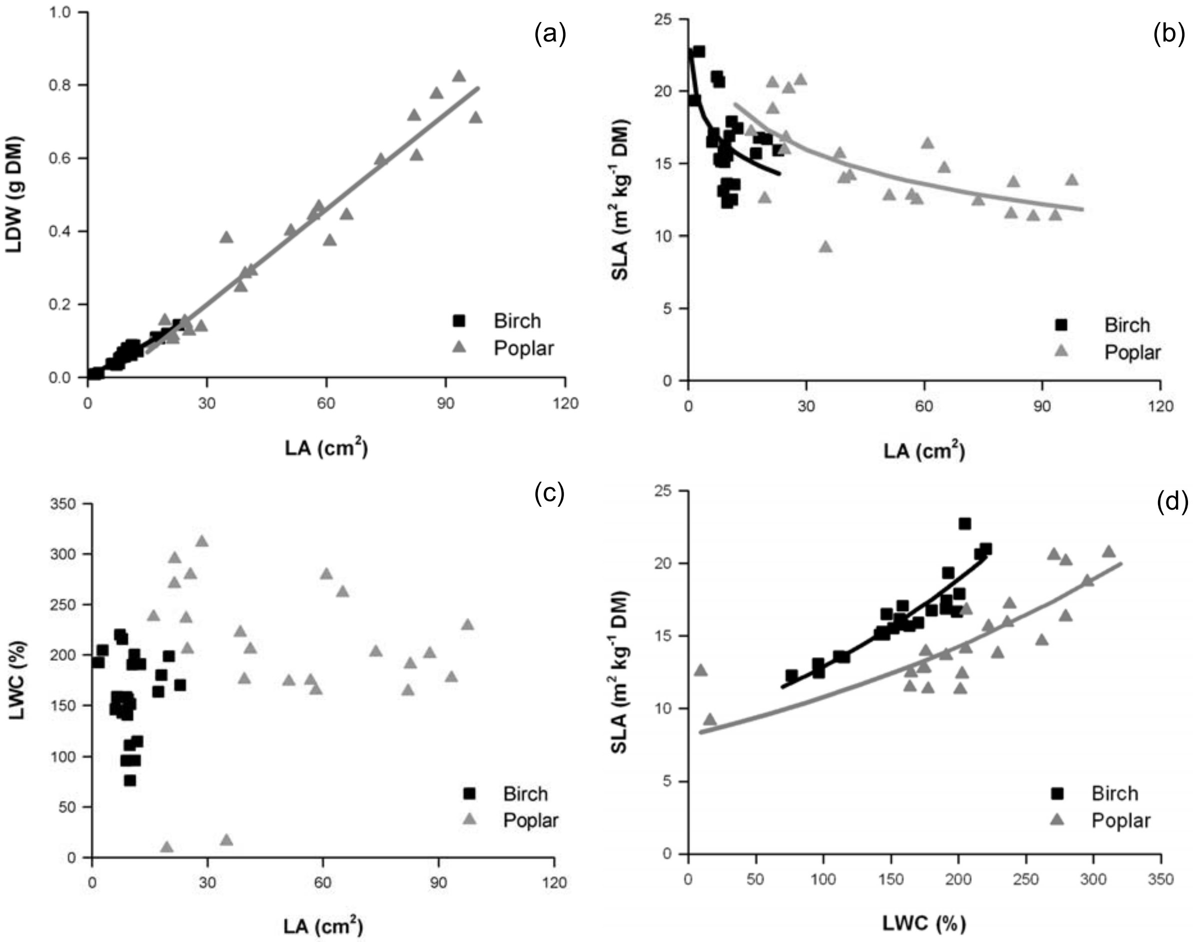

Figure 2.2. *Relationship between (a) leaf dry weight LDW and individual leaf area LA, (b) specific leaf area SLA and individual leaf area LA, (c) leaf water content LWC and individual leaf area LA and (d) specific leaf area SLA and leaf water content LWC for birch and poplar (based on mean data per plot and per date, pooled per species, for all three years). Regression equations are for (a) : LDW = 0.0063 . LA for birch (R^2_{adj} = 0.93) and LDW = -0.061 + 0.0087 . LA for poplar (R^2_{adj} = 0.95); for (b) : SLA = 21.31 - 2.23 . ln (LA) for birch (R^2_{adj} = 0.19) and SLA = 27.62 - 3.43 . ln (LA) for poplar (R^2_{adj} = 0.35), and for (d) : SLA = 8.79 . (exp (0.0038 . LWC)) for birch (R^2_{adj} = 0.82) and SLA = 8.16 . (exp (0.0028 . LWC)) for poplar (R^2_{adj} = 0.65).*

2.3.2. Empirical SLA model

For the establishment of the SLA models, individual data points represented mean SLA values for a single plot on a specific date. The four independent variables considered for the building of the SLA model (*i.e.,* DOY, DsBB, RI and TT) were strongly correlated with each other (R^2 ranged from 0.95 to 0.98). Therefore, adding more than one of these variables did not improve the model. Non-linear regression did not greatly improve the adjusted coefficient of determination (R^2_{adj}) neither. Initially, a simple linear model was established for both birch and poplar. Coefficients of these models are given in Table 2.2. Afterwards, an additional variable, the maximum observed SLA value, determined at plot level, or $\mathrm{SLA}_{\mathrm{max}}$, was added to the model. For birch, the coefficient of this extra variable was not significant, which can be explained by the fact that SLA values in the two birch plots were very similar. For poplar, however, a significant coefficient and a higher R^2_{adj} were obtained for this extended SLA model (Table 2.2), because of the differences in SLA values found for the two poplar plots. In Fig. 2.3, the SLA model with the highest R^2_{adj} value is presented for poplar.

Table 2.2. *Regression coefficients* a *and* b *(and s.e.) and the adjusted* R^2 *(*R^2_{adj}*) for the linear SLA model determined for birch and poplar at Zwijnaarde; DOY : day of the year, DsBB : number of days since budburst, RI : radiation integral (MJ* m^{-2} *since 1st March), TT : thermal time, based on mean air temperature,* T_{base} *= 5 °C, and starting from 1st March (°C d); SLA*$_{max}$ *: maximum observed SLA value (*m^2 kg^{-1} *DM). All coefficients are significantly different from 0 at the* p *= 0.05 level.*

	X	*a*	*b*	R^2_{adj}
Model : SLA = *a* + *b* . X				
Birch	DOY	21.6 (1.3)	-0.025 (0.006)	0.40
	DsBB	19.6 (0.8)	-0.030 (0.007)	0.49
	RI	20.0 (0.9)	-0.002 (0.000)	0.45
	TT	19.2 (0.8)	-0.002 (0.001)	0.43
Poplar	DOY	23.5 (1.2)	-0.042 (0.005)	0.73
	DsBB	20.0 (0.7)	-0.047 (0.006)	0.78
	RI	20.7 (0.8)	-0.003 (0.000)	0.74
	TT	19.5 (0.7)	-0.004 (0.000)	0.74
Model : SLA = *a* . SLA$_{max}$ + *b* . X				
Poplar	DOY	1.18 (0.05)	-0.041 (0.004)	0.83
	DsBB	1.01 (0.03)	-0.047 (0.004)	0.89
	RI	1.05 (0.03)	-0.003 (0.000)	0.85
	TT	0.99 (0.03)	-0.004 (0.000)	0.84

The models described above were used to calculate the SLA for birch and poplar on the dates that the photosynthesis measurements were performed in 2004. As to validate the SLA models, the modelled values were compared to the measured SLA. The measured SLA value was calculated as the mean SLA value of all leaves (upper and lower canopy layer together) used for photosynthesis measurements in one measurement period (June, July or September) and in one plot. As can be seen in Fig. 2.4, there was a good correspondence between modelled and measured SLA values. For birch, the best correspondence, indicated by the lowest Mean Squared Error (MSE) (Kramer *et al.* 2002), between modelled and measured SLA was found for the model with the radiation integral RI as independent variable. The same was the case for poplar, when the model without SLA$_{max}$ was used. In the case SLA$_{max}$ was included in the model, the DOY model performed the best, although differences in modelled values were sometimes very small (Fig. 2.4). For birch (Fig. 2.4a), the modelled SLA was in general slightly higher than the measured values. The highest overestimation of the measured SLA (21 %) was found for the model based on TT. For poplar, both higher and lower values were modelled (Fig. 2.4b and 2.4c). In this case, the maximum over- and underestimation were 20 % and 11 %, respectively.

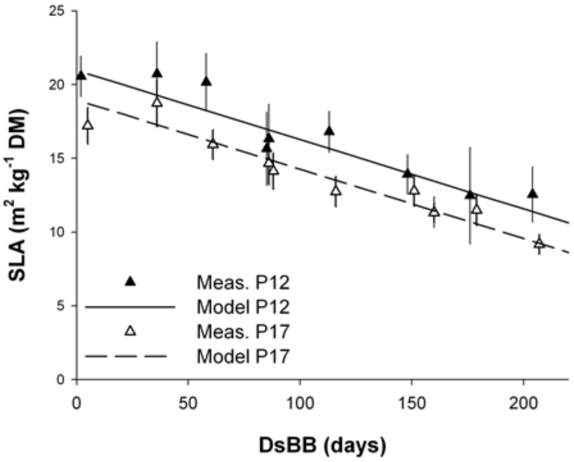

Figure 2.3. *Overall regression equation for poplar : SLA = 1.01 . SLA_{max} - 0.047 . DsBB, with SLA_{max} : maximum observed SLA and DsBB : number of days since budburst. The maximum observed SLA equalled 20.7 m^2 kg^{-1} DM for plot P12 and 18.7 m^2 kg^{-1} DM for plot P17. Error bars represent ± 1 standard deviation.*

2.3.3. Photosynthetic characteristics

Average values of the photosynthetic characteristics A_{max}, R_d, α_c, I_c, V_{cmax} and J_{max} are listed in Table 2.3. For both birch and poplar, t-tests revealed significantly higher A_{max}, R_d, I_c, V_{cmax} and J_{max} values in the upper layer compared to the bottom of the canopy. No significant differences were found in the quantum efficiency α_c of the two canopy layers of birch, in contrast to poplar, where a significantly higher as well as lower α_c was found in the upper tree layer (Table 2.3). When mean photosynthetic leaf characteristics of a canopy layer were compared between the two birch plots, assuming that these two plots are independent, only three significant differences were found (Table 2.3). A significantly higher R_d and I_c were found in July for the upper layer of plot B5, and a significantly higher α_c in July in the lower layer of plot B8. When the two poplar plots were compared, more significant differences were detected (Table 2.3). In all cases, highest values for A_{max}, R_d, I_c, V_{cmax} and J_{max} were found in plot P17.

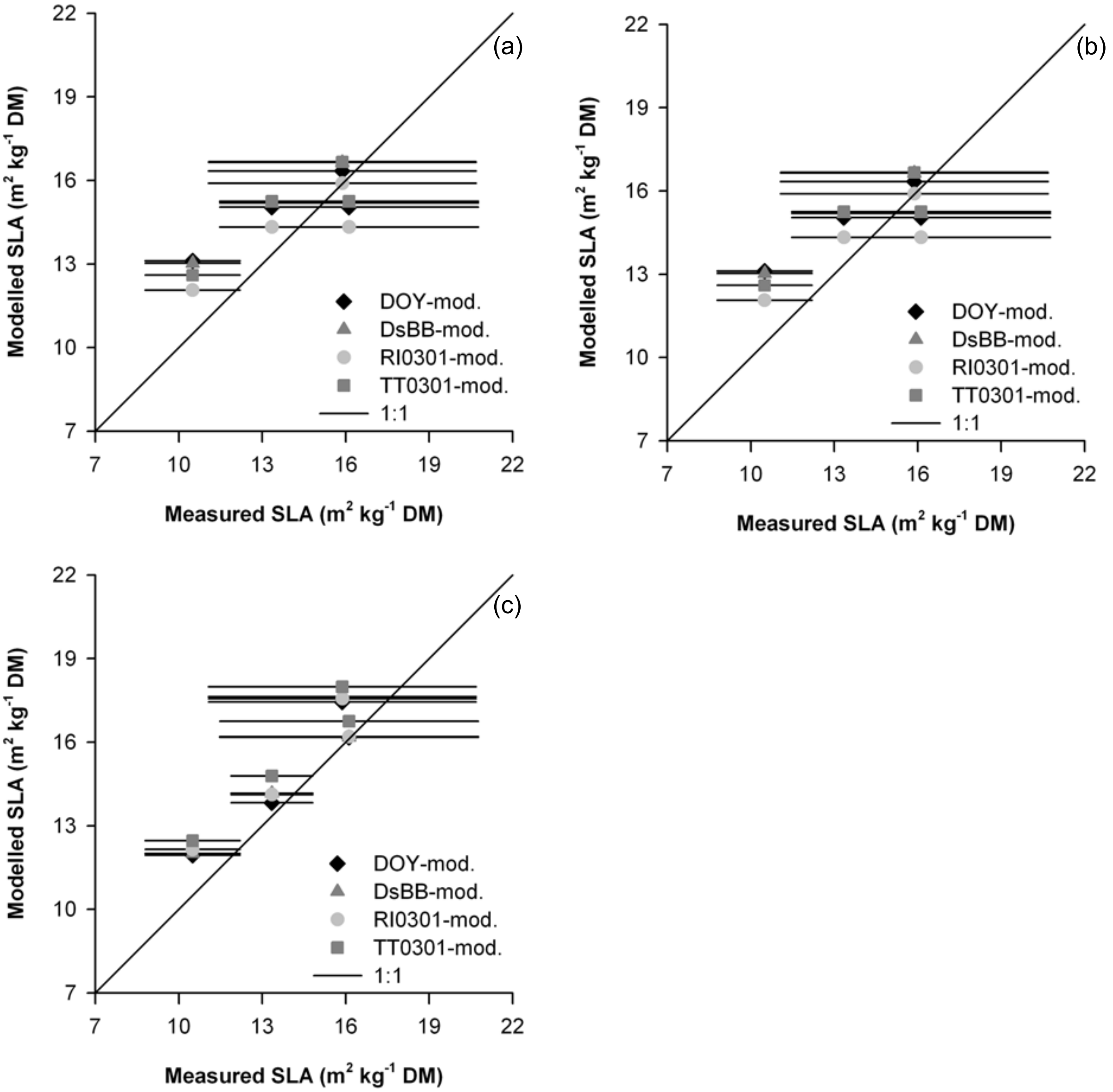

Figure 2.4. *Comparison of modelled and measured SLA values for birch (a, n = 5) and poplar (b and c, n = 4). In (b), the SLA model was : SLA* = a + b . *X, while in (c), the SLA model was : SLA* = a . *SLA_{max}* + b . *X. For model equations and explanation of the abbreviations, see Table 2.2. Error bars represent ± 1 standard deviation.*

Table 2.3. *Average values (and standard deviation) of leaf photosynthetic characteristics (from upper and lower canopy layers) for 2 birch and 2 poplar plots determined during 3 measuring campaigns in the growing season of 2004 (n = 6)*

			A_{max} (µmol CO_2 m^{-2} s^{-1})		
Species	**Plot**	**Layer**	**June**	**July**	**September**
Birch	B5	upper		14.5 $(5.2)^{a,j}$	17.6 $(1.9)^{b,j}$
		lower		9.0 $(4.8)^{a,j}$	9.5 $(3.5)^{a,j}$
	B8	upper	15.8 $(3.7)^{b}$	14.6 $(3.2)^{b,j}$	14.6 $(3.0)^{b,j}$
		lower	9.8 $(4.0)^{a}$	10.6 $(1.4)^{a,j}$	9.6 $(2.8)^{a,j}$
Poplar	P12	upper	18.9 $(2.7)^{b}$	16.8 $(4.2)^{b,j}$	
		lower	9.9 $(3.2)^{a}$	7.1 $(1.1)^{a,j}$	
	P17	upper		21.2 $(3.2)^{b,j}$	8.8 $(1.7)^{a}$
		lower		12.8 $(2.4)^{a,k}$	7.1 $(0.9)^{a}$

			R_d (µmol CO_2 m^{-2} s^{-1})		
			June	**July**	**September**
Birch	B5	upper		2.0 $(0.1)^{b,k}$	2.1 $(0.3)^{b,j}$
		lower		1.2 $(0.3)^{a,j}$	1.5 $(0.4)^{a,j}$
	B8	upper	2.0 $(0.4)^{b}$	1.5 $(0.4)^{a,j}$	1.6 $(0.4)^{b,j}$
		lower	1.5 $(0.4)^{a}$	1.5 $(0.2)^{a,j}$	1.2 $(0.3)^{a,j}$
Poplar	P12	upper	2.7 $(1.0)^{b}$	1.5 $(0.4)^{b,j}$	
		lower	1.0 $(0.2)^{a}$	0.8 $(0.1)^{a,j}$	
	P17	upper		2.4 $(0.6)^{b,k}$	3.7 $(0.6)^{b}$
		lower		1.0 $(0.2)^{a,k}$	2.1 $(0.4)^{a}$

			α_c (µmol CO_2 $µmol^{-1}$ PAR)		
			June	**July**	**September**
Birch	B5	upper		0.059 $(0.003)^{a,j}$	0.055 $(0.002)^{a,j}$
		lower		0.054 $(0.007)^{a,j}$	0.052 $(0.004)^{a,j}$
	B8	upper	0.057 $(0.006)^{a}$	0.056 $(0.009)^{a,j}$	0.052 $(0.004)^{a,j}$
		lower	0.053 $(0.009)^{a}$	0.062 $(0.002)^{a,k}$	0.052 $(0.004)^{a,j}$
Poplar	P12	upper	0.049 $(0.008)^{b}$	0.056 $(0.002)^{a,j}$	
		lower	0.062 $(0.011)^{a}$	0.061 $(0.003)^{b,j}$	
	P17	upper		0.058 $(0.003)^{a,j}$	0.049 $(0.006)^{a}$
		lower		0.061 $(0.002)^{a,j}$	0.055 $(0.002)^{b}$

[a, b] *denote significant differences (*p *< 0.05) between the upper and the lower canopy layer within each characteristic, plot and measurement period.*

[j, k] *denote significant differences (*p *< 0.05) between the two plots of a species, within each characteristic, layer and measurement period.*

Table 2.3. *(continued)*

			I_c (µmol PAR m^{-2} s^{-1})		
Species	**Plot**	**Layer**	**June**	**July**	**September**
Birch	B5	upper		36.0 $(2.4)^{a,k}$	39.8 $(5.0)^{a,j}$
		lower		26.5 $(11.9)^{a,j}$	38.2 $(28.8)^{a,j}$
	B8	upper	38.1 $(7.3)^{a}$	28.5 $(4.4)^{a,j}$	32.8 $(4.5)^{b,j}$
		lower	31.0 $(8.1)^{a}$	25.5 $(2.2)^{a,j}$	25.0 $(4.9)^{a,j}$
Poplar	P12	upper	62.5 $(30.5)^{b}$	28.8 $(7.2)^{b,j}$	
		lower	16.8 $(5.3)^{a}$	13.1 $(2.0)^{a,j}$	
	P17	upper		42.9 $(13.3)^{b,k}$	103.0 $(23.2)^{b}$
		lower		17.4 $(3.1)^{a,k}$	46.2 $(8.8)^{a}$

			V_{cmax} (µmol CO_2 m^{-2} s^{-1})		
			June	**July**	**September**
Birch	B5	upper		67.8 $(10.4)^{b,j}$	75.7 $(3.9)^{b,j}$
		lower		45.1 $(17.2)^{a,j}$	41.6 $(17.5)^{a,j}$
	B8	upper	64.7 $(6.7)^{b}$	68.5 $(9.5)^{b,j}$	69.9 $(11.5)^{b,j}$
		lower	39.3 $(12.0)^{a}$	49.8 $(2.5)^{a,j}$	45.5 $(8.6)^{a,j}$
Poplar	P12	upper	68.3 $(20.5)^{b}$	74.3 $(21.1)^{b,j}$	
		lower	43.7 $(5.8)^{a}$	33.1 $(7.4)^{a,j}$	
	P17	upper		94.3 $(15.3)^{b,j}$	28.0 $(6.6)^{a}$
		lower		56.5 $(16.7)^{a,k}$	22.4 $(2.7)^{a}$

			J_{max} (µmol e^- m^{-2} s^{-1})		
			June	**July**	**September**
Birch	B5	upper		93.3 $(31.5)^{a,j}$	116.2 $(10.4)^{b,j}$
		lower		59.3 $(30.7)^{a,j}$	62.3 $(23.2)^{a,j}$
	B8	upper	114.2 $(29.7)^{b}$	97.3 $(22.4)^{b,j}$	98.3 $(16.8)^{b,j}$
		lower	65.5 $(24.8)^{a}$	66.6 $(7.7)^{a,j}$	62.9 $(17.5)^{a,j}$
Poplar	P12	upper	139.1 $(25.0)^{b}$	112.6 $(31.5)^{b,j}$	
		lower	66.0 $(22.0)^{a}$	44.2 $(8.4)^{a,j}$	
	P17	upper		143.9 $(20.1)^{b,j}$	56.2 $(11.0)^{b}$
		lower		79.9 $(16.1)^{a,k}$	42.6 $(5.9)^{a}$

$^{a,\,b}$ *denote significant differences (*$p < 0.05$*) between the upper and the lower canopy layer within each characteristic, plot and measurement period.*

$^{j,\,k}$ *denote significant differences (*$p < 0.05$*) between the two plots of a species, within each characteristic, layer and measurement period.*

Pearson's (for normally distributed data) and Spearman's rho (for non-normally distributed data) correlation coefficients together with simple linear regression analysis (SPSS 11.5) were used to check the mutual correlation between leaf photosynthetic characteristics. Analyses were based on individual parameter values for all studied leaves. As can be seen in Fig. 2.5 and Table 2.4, a significant positive linear relationship was found between A_{max}, V_{cmax}, and J_{max}, for both birch and poplar.

ANCOVA was applied to check differences of slopes (Table 2.4). Other mutual relationships between leaf photosynthetic characteristics were not evident.

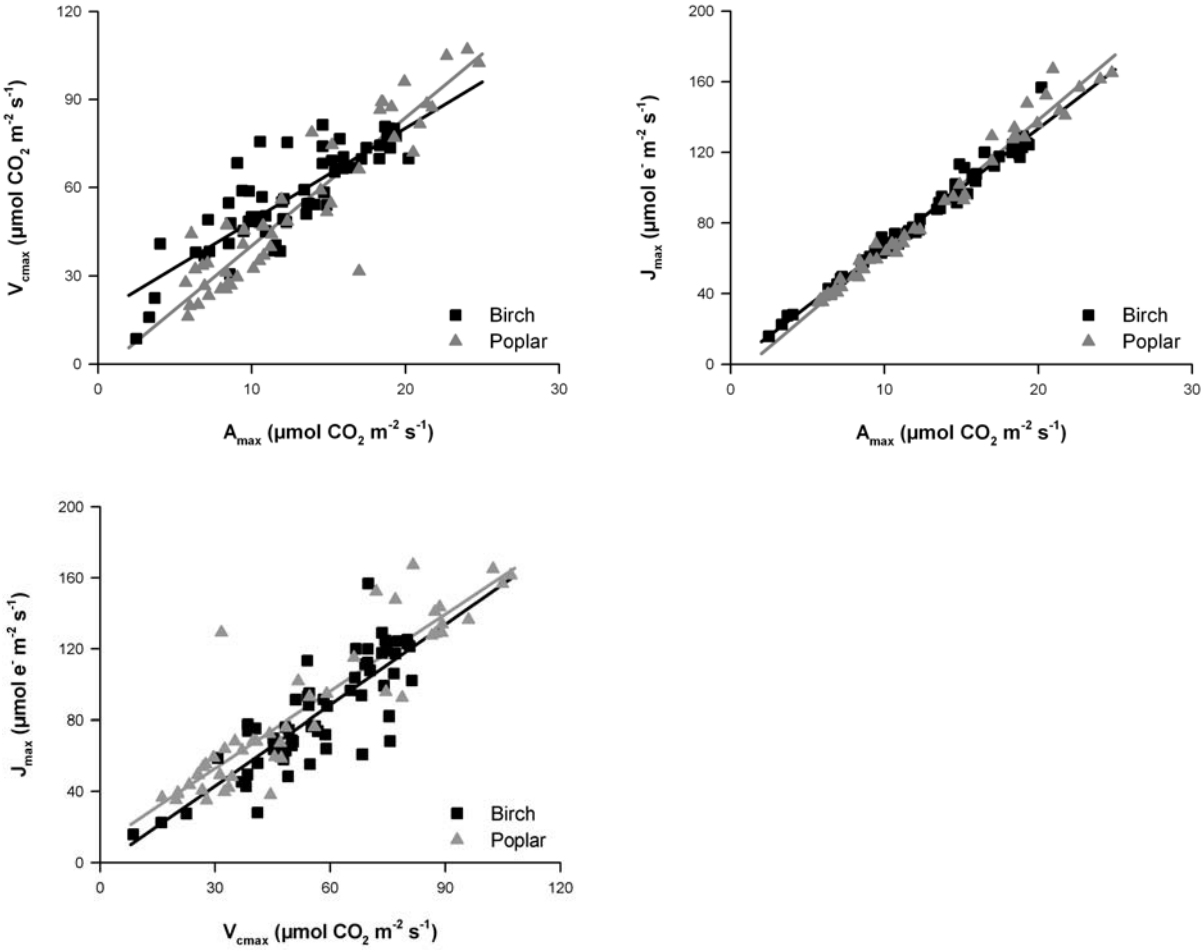

Figure 2.5. *Relationships between leaf photosynthetic characteristics for birch (n = 60) and poplar (n = 48); all individual leaf data for the three measurement periods and both canopy layers were considered. The linear regression equations are given in Table 2.4.*

Table 2.4. *Coefficients (and s.e.) of the linear relationship between the dependent variable Y and the independent variable X for birch (n = 60) and poplar (n = 48); see text for description of the variables.*

Y	X	Species	Intercept	Slope	R^2_{adj}
V_{cmax}	A_{max}	Birch	17.1 (3.52)	3.15 (0.27)^a	0.71
		Poplar	-3.17 (3.65)*	4.35 (0.26)^b	0.86
J_{max}	A_{max}	Birch	-0.61 (1.99)*	6.70 (0.15)^a	0.97
		Poplar	-8.73 (2.20)	7.36 (0.16)^b	0.98
J_{max}	V_{cmax}	Birch	-2.04 (7.85)*	1.51 (0.13)^a	0.69
		Poplar	9.94 (5.76)*	1.44 (0.10)^a	0.82

* *denotes non-significant coefficients.*

$^{a,\,b}$ *denote significantly different slopes* (p *< 0.05) between the two species for a specific linear regression.*

2.3.4. Relation between photosynthetic characteristics and other leaf traits

Pearson's and Spearman's rho correlation coefficient were also applied to find significant correlations between leaf photosynthetic characteristics and SLA, LA and LWC for birch and poplar separately. When the two canopy layers were studied individually, almost no significant correlations were detected. Pooling the data from both layers together revealed a significant positive correlation between A_{max}, V_{cmax} or J_{max} on the one hand, and LA on the other hand, for both species. For birch, correlation coefficients amounted to 0.58, 0.72 and 0.56 for A_{max}, V_{cmax} and J_{max}, respectively, while for poplar, correlation coefficients were 0.52, 0.39 and 0.53 for A_{max}, V_{cmax} and J_{max}, respectively. Only a weak correlation between leaf photosynthetic characteristics and LWC ($r < 0.4$) was found for both species, and finally, no significant correlation with SLA could be detected.

2.4. Discussion

2.4.1. Seasonal dependence of leaf traits

Comparison of SLA values published in literature is sometimes difficult, as the exact methodology applied (rehydration, removal of petioles, …) influences the SLA value (Garnier *et al.* 2001). However, SLA values found in the short-rotation plantation at Zwijnaarde (Fig. 2.1, Table 2.1) fitted well within the range of values mentioned in literature. SLA reported for deciduous tree species varies from 8 to 38 m^2 kg^{-1} DM (Abrams *et al.* 1994, Bartelink 1997, Niinemets *et al.* 1998, Cornelissen *et al.* 1999, Reich *et al*. 1999, Verwijst and Telenius 1999, Gonzalez-Real and Baille 2000, Evans and Poorter 2001, Garnier *et al.* 2001, Koike *et al.* 2001, Bouriaud *et al.* 2003, Al Afas *et al.* 2005, Laureysens *et al.* 2005), with some exceptional high values up to 50 m^2 kg^{-1} DM (Niinemets *et al.* 2004). The LWC values found here (Table 2.1) were also comparable to those mentioned in literature (Al Afas *et al.* 2005). Individual leaf areas LA of poplar (Table 2.1) were low compared to the study of Barigah *et al.* (1994), a study conducted under comparable climatic conditions. LA measured in Zwijnaarde was more comparable to the results of Laureysens *et al.* (2005), who studied 17 poplar clones in a short-rotation plantation near Boom (Belgium). In our study, a decreasing trend of SLA with time was observed (Fig. 2.1, Table 2.2), as was also seen in other studies (Gun *et al.* 1999, Niinemets *et al.* 2004, Simioni *et al.* 2004). This decrease in SLA due to leaf senescence can be attributed to an increase in leaf thickness and/or leaf tissue density, two components determining SLA (Niinemets 1999, Reich *et al*. 1999, Lafarge and Hammer 2002).

Values for A_{max} in this study (Table 2.3) were comparable to values reported in literature for temperate broad-leaved species (Abrams *et al.* 1994, Barigah *et al.* 1994, Deraedt and Ceulemans 1998, Le Goff *et al.* 2004). V_{cmax} and J_{max} also corresponded well with reported values (Deraedt and Ceulemans 1998, Medlyn *et al.* 1999, Wilson *et al.* 2001, Dungan *et al.* 2004, Niinemets *et al.* 2004). As an example, Casella and Ceulemans (2002) mentioned V_{cmax} values of 17 to 93 µmol CO_2 m^{-2} s^{-1}

and J_{max} values ranging from 45 to 186 µmol e^{-} m^{-2} s^{-1}, for three poplar clones growing in a short-rotation plantation.

When significant differences were found between the SLA values of the two poplar plots, it appeared that plot P17 always had the lowest SLA (Fig. 2.1). In addition to this, it can be remarked from Table 2.3 that the measurements in July revealed higher values for A_{max}, R_d, I_c, V_{cmax} and J_{max} for plot P17 compared to plot P12. Measurements of soil matrix potential up to 1 m depth (Van de Casteele 2004) gave rise to the supposition that plot P17 was in general drier than plot P12. As such, the observed differences in leaf characteristics for the two poplar plots confirmed the studies of other authors (Abrams *et al.* 1994, Pierce *et al.* 1994, Reich *et al.* 1999, Wright and Westoby 2002, Wright *et al.* 2002), who found lower SLA values and higher net photosynthesis rates on drier sites. For the two birch plots, there was no evidence that one of the two plots was drier than the other, which might explain the more comparable SLA values (Fig. 2.1) and leaf photosynthetic characteristics (Table 2.3) found in plot B5 and B8.

It could be expected that plots with high photosynthetic parameters show a high biomass production. However, as described in Chapter 3, the actual aboveground biomass production was higher in plot P12 compared to P17 : 5.0 and 3.2 t DM ha^{-1} $year^{-1}$ after four growing seasons for P12 and P17, respectively (Table 3.3). This contradiction can be partly explained by drought stress, which is probably more pronounced on the drier plot P17. The photosynthetic leaf characteristics listed in Table 2.3 are determined under optimal conditions of water supply, while in reality, it can be expected that trees will suffer more severely from drought stress on the driest plot P17. The difference in maximal LAI measured in the summer of 2004 on P12 (3.7 m^2 m^{-2}) compared to P17 (3.5 m^2 m^{-2}), was rather low, and it can be supposed that this low difference cannot explain the higher aboveground production found in P12. Possibly, trees growing on plot P17 invest more carbon in the belowground tree compartments, as this seems to be a strategy of adaptation to drought, or resistance to diseases, as suggested by Barigah *et al.* (1994) and Tardieu *et al.* (1999). However, this belowground carbon storage was not specifically monitored in our study. An additional explanation for the lower production in plot P17 could be the fact that trees in plot P17 suffered more severely from an infection of *Colletotrichum gloeosporioides* than trees in plot P12.

Wilson *et al.* (2001) emphasized the importance of considering the seasonal dependence of V_{cmax} when estimating net ecosystem C uptake. They found that using the maximum value for V_{cmax} instead of the observed temporal pattern overestimated the annual net ecosystem exchange by nearly 50 %. Medlyn *et al.* (1999) and Urban and Marek (1999) also mentioned a decreasing trend in photosynthetic characteristics over time. On the other hand, the study of Niinemets *et al.* (2004) demonstrated that foliar photosynthetic capacity is less variable in field conditions than thought previously. A lack of seasonal variation in ecophysiological parameters of deciduous tree species was also mentioned by Abrams *et al.* (1994). At Zwijnaarde, the evolution with time of the photosynthetic characteristics was not obvious (Table 2.3). However, when concentrating on A_{max}, V_{cmax} and J_{max} (Table 2.3), a different pattern could be distinguished for birch and poplar. For birch, almost no difference was observed for the three measurement periods, while for poplar, distinct lower parameter values were measured in September compared to June and July. This lack of seasonal change in leaf photosynthetic parameters for birch can be caused by the rather limited time period (June-September) during which measurements took place. Koike *et al.* (2001) signalled a substantial decline of A_{max} values in October, at the onset of senescence, compared to mid-summer, for different deciduous broadleaved species. Leaf senescence of birch only started after our last measurement period. For poplar on the contrary, leaf senescence started much earlier, as is illustrated by the fact that all leaves in plot P12 were already dead by the time of the September measurements. As such, measurements in this period reflected much more the autumnal senescence period as described by Wilson *et al*. (2001) than the measurements for birch. Additional measurements of leaf photosynthetic characteristics of birch in the leaf development period (May) and during the period of leaf senescence (October-November) are needed to get a complete overview of the evolution of these characteristics. However, our results seemed to confirm the study of Wilson *et al*. (2001), who stated that seasonality of leaf photosynthetic characteristics should be approximated and incorporated in the simulations of net ecosystem C uptake.

2.4.2. Height dependence of leaf photosynthetic characteristics

A prime factor governing a leaf's photosynthetic productivity is its position in the plant canopy, which determines its light environment and its rate of net CO_2 uptake (Boardman 1977, Lambers *et al.* 1998). The observed differences in photosynthetic characteristics of the upper and lower canopy layer (Table 2.3) are in accordance to the consensus that sun leaves, situated on top of the canopy, have in general higher A_{max}, I_c, R_d, V_{cmax} and J_{max} than shade leaves, growing at the bottom canopy layers (Boardman 1977, Lambers *et al.* 1998, Niinemets *et al.* 1998, Gonzalez-Real and Baille 2000, Koike *et al.* 2001, Stenberg *et al.* 2001, Casella and Ceulemans 2002, Le Goff *et al.* 2004, Niinemets *et al.* 2004). The decrease in net assimilation from the upper to the lower crown layers can be explained by the general decrease in the level of global radiation and also by the decrease in N concentration in leaves from the top to the bottom of the crown (Le Goff *et al.* 2004). However, the present study was not designed to investigate leaf chemical characteristics, neither the light profile in the canopy. As such, the previous statement cannot be denied nor confirmed. The similarity of the quantum efficiency α_c from upper and lower canopy layers also confirmed previous studies (Boardman 1977, Stenberg *et al.* 2001, Whitehead and Gower 2001, Niinemets *et al.* 2004). This similarity indicates that there was a constant quantum efficiency for photosynthetic CO_2 reduction irrespective of the light intensity for growth (Boardman 1977). If multilayer models are used to scale up CO_2 fluxes from the leaf to the canopy level, this height dependence of leaf photosynthetic characteristics is an important factor that should certainly be taken into account.

In general, a trend of SLA increasing towards the lower crown layers is observed (Niinemets and Kull 1994, Maguire and Bennett 1996, Walcroft *et al.* 1997, Gun *et al.* 1999, Meziane and Shipley 1999, Tardieu *et al.* 1999, Verwijst and Telenius 1999, Gonzalez-Real and Baille 2000, Koike *et al.* 2001, Casella and Ceulemans 2002, Marshall and Monserud 2003, Burns 2004, Le Goff *et al.* 2004, Niinemets *et al.* 2004, Simioni *et al.* 2004). This variation in SLA is due to morphological differences between sun and shade leaves, caused by differences in light conditions within the canopy (Bartelink 1997, Poorter and De Jong 1999, Al Afas *et al.* 2005). Mean values of SLA measured on the leaves used for the determination of photosynthetic characteristics are shown in Fig. 2.6.

For birch, no significant differences between upper and lower layer SLA values were noted, while for poplar, significantly higher SLA values were found in the lower canopy layer in 3 of the 4 cases. Cermák *et al.* (1998) explained the lack of SLA differentiation with height by the rather limited crown depth and the small gradient in light profile in their study. Probably, the limited crown depth in the young tree plantation at Zwijnaarde can therefore partly explain the similarity in SLA values for upper and lower canopy layers in the case of birch. The maximum LAI of the two birch plots, measured in the summer of 2004, was 2.5 $m^2 m^{-2}$ and 2.3 $m^2 m^{-2}$ for B5 and B8, respectively. This is obviously lower than the LAI values of 3.7 $m^2 m^{-2}$ for P12 and 3.5 $m^2 m^{-2}$ for P17. The higher LAI of poplar probably caused a more pronounced light profile in the canopy than for birch, and, as a consequence, SLA differences between upper and lower canopy layers were more prominent. For both species, larger leaves were found in the upper layer of the canopy (Fig. 2.6), confirming the study of Niinemets (1996) who noted larger LA values in higher canopy layers of 60 woody taxa. For poplar, this phenomenon was also observed by Al Afas *et al.* (2005). Moreover, these authors found higher LWC values in the lower canopy layers. In our study however, these differences were in most cases not significant, and there was no clear tendency for a higher or lower LWC in the lower layers (Fig. 2.6). Concluding, we can state that differences in photosynthetic characteristics between upper and lower canopy layers were much more prominent than differences in morphological leaf traits.

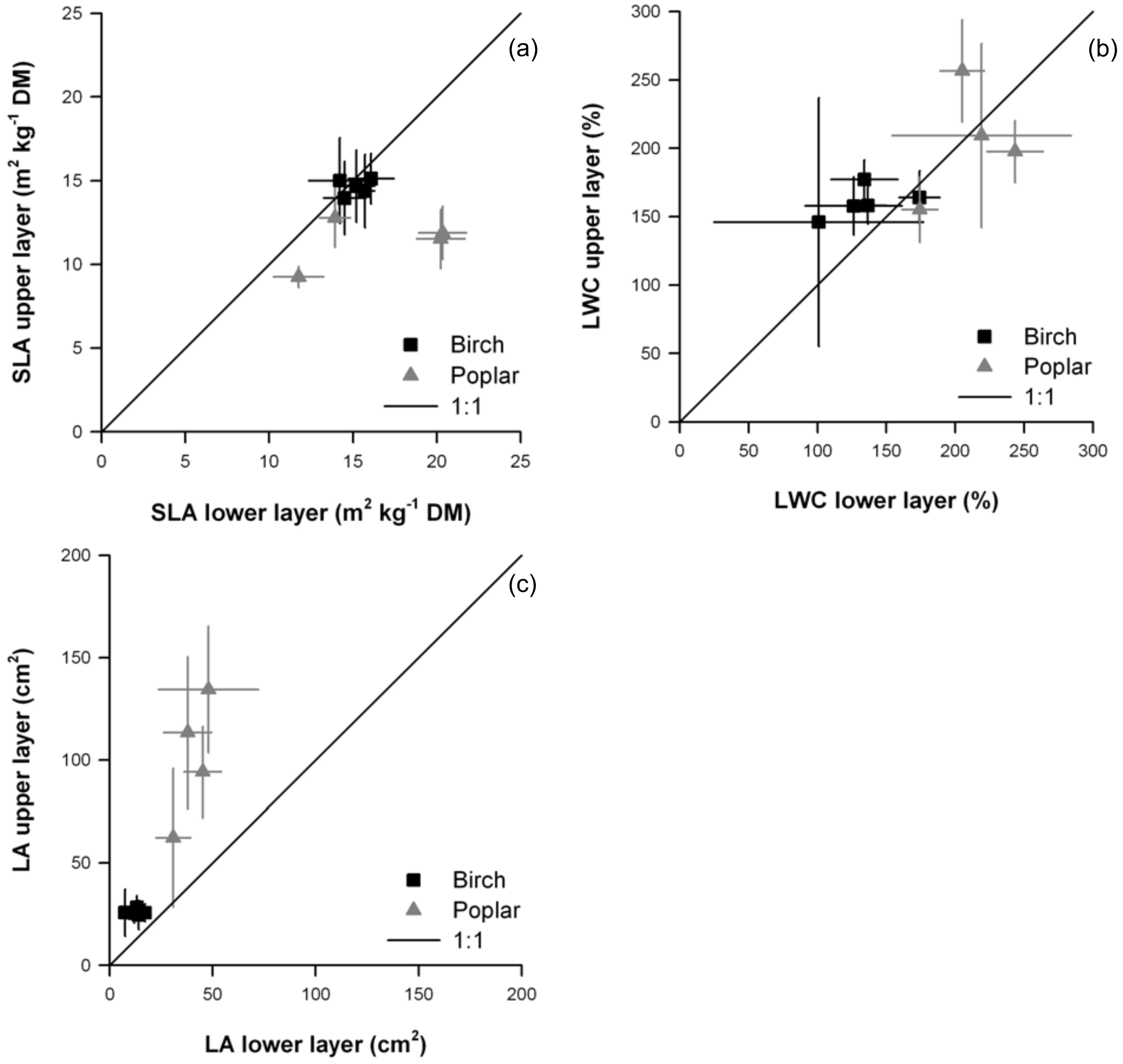

Figure 2.6. *Comparison of mean values (n = 6) per plot of a) specific leaf area SLA, b) leaf water content LWC and c) individual leaf area LA of upper and lower canopy layer leaves, used for determination of photosynthetic characteristics during the same measurement period (June, July or September 2004); error bars represent ± 1 standard deviation.*

2.4.3. Relationship between leaf characteristics

The negative relationships between SLA and LA, and the increase of SLA for leaves with a higher LWC (Fig. 2.2), were in agreement with the study of Al Afas *et al.* (2005). These relationships, together with the decreasing trend of SLA over the growing season (Fig. 2.1), reflect the fact that the mean individual leaf area increases over the growing season, and that the mean LWC decreases. The strong positive relationships between A_{max}, V_{cmax} and J_{max} (Table 2.4) confirmed previous research (Leuning 1997, Niinemets *et al.* 1998). The strong correlation between V_{cmax} and J_{max} reflects the optimum allocation of resources, which guarantees that Rubisco and electron transport co-limit photosynthesis, such that no excess capacities remain (Niinemets *et al.* 1998).

Within a single species, SLA is generally negatively related to leaf photosynthetic capacity (Abrams *et al.* 1994, Niinemets 1999, Evans and Poorter 2001, Grubb 2002). In our study, we found a significantly higher A_{max} value for the upper canopy layers, for both tree species. However, for birch, there was no clear difference in SLA between these two layers (Fig. 2.6). For poplar, the higher SLA values found in the lower layer confirmed the results mentioned in literature. As stated before, the higher LAI in the two poplar plots compared to the birch plots probably caused a more distinct light profile, influencing the SLA at specific depths in the canopy.

Foliage photosynthetic characteristics as A_{max}, V_{cmax} and J_{max} are mentioned to have a strong positive correlation with nitrogen (N) concentrations (expressed on an area base), while SLA scales negatively with leaf N content (Pierce *et al.* 1994, Walcroft *et al.* 1997, Niinemets 1999, Reich *et al.* 1999, Evans and Poorter 2001, Meziane and Shipley 2001, Warren and Adams 2001, Casella and Ceulemans 2002, Al Afas *et al.* 2005). These N relationships are often used to estimate the parameters V_{cmax} and J_{max}, although the prediction of leaf N content remains a difficult problem (Medlyn *et al.* 1999, Whitehead and Gower 2001). As there was no clear relationship between leaf photosynthetic characteristics and SLA in this study, it can be doubted that a clear link with leaf nitrogen would be found. However, as it was beyond the scope of this investigation to study leaf chemical characteristics, this point is not further discussed here.

2.4.4. SLA models

Lafarge and Hammer (2002) found a rapid decline of SLA of sorghum leaves in the beginning of the growing season, with a smoother decrease afterwards. The SLA models established for Zwijnaarde showed a linear decrease of SLA over (thermal) time (Table 2.2, Fig. 2.3).

The deviation of modelled SLA compared to measured SLA (Fig. 2.4) can be due to the sampling method. During the SLA measuring campaigns in 2001, 2002 and 2003, leaves of different sizes (and ages) were collected. In 2004 on the contrary, photosynthetic characteristics (and SLA) were determined on fully expanded leaves. The SLA of these leaves will have been lower than the SLA of young, not-fully expanded leaves. As such, the mean SLA determined in 2004 could be expected to be lower than the modelled value. A better set of validation data could be obtained when the same sampling technique as in 2001, 2002 and 2003 would be applied. It can be expected that SLA determined in this way will better correspond to the model output. However, the simple SLA models presented here seemed to perform quite well for birch and poplar at the short-rotation plantation at Zwijnaarde.

2.5. Conclusion

In recent years, more and more attention is given to short-rotation forestry (SRF) plantations, as they offer a possibility to sequester carbon from the atmosphere. Poplar is one of the most widely used species in SRF, while birch offers an interesting alternative (Vande Walle *et al.* 2007a and 2007b). Therefore, leaf morphological (specific leaf area, individual leaf area and leaf water content) and photosynthetic characteristics were determined for these two species, growing in a short-rotation plantation at Zwijnaarde (Belgium). Appropriate parameter values are needed when one wants to calculate carbon uptake of a whole plantation by upscaling photosynthesis from leaf to canopy level (Medlyn *et al.* 1999).

A distinct decreasing trend of SLA over time was observed for both species, due to leaf senescence. This is an important conclusion towards the calculation of LAI based on leaf biomass weight and SLA. If the seasonal evolution of SLA is not taken into account, an erroneous LAI will be obtained. This can have large consequences for the simulation of carbon fluxes of the whole canopy (Pierce *et al.* 1994, Ceulemans *et al.* 1996). Leaf photosynthetic characteristics exhibited a less pronounced change over time. However, a clear distinction could be made between upper and lower canopy layers. Leaves from the upper canopy layer had photosynthetic characteristics which are typical for sun leaves, while at the bottom of the canopy, shade leaves were growing. This vertical variation is important in the view of building accurate canopy carbon cycling models. The seasonal variation of SLA and height dependence of leaf photosynthetic characteristics was noted for both tree species under consideration.

The simple empirical SLA models that were established, link SLA with easily measurable variables as day of the year, number of days since budburst, mean air temperature or global radiation. The models described here appeared to work well for the particular short-rotation plantation of birch and poplar used in this study. However, they could easily be calibrated for other species or other locations.

The obtained information on leaf morphological and photosynthetic characteristics will be used in further applications and upscaling procedures, more particularly to fine-tune the FORUG model (Boonen *et al.* 2002, Verbeeck *et al.* 2006) to scale up carbon fluxes from leaf to stand level for short-rotation plantations.

Chapter 3

Biomass production at the short-rotation forestry plantation at Zwijnaarde after 4 years of tree growth

Adapted from : Vande Walle, I., N. Van Camp, L. Van de Casteele, K. Verheyen and R. Lemeur. 2007. Short-rotation forestry of birch, maple, poplar and willow in Flanders (Belgium) I. Biomass production after 4 years of tree growth. Biomass Bioenerg. 31 (in press).

Abstract

During the last three decades, oil crises, agricultural surpluses and global climate change enhanced the interest in short-rotation forestry (SRF). In this study, the biomass production of birch (*Betula pendula* Roth), maple (*Acer pseudoplatanus* L. - Tintigny), poplar (*Populus trichocarpa* x *deltoides* - Hoogvorst) and willow (*Salix viminalis* - Orm) growing under a short-rotation management system were compared after a period of 4 years. The plantation was established on former agricultural land. The sandy soil had a mean pH of 4.5 and a mean carbon content of 1.0 %. Survival rates after 4 years were 75.8, 96.8, 86.3 and 97.6 % for birch, maple, poplar and willow, respectively. Mean actual annual biomass production for these four species amounted to 2.6, 1.2, 3.5 and 3.4 t dry mass ha^{-1} year^{-1}, respectively. The large variation in biomass production at the different plots of the plantation could not be explained by the measured soil parameters. Biomass production results found here were in the lower range of values reported in literature. However, in contrast to most other studies, no weed control, fertilization or irrigation was applied in this experiment. As marginal agricultural soils are suboptimal for the growth of poplar and willow, birch can be considered as an interesting alternative for the establishment of short-rotation plantations in Flanders.

3.1. Introduction

Since the energy crisis of the 1970s, many European countries have shown an increased interest in renewable energy sources. The use of biomass produced by either crops or short-rotation plantations was one of the options studied at that time (Tahvanainen and Rytkönen 1999). In the late 1980s, the large agricultural surpluses in Western Europe had to be reduced, which resulted in large areas of land being released from agricultural production. The establishment of short-rotation plantations on this arable land therefore fits in the European set-aside policy. The growing concern about the consequences of burning fossil fuels on the global climate system enhanced the attention for short-rotation forestry (SRF) during the last decade. Under the Kyoto Protocol, the European Union committed itself to an 8 % reduction in annual greenhouse gas emissions by the first commitment period (2008-2012), compared to the reference year 1990 (Schulze *et al.* 2002). The 'White Paper of the European Commission on renewable sources of energy' (EC 1997) gave a clear political signal by setting the target at increasing the renewable energy sources, including biomass, to reach 12 % of the European gross energy consumption by 2010. This is a doubling compared to the situation in 1997 (EC 1997, EC 2004). An extensive literature review showed that it can be expected that in the future, bioenergy plantations will become the most important source of biomass for energy on a global scale (Berndes *et al.* 2003). In short-rotation forestry systems, fast-growing species are grown to attain high yields of biomass. Highest yields are obtained under intensive management systems, including weed control, fertilizer application and irrigation (Labrecque *et al.* 1997, Ceulemans and Deraedt 1999, Tahvanainen and Rytkönen 1999, Adegbidi *et al.* 2001, Mead 2005).

From a survey executed by Meiresonne *et al.* (pers. comm.), it appeared that, in principle, the Flemish farmers are not against the implementation of SRF on their former fields. However, it becomes clear though that farmers, willing to introduce plantations of woody biomass, will certainly not use their best agricultural soils for this purpose. This is an important element in the choice of a suitable species to be used in biomass plantations. It can be expected that the water and nutrient status of these marginal fields will not be optimal for tree growth at all.

Most often, poplar and willow are recommended for planting in short-rotation systems. Some research focussed on the use of birch (Ledin 1996a, Hytönen and Kaunisto 1999, Telenius 1999, Hytönen and Issakainen 2001, Jõgiste *et al.* 2003) and alder (Telenius 1999, Proe *et al.* 2002, Uri *et al.* 2002). In this study, four tree species were used. Poplar and willow were selected as the most frequently used species in short-rotation plantations, while birch and maple represented possible alternatives. Birch is an indigenous species in Flanders, while both birch and maple, two shade-intolerant species, are known to have a high regeneration capacity and a high growth rate, even on poor soils. In this chapter, the aboveground biomass production of the four tree species under a short-rotation forestry system were compared after four years of tree growth, and production results of this plantation were compared to results from other (European) studies.

3.2. Material and methods

3.2.1. Short-rotation plantation

The plantation that served as study site is extensively described in § 1.6.1. It should be mentioned here that the plantation activity was preceded by tillage of the upper 20 cm of the soil, while afterwards, no specific management activities such as weed control, fertilization or irrigation were performed on the plantation. As such, biomass production was examined under non-ideal growing conditions, and the SRF system can be considered as an extensively managed plantation.

3.2.2. Allometric relationships

At the end of 2003, site- and species-specific allometric relationships were established by destructive sampling of a number of trees per species and per diameter class. Trees were cut at a height of 10 cm. Stems and branches were dried until constant weight. Allometric relationships of the form AGDM = $a \cdot d_{30}^{b}$ were established for each species, with AGDM the aboveground dry mass of the tree (stems and branches, in g dry mass or g DM) and d_{30} the diameter at 30 cm (in mm). This power function is biologically reasonable and in accordance to literature (Bergkvist and Ledin 1998, Tahvanainen and Rytkönen 1999, Laureysens *et al.* 2004, Nordh and Verwijst 2004). The number of trees harvested per species and the values for the coefficients *a* and *b* are shown in Table 3.1. The adjusted R^2 values were as high as 0.98 for all four species.

Table 3.1. *Number of sample trees (n), regression coefficients* a *and* b *(with standard error) of the allometric power equation AGDM =* $a \cdot d_{30}^{b}$*, with AGDM : aboveground dry mass of the tree (g DM) and* d_{30} *: diameter at 30 cm (mm)*

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

3.2.3. Aboveground biomass production

In January 2005, the diameter at 30 cm height (d_{30}) of 20 randomly chosen trees per plot was measured with a mechanical calliper. Trees growing in the outer two rows of a plot were excluded from the measurements in order to reduce edge effects. The allometric relationships described above were used to convert the diameter d_{30} in total aboveground dry mass (AGDM) of the tree.

The potential and the actual biomass production of each plot were calculated according to eq. 3.1 :

$$PROD = \frac{(AGDM_{mean} - PDM_{mean})}{1000000} \cdot D \qquad \textbf{(3.1)}$$

where PROD is biomass production of the plot (t DM ha^{-1}), $AGDM_{mean}$ is mean aboveground dry mass per tree for a specific plot, determined in January 2005 (g DM $tree^{-1}$), PDM_{mean} is mean planted aboveground dry mass per tree, determined in March or April 2001 (g DM $tree^{-1}$), D is stem density of the plot (# trees ha^{-1}) and 1000000 is conversion factor from g DM to t DM.

For poplar and willow, which were planted as cuttings, the mean planted dry mass, PDM_{mean}, was equal to zero. PDM_{mean} was 3.0 and 5.1 g DM $tree^{-1}$ for birch and maple, respectively.

To calculate the potential biomass production $PROD_{pot}$, D equalled the initial tree density D_{ini} (6667 trees ha^{-1} for birch and maple, 20000 trees ha^{-1} for poplar and willow). The actual biomass production $PROD_{act}$ of the different plots was calculated based on the actual tree density D_{act}. This actual density was determined in the field in January 2005, by counting all surviving trees per plot. The survival rate (SuR) per plot was calculated as the ratio between D_{act} and D_{ini}, expressed in terms of percentage.

3.3. Results

3.3.1. Diameter d_{30} and aboveground dry mass AGDM

In Table 3.2, the mean diameter at 30 cm (d_{30}) and the mean aboveground dry mass per tree (AGDM) is given for all plots of the short-rotation plantation at Zwijnaarde. There were no results available for plot 2. For willow, there was a negative relationship between the mean diameter and the number of shoots : the highest mean diameters were found in the plots with the lowest mean number of shoots per tree. ANOVA and post-hoc tests (Duncan and Tamhane's test) (SPSS 11.5) revealed significant differences in d_{30} between plots for all four species. Regarding the AGDM, no significant differences were found between the four birch plots (Table 3.2), while for the other species, two (maple and poplar) or more (willow) groups of plots could be distinguished.

Table 3.2. *Mean diameter at 30 cm height (d_{30}) and mean aboveground dry mass per tree (AGDM) for all plots of the SRF plantation at Zwijnaarde after four growing seasons; s.e. on the mean is given between brackets. Measurements were performed in January 2005. Different letters indicate significant differences in d_{30} or AGDM of the plots for a specific species ($p = 0.05$).*

Species	Plot nr.	Mean d_{30} (mm)		Mean AGDM (g DM tree^{-1})	
Birch	4	49.0	(2.9)^{a,b}	1968	(284)^{a}
	5	46.8	(3.1)^{a}	1810	(262)^{a}
	8	56.2	(3.1)^{b}	2636	(314)^{a}
	11	45.2	(2.5)^{a}	1616	(183)^{a}
Maple	13	25.2	(1.4)^{a}	401	(56)^{a}
	18	32.5	(1.7)^{b}	787	(96)^{b}
	22	35.4	(2.2)^{b}	1025	(147)^{b}
Poplar	1	29.0	(1.3)^{a}	552	(52)^{a}
	6	29.7	(3.4)^{a,b}	749	(197)^{a,b}
	9	33.2	(1.4)^{a,b}	745	(71)^{a,b}
	12	39.4	(1.5)^{b}	1079	(89)^{b}
	16	38.5	(2.0)^{b}	1073	(116)^{b}
	17	32.4	(3.0)^{a,b}	820	(158)^{a,b}
	20	34.7	(2.9)^{a,b}	918	(153)^{a,b}
	21	31.7	(2.1)^{a,b}	709	(84)^{a,b}
Willow*	3	11.6	(0.6)^{a}	283	(37)^{a}
	7	15.6	(1.0)^{b}	523	(65)^{b}
	10	24.0	(1.1)^{d}	1170	(125)^{d}
	14	21.5	(1.2)^{c,d}	870	(142)^{b,c,d}
	15	18.0	(1.1)^{b,c}	771	(91)^{c}
	19	20.1	(1.0)^{c,d}	606	(75)^{b,c}

* *number of shoots was 59, 50, 44, 42, 51 and 36 for the plots 3, 7, 10, 14, 15 and 19, respectively.*

3.3.2. Potential and actual biomass production

The annual potential ($PROD_{pot}$) and actual biomass production ($PROD_{act}$) after four years of tree growth are listed in Table 3.3 for all plots. Survival rates of more than 90.0 % were noted for all maple and willow plots (Table 3.3), which resulted in a very good agreement between the potential and actual biomass production values for the plots of these two species. For birch and poplar, the mean actual biomass production was clearly lower than the potential one, because of the lower survival rates (mean of 75.8 % for birch, and 86.3 % for poplar). As can be concluded from the same table, a large within-species heterogeneity of biomass production was observed.

ANOVA combined with post-hoc tests (Duncan) revealed a significantly lower mean potential biomass production of maple (1.2 t DM ha^{-1} year^{-1}) compared to birch (3.3 t DM ha^{-1} year^{-1}), willow (3.5 t DM ha^{-1} year^{-1}) and poplar (4.2 t DM ha^{-1} year^{-1}). On the other hand, the actual biomass production of maple (1.2 t DM ha^{-1} year^{-1}) and birch (2.6 t DM ha^{-1} year^{-1}) were significantly lower than the $PROD_{act}$ of willow (3.4 t DM ha^{-1} year^{-1}) and poplar (3.5 t DM ha^{-1} year^{-1}).

Table 3.3. *Survival rate SuR, potential ($PROD_{pot}$) and actual biomass production ($PROD_{act}$) of all plots of the SRF plantation at Zwijnaarde after four years of tree growth*

Species	**Plot nr.**	**SuR (%)**	**$PROD_{pot}$ (t DM ha^{-1} $year^{-1}$)**	**$PROD_{act}$ (t DM ha^{-1} $year^{-1}$)**
Birch	4	69.5	3.3	2.3
	5	69.5	3.0	2.1
	8	83.5	4.4	3.7
	11	80.8	2.7	2.2
	mean	*75.8*	*3.3*	*2.6*
Maple	13	93.8	0.7	0.6
	18	97.7	1.3	1.3
	22	98.9	1.7	1.7
	mean	*96.8*	*1.2*	*1.2*
Poplar	1	94.9	2.8	2.6
	6	81.3	3.7	3.0
	9	95.4	3.7	3.6
	12	92.9	5.4	5.0
	16	68.3	5.4	3.7
	17	77.9	4.1	3.2
	20	85.4	4.6	3.9
	21	94.7	3.5	3.4
	mean	*86.3*	*4.2*	*3.5*
Willow	3	97.1	1.4	1.4
	7	98.4	2.6	2.6
	10	98.5	5.9	5.8
	14	98.9	4.4	4.3
	15	97.5	3.9	3.8
	19	95.0	3.0	2.9
	mean	*97.6*	*3.5*	*3.4*

3.4. Discussion

3.4.1. Individual tree growth

Ferm (1993) reported a mean diameter of 92 mm in a *Betula pendula* energy stand in Finland after 14 years of tree growth, for a density of 6475 stems ha^{-1}. This diameter is larger than the value found in Zwijnaarde after a rotation period of 4 years. However, comparison is difficult as the measuring height is not indicated in the Finnish study. In a study on 8 poplar and 2 willow clones, Bungart and Hüttle (2001) found a mean diameter range (height undefined) from 20 to 41 mm after 4 years of tree growth. Values found for poplar and willow in Zwijnaarde were comparable to these results (Table 3.2).

In a Swedish study (Telenius 1999), the mean dry weight of an individual birch tree after four growing seasons was 1247 to 1522 g; values found here (Table 3.2) were slightly higher. Individual values of tree biomass production of poplar clones ranged from 1231 to 4226 g DM $tree^{-1}$ in the study of Telenius (1999), while Barigah *et al.* (1994) found a mean aboveground biomass between 111 and 408 g DM $tree^{-1}$ for five poplar clones during their establishment year. Values found in Zwijnaarde after four years of tree growth were situated in between these results. Bergkvist and Ledin (1998) reported for *S. viminalis* a mean dry weight between 1000 and 2200 g DM stool^{-1} after four years of tree growth. The mean tree dry weight of *S. fragilis* was 1256 g DM in the Swedish study of Telenius (1999), while a value of 1000 to 2100 g DM tree^{-1} was found for a coppiced willow plantation in Ireland (Dawson and McCracken 1995), with a planting density identical to the one in Zwijnaarde, and after a 3 years growing period. Nordh and Verwijst (2004) reported tree dry mass values for different willow clones between 1600 and 2070 g DM tree^{-1} after 4 years of tree growth. Except for plot 10, values found in Zwijnaarde were considerably lower than these literature values (Table 3.2). Possible causes of the mentioned differences are discussed below.

3.4.2. Survival rate

Both maple and willow had a very high survival rate, in all plots (Table 3.3). The low survival rate of birch (mean of 75.8 %) was mainly caused by the dieback of a number of birches during the first year, caused by the exposure of the root systems to strong wind and cold temperatures at the moment of tree planting. Unfortunately, we were not able to replant these trees later on. The mean survival rate for poplar was 86.3 %, but ranged from 68.3 % in plot 16 to 95.4 % in plot 9. Some of the poplar plots suffered severely from an infection of *Colletotrichum gloeosporioides*. As such, a number of poplars died during the four years of the experiment. However, the survival rates in Zwijnaarde were higher than most results found in literature (Bergkvist and Ledin 1998, Armstrong *et al.* 1999, Kopp *et al.* 2001, Nordh and Verwijst 2004). The poplars in Zwijnaarde had a very high and uniform survival rate during the establishment year, as was also reported by Ceulemans and Deraedt (1999). These authors (Deraedt and Ceulemans 1998) also mentioned an infection of poplar species with rust and other pathogens, in a short-rotation plantation situated in Boom, at a distance of 60 km from our study site.

3.4.3. Intra-species variation in biomass production

As indicated in Table 3.3, there was a large variation in biomass production between the different plots of the four species. These differences in production were possibly due to a large heterogeneity of the soil physical and soil chemical characteristics, as is also mentioned by Venendaal *et al.* (1997) and Tahvanainen and Rytkönen (1999). To test this hypothesis, a number of abiotic soil parameters were studied. These comprised the carbon (%) and nitrogen content (ppm), the C:N ratio, the pH_{KCl} and the bulk density (g cm^{-3}) of the soil layers 0-5 cm, 5-15 cm, 15-30 cm, 30-50 cm and 50-100 cm; mean values of these parameters were also calculated for the layers 0-30 cm, 0-50 cm and 0-100 cm (Van de Casteele 2004). However, neither regression analysis nor PCA resulted in a significant relationship between one or more of these soil parameters and the potential or actual biomass production of one of the tree species. This means that probably other characteristics, which were not measured, caused the variation in biomass production. These can include parameters related to micro-scale differences in soil water content or other element concentrations than the ones measured. Tahvanainen and Rytkönen (1999) stated that there were also non-climatic and non-soil-related factors which had an influence on the biomass production of *Salix viminalis* clones in their study. Our results also confirmed the findings of Laureysens *et al.* (2004), who found no clear relationship between the production of 17 poplar clones, among which the clone 'Hoogvorst', and a whole range of soil characteristics.

3.4.4. Comparison between the biomass production at Zwijnaarde and at other European sites

Most biomass production results of SRF systems described in literature refer to poplar or willow plantations (Barigah *et al.* 1994, Labreque *et al.* 1997, Bergkvist and Ledin 1998, Deraedt and Ceulemans 1998, Armstrong *et al.* 1999, Hofmann-Schielle *et al.* 1999, Tahvanainen and Rytkönen 1999, Telenius 1999, Adegbidi *et al.* 2001, Bungart and Hüttl 2001, Kopp *et al.* 2001, Deckmyn *et al.* 2004a, Laureysens *et al.* 2004, Nordh and Verwijst 2004, Fisher *et al.* 2005), only a few are related to birch plantations (Ledin 1996a, Telenius 1999, Hytönen and Issakainen 2001), and no results were found for maple, except for the reference by Ceulemans *et al.* (1996) to unpublished results. In general, biomass production results for poplar are in the range between 10 and 15 t DM ha^{-1} year^{-1}, while for willow, a range from 10 to 12 t DM ha^{-1} year^{-1} is most often referred to (Ceulemans *et al.* 1996). Some extremely high productivity numbers are also quoted : 27.5 t DM ha^{-1} year^{-1} for a new hybrid poplar clone under optimal conditions, 28.5 t DM ha^{-1} year^{-1} for *Salix viminalis* in Canada and 36 t DM ha^{-1} year^{-1} for *Salix dasyclados* in intensively irrigated and fertilized small plots in Sweden. However, in practice, biomass production is often lower than these potential values (Ceulemans *et al.* 1996).

In Table 3.4, the mean actual biomass production $PROD_{act}$ of birch in Zwijnaarde is compared to results from other European studies; only results for a rotation length of at least 4 years were presented. Table 3.5 gives an overview of production results for poplar; here, only results from European studies were selected, from a first rotation with a minimum length of 4 years and a minimum planting density of 5000 trees ha^{-1}; results from plantations on peat soils were not listed. Results from European *Salix viminalis* plantations can be found in Table 3.6; the same restrictions as for the poplar plantations were applied. From these three tables, it can be concluded that the results found in Zwijnaarde fell in the lower range of the results from other European studies. However, in SRF plantations, cultural factors such as planting density, rotation length, fertilization and irrigation all affect biomass production (Adegbidi *et al.* 2001). Moreover, site characteristics as soil type and climate also result in different production potentials.

Table 3.4. *Biomass production of European SRF plantations of birch; Species code (Sp.) 1 :* Betula pubescens*, 2 :* Betula pendula*, 3 :* Betula verrucosa*; Country code (C.) A : Austria, B : Belgium, F : Finland, S : Sweden; Management code (Manag.) W : weed control, F : fertilizers; planting density (Plant. dens.), rotation length (Rot. length) and biomass production (Biomass prod.) are also listed; n.i. : no information.*

Sp.	**C.**	**Soil type**	**pH**	**Manag.**	**Plant. dens. (# trees ha^{-1})**	**Rot. length (years)**	**Biomass prod. (t DM ha^{-1} year^{-1})**	**Reference**
1	F	peat	n.i.	n.i.	30000	4	0.5	Hytönen and Issakainen (2001)
1	F	peat	n.i.	n.i.	30000	8 (2 x 4)	0.7	Hytönen and Issakainen (2001)
1	F	peat	n.i.	n.i.	33000	4	1.2	Hytönen and Issakainen (2001)
1	F	peat	n.i.	n.i.	30000	8	1.3	Hytönen and Issakainen (2001)
1	F	peat	n.i.	n.i.	33000	8 (2 x 4)	1.5	Hytönen and Issakainen (2001)
1	F	peat	n.i.	n.i.	30000	12	1.7	Hytönen and Issakainen (2001)
1	F	peat	n.i.	n.i.	30000	16	1.9	Hytönen and Issakainen (2001)
2	S	clay	6-7	W	5000	6	2.3	Telenius (1999)
1	F	peat	n.i.	n.i.	33000	8	2.6	Hytönen and Issakainen (2001)
2	B	sand	4.5	none	6667	4	2.6	This study
3	A	brown soil	5.4	none	3333	12	3.7	Ledin 1996a
3	A	brown soil	5.4	none	4444	12	4.2	Ledin 1996a
3	A	brown soil	5.4	none	10000	12	5.6	Ledin 1996a
3	A	brown soil	5.4	F	3333	12	5.9	Ledin 1996a
3	A	brown soil	5.4	F	4444	12	6.1	Ledin 1996a
3	A	brown soil	5.4	F	10000	12	7.3	Ledin 1996a

Table 3.5. *Biomass production of European SRF plantations of poplar; Species (Sp.) refers to the parentage of the poplar clone used : B :* P. balsamifera*, D. :* P. deltoides*, N :* P. nigra*, T :* P. trichocarpa*, Ta :* P. tremula*, To :* P. tremuloides*; Country code (C.) B : Belgium, D : Germany, GB : England, S : Sweden; Management code (Manag.) W : weed control, F : fertilizers; planting density (Plant. dens.), rotation length (Rot. length) and biomass production (Biomass prod.) are also listed; rotation length : C means coppiced after establishment year; n.i. : no information.*

Sp.	C.	Soil type	pH	Manag.	Plant. dens. (# trees ha^{-1})	Rot. length (years)	Biomass prod. (t DM ha^{-1} $year^{-1}$)	Reference
D x N	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	2.2	Laureysens *et al.* (2004)
-[a]	D	sand	7.2-7.7	F	8333	4	2.3[a]	Bungart and Hüttl (2001)
D x T	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	2.8	Laureysens *et al.* (2004)
Ta x To	D	sand	n.i.	none	5000	5	3.0	Hofmann-Schielle *et al.* (1999)
Ta x To	D	sand	n.i.	F	8333	5	3.0	Hofmann-Schielle *et al.* (1999)
Ta x To	D	sand	n.i.	F	5000	5	3.2	Hofmann-Schielle *et al.* (1999)
T x D	B	sand	4.5	none	20000	4	3.5	This study
Ta x To	D	sand	n.i.	none	8333	5	3.6	Hofmann-Schielle *et al.* (1999)
D x T	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	3.6	Laureysens *et al.* (2004)
T x D	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	3.7	Laureysens *et al.* (2004)
D x T	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	3.7	Laureysens *et al.* (2004)
T	D	sand	n.i.	F	8333	5	3.7	Hofmann-Schielle *et al.* (1999)
T	D	sand	n.i.	F	16667	5	4.1	Hofmann-Schielle *et al.* (1999)
D x N	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	4.7	Laureysens *et al.* (2004)
T	D	-[b]	n.i.	none	16667	5	4.8[b]	Hofmann-Schielle *et al.* (1999)
T	D	sand	n.i.	none	8333	5	4.8	Hofmann-Schielle *et al.* (1999)
T x B	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	4.8	Laureysens *et al.* (2004)
Ta x To	D	sandy clay	n.i.	none	16667	5	5.2	Hofmann-Schielle *et al.* (1999)
D x N	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	5.2	Laureysens *et al.* (2004)
Ta x To	D	sandy clay	n.i.	F	16667	5	5.5	Hofmann-Schielle *et al.* (1999)
T x D	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	5.8	Laureysens *et al.* (2004)
T	D	sandy clay	n.i.	F	16667	5	5.9	Hofmann-Schielle *et al.* (1999)
T x D	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	5.9	Laureysens *et al.* (2004)
T x D	GB	heavy clay loam	n.i.	W	10000	C + 4	6.4	Armstrong *et al.* (1999)
T x D	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	6.6	Laureysens *et al.* (2004)
T x D	S	clay	6-7	W	5000	6	7.3	Telenius (1999)
T	GB	heavy clay loam	n.i.	W	10000	C + 4	7.5	Armstrong *et al.* (1999)
T x D	S	clay	6-7	W	5000	6	7.6	Telenius (1999)

Table 3.5. *(continued)*

Sp.	C.	Soil type	pH	Manag.	Plant. dens. (trees ha^{-1})	Rot. length (years)	Biomass prod. (t DM ha^{-1} year^{-1})	Reference
T	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	8.0	Laureysens *et al.* (2004)
T x D	GB	heavy clay loam	n.i.	W	10000	C + 4	8.1	Armstrong *et al.* (1999)
N	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	8.2	Laureysens *et al.* (2004)
T	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	8.3	Laureysens *et al.* (2004)
T	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	8.5	Laureysens *et al.* (2004)
T	GB	argilic brown earth	n.i.	W	10000	C + 4	9.2	Armstrong *et al.* (1999)
-[c]	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	10.1[c]	Laureysens *et al.* (2004)
T x D	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	10.4	Laureysens *et al.* (2004)
T x D	B	loam / sandy-loam	7.3-8.0	W	10000	C + 4	11.4	Laureysens *et al.* (2004)
T x D	GB	argilic brown earth	n.i.	W	10000	C + 4	12.8	Armstrong *et al.* (1999)
T x D	GB	argilic brown earth	n.i.	W	10000	C + 4	13.6	Armstrong *et al.* (1999)

[a] : mean of 8 poplar clones; [b] : mean of 2 sites; [c] : mixture of clones

Table 3.6. *Biomass production of European SRF plantations of willow; all results are for studies of* Salix viminalis *clones (species code Sp. 1); Country code (C.) B : Belgium, D : Germany, S : Sweden; Management code (Manag.) W : weed control, F : fertilizers; planting density (Plant. dens.), rotation length (Rot. length) and biomass production (Biomass prod.) are also listed; rotation length : C means coppiced after establishment year; n.i. : no information.*

Sp.	**C.**	**Soil type**	**pH**	**Manag.**	**Plant. dens. (trees ha^{-1})**	**Rot. length (years)**	**Biomass prod. (t DM ha^{-1} year^{-1})**	**Reference**
1[a]	D	sand	7.2-7.7	F	8333	4	2.3[a]	Bungart and Hüttl (2001)
1	D	sand	n.i.	F	8333	5	2.4[b]	Hofmann-Schielle *et al.* (1999)
1	D	-[c]	n.i.	none	16667	5	2.7[c]	Hofmann-Schielle *et al.* (1999)
1	S	clay	5.5-6.2	W, F	10000	C + 4	2.8[d]	Bergkvist and Ledin (1998)
1	D	sand	n.i.	W, F	16667	5	3.1[e]	Hofmann-Schielle *et al.* (1999)
1	S	clay	5.5-6.2	W, F	15000	C + 4	3.4[f]	Bergkvist and Ledin (1998)
1	D	sand	n.i.	none	8333	5	3.4	Hofmann-Schielle *et al.* (1999)
1	B	sand	4.5	none	20000	4	3.4	This study
1	D	sand	n.i.	F	16667	5	4.1[g]	Hofmann-Schielle *et al.* (1999)
1	S	clay	5.5-6.2	W, F	20000	C + 4	4.4[h]	Bergkvist and Ledin (1998)
1	S	clay	5.5-6.2	W, F	25000	C + 4	4.5	Bergkvist and Ledin (1998)
1	S	clay	5.5-6.2	W, F	20000	C + 4	4.5	Bergkvist and Ledin (1998)
1[i]	S	clay	6.5	W, F	20000	4	5.8[i]	Nordh and Verwijst (2004)
1[j]	S	clay	6.5	W, F	20000	4	9.5[j]	Nordh and Verwijst (2004)

[a] : mean of two clones; [b] : minimum of 12 clones; [c] : maximum of 12 clones; [d] : mean of 2 sites; [e] : mean of 3 fertilizer treatments; [f] : mean of 4 planting designs; [g] mean of 3 fertilizer treatments; [h] : mean of 6 planting designs; [i] : mean of 8 fertilizer treatments; [j] : mean of 3 planting designs

3.4.4.1. Planting density and rotation length

Optimal planting densities recommended for willow are 10000 to 20000 trees ha^{-1} (Ledin 1996b, Bergkvist and Ledin 1998, Hofmann-Schielle *et al.* 1999), while Ceulemans and Deraedt (1999) stated that for poplar SRF plantations, optimal densities range from 2500 to 10000 trees ha^{-1}. Optimum rotation length for poplar and willow short-rotation forestry seems to be 4 or 5 years (Ledin 1996b, Ceulemans and Deraedt 1999). Planting densities used at Zwijnaarde were 20000 trees ha^{-1} for both poplar and willow, and results were calculated after a 4 years rotation length. As such, there is no reason to suspect that one of these two factors will have caused the low biomass production of poplar and willow. Hytönen and Issakainen (2001) reported that the highest mean annual biomass production of downy birch (*B. pubescens*) was found for rotation lengths of 8 years and more, while the highest values listed in Table 3.4 were found for a rotation period of 12 years (Ledin 1996a). This leads us to suspect that after the rotation length of only 4 years, the mean annual biomass production of birch in Zwijnaarde is not at its maximum yet, and will enhance if the rotation length will increase. Moreover, sowing birches instead of planting them could enhance the tree density compared to the density applied in Zwijnaarde (6667 trees ha^{-1}). A higher tree density combined with a longer rotation period will most probably improve the biomass production of birch.

3.4.4.2. Weed control and fertilization

In most studies reported in Table 3.4, 3.5 and 3.6, herbicides and/or fertilizers were applied. Weed control is indicated as an important factor during the establishment phase of SRF plantations (Ledin 1996b, Mitchell *et al.* 1999). Although the use of fertilizers is advised by some authors (Ledin 1996b, Hofmann-Schielle *et al.* 1999), fertilization did not always result in a higher biomass production (Telenius 1999, Kopp *et al.* 2001, Deckmyn *et al.* 2004a). The response on nutrients in soils and fertilizers is strongly dependent on a large variety of factors. Nutrient amounts in the soil from former land use, soil texture, rooting behaviour, moisture availability and specific clone or cultivar properties will effect growth and nutrition of the stand, and therefore, have to be considered (Hofmann-Schielle *et al.* 1999). The regular removal of woody biomass may require additional fertilizers to be added, to maintain soil fertility and to sustain high production rates (Mitchell *et al.* 1999, Bungart and Hüttl 2001). If we would have applied chemical or mechanical weed control, biomass production could have been somewhat higher. However, the use of additional fertilizers will probably not have affected the production rates in Zwijnaarde to a large extent, as the plantation was established on former agricultural land, and we only studied the first rotation cycle.

3.4.4.3. Soil characteristics

The optimal pH for the growth of poplar and willow ranges from 5 to 7.5 (Ledin 1996b, Deraedt and Ceulemans 1998, Mitchell *et al.* 1999, Tahvanainen and Rytkönen 1999). The mean pH of the top soil in Zwijnaarde was only 4.5, a value lower than this optimal range. Poplar and willow are known to be species with a high water requirement, and often, water availability is the limiting factor for the biomass production of these species (Venendaal *et al.* 1997, Faaij *et al.* 1998, Hofmann-Schielle *et al.* 1999, Mitchell *et al.* 1999, Deckmyn *et al.* 2004a, Hoffmann and Weih 2005). Birches on the other hand are moderately high consumers of water, and they have a higher drought stress resistance than willow (Ferm 1993). As the plantation in Zwijnaarde is situated on a sandy soil, it can be expected that water availability was too low for an optimal growth of both poplar and willow. An additional explanation of the low biomass production results in Zwijnaarde is the relative large plot size of 400 m^2. Most other studies used much smaller plot sizes, and Pontailler *et al.* (1999) stated that results from small plots are most probably overestimating the real production potential of poplar.

3.4.5. Reflections towards the establishment of SRF plantations in Flanders

The fields that will become available for SRF in Flanders will be marginal for agriculture production, as is the case in Germany (Hofmann-Schielle *et al.* 1999). As such, the site in Zwijnaarde demonstrates fairly well the biomass production that can be expected on these fields in the future. Birch can be considered as a very interesting alternative for the establishment of SRF plantations, as this species has a high adaptability to poor- or medium-quality soils. A rotation period of 8 to 12 years, combined with sowing birches instead of planting, seems promising for higher biomass productions in SRF plantations of this species (Ferm 1993). Further research is needed to test this hypothesis. If poplar is used, a multiclonal plantation structure should be preferred, as this diminishes the risk on severe disease attacks (ODE-Vlaanderen 2001).

In our plantation, neither weed control nor fertilizers were applied. Although the effects of these cultural activities on biomass production are ambiguous, it can have had a negative effect on the biomass production. On the other hand, avoiding the use of herbicides, pesticides and fertilizers averts environmental hazards as groundwater pollution by leaching of nitrogen (Ledin 1996b, Deckmyn *et al.* 2004a, Fisher *et al.* 2005). Moreover, lower (or no) fertilization and decreased (or no) use of herbicides and pesticides can enhance the diversity of both fauna and flora populations in SRF plantations (Sage 1998, Weih *et al.* 2003, Londo *et al.* 2005).

3.5. Conclusion

Biomass production results for birch, maple, poplar and willow found in this study were in the lower range of results from other European studies. The low production for poplar and willow in Zwijnaarde was mainly caused by the limiting soil characteristics, as the combination of a low pH and a sandy soil, implying a suboptimal water availability, are not favourable for the growth of these two species. The low production of birch was mainly attributable to the rotation length of only

4 years and the low planting density in comparison to other experiments. An enhanced mean annual biomass production can be expected for this species when the rotation length and the planting density are increased. Because of the high water consumption of poplar and willow, and the sensitivity of especially poplar to different types of diseases, birch seems to offer an interesting alternative for SRF plantations in Flanders.

However, to gain optimal profit from all functions of SRF plantations, including soil water protection and provision of a (temporary) habitat, it is important that indigenous species, suited to the site, are chosen, and that as less herbicides and fertilizers as possible (by preference : none of them) are applied. In addition, multiclonal plantations offer a better protection against severe pathogen or disease attacks than monoclonal sites, and should therefore be preferred.

Chapter 4

Energy production and CO_2 emission reduction potential of short-rotation forestry plantations in Flanders

Adapted from : Vande Walle, I., N. Van Camp, L. Van de Casteele, K. Verheyen and R. Lemeur. 2007. Short-rotation forestry of birch, maple, poplar and willow in Flanders (Belgium) II. Energy production and CO_2 emission reduction potential. Biomass Bioenerg. 31 (in press).

Abstract

Belgium, being an EU country, has committed itself to a 7.5 % reduction of greenhouse gas emissions during the first commitment period of the Kyoto Protocol. Within this framework, the Flemish government aims at reaching a share of 6 % of renewable electricity in the total electricity production by 2010. In this study, the biomass production of birch, maple, poplar and willow in a short-rotation forestry (SRF) plantation after a growth period of 4 years served as the base to calculate the amount of (electrical) energy that could be produced by this type of bioenergy crop in Flanders. The maximum amount of electricity that could be provided by SRF biomass was estimated at 72.9 GWh_e $year^{-1}$, which only accounts for 0.16 % of the total electricity production in this region. Although the energy output was rather low, the bioenergy production process under consideration appeared to be more energy efficient than energy production processes based on fossil fuels. The high efficiency of birch compared to the other species was mainly due to the high calorific value of the birch wood. The maximum CO_2 emission reduction potential of SRF plantations in Flanders was estimated at only 0.09 % of the total annual CO_2 emission. The most interesting application of SRF in Flanders seemed to be the establishment of small-scale plantations, linked to a local combined heat and power plant. These plantations could be established on marginal arable soils or on polluted sites, and they could be of importance in the densely populated area of Flanders because of other environmental benefits, among which their function as (temporary) habitat for many species.

4.1. Introduction

The European Kyoto target is an 8 % reduction in annual greenhouse gas emissions by the first commitment period (2008-2012), compared to the reference year 1990 (Schulze *et al.* 2002). In this context, the European Commission set the target to increase renewable energy sources to reach 12 % of the European gross energy consumption by 2010 (EC 1997). Short-rotation plantations can be expected to play a major role in the production of biomass for bioenergy. Biomass produced in short-rotation plantations can serve as a substitute for fossil fuels, reducing as such the emission of greenhouse gases to the atmosphere and helping to attain the greenhouse gas emission reduction target (EC 1997, Dubuisson and Sintzoff 1998, Perttu 1998, Matthews 2001, Wirth *et al.* 2004a). Moreover, short-rotation plantations are one of the measures indicated by the Art. 3.4 (Additional activities) of the Kyoto Protocol (Watson *et al.* 2000, Schulze *et al.* 2002), which can lead to a significant uptake of carbon from the atmosphere.

Belgium, being an EU country, has committed itself to a 7.5 % reduction of greenhouse gas emissions during the first commitment period of the Kyoto Protocol. Within this framework, the government of the Flemish region aims at reaching a share of 6 % of renewable electricity in the total electricity production by 2010. To reach this objective, the Flemish energy market has been fully opened for external competition. Therefore, a green certificate system for electricity has been established, and a similar system for 'green heat' production is under development actually. In this way, the Flemish government hopes to stimulate the use of biomass as an energy source (ODE-Vlaanderen 2001, CEC 2004).

In the present study, the potential use of biomass from short-rotation forestry (SRF) in Flanders is evaluated. Actually, only a few hectares of experimental plantations exist in Belgium (Laureysens *et al.* 2005). One of these plantations was established on former agricultural land in 2001, and the growth of four tree species (birch, maple, poplar and willow) was studied there. Based on biomass production results of this plantation (see Chapter 3), the possible contribution of SRF plantations to the total electricity production in Flanders was determined. Furthermore, the potential of SRF plantations to reduce the CO_2 emissions in this region was assessed, within the overall scope of achieving the Kyoto Protocol targets.

4.2. Material and methods

4.2.1. Short-rotation plantation

The short-rotation plantation at Zwijnaarde served as study site for this investigation. An extensive description of soil characteristics, plant material, planting scheme and management activities can be found in § 1.6.1 and § 3.2.1.

4.2.2. Aboveground biomass production

In January 2005, the potential biomass production $PROD_{pot}$ (t dry mass ha^{-1} $year^{-1}$ or t DM ha^{-1} $year^{-1}$) of 21 plots was determined. The methodology applied to calculate $PROD_{pot}$ is described in Chapter 3. In Table 4.1, a summary of annual production results after 4 years of tree growth is given. In this table, both the mean biomass production and the maximum result are presented for each species.

Table 4.1. *Number of plots (n), mean (with standard error between brackets) and maximum (Max.) potential biomass production $PROD_{pot}$ (t DM ha^{-1} $year^{-1}$), and coefficient of variation CV (%) of birch, maple, poplar and willow after four years of tree growth at the short-rotation plantation in Zwijnaarde*

		$PROD_{pot}$ (t DM ha^{-1} $year^{-1}$)		
Species	**n**	**Mean**	**Max.**	**CV (%)**
Birch	4	3.3 (0.4)	4.4	29.2
Maple	3	1.2 (0.3)	1.7	45.0
Poplar	8	4.2 (0.3)	5.4	20.2
Willow	6	3.5 (0.6)	5.9	44.2

4.2.3. Calorific value of the wood

The calorific value of wood (CAL, kJ g^{-1} DM) indicates its energy content. It represents the amount of energy liberated when the wood is burned (WSUEE 2003). Four trees, with a mean diameter, per species were cut in December 2003 in order to determine the calorific value. Wood samples were taken all over the stem for this purpose. Samples of stem and branch wood were dried at 40 °C until constant weight, and subsequently ground on a 1 mm sieve. After drying again for 24 h, the calorific value of the samples was experimentally determined with an oxygen bomb calorimeter (model IKA C7000). The calorific value of stem wood and branch wood was determined separately, and a mean value for the whole tree was calculated based on the relative dry mass contribution of branches and stems (Van de Casteele 2004). The bark was not removed, since this is mostly no option in commercial SRF plantations.

4.2.4. Energy production

The energy stored in woody biomass can be transformed into 'usable' energy. Usable energy is energy in a form that is sold (electricity, heat, …). In policy documents, reference is made to this usable energy (EC 1997). Before transformation can take place, the wood has to be dried, to improve the conversion efficiency. Two drying procedures are used in practice. The first option is to transport the harvested wood immediately to a special location or building where the drying process takes place under controlled conditions. The second option is the on-site storage and drying of harvested stems. For the calculations in this study, the second option was chosen, as the energy input needed for this drying procedure is much lower than for off-site drying. However, drying under field conditions results in a small decrease of biomass

(4 % for willow, 13 % for poplar) (Nellist 1997, Sintzoff *et al.* 2001). For birch and maple, a biomass decrease of 2 % was assumed by field drying, as the moisture content of these two species (birch : 45.7 % and maple : 45.3 %) was lower than for willow (50.3 %) and poplar (54.2 %) (Vanlerberghe 2004).

The energy stored in the biomass, EN_{DM}, was calculated according to eq. 4.1 :

$$EN_{DM} = \alpha \cdot PROD_{pot} \cdot CAL \tag{4.1}$$

where EN_{DM} is energy stored in the biomass after 4 years of tree growth (GJ ha^{-1} $year^{-1}$), α is a correction factor for the loss of biomass due to drying (α equals 0.98 for birch and maple, 0.87 for poplar and 0.96 for willow), $PROD_{pot}$ is potential biomass production after 4 years of tree growth (t DM ha^{-1} $year^{-1}$) and CAL is calorific value of the wood (kJ g^{-1} DM).

In this study, three types of conversion processes were considered, which can be used to transform biomass energy EN_{DM} into usable energy : co-burning, burning and gasification. Conversion efficiencies (CE) of these three processes were given by García Cidad *et al.* (2003), and are presented in Table 4.2. Both efficiencies for converting biomass energy into electricity (CE_e) and heat (CE_{th}) are given.

Table 4.2. *Conversion efficiencies (GJ GJ^{-1}) of co-burning, burning and gasification processes applied to transform biomass energy into electricity (CE_e) or heat (CE_{th}) (García Cidad* et al. *2003)*

Conversion technique	**CE_e**	**CE_{th}**
Co-burning	0.37	0.50
Burning	0.16	0.69
Gasification	0.27	0.53

The electrical energy that can be produced from the biomass of a specific species, EN_e (GJ ha^{-1} $year^{-1}$), was calculated according to eq. 4.2, while the thermal energy, EN_{th} (GJ ha^{-1} $year^{-1}$), is given by eq. 4.3 :

$$EN_e = CE_e \cdot EN_{DM} \tag{4.2}$$

and $EN_{th} = CE_{th} \cdot EN_{DM}$ **(4.3)**.

4.2.5. Energy efficiency

When bioenergy production systems are compared, not only the biomass production, expressed as t DM ha^{-1} $year^{-1}$, but also the energy efficiency (EE) of the systems is an important characteristic. The energy efficiency EE expresses the number of energy units produced by the system per unit of energy input needed to drive the system (Dubuisson and Sintzoff 1998, Dalgaard *et al.* 2001, Matthews 2001, García Cidad *et al.* 2003) :

$$EE = \frac{EN_{output}}{EN_{input}} \tag{4.4}$$

where EE is energy efficiency of the energy production process (dimensionless), EN_{output} is usable energy produced (GJ ha^{-1} $year^{-1}$) and EN_{input} is input of energy needed to produce and transport biomass, and to convert biomass into usable energy (GJ ha^{-1} $year^{-1}$).

Energy efficiencies for fossil fuel based energy production systems are typically lower than 1 : 0.74 to 0.84 for petrol and 0.88 for diesel (García Cidad *et al.* 2003). The energy output, EN_{output}, can be equal to the amount of electricity produced (EN_e), the heat production (EN_{th}) or the total energy produced ($EN_e + EN_{th}$). Here, the EN_{output} was calculated for each species based on the mean potential biomass production results (Table 4.1).

The bioenergy production process can be subdivided in three stages where energy input is needed. This is reflected in eq. 4.5 :

$$EN_{input} = EN_{est} + EN_{trans} + EN_{conv} \tag{4.5}$$

where EN_{est} is energy input needed for the establishment of the SRF plantation, including the application of herbicides and fertilizers, and the harvest of the wood (GJ ha^{-1} $year^{-1}$), EN_{trans} is energy input needed to prepare the harvested wood and to transport it to the installation where the conversion will take place (GJ ha^{-1} $year^{-1}$), and EN_{conv} is energy input needed for the conversion process (GJ ha^{-1} $year^{-1}$).

The energy input needed during the establishment phase of the plantation, EN_{est}, can on its turn be subdivided in direct and indirect energy use. In Table 4.3, an overview is given of energy input values used in this study. Human labour and solar energy were ignored. The assumption was made here that the plantation lasts for a period of 20 years, and that a rotation length of 4 years is applied. As was the case for the plantation in Zwijnaarde, it was assumed that no fertilizers, lime or herbicides were applied.

To calculate EN_{trans}, an energy input of 0.8 MJ t^{-1} DM km^{-1} for the transport of the harvested wood to the conversion installation was assumed (Biewinga and van der Bijl 1996). The mean distance between a SRF plantation and a co-burning, burning or gasification installation in Flanders was estimated by García Cidad *et al.* (2003) at 25, 22 and 12.5 km, respectively.

Table 4.3. *Energy use during the establishment phase of a SRF plantation; information extracted from Sintzoff* et al. *(2001), García Cidad* et al. *(2003), Dalgaard* et al. *(2001) and Hülsbergen* et al. *(2001)*

Activity	Value	Unit
1/ Direct energy use		
Ploughing and preparation of the soil	37.0	l diesel ha^{-1}
Use of machines at harvest	14.0	l diesel ha^{-1}
Chipping of the wood	13.9	l diesel t^{-1} DM
Transport of machines to the field	0.1	l diesel km^{-1}
Loading of the biomass	2.9	l diesel t^{-1} DM
Energy use due to diesel consumption	40.9	MJ l^{-1} diesel
Energy use due to the use of lubricating oil	3.6	MJ l^{-1} diesel
2/ Indirect energy use		
Production of machines	12.0	MJ l^{-1} diesel
Production of plant material (cuttings or young trees)	300.0	MJ ha^{-1}

The energy input for the conversion process itself, EN_{conv}, was estimated at 7 % of the energy stored in the woody biomass, EN_{DM}, added to a co-burning system, and 5 % for a burning or gasification system (García Cidad *et al.* 2003).

4.2.6. CO_2 emission reduction potential

The burning of biomass from SRF plantations can be considered as a CO_2 neutral process (Proe *et al.* 2002), since the CO_2 liberated during the burning process will be sequestered during the next rotation period (IEA 2002). The production of energy from biomass, however, requires a certain input of fossil energy, emitting a corresponding quantity of carbon dioxide to the atmosphere. On the other hand, SRF biomass is only partly harvested, as the roots and stumps remain on the site as unutilized biomass. The carbon stored in these parts slowly decomposes, and becomes incorporated into the soil organic matter (SOM). It was assumed here, in accordance to Sáez *et al*. (1998), that the carbon input into the soil will be comparable to the amount of C released during cultivation and transport, so that the biomass fuel cycle may be considered to be carbon neutral, a statement supported by the studies of Matthews (2001) and Lettens *et al.* (2003).

When bioenergy is used as a substitute for energy produced from fossil fuels, carbon emissions to the atmosphere are reduced (Schwaiger and Schlamadinger 1998). The CO_2 emission reduction potential ER_c (kg CO_2 year^{-1}) of SRF plantations in Flanders was calculated as given by eq. 4.6 :

$$ER_c = EN_e \cdot TA \cdot EM_c \qquad \textbf{(4.6)}$$

where EN_e is amount of electricity that can be produced from biomass of a specific species (GJ ha^{-1} year^{-1}), TA is total area of plantations (ha) and EM_c is amount of CO_2 emitted during a traditional electricity production process (kg CO_2 GJ^{-1}).

As in Flanders there is actually only a green certificate system for electricity, and not for thermal energy, we focussed the calculations on the use of bioelectricity. Sintzoff *et al.* (2001) reported a CO_2 emission of 263.9 kg CO_2 per GJ electrical energy produced in the oldest electrical coal plant in Belgium, while the emission was estimated at 136.1 kg CO_2 GJ_e^{-1} produced in the most modern gas turbine. These values are comparable to the ones mentioned by Matthews and Robertson (2001). In the Spatial Structure Plan of the regional government of Flanders, 10000 ha are intended for the establishment of energy forests (ODE-Vlaanderen 2001, RSV 2004). This number of 10000 ha can be interpreted as the maximum area that will become available for the establishment of SRF plantations in Flanders. We determined the CO_2 emission that could be avoided by using the biomass grown on this maximal area of 10000 ha, based on the mean potential production results of the plantation at Zwijnaarde.

4.3. Results

4.3.1. Energy content of the biomass stock

The mean calorific value of the wood was lowest for maple (19.41 kJ g^{-1} DM), intermediate for poplar (19.63 kJ g^{-1} DM) and willow (19.92 kJ g^{-1} DM), and highest for birch (21.30 kJ g^{-1} DM). Although the mean potential biomass production of birch was lower than the production of poplar and willow (Table 4.1), the total amount of energy stored in the biomass of birch was higher than that of willow, and was almost equal to the energy stored in the poplar trees (Table 4.4). As could be expected on base of the low biomass production and the low calorific value, maple had the lowest amount of energy stored in the biomass.

Table 4.4. *Energy stored in the biomass and usable energy produced from the biomass of four species of the short-rotation plantation in Zwijnaarde after four growing seasons. Three types of conversion processes are considered. Max. refers to the willow plot with the highest potential biomass production. All values are expressed in GJ ha^{-1} $year^{-1}$.*

		Birch	**Maple**	**Poplar**	**Willow**	**Max.**
Biomass energy		69.7	23.2	70.9	67.3	111.9
Usable energy						
Co-burning	electricity	25.8	8.6	26.2	24.9	41.4
	heat	34.9	11.6	35.5	33.7	56.0
	total	*60.7*	*20.2*	*61.7*	*58.6*	*97.4*
Burning	electricity	11.2	3.7	11.3	10.8	17.9
	heat	48.1	16.0	48.9	46.4	77.2
	total	*59.3*	*19.7*	*60.3*	*57.2*	*95.1*
Gasification	electricity	18.8	6.3	19.2	18.2	30.2
	heat	37.0	12.3	37.6	35.7	59.3
	total	*55.8*	*18.6*	*56.7*	*53.9*	*89.5*

The amount of usable energy depends on the conversion technique applied (Table 4.4). If both electrical and thermal power can be used, values between 18.6 and 61.7 GJ ha^{-1} $year^{-1}$ were found. If only the electrical energy can be utilized, usable energy values ranged from 3.7 to 26.2 GJ_e ha^{-1} $year^{-1}$, less than half of the total usable energy. As there was a large variation in biomass production all over the plantation at Zwijnaarde (Table 4.1), the total amount of usable electricity that can be produced was also calculated based on the results of the plot with the highest amount of energy stored in the biomass. This maximum biomass energy stock was found on one of the willow plots, as can be seen in Table 4.1. Based on this best growing plot, the maximum amount of electricity that can be produced was 41.4 GJ_e ha^{-1} $year^{-1}$, while the total usable energy production was assessed at 97.4 GJ ha^{-1} $year^{-1}$ (Table 4.4).

4.3.2. Energy efficiency

The energy input and energy efficiency of the three conversion processes are listed in Table 4.5. For all species and all conversion systems together, the energy efficiency was at least 1.4, and went up to 8.4. The efficiency of a combined heat and power system was more than double the efficiency of a system where only electricity is produced. Efficiencies were highest for birch, compared to the other species.

4.3.3. CO_2 emission reduction potential

In Fig. 4.1, the CO_2 emission reduction potential of 10000 ha of SRF plantations of birch, maple, poplar or willow are presented. Emission reductions were highest when bioenergy production systems were compared to the oldest Belgian electricity plant. Depending on the conversion process chosen (co-burning, burning or gasification), using biomass instead of fossil-fuel based electricity could reduce CO_2 emissions with 9.8 to 69.3 kt CO_2 $year^{-1}$ (Fig. 4.1a). Reductions compared to a modern gas turbine ranged from 5.1 to 35.8 kt CO_2 $year^{-1}$ (Fig. 4.1b). Results were of course strongly dependent on the amount of energy stored in the biomass, which resulted from the combination of both the mean potential biomass production and the calorific value of the wood of a specific tree species.

Table 4.5. *Mean energy input (GJ ha^{-1} $year^{-1}$) for three types of conversion processes during three stages of the energy production process; mean energy efficiency (-), calculated as the amount of usable energy produced divided by the energy input; a rotation period of 4 years and a plantation length of 20 years were assumed.*

	Birch	**Maple**	**Poplar**	**Willow**
ENERGY INPUT (GJ ha^{-1} $year^{-1}$)				
1/ Establishment phase	3.436	1.461	3.759	3.536
2/ Transportation				
Co-burning	0.131	0.048	0.145	0.135
Burning	0.115	0.042	0.127	0.119
Gasification	0.065	0.024	0.072	0.068
3/ Conversion process				
Co-burning	4.882	1.626	4.966	4.712
Burning	3.487	1.161	3.547	3.366
Gasification	3.487	1.161	3.547	3.366
TOTAL INPUT				
Co-burning	8.449	3.134	8.869	8.383
Burning	7.039	2.664	7.433	7.021
Gasification	6.989	2.646	7.378	6.969
ENERGY EFFICIENCY (-)				
1/ Electricity production				
Co-burning	3.1	2.7	3.0	3.0
Burning	1.6	1.4	1.5	1.5
Gasification	2.7	2.4	2.6	2.6
2/ Heat production				
Co-burning	4.1	3.7	4.0	4.0
Burning	6.8	6.0	6.6	6.6
Gasification	5.3	4.7	5.1	5.1
3/ Combined heat and power				
Co-burning	7.2	6.4	7.0	7.0
Burning	8.4	7.4	8.1	8.1
Gasification	8.0	7.0	7.7	7.7

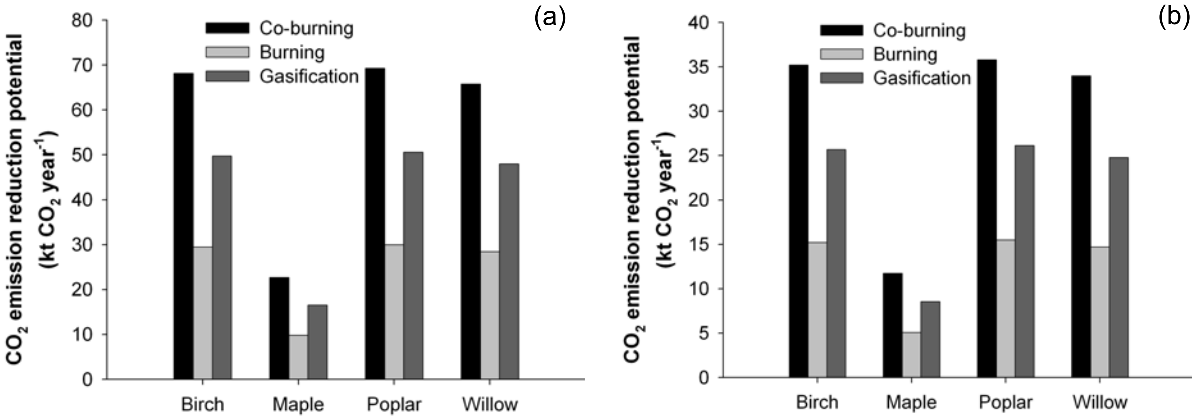

Figure 4.1. *CO_2 emission reduction potential (kt CO_2 $year^{-1}$) of 10000 ha of birch, maple, poplar or willow SRF plantations compared to (a) the oldest electrical coal plant of Belgium and (b) the most modern gas turbine*

4.4. Discussion

4.4.1. *Energy production capacity of SRF in Flanders*

Calorific values for different types of biomass reported in literature range from 13.5 to 24.0 kJ g^{-1} DM (Dawson and McCracken 1995, Goel and Behl 1996, Ledin 1996b, Labrecque *et al.* 1997, Senelwa and Sims 1999, Bungart and Hüttl 2001, Hülsbergen *et al.* 2001, Kataki and Konwer 2002, Klasnja *et al.* 2002, Faúndez 2003, Sheng and Azevedo 2005). Values determined for birch, maple, poplar and willow in Zwijnaarde fell within this range. As stated above, the maximum area that can be expected to become available for the establishment of energy forests in Flanders amounts to 10000 ha (ODE-Vlaanderen 2001, RSV 2004). The maximum amount of electrical energy that can be provided by this area ranges from 23.9 GWh_e $year^{-1}$ for maple to 72.9 GWh_e $year^{-1}$ for poplar, if co-burning is considered. Assuming a mean energy need per household of 3300 kWh_e $year^{-1}$, the woody biomass of these 10000 ha plantations can provide the electrical energy need of 7234 to 22095 households per year. In Flanders, there are approximately 2.2 million households (RSV 2004), which means that the number of households that can be provided by biomass electricity only accounts for 1.0 % of all households in Flanders. The total electrical energy production in Flanders amounted to 46233 GWh_e in 2002 (Vito 2005). The highest result (co-burning of poplar wood) based on mean potential biomass production values of the plantation at Zwijnaarde showed that only 0.16 % of this total electricity production could come from SRF biomass in Flanders. If the production result of the best (willow) plot of the plantation at Zwijnaarde was used as reference, the total number of households that could use biomass electricity was 34857 per year, and SRF biomass could provide about 0.25 % of the total electricity production in Flanders. As these last values are based on the result of one single plot, they are probably not as representative as the results based on the mean production values.

Although the total amount of energy produced is not that high, the efficiency of the bioenergy systems studied was always higher than 1, indicating that the systems can be considered as being energy efficient (García Cidad *et al.* 2003). As the definition of the system boundaries are not always equal, it is difficult to compare energy ratios found here with values mentioned in literature (Venendaal *et al.* 1997, Acaroglu and Semi Aksoy 2005). Lettens *et al.* (2003), however, also concluded that low-input bioenergy crops are highly energy efficient. The high efficiency of birch compared to the other species could be attributed to the high calorific value of this species. From Table 4.5, it can also be concluded that research should be focussed on the development of new combined heat and power systems, as these clearly have the highest energy efficiency.

The energy input related to the transportation phase was only minor compared to the input needed during the establishment phase and the conversion process itself (Table 4.5). However, if drying is done off-site, this transportation phase, which includes drying of the biomass, will require much more energy than was assumed here (Matthews 2001). As the contributions of the establishment phase and the conversion process to the overall energy input were more or less comparable, enhancement of the energy efficiency of the bioenergy production process can be achieved by reducing the energy needed during the set-up of the plantation, as well as by fine-tuning existing or searching for new conversion processes that need less energy input.

In Flanders, there is a high population density of 430 residents km^{-2} (RSV 2004). This results in large claims on available land for all types of land use. As such, it can be expected that it will not be possible to establish large-scale SRF plantations in Flanders. Therefore, the combination of smaller SRF fields with small-scale local gasification installations seems to be the most promising option for using SRF biomass as energy source in Flanders. These installations are often of the combined heat and power type, and are as such more efficient than larger burning or co-burning installations from which only the electricity produced can be used (Table 4.5).

Biomass production results of birch in Zwijnaarde were lower than for poplar and willow (Table 4.1). However, birch was planted at a much lower density (6667 trees ha^{-1}) than the two latter species (20000 trees ha^{-1}), and it can be expected that a rotation length of more than 4 years will result in higher mean annual production values for birch (see Chapter 3). Moreover, the calorific value of birch is higher than the one of poplar and willow, which makes the production of electricity from birch wood more energy efficient. Combined with a higher water stress tolerance, birch is therefore an interesting species for establishing SRF plantations in Flanders.

4.4.2. CO_2 emission reduction potential of SRF in Flanders

In 2000, the total CO_2 emissions in Flanders amounted to 76264 kt CO_2 (Vito 2005). The maximum CO_2 emission reduction potential found here amounted to 69.3 kt CO_2 year^{-1}, for 10000 ha of poplar plantation (Fig. 4.1), or only 0.09 % of the total annual CO_2 emissions. If calculations were based on the (willow) plot with the highest biomass production, the CO_2 emission reduction potential was 109.3 kt CO_2 year^{-1}, or still only 0.14 % of the total Flemish CO_2 emissions in 2000. From this, it can be concluded that the use of SRF energy will only be of minor significance in the view of reaching the Kyoto Protocol target for Flanders. The critical parameters for this conclusion are the low biomass production levels found at our plantation, and the land scarcity in Flanders, which inhibits the extension of the area that will become available for SRF plantations. It can be doubted that the establishment of SRF plantations can create a large number of new jobs in Flanders, as was reported for other regions (Sáez *et al.* 1998, Domac *et al.* 2005). However, biomass plantations can have other benefits than only the reduction of CO_2 emissions and employment creation. Small-scale, extensively managed SRF plantations can prevent soil erosion (Sáez *et al.* 1998) or improve the physical properties of the soil (Perttu 1998, Sage

1998); they can be established on marginal agricultural land or on contaminated sites unsuitable for food crops; they can have a recreational function or be a (temporary) habitat for many species, including birds (Hoffmann and Weih 2005, Londo *et al.* 2005), and thus serve as stepping stones or corridors between populations in protected areas (Faaij *et al.* 1998, Trinkaus 1998). Most probably, the highest benefits from SRF plantations in the densely populated and intensively managed region of Flanders will come from these additional characteristics of SRF plantations.

4.5. Conclusion

Although energy production based on SRF biomass has a high efficiency, the total amount of electrical and thermal SRF energy that could be produced in Flanders is low. The CO_2 emission reduction potential of SRF plantations in Flanders seems to be very restricted as well. Main causes of these two phenomena are the land scarcity in this region as a result of the high population density, and the low biomass production values found at the plantation studied here. The most interesting option seems to be the combination of a combined heat and power installation with a relatively small SRF plantation in the close neighbourhood. As such, both electrical and thermal energy can be used, and transport costs are kept to a minimum. Other possible functions of SRF plantations are the prevention of soil erosion and the protection of soil water; moreover, SRF plantations can serve as (temporary) habitats for many species. However, to reach these objectives, it is important that SRF plantations are established on formerly intensively used agricultural land, and that as less herbicides and fertilizers as possible (by preference : none of them) are applied. The tree species used should be indigenous, and suited to the site. Because of the high calorific value of birch wood and a higher water stress tolerance of this species compared to poplar and willow, birch could be a good choice for establishing SRF plantations on marginal agricultural soils in Flanders.

Chapter 5

Temporal variation and high resolution spatial heterogeneity in soil CO_2 efflux in a short-rotation tree plantation

Adapted from : Vande Walle, I., R. Samson, B. Looman, K. Verheyen and R. Lemeur. 2007. Temporal variation and high-resolution spatial heterogeneity in soil CO_2 efflux in a short-rotation tree plantation. Tree Physiol. 27:837-848.

Abstract

Short-rotation forestry plantations can become a means to fulfil the commitments under the Kyoto Protocol, by sequestering carbon (C) from the atmosphere or by preventing CO_2 emissions by fossil fuels burning. As a future increase of the total area of short-rotation forestry plantations can be expected, more knowledge is needed about carbon fluxes in this type of ecosystem, as to calculate regional carbon balances. Soil CO_2 efflux is the second largest carbon flux on earth. The magnitude of this flux determines for a large part the carbon sink or source strength of a forest ecosystem. Therefore, it is important to get insight in temporal and spatial variations in this carbon flux.

In this study, the driving factors of the seasonal change and the short-distance spatial variation in soil CO_2 effluxes in a short-rotation plantation of willow (*Salix viminalis* - Orm) were investigated. Total annual soil CO_2 effluxes ranged from 723 to 1149 g C m^{-2} $year^{-1}$, which was comparable to values mentioned in literature. Both an exponential and a logistic model were fitted to the available data, with soil temperature at 5 cm depth as independent variable. As R^2 values for individual sampling points ranged from 0.83 to 0.95 and from 0.85 to 0.93 for the exponential and logistic model, respectively, it appeared that soil temperature largely determined the seasonal variation in soil CO_2 efflux. Modelled soil CO_2 effluxes at 10 °C ranged from 1.22 to 1.95 µmol CO_2 m^{-2} s^{-1}, whereas annual Q_{10} values between 3.31 and 6.13 were derived from the models. These values for Q_{10} were high in comparison to those found in other studies, which was attributed to the absence of drought stress during the study period (2005).

The coefficients of the general soil CO_2 efflux models were replaced by linear dependencies on soil and vegetation-related characteristics. The spatially-explicit soil CO_2 efflux models explained 85 % and 86 % (exponential and logistic model, respectively) of the variability within the available data set. From this analysis, it appeared that soil carbon content, leaf area index, soil pH and root biomass caused differences in soil CO_2 effluxes at the short distances considered in this study. However, incorporating information on variables considered to account for spatial variability in the model did not result in a higher R^2 compared to a simple temperature function. When the general soil CO_2 efflux models were applied to independent data sets from the same plantation, it appeared that the logistic model fitted the data better than the exponential model when drought stress occurred. In that situation, annual Q_{10} values of soil CO_2 efflux were much lower than in the absence of drought stress.

5.1. Introduction

Young tree plantations may offer an opportunity to sequester carbon (C), *e.g.*, by increasing soil carbon stocks (Palmer 2003), or may prevent CO_2 emissions from burning fossil fuels when the biomass produced in these plantations is used as a substitute for fossil fuels. Bioenergy plantations are characterized by high standing densities, and the emphasis on these intensively managed ecosystems is placed on maximizing biomass production over a relatively short time scale. Establishing short-rotation forestry (SRF) plantations on former agricultural land fits in the European set-aside policy. Therefore, getting more insight in the carbon cycle of such ecosystems is important in order to establish regional carbon balances, which is needed in the view of the Kyoto Protocol obligations of European governments. Although many studies have investigated soil respiration processes in forest ecosystems, only a few have focussed on bioenergy tree plantations (*e.g.*, Scarascia-Mugnozza *et al.* 2005).

Forest ecosystems play an important role in the global carbon cycle, because of their large carbon stocks and the large amounts of carbon exchanged between forests and the atmosphere (Bolin and Sukumar 2000). The Net Ecosystem Exchange (NEE) of a forest is the balance between two large fluxes : CO_2 fixation by photosynthesis and release of carbon as CO_2 by ecosystem respiration (Matteucci *et al.* 2000, Högberg *et al.* 2001). This balance determines the carbon sink or source strength of a forest. Over two-thirds of the ecosystem respiratory flux might be attributed to soil CO_2 efflux (SR) (Epron *et al.* 1999a, Curiel Yuste *et al.* 2005). As such, soil CO_2 efflux is the second largest flux in the global C cycle, after gross primary productivity (GPP) (Conant *et al.* 2000, Pregitzer 2003). Soil CO_2 efflux is on its turn the result of belowground CO_2 production through the respiratory activities of plant roots, their symbiotic mycorrhizal fungi and the free-living microbial and faunal populations of the soil, and its transport from the soil to the atmosphere (Janssens *et al.* 2001a). Soil CO_2 efflux for a specific ecosystem can be characterized by its magnitude and its temporal and spatial variability. Spatial variation of soil CO_2 efflux beneath a forest canopy has been studied less often than temporal variation, although high spatial variability of soil CO_2 effluxes has been reported in some forest ecosystems (Fang *et al.* 1998, Rayment and Jarvis 2000, Scott-Denton *et al.* 2003, Perrin 2005). Detailed knowledge of the factors driving both temporal and spatial variability of soil CO_2 effluxes is necessary in order to understand ecosystem behaviour, develop process-based models and formulate appropriate experimental designs.

In literature, the rate of soil CO_2 efflux is mentioned to depend upon many parameters. Soil temperature and soil moisture content are often indicated as the primary drivers of temporal variability of soil CO_2 efflux (Kutsch and Kappen 1997, Hom 2003, Kane *et al.* 2003, Pregitzer 2003). Root biomass, biomass of mycorrhizae, the amount and availability of organic substrates, litter content or thickness, soil bulk density, nitrogen (N) availability, phosphorus (P) content and distance from surrounding trees are all mentioned to cause spatial differences in soil CO_2 effluxes (Fang *et al.* 1998, Matteucci *et al.* 2000, Stoyan *et al.* 2000, Xu and Qi 2001, Irvine and Law 2002, Pregitzer 2003, Perrin 2005, Tang *et al.* 2005).

Much effort has been spent during recent years in developing of soil CO_2 efflux models. In many models, soil CO_2 efflux is described to increase exponentially with increasing soil temperature (Buchmann 2000, Lee *et al.* 2002, Pregitzer 2003). Different types of models are used to describe the influence of soil moisture on soil CO_2 efflux (Janssens *et al.* 2001a, Lee *et al.* 2002, Raich *et al.* 2002, Perrin 2005). An extensive overview of frequently used soil CO_2 efflux models has been presented by Fang and Moncrieff (2001). Because soil CO_2 efflux can lead to as much as 63 % of the uncertainty of the model output regarding Net Ecosystem Exchange (Verbeeck *et al.* 2006), fine-tuning of soil CO_2 efflux submodels can help to optimize carbon cycling models.

Information on soil respiration processes in bioenergy plantations is scarce. Therefore, the aim of this research project was to study the temporal and spatial variability of soil CO_2 efflux in a short-rotation tree plantation at Zwijnaarde (Belgium), and to identify factors that control this temporal and spatial variability. For that purpose, soil temperature, soil moisture content, soil physical and chemical characteristics and vegetation-related characteristics were all determined at a small spatial scale. The obtained relationships were translated into both general and spatially-explicit soil CO_2 efflux models, which were validated by the use of independent data sets from the same plantation.

5.2. Material and methods

5.2.1. Short-rotation tree plantation

An extensive description of the short-rotation plantation at Zwijnaarde is given in § 1.6.1. For the study presented here, measurements were performed in three willow plots : W3, W15 and W19. Model validation data were available for plot W19 and for birch plot B8.

5.2.2. Measurement of soil CO_2 efflux

Soil CO_2 efflux, often called soil respiration or SR, was measured with a portable infrared gas analyser (EGM-1), connected to a soil respiration chamber (SRC-1) (PP systems 1994). This system is based on the closed dynamic system principle. The SRC-1 had an inner diameter of 10 cm. An additional PVC rim attached to the bottom of the chamber enlarged the soil surface under consideration to 277.6 cm^2 (Curiel Yuste *et al.* 2003). The bottom side of the PVC rim had a slot in which a rubber joint provided an airtight seal with the soil collars. For measuring soil CO_2 efflux, the chamber was placed on a 12 cm high PVC collar with a 18.8 cm internal diameter. Collars were inserted 6 cm in the soil 2 months prior to the onset of the measurements, and remained in place for the duration of the experiment. All live vegetation was removed from inside the collar at the time of insertion; emerging vegetation was clipped at the soil surface during the experiment. During measurements, a maximum CO_2 concentration increase of 50 ppm inside the chamber headspace was allowed, and a maximum measurement duration of 2 minutes. Preliminary measurements showed that during this measurement duration, rate of change in CO_2 concentration was constant. Coefficients of the relationship between the increasing CO_2 concentration and elapsed time were used to determine the soil CO_2 efflux rate (µmol CO_2 m^{-2} s^{-1}). Janssens *et al.* (2000) reported that the SRC-1 system tends to overestimate soil CO_2 effluxes. However, Giardina and Ryan (2002) found no significant differences between the SRC-1 and the LI-6400 system. Moreover, an extensive comparative study by Pumpanen *et al.* (2004), performed on a calibration tank, failed to demonstrate that the modified SRC-1 system (as we used) systematically over- or underestimated CO_2 effluxes.

Soil CO_2 efflux was measured fortnightly from January 2005 to January 2006 in three willow plots of the plantation at Zwijnaarde (plots W3, W15 and W19). In each plot, 8 collars were installed. Four of these collars were installed very close to each other (within 1 m^2), whereas the others were randomly spread over the plot. Because preliminary measurements revealed a distinct daily pattern in soil CO_2 efflux, measurements were performed between 9.00 and 11.00 a.m., as to get an idea of the mean daily soil CO_2 efflux rate (unpublished data). This procedure was in accordance to the methodology described by Xu and Qi (2001). Single soil CO_2 efflux measurements were made at each of the sampling points on each sampling date. In total, soil CO_2 efflux measurements were performed on 29 dates during 2005. As these measurements were spaced representatively in time, they reflected well the real seasonal trend of soil CO_2 effluxes. As such, interpolation was used to calculate the annual budget of the amount of carbon lost by soil respiration processes. Therefore, instantaneous soil CO_2 efflux rates were weighted according to half the number of days between the former and the next measurement in relation to the actual sampling date.

5.2.3. Soil temperature and soil moisture content

Concurrently with soil CO_2 efflux, soil temperature (ST) and volumetric soil moisture content (SMC) were measured. An STP-1 sensor was connected to the EGM-1 to measure soil temperature (°C) at a depth of 5 cm, next to the PVC collar. Volumetric soil moisture content (cm^3 cm^{-3}) at a depth of 6 cm was determined with a portable frequency domain reflectometry sensor (Theta Probe, model ML2-x, Delta-T Devices, Cambridge, UK) from April 2005 onwards. Three soil moisture measurements were made around each collar. A plot-specific calibration function was established for this sensor before the onset of the measurements. The weighted mean annual soil temperature and soil moisture content were calculated as described for soil CO_2 efflux.

5.2.4. Soil bulk density

Soil bulk density and porosity affect soil CO_2 efflux rates (Raich and Schlesinger 1992). We determined soil bulk density (g cm^{-3}) in 0-5 cm and 5-15 cm mineral soil layers using Kopecky rings to take samples of a known volume. Soil samples were taken within the collars after the measuring campaign, when litter lying on top of the soil in the collars was already removed. Soil samples were dried at 105 °C for 24 h, and bulk density was determined as dry weight (g soil) divided by volume (cm^3).

5.2.5. Chemical soil characteristics

In January 2006, soil samples were also taken within the collars to determine pH_{KCl}, organic carbon (C) content, total nitrogen (N) content and C:N ratio in three layers : 0-5 cm, 5-15 cm and 15-30 cm. The pH_{KCl} was determined with an Orion pH 15 E Meter (model 920A), in combination with an Orion pH-electrode Model Ross sure-flow 8172. Organic C was determined by the Walkley and Black method (Kalra and Maynard 1991) and N by the modified Kjeldahl method (Bremner 1996). C:N ratios were calculated as organic C content (%) divided by total N content (ppm) and multiplied by 10000. Bulk density values of the layer 5-15 cm were used for the lower layer too (15-30 cm), as to express soil C and N content on an area base.

5.2.6. Vegetation-related characteristics

On 17th August 2005, maximum leaf area index (LAI_{max}, m^2 m^{-2}) was determined above each sampling point as the mean of two measurements, made by holding a SunScan system (Delta-T Devices, Cambridge, UK) immediately above the collar in two perpendicular directions (N-S and E-W). In January 2006, distances (cm) between the centre of the collars and the four nearest trees were determined. Mean distances (cm) to the two, three and four nearest trees were calculated. Stem diameters at 30 cm height of these trees were also measured, and site-specific allometric relationships (see Table 3.1) were used to determine aboveground (AG) biomass of these trees (g AG dry mass tree^{-1} or g AGDM tree^{-1}). Total aboveground biomass within a radius of 1 m around the centre of each collar was calculated too (g DM m^{-2}). In January 2006, litter lying on top of the soil was gathered in each collar. After drying at 50 °C until constant weight, the amount of litter was expressed as g DM m^{-2}. Root biomass in the layers 0-15 cm and 15-30 cm was determined from soil samples taken in the collars in January 2006. Samples were stored in a freezer and analysed within the next month. Defrosted soil samples were spread on a sieve with a mesh width of 0.5 mm. After washing the samples gently with water, roots were removed manually from the sieve, and were divided in 4 diameter classes : < 1 mm, 1-2 mm, 2-5 mm, > 5 mm. After drying at 50 °C until constant weight, root dry weight was determined (g DM m^{-2}).

5.2.7. Soil CO_2 efflux model

Two model types were selected to describe the relationship between soil CO_2 efflux and soil temperature : a simple first-order exponential model (eq. 5.1), and a logistic function (eq. 5.2) :

$$SR = a_e \cdot e^{b_e \cdot ST} \tag{5.1}$$

$$SR = \frac{a_l}{1 + b_l \cdot e^{-k_l \cdot ST}} \tag{5.2}$$

where SR is soil CO_2 efflux (µmol CO_2 m^{-2} s^{-1}), ST is soil temperature at 5 cm depth (°C), a_e and b_e are coefficients of the exponential model and a_l, b_l and k_l are coefficients of the logistic model. The a_l coefficient represents the maximum soil CO_2 efflux, b_l determines the elongation along the x axis and k_l affects the steepness of the curve at its inflection point (Rodeghiero and Cescatti 2005). This logistic function is symmetric about the inflection point (coordinates $x = (\ln b_l) / k_l$, $y = a_l / 2$) and has an asymptote at $y = a_l$. Moreover, since this model is S-shaped, it can fit soil CO_2 efflux data affected by drought stress. In that case, soil CO_2 effluxes at high temperatures will decrease because of drought. Model coefficients were determined for each sampling point using the non-linear regression option of SPSS 12.0.

Q_{10} is the factor by which the respiration rate differs for a temperature interval of 10 °C, and can be defined as :

$$Q_{10} = \frac{SR_{ST+10}}{SR_{ST}} \tag{5.3}$$

where SR_{ST} and SR_{ST+10} are soil CO_2 efflux rates at soil temperature ST and ST + 10, respectively (Fang and Moncrieff 2001, Smith *et al.* 2003). For the first-order exponential model, Q_{10} values are constant over the whole temperature range under consideration, while the logistic model suggests a change in Q_{10} over this temperature range.

To visualize the spatial correlation of mean soil CO_2 efflux and of weighted mean soil temperature, semivariograms were constructed. For this purpose, the semivariance SVAR for each pair of collars (i, j), separated by a specific distance h(i, j), was calculated as :

$$SVAR(i,j) = \frac{(z_i - z_j)^2}{2} \qquad \textbf{(5.4)}$$

where z is the value of the parameter at location i and j. Graphing SVAR across all separation distances provides the semivariogram, which summarizes both the degree of autocorrelation present and the geographic range over which it is significant (Stoyan *et al.* 2000, Fortin *et al.* 2002, Omonode and Vyn 2006). For a spatially dependent variable, the semivariogram should theoretically increase asymptotically from the origin.

We followed the procedure described by Rodeghiero and Cescatti (2005) to establish spatially-explicit soil CO_2 efflux models. After model coefficients for eq. 5.1 and 5.2 were fitted separately for each sampling point, both models were applied to the whole data set (24 sampling points, 648 data points). At first, the original formulations as given by eq. 5.1 and 5.2 were used. As such, soil CO_2 efflux depended entirely on soil temperature. The coefficient of determination for the whole data set was 0.85 for both models. Afterwards, the coefficients of the models (a_e, b_e, a_l, b_l and k_l) were replaced by linear dependencies on characteristics other than soil temperature. As such, more insight in the driving factors of spatial differences in soil CO_2 efflux is obtained (Rodeghiero and Cescatti 2005). The total number of available soil and vegetation-related parameters was 79. To reduce the number of characteristics considered for the construction of a spatially-explicit soil CO_2 efflux model, correlation among and between these characteristics was first checked by Pearson or Spearman's rho correlation coefficients. Only one of strongly mutually correlated characteristics was taken into account for model construction. Following 25 characteristics were selected as possible predictors : maximum soil moisture content, interquartile range of soil moisture content, soil bulk density of the layers 0-5 cm and 5-15 cm, pH of the layer 0-30 cm, C and N content of the layer 0-30 cm (both on a mass and an area base), C:N ratio of the layers 0-5 cm, 5-15 cm and 15-30 cm, LAI_{max}, amount of litter, distance and biomass of the nearest tree, biomass within a circle with a radius of 1 m around the collar, and root biomass of the four diameter classes, for the layers 0-15 cm and 15-30 cm. A forward linear regression procedure

(SPSS 12.0) was used to determine the linear relationship that best described the coefficients a_e, b_e, a_l, b_l and k_l of the soil CO_2 efflux models as a function of the just mentioned characteristics, based on data from the 24 collars in plots W3, W15 and W19. After replacing the coefficients by the linear dependencies on soil and/or vegetation-related characteristics, the spatially-explicit models were calibrated using all available soil CO_2 efflux measurements ($n = 648$).

5.3. Results

5.3.1. Seasonal variability of soil CO_2 efflux

In Fig. 5.1, the evolution of soil CO_2 efflux (SR), soil temperature (ST) and volumetric soil moisture content (SMC) during the year 2005 is shown. On these graphs, mean values for all collars ($n = 24$) are presented. From Fig. 5.1, it can already be seen that the seasonal variation in soil CO_2 efflux was strongly correlated with the variation in soil temperature, with high values in summer, and low values in winter. The period with higher temperatures (July to September) corresponded with a period of relatively low soil moisture content.

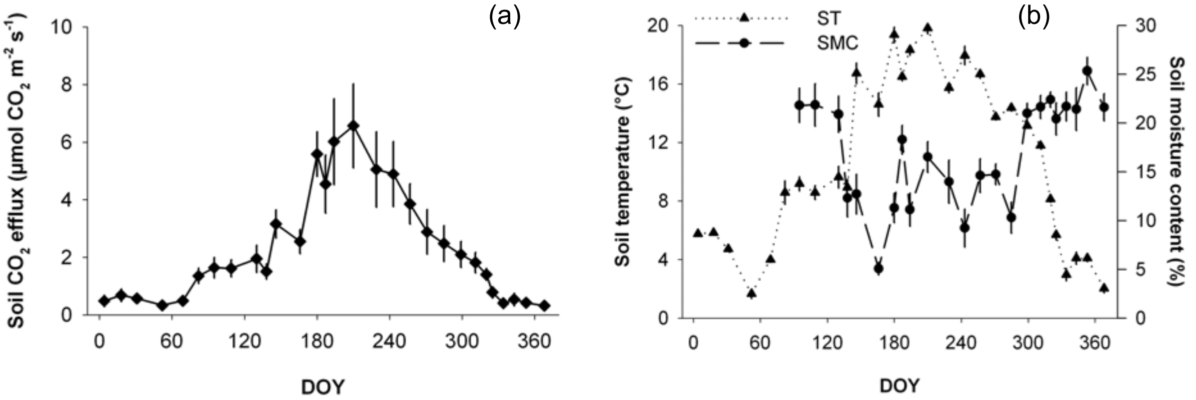

Figure 5.1. *Seasonal pattern of a) soil CO_2 efflux and b) soil temperature (ST) and volumetric soil moisture content (SMC); mean for all rings installed in willow plots W3, W15 and W19 ($n = 24$); error bars indicate one standard deviation; DOY : day of the year 2005.*

The annual evolution of the coefficient of variation (CV, calculated as standard deviation divided by the mean, and expressed in terms of percentage) for all sampling points together ($n = 24$) is given in Fig. 5.2 for soil CO_2 efflux, soil temperature and soil moisture content. For soil CO_2 efflux and soil temperature, highest CV values were found in winter. It appeared that the coefficient of variation was definitely higher for soil CO_2 efflux than for soil temperature and soil moisture content (Fig. 5.2). These results already suggested that spatial variation in neither soil temperature nor soil moisture content would be able to account for the observed spatial variability in soil CO_2 effluxes.

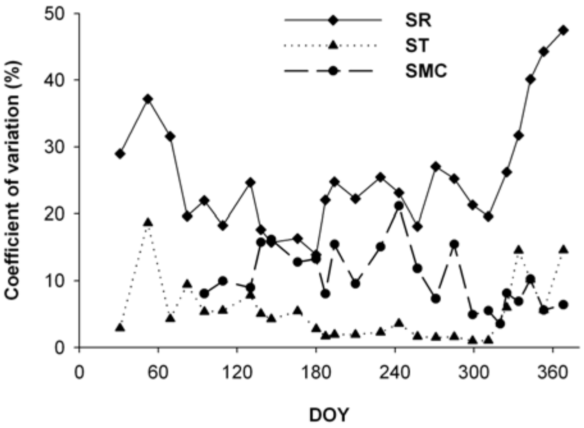

Figure 5.2. *Coefficients of variation for soil CO_2 efflux (SR), soil temperature (ST) and volumetric soil moisture content (SMC) (n = 24); DOY : day of the year 2005*

5.3.2. Total annual soil CO_2 efflux

Mean total soil CO_2 efflux of all individual sampling points calculated for the year 2005 was 896 g C m^{-2} $year^{-1}$ (n = 24), with a CV of 13.8 %. Minimum and maximum total soil CO_2 efflux amounted to 723 and 1149 g C m^{-2} $year^{-1}$, respectively. In comparison, the CV of the weighted mean soil temperature was only 1.6 %, whereas the CV of the weighted mean soil moisture content was 6.6 %.

In a first attempt to explain spatial differences in soil CO_2 effluxes, linear regressions between total annual soil CO_2 efflux and weighted mean soil temperature and soil moisture content were established (SPSS 12.0). However, the coefficients of these relationships were not significantly different from zero. Furthermore, no significant coefficients were found when the range in ST or the range in SMC (calculated as the maximum minus the minimum value of ST and SMC, respectively) was the independent variable in the linear regression.

5.3.3. Relationship between soil CO_2 efflux and soil temperature

Values of R^2 for the exponential model (eq. 5.1), calculated for all sampling points separately, ranged from 0.83 to 0.94, while for the logistic model (eq. 5.2), R^2 ranged from 0.85 to 0.93. This indicated the high correlation between soil temperature and soil CO_2 efflux. For some collars, non-linear regression analysis resulted in unrealisticly high values for the coefficients of the logistic model (*e.g.,* a_l-values from 624 to 110978500 µmol CO_2 m^{-2} s^{-1}). For these sampling points, nine in total, results of the logistic model were omitted from the data set. Fig. 5.3a illustrates the relationship between soil CO_2 efflux and soil temperature for measurements made in one of the collars in plot W19. The exponential and logistic models are indicated on the same graph. In general, residuals were larger for higher soil CO_2 effluxes. However, Fig. 5.3b shows that both models captured very well the seasonal variation as well as the magnitude of soil CO_2 effluxes measured in this sampling point. Graphs for other collars were similar. The overall exponential and logistic models are given by eq. 5.5 and 5.6, respectively :

$$SR = 0.3654 \cdot e^{(0.1451 \cdot ST)} \tag{5.5}$$

$$SR = \frac{28.5984}{1 + 90.5849 \cdot e^{(-0.1642 \cdot ST)}} \tag{5.6}$$

where SR is soil CO_2 efflux (µmol CO_2 m^{-2} s^{-1}) and ST is soil temperature at 5 cm depth (°C). For both models, R^2 was 0.85 and RMSE was 0.79.

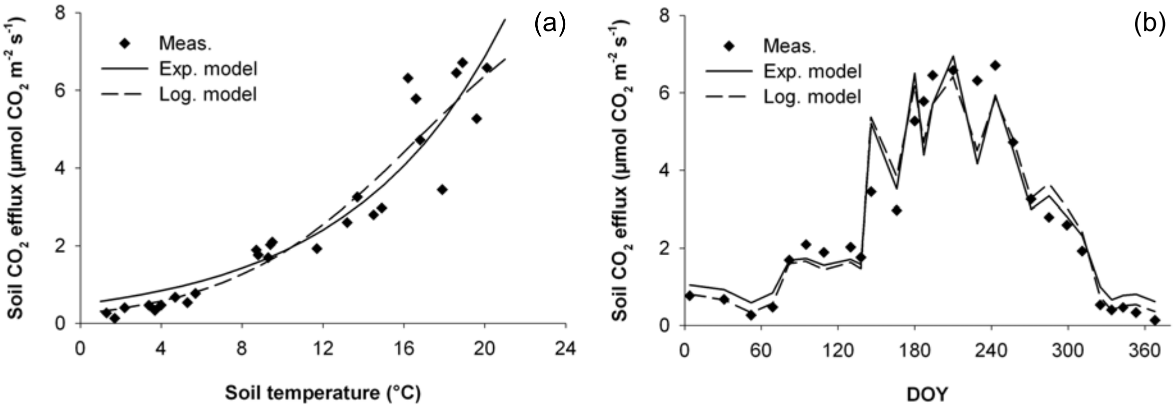

Figure 5.3. *Measured (Meas.) soil CO_2 efflux and soil CO_2 efflux modelled with the exponential (Exp. model, R^2 = 0.88) and logistic model (Log. model, R^2 = 0.90); results for a sampling point in plot W19 (n = 27); DOY : day of the year 2005*

After obtaining the model parameters, soil CO_2 efflux rate at a reference soil temperature of 10 °C was calculated; this temperature was close to the weighted mean soil temperature observed during 2005 (*i.e.*, 10.7 °C). Soil CO_2 effluxes at reference soil temperature ranged from 1.22 to 1.95 µmol CO_2 m^{-2} s^{-1}. Values of Q_{10}, calculated from modelled soil CO_2 efflux rates at a soil temperature of 10 °C and 20 °C (eq. 5.3), ranged between 3.47 and 6.14 for the exponential model, and between 3.31 and 4.97 for the logistic model.

5.3.4. Correlation among and between soil and vegetation-related characteristics

After testing normality of data, Pearson or Spearman's rho correlation coefficients were calculated (SPSS 12.0). These analyses revealed no obvious correlation between soil bulk density and pH on the one hand, and C content, N content or C:N ratio on the other hand. Carbon and nitrogen content were strongly and positively correlated within soil layers, even as C content and C:N ratio. Mean, minimum and maximum values for these parameters can be found in Table 5.1.

Table 5.1. *Overview of mean (and standard deviation), minimum (Min.) and maximum (Max.) values of soil and vegetation-related characteristics determined in three willow plots of the short-rotation plantation at Zwijnaarde; CV : coefficient of determination (in %, n = 24); LAI_{max} : maximum leaf area index; d : diameter*

Variable	Unit	Mean	CV	Min.	Max.
Soil characteristics					
Bulk density					
0-5 cm	g cm^{-3}	1.16 (0.11)	9.9	0.89	1.30
5-15 cm	g cm^{-3}	1.23 (0.06)	4.8	1.10	1.36
pH					
0-5 cm		5.07 (0.53)	10.4	4.01	5.62
5-15 cm		4.67 (0.50)	10.8	3.83	5.40
15-30 cm		4.76 (0.57)	12.0	3.83	5.47
Carbon content					
0-5 cm	%	1.59 (0.26)	16.1	1.03	2.10
5-15 cm	%	1.06 (0.14)	13.2	0.77	1.29
15-30 cm	%	0.97 (0.13)	13.3	0.79	1.27
Nitrogen content					
0-5 cm	ppm	1328 (155)	11.7	1009	1655
5-15 cm	ppm	1030 (101)	9.8	844	1217
15-30 cm	ppm	969 (85)	8.8	799	1113
C:N ratio					
0-5 cm		12.0 (1.2)	9.6	10.2	14.2
5-15 cm		10.3 (0.6)	5.9	8.8	11.2
15-30 cm		10.0 (0.7)	6.8	9.3	12.2
Vegetation-related characteristics					
LAI_{max}	m^2 m^{-2}	3.40 (0.54)	15.8	1.85	4.35
Distance to tree					
Nearest tree	cm	38 (12)	30.7	16	56
Second nearest tree	cm	46 (13)	27.8	26	79
Third nearest tree	cm	64 (9)	14.1	49	85
Fourth nearest tree	cm	72 (11)	15.0	56	100
Aboveground tree biomass					
Nearest tree	g $tree^{-1}$	716 (666)	93.1	10	2658
Second nearest tree	g $tree^{-1}$	742 (612)	82.5	119	2413
Third nearest tree	g $tree^{-1}$	557 (469)	84.3	118	1990
Fourth nearest tree	g $tree^{-1}$	642 (524)	81.7	127	2507
Litter	g DM m^{-2}	198 (74)	37.6	103	466
Root biomass					
0-15 cm					
d : < 1 mm	g DM m^{-2}	709 (336)	47.4	229	1487
d : 1-2 mm	g DM m^{-2}	32 (22)	70.7	4	69
d : 2-5 mm	g DM m^{-2}	57 (55)	96.6	0	199
d : > 5 mm	g DM m^{-2}	0 (0)		0	0
15-30 cm					
d : < 1 mm	g DM m^{-2}	114 (54)	47.3	27	257
d : 1-2 mm	g DM m^{-2}	15 (18)	126.6	0	70
d : 2-5 mm	g DM m^{-2}	46 (55)	119.6	0	227
d : > 5 mm	g DM m^{-2}	37 (79)	211.6	0	264

No significant correlation was found between different root diameter classes within one soil layer (0-15 cm or 15-30 cm). Moreover, there was no significant correlation between the biomass of roots of a specific diameter class present in the two soil layers. The only exception was a weak but significant correlation between the biomass of the finest roots (diameter < 1 mm) found in the soil layer from 0 to 30 cm, and the biomass of the roots with a diameter from 1 to 2 mm, for the same layer (r = 0.444). In Table 5.1, mean values of root biomass in both soil layers are given for the four diameter classes. The largest part of root biomass existed of very fine roots (diameter < 1 mm), which were most abundant in the upper soil layer, while roots with a diameter larger than 5 mm were found exclusively in the layer from 15 to 30 cm. Roots of intermediate diameters (1-2 mm and 2-5 mm) were almost equally present in both layers.

No obvious correlation could be found between LAI_{max} and distance to the surrounding trees, biomass of the surrounding trees, or amount of litter in the collars. This last variable was not correlated with the distance to or biomass of surrounding trees either. Mean, minimum and maximum values of these variables are given in Table 5.1. No significant correlation was found between root biomass and tree biomass, nor between root biomass and LAI_{max}. The amount of very fine roots (diameter < 1 mm) in the upper soil layer (0-15 cm) was positively correlated with the mean distance to the two nearest trees (r = 0.596), and negatively with the amount of litter (r = -0.667). For other root diameter classes and for the lower soil layer (15-30 cm), statistical analysis revealed no significant correlations.

Few significant correlations were found between soil and vegetation-related characteristics. Although one could expect correlations between LAI_{max}, amount of litter and root biomass on the one hand and soil C or N content on the other hand, none of these relationships were significant in this young tree plantation. The only positive correlation was found between pH of the upper soil layer and the aboveground tree biomass within a radius of 1 m (r = 0.739).

5.3.5. Spatially-explicit soil CO_2 efflux model

Semivariograms for all pairs of collars situated within one willow plot (distance between collars < 12 m) are presented in Fig. 5.4a and 5.4b for annual mean soil CO_2 efflux and weighted mean soil temperature, respectively. From these graphs, it can be concluded that these two variables were not spatially correlated, in contrast to the study of Rayment and Jarvis (2000), where there was a close correlation for distances less than 1 meter. Semivariograms for all available pairs of collars, *i.e.*, for distances up to 200 m, revealed no spatial correlation for mean soil CO_2 efflux and weighted mean soil temperature within the plantation.

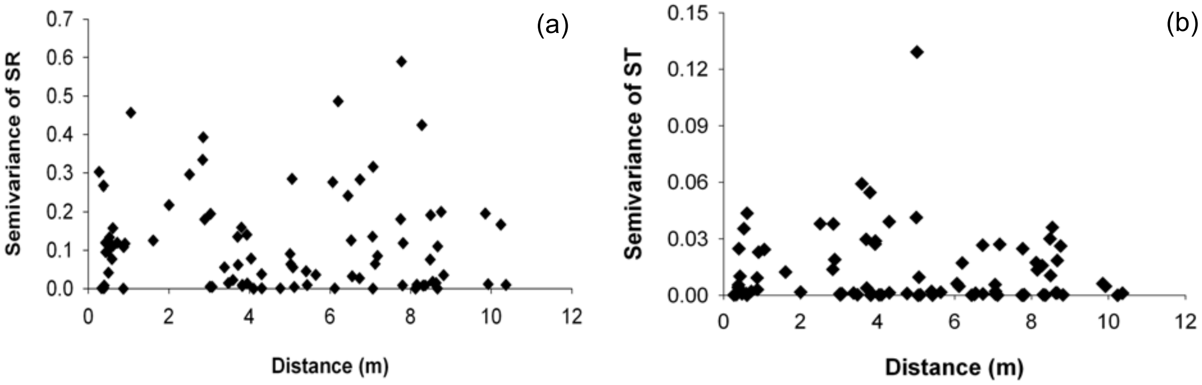

Figure 5.4. *Semivariograms for a) annual mean soil CO_2 efflux (SR) and b) weighted mean annual soil temperature (ST); measurements made in three willow plots (W3, W15 and W19) of the short-rotation plantation at Zwijnaarde during the year 2005*

Pearson or Spearman's rho correlation coefficients were used to investigate the relationship between total annual soil CO_2 efflux and soil and vegetation-related characteristics. As illustrated in Fig. 5.5, a significant positive correlation was found between total annual soil CO_2 efflux and root biomass (diameter < 2 mm), present in the upper soil layer (0-15 cm). Total annual soil CO_2 efflux was not correlated with soil characteristics as C or N content.

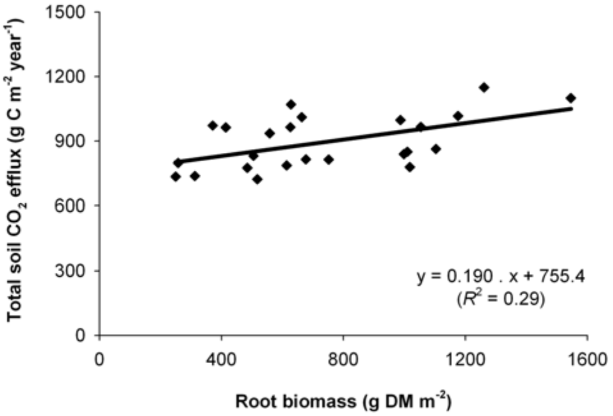

Figure 5.5. *Relationship between total annual soil CO_2 efflux (SR_{total}, g C m^{-2} year^{-1}) in the year 2005 and biomass of roots present in the upper soil layer (0-15 cm) and with a diameter < 2 mm (r_{a02}, g DM m^{-2}); linear regression is given by SR_{total} = 0.190 . r_{a02} + 755.4 (R^2 = 0.29); data from three willow plots (W3, W15 and W19) of the short-rotation plantation at Zwijnaarde.*

After replacing the model coefficients a_e, b_e, a_l, b_l and k_l by linear dependencies on soil and/or vegetation-related characteristics and calibrating the models with all available data (n = 648), following spatially-explicit soil CO_2 efflux models were obtained :

$$SR = 0.356 \cdot e^{(0.172-0.023 \cdot C_{0-30}) \cdot ST} \tag{5.7}$$

$$SR = \frac{46.50 - 6.444 \cdot LAI_{max}}{1 + (170.2 - 26.45 \cdot LAI_{max}) \cdot e^{-(0.1565 + 0.00063 \cdot pH_{0-30} + 0.00026 \cdot r_{a1\text{-}2}) \cdot ST}} \qquad \textbf{(5.8)}$$

where SR is soil CO_2 efflux (µmol CO_2 m^{-2} s^{-1}), ST is soil temperature at 5 cm depth (°C), $C_{0\text{-}30}$ is the mean soil carbon content in the layer 0-30 cm, expressed in %, LAI_{max} is maximum leaf area index (m^2 m^{-2}), $pH_{0\text{-}30}$ is the mean pH of the soil layer 0-30 cm and $r_{a1\text{-}2}$ is the biomass of roots present in the upper soil layer (0-15 cm) and with a diameter between 1 and 2 mm (g DM m^{-2}). As can be seen from eq. 5.7, no clear relationship emerged between a_e and soil or vegetation-related characteristics.

Values of R^2 were 0.85 for the spatially-explicit exponential model and 0.86 for the spatially-explicit logistic model, and RMSE was 0.77 and 0.75, respectively. For both general and spatially-explicit models, higher (absolute) values of residuals were observed for higher soil temperatures. The slopes of the linear regressions between residuals and soil temperatures were not significantly different from zero, indicating no significant under- or overestimation of soil CO_2 effluxes with increasing temperature for the four models.

5.4. Discussion

5.4.1. Temporal variation of soil CO_2 efflux

Inserting collars certainly disturbs the soil environment. When the soil within the collars is not recolonized by roots, total soil CO_2 effluxes are underestimated. At Zwijnaarde, soil collars were installed two months before the measurements began. As can be seen from Table 5.1, most (fine) roots were found in the upper soil layer. At the time root biomass was determined, we observed that most roots were alive. Mean root biomass (diameter < 5 mm) found in the collars in plot W19 amounted to 803 (± 359) and 200 (± 93) g DM m^{-2} in the layer from 0 to 15 cm and from 15 to 30 cm, respectively (n = 8; samples taken in January). Devos (2006) determined root biomass in the same plot, on places without soil collars (samples taken in October). Using the soil core method (n = 9), he reported root biomasses of 1103 (± 275) and 227 (± 91) g DM m^{-2} for the upper and the lower soil layer, respectively. He found a lower root biomass by applying the soil pit method : 561 (± 133) and 188 (± 23) g DM m^{-2} for the upper and lower soil layer, respectively. This supports our statement that roots recolonized soil collars well at our plantation, probably because the study stand comprised fast-growing young willow trees, planted at a high density.

The observed annual pattern of soil CO_2 effluxes in the plantation at Zwijnaarde, with high SR in summer and autumn-winter minima, corresponded well to other studies (Davidson *et al.* 1998, Rey *et al.* 2002, Curiel Yuste *et al.* 2003, Scott-Denton *et al.* 2003, Perrin 2005). The highest soil CO_2 efflux measured at Zwijnaarde amounted to 10.0 µmol CO_2 m^{-2} s^{-1}, but only 5 % of all measured soil CO_2 effluxes were higher than 6.3 µmol CO_2 m^{-2} s^{-1}. Soil CO_2 efflux was less than 3.5 µmol CO_2 m^{-2} s^{-1} for

75 % of all measurements. This corresponded well to the range given by Matteucci *et al.* (2000), who stated that in general, soil CO_2 effluxes in temperate forests vary between 0.6 and 14.8 µmol CO_2 m^{-2} s^{-1}, with the majority of sites ranging between 0.6 and 3.2 µmol CO_2 m^{-2} s^{-1}. Soil CO_2 efflux values observed at Zwijnaarde were also well within the range of values reported for temperate forests by Epron *et al.* (1999a), Janssens *et al.* (2001a), Irvine and Law (2002), Perrin (2005) and Rodeghiero and Cescatti (2005). As in our study, these authors all used closed dynamic systems to measure soil CO_2 efflux.

Seasonal variability in soil CO_2 effluxes was largely accounted for by variability in soil temperature at a depth of 5 cm in the short-rotation plantation at Zwijnaarde. Similarly, Janssens *et al.* (2001b) found that temperature was the dominant factor controlling the seasonal variation in soil CO_2 effluxes in 16 forests of the Euroflux network. Many other authors also noted a distinctly coherent seasonal variation of soil CO_2 efflux and soil temperature (Matteucci *et al.* 2000, Raich *et al.* 2002).

The high R^2 values of both soil CO_2 efflux models suggested that soil moisture was not limiting soil CO_2 effluxes in the three studied willow plots during the year 2005, as illustrated in Fig. 5.6. From this graph, it is clear that no situation of excess water occurred in the willow plots, as both soil temperature and soil CO_2 efflux decreased with increasing soil moisture content, indicating the autumn and winter period (Fig. 5.1). At the other end of the soil moisture content scale, patterns of soil CO_2 efflux and soil temperature diverged, indicating drought stress. However, these measurements represented less than 5 % of all measurements made during the year 2005. Other authors have noted a comparable divergence between soil CO_2 efflux and soil temperature under dry soil conditions (Epron *et al.* 1999a, Xu and Qi 2001, Scott-Denton *et al.* 2003). In other studies, soil temperature also explained more than 80 % of the temporal variation of soil CO_2 efflux, when water stress was negligible (Maljanen *et al.* 2001, Borken *et al.* 2002).

Soil CO_2 efflux and soil temperature at the inflection point of the logistic function ranged from 3.9 to 16.4 µmol CO_2 m^{-2} s^{-1} and from 16.6 to 28.1 °C, respectively. Only 20 % of all SR measurements were higher than 4.1 µmol CO_2 m^{-2} s^{-1}, whereas soil temperature was higher than 16.6 °C in only 20 % of all measurements, and the maximum soil temperature measured was 20.6 °C. This means that most measurements were made in the linear part of the S-shaped logistic function, and as such, exponential and logistic functions described the relationship between soil temperature and soil CO_2 efflux equally well. The absence of drought stress can also have resulted in the impossibility to find a reasonable asymptotic soil CO_2 efflux value for some of the sampling points, as measurements in the field did not level off at higher temperatures, because soil moisture content was not limiting soil CO_2 efflux.

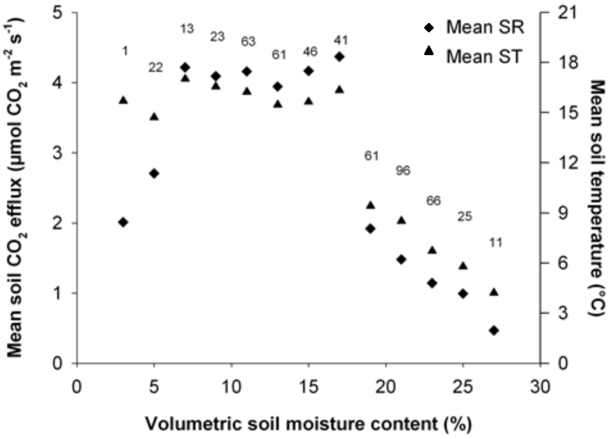

Figure 5.6. *Mean soil CO_2 efflux (SR) and mean soil temperature (ST) in function of volumetric soil moisture content classes of 2 %; values above the symbols indicate the number of measurements within each soil moisture content class; measurements made in three willow plots of the short-rotation plantation at Zwijnaarde during the year 2005.*

One might suppose that interpolation of soil CO_2 efflux measurements made during a limited time period of the day (9.00 to 11.00 a.m. in our case) can lead to an over- or underestimation of total annual soil CO_2 effluxes. Hourly soil temperature data (5 cm depth) were available for 2003 and 2004 for the plantation. Applying the collar-specific logistic model for 18 collars in birch plot B8 (see further) gave a mean total annual soil CO_2 efflux (August 2003 to July 2004) of 658.9 (± 36.0) g C m^{-2} $year^{-1}$, which was higher although not significantly different from the mean value of 607.6 (± 32.0) g C m^{-2} $year^{-1}$ obtained by the interpolation procedure. Mean total annual soil CO_2 efflux in 2004 according to the modelling approach amounted to 676.6 (± 27.5) g C m^{-2} $year^{-1}$ for the three rings in plot W19 (see further), which was not significantly different from the mean value of the interpolated data, amounting to 718.7 (± 48.2) g C m^{-2} $year^{-1}$. Therefore, we concluded that the interpolation method gave a reliable estimate of total annual soil CO_2 effluxes for our study site. Moreover, we could not apply the modelling approach for 2005, as no soil temperature data were available. Measurements after the first four years of tree growth indicated a decrease of soil carbon stocks in the upper 30 cm of 42 % (mean for the plantation). Soil carbon stock changes calculated on base of soil CO_2 efflux measurements made all over the plantation, assessed even larger carbon losses from the soil (see Chapter 6). This indicated that the methodology applied here was not underestimating total annual soil CO_2 effluxes.

Total annual soil CO_2 efflux values determined for individual collars at Zwijnaarde in the year 2005, ranging from 723 to 1149 g C m^{-2} $year^{-1}$, fell within the range from 304 to 1414 g C m^{-2} $year^{-1}$ mentioned in literature for temperate forests (Raich and Schlesinger 1992, Buchmann 2000, Janssens *et al.* 2001b). Curiel Yuste *et al.* (2005) measured total annual soil CO_2 effluxes from 410 g C m^{-2} $year^{-1}$ in Scots pine plots to 1360 g C m^{-2} $year^{-1}$ in pedunculate oak plots of a mixed forest at Brasschaat, while Longdoz *et al.* (2000) determined soil CO_2 efflux rates of 438 g C m^{-2} $year^{-1}$ and 870 g C m^{-2} $year^{-1}$ in a Douglas fir and a beech stand at Vielsalm, respectively. Although the plantation at Zwijnaarde was much younger and had a much higher planting density than most forests considered in other studies, the amount of carbon released annually from the soil was comparable to the amount measured in these older ecosystems. Values of annual soil CO_2 effluxes measured in an 11 years old loblolly pine plantation (1263 to 1576 g C m^{-2} $year^{-1}$, Maier and Kress 2000) exceeded our values. However, the climate in that region was much warmer (mean annual temperature of 17 °C) and wetter (mean annual rainfall of 1210 mm) than at Zwijnaarde, which can be expected to stimulate soil respiration processes.

Raich and Schlesinger (1992) found a relationship between total annual soil CO_2 efflux and mean annual air temperature in temperate deciduous forests, on a global scale. In contrast, Janssens *et al.* (2001b) found no significant relationship between total annual soil CO_2 efflux and mean annual soil temperature in temperate forests, at a smaller scale. Following their conclusion, the very small range of weighted mean temperatures (only 0.7 °C between the different collars) can explain why no relationship between total annual soil CO_2 efflux and weighted mean soil temperature was found at Zwijnaarde.

By definition, determination of Q_{10} values is based on measurements at constant conditions, with only temperature varying. This precondition is rarely, if ever, true for field data, especially when measurements are performed over longer time periods, as in our study. Q_{10} values calculated here are therefore no 'real' Q_{10} values *s.s.*, but should be considered as 'annual' Q_{10} values, which means that they are likely influenced by phenological changes occurring throughout the year. Despite the violation of the basic assumptions, Q_{10} values are often applied in soil CO_2 efflux studies. A whole range of factors influence calculated (annual) Q_{10} values : soil depth at which soil temperature was measured (Irvine and Law 2002), soil CO_2 efflux model type (Fang and Moncrieff 2001), length of the measuring campaign (whole year or shorter) (Kutsch and Kappen 1997) and soil moisture content (Xu and Qi 2001, Curiel Yuste *et al.* 2003). Annual Q_{10} values of both the exponential and the logistic model found in this study were higher than values reported by Fang *et al.* (1998), Buchmann (2000) and Matteucci *et al.* (2000), who all measured soil temperature at 5 cm depth, as was done here. Their research was situated in forest ecosystems that were considerably older than the short-rotation plantation at Zwijnaarde. However, it should also be mentioned that the annual Q_{10} value determined by the logistic model was only 2.7 for willow plot W19 in 2004, and 2.2 for birch plot B8 in the period August 2003 to July 2004 (see further). These lower Q_{10} values are probably reflecting the interannual variability in climatic conditions. Mean monthly air temperature and total monthly precipitation for 2003, 2004 and 2005 were calculated from measurements of the Royal Meteorological Institute of Belgium (RMI 2003, 2004, 2005) in the measuring station at Kruishoutem, situated 20 km from

Zwijnaarde. Mean monthly air temperatures for July and August were higher in 2003 (19.4 °C and 20.3 °C) than in 2004 (17.6 °C and 19.6 °C) and 2005 (18.6 °C and 17.1 °C), while precipitation in July was clearly lower in 2003 (60.7 mm) and 2004 (65.1 mm) than in 2005 (129.6 mm). The lower annual Q_{10} values observed in 2003 and 2004 probably resulted from drought stress occurring during these two summers, while the higher annual Q_{10} value for 2005 suggests the absence of drought stress. In this latter case, soil CO_2 efflux is much more sensitive to soil temperature than if soil moisture is limiting soil respiration processes (Xu and Qi 2001).

5.4.2. Spatial heterogeneity of soil CO_2 efflux

Several studies revealed a high small-scale variability of soil CO_2 effluxes (Matteucci *et al.* 2000), even at the centimeter scale (Stoyan *et al.* 2000). Spatial variability of soil CO_2 effluxes, expressed as coefficient of variation, ranged mainly from 20 to 30 % in the plantation at Zwijnaarde (Fig. 5.2). Whereas mean air temperature and precipitation cause differences in soil CO_2 efflux rates between biomes (Raich *et al.* 2002), other factors such as root biomass (Fang *et al.* 1998), plant productivity (Raich and Schlesinger 1992), soil pH, N and P content or C:N ratio (Xu and Qi 2001, Borken *et al.* 2002) are reported to determine spatial variation in soil CO_2 effluxes at shorter distances.

The positive correlation between total annual soil CO_2 efflux and root biomass, and the lack of correlation between total annual soil CO_2 efflux and soil C or N content, led us to assume that root respiration differed more from collar to collar than soil microbial respiration (*i.e.*, heterotrophic soil respiration). As can be seen in Table 5.1, the coefficient of variation was much larger for root biomass than for soil C and N content. This could be related to the young age of the ecosystem we studied, where root distribution could still be highly variable and the former agricultural land use can have caused uniform conditions in the upper soil layer. The five years of tree growth will not have been long enough to change the soil carbon content considerably yet. However, it can be expected that on the longer term, the turnover of root biomass into soil carbon will cause a larger spatial variation in this soil carbon content.

The lack of correlation between soil and vegetation-related characteristics was probably also associated with the young age of the plantation at Zwijnaarde. Previously, no correlation could be found between biomass production within the 21 plots and soil characteristics of this plantation either (see Chapter 3). Most other studies on soil CO_2 efflux considered older forest ecosystems, which means that vegetation already had more time to influence physical, chemical as well as biological characteristics of the soil (Maljanen *et al.* 2001). It can be expected that after several years of tree growth, stronger correlations between vegetation-related characteristics as root biomass, amount of litter or LAI and soil characteristics will be found. Moreover, we studied a bioenergy plantation, with a very high planting density, in contrast to most other studies. Possibly, this high planting density caused a more homogeneous situation than in forests with lower planting densities, as suggested by Xu and Qi (2001). However, although CV of aboveground tree biomass and of root biomass was much larger than the CV of soil CO_2 efflux in our study (Table 5.1), no obvious relationship could be found between these productivity variables and soil CO_2 effluxes. Other soil or vegetation-related characteristics than the ones included in our study should be the main causes of short-distance spatial variation in soil CO_2 effluxes in the plantation at Zwijnaarde. Phosphorus content of the soil can be one of these characteristics, as soil CO_2 efflux rates in European forests appear to be more P than N limited (Borken *et al.* 2002). The study of Vanlerberghe (2004), performed on the plantation at Zwijnaarde, showed that nitrogen availability was optimal for birch and willow and suboptimal for poplar, while the phosphorus content appeared to be suboptimal for birch and willow, and insufficient for poplar.

5.4.3. Validation of the soil CO_2 efflux model

The two general soil CO_2 efflux models established for the willow plots W3, W15 and W19 were used to model the soil CO_2 efflux measured in willow plot W19 during 2004 (n = 111). During that year, soil CO_2 efflux was measured fortnightly on three sampling points in plot W19, according the methodology described in § 5.2. The three sampling points differed from the points where measurements were performed in 2005. R^2 values, illustrating the relationship between modelled and measured soil CO_2 effluxes, were 0.67 and 0.70 for the exponential and the logistic function, respectively. The spatially-explicit exponential model (eq. 5.7) was applied too, and gave an R^2 of 0.67. The spatially-explicit logistic model (eq. 5.8) could not be used, as no information on LAI was available. RMSE was 1.22, 1.17 and 1.27 for the general exponential, the general logistic and the spatially-explicit model, respectively. All three original models overestimated soil CO_2 effluxes measured in plot W19 during 2004. After the models were parameterized for plot W19, R^2 values increased to 0.73, 0.81 and 0.73 and RMSE decreased to 0.63, 0.53 and 0.63 for the general exponential, general logistic and the spatially-explicit model, respectively. The higher R^2 values obtained after parameterization reflected the fact that the established relationship was not transferable from one year to another because of changing biotic and abiotic conditions. If one wants to establish a model that can be used on the longer term, the considered data set has to cover a relative long time series reflecting a variety of biotic and abiotic conditions.

Soil CO_2 efflux measurements were also performed on 18 sampling points in birch plot B8, from August 2003 to August 2004 (n = 433). After soil CO_2 efflux models were parameterized, R^2 values were only 0.56, 0.65 and 0.57 in this plot for the general exponential, the general logistic and the spatially-explicit model, respectively, while RMSE values were 0.96, 0.86 and 0.96, respectively. Model fits were clearly lower for the birch plot than for the willow plots. This birch plot is quite different from the willow plots, as planting density was much lower and as such, canopy structure was different too. As was the case for the three willow plots, no significant relationships between soil C or N content and root biomass were observed for the birch plot. Moreover, relationships between biomass of different root diameter classes within one soil layer or between the root biomass of a specific diameter class within the two soil layers were not significant. In contrast to the willow plots, aboveground biomass within a radius of 1 m around the soil collars was significantly related to the amount of litter in the collars, and to the amount of the finest roots (diameter < 1 mm) within the upper soil layer (0-15 cm). This can be the consequence of the lower planting density (6667 trees ha^{-1}) of birch compared to willow (20000 trees ha^{-1}). Larger distances between trees in the birch plot also induced larger variation in litter (CV of 60.1 %) and in aboveground biomass within 1 m (CV of 77.1 %) than in the willow plots (CV of 37.6 and 53.1 %, respectively).

For both willow plot W19 and birch plot B8, the logistic model fitted better to the measured data than the exponential model. This is illustrated in Fig. 5.7 for plot W19. As mentioned before, summer periods of 2003 and 2004 were drier and warmer than the summer period of 2005. Total precipitation from 1st July to 31st August amounted to 105.8 mm, 176.5 mm and 240.7 mm in 2003, 2004 and 2005, respectively. Mean air temperatures for this period were 19.9 °C, 18.6 °C and 17.8 °C for 2003, 2004 and 2005, respectively. The higher R^2 values for the logistic model suggested that drought stress occurred in 2004 in plot W19 and in the period from August 2003 to July 2004 in plot B8, following the statement of Rodeghiero and Cescatti (2005) that the logistic model is more powerful under conditions of limited soil moisture content than the exponential model. The lack of drought stress in 2005 was also reflected by the high correlation between soil temperature and soil CO_2 efflux observed for that year (R^2 = 0.85). In the study of Rodeghiero and Cescatti (2005), R^2 of the logistic model was higher than 0.80, except for a site where drought stress occurred (R^2 = 0.62). These authors mentioned that drought stress leads to a higher scatter of soil CO_2 efflux rates than when no drought stress occurs, leading to a somewhat lower R^2 value.

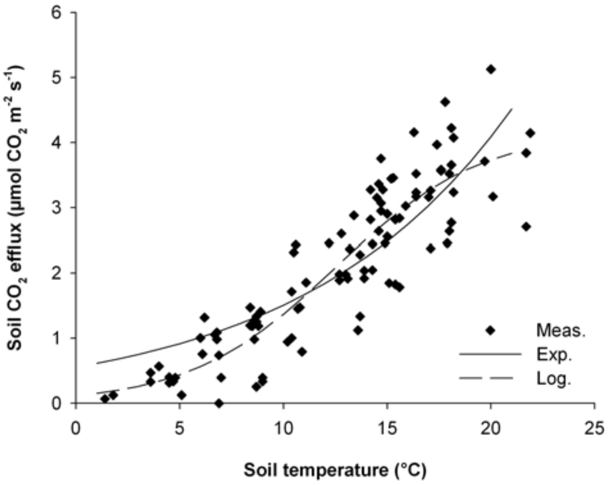

Figure 5.7. *Measured soil CO_2 effluxes (Meas.) for willow plot W19 during 2004; plot-specific exponential (Exp., R^2 = 0.73) and logistic (Log., R^2 = 0.81) soil CO_2 efflux models, parameterized for 2004, are also given.*

5.5. Conclusion

As was reported for many other forest ecosystems, temporal variations in soil CO_2 efflux observed in the young tree plantation at Zwijnaarde were strongly correlated with the seasonal variation in soil temperature. As no pronounced drought stress occurred during the summer of 2005, an exponential soil CO_2 efflux model fitted the data as well as a logistic model did for that year. The high annual Q_{10} values for soil CO_2 efflux determined for the three willow plots during 2005 confirmed this lack of drought stress. Under conditions of drought stress, annual Q_{10} values were much lower, and a logistic model appeared to be a good tool to capture the impact of drought stress on soil CO_2 effluxes. Total annual soil CO_2 efflux was comparable in this young tree plantation and in older temperate forest ecosystems.

A whole range of soil physical and chemical characteristics and vegetation-related parameters were considered in this study. These parameters were known to cause spatial differences in soil CO_2 efflux between ecosystems or biomes, or at a global scale. However, none of these characteristics was an obvious driver of spatial heterogeneity of soil CO_2 effluxes measured in the short-rotation plantation at Zwijnaarde, indicating that soil or vegetation-related characteristics other than the ones involved in this study determine the spatial variation of soil CO_2 effluxes in this ecosystem. Sampling points in this study were situated within a distance of 200 m, whereas sampling distances in most other studies are much greater. A slight positive correlation was found between total annual soil CO_2 efflux and root biomass. The lack of correlation between soil and vegetation-related characteristics observed at this short-rotation plantation may be associated with the very high planting density or the young age of the stand. Its former agricultural use can have caused relatively homogeneous soil characteristics at the moment of planting. The absence of spatial correlation of soil CO_2 effluxes in this study indicated that the choice of sampling sites is of minor importance to obtain reliable estimates of soil CO_2 effluxes and their spatial variation in this young, densely planted, short-rotation plantation. The choice of the sampling sites might have a larger impact on measured soil CO_2 effluxes in better developed plantations, or in mature forest ecosystems.

Chapter 6

Carbon balance of a short-rotation tree plantation during the first five years of tree growth

Adapted from : Vande Walle, I., R. Samson, N. Van Camp, K. Verheyen and R. Lemeur. Carbon balance of a short-rotation tree plantation during the first five years of tree growth. *Submitted to Tree Physiology.*

Abstract

Actually, more and more agricultural land is being planted with short-rotation forestry (SRF) plantations. This afforestation can offer an opportunity to sequester carbon (C) from the atmosphere, by increasing carbon stocks in vegetation and soil. In this study, the carbon balance, or net ecosystem productivity (NEP), of a SRF plantation of birch, poplar, maple and willow was determined during the first five years of tree growth, with two different approaches (mass-balance and ecological approach).

The low net primary productivity (NPP) observed at the plantation can be explained by the sandy texture of the soil, and by the absence of irrigation, fertilization and/or weed control. Birch appeared to be a good alternative for poplar and willow for the establishment of SRF plantations, especially on marginal agricultural soils. Soil CO_2 effluxes at the plantation were as high as fluxes observed in mature forest stands. Root respiration only contributed 5 % to soil CO_2 efflux. As such, the plantation was a carbon source during the first years of tree growth.

Depending on the approach used for NEP calculation, total carbon loss over the first four years of tree growth was 5.4 t C ha^{-1} (mass-balance approach) or 9.7 t C ha^{-1} (ecological approach). Applying the latter method revealed that NEP became less negative with time. Where the plantation was a source of 4.8 t C ha^{-1} during the first year, NEP was only -0.1 t C ha^{-1} during the fourth year. Therefore, it can be expected that the plantation will become a carbon sink within the next years.

Net primary productivity was increasing with age of the plantation, while interannual differences in total soil CO_2 efflux were related to variability of temperature and precipitation, especially during the leafed period (1st April to 30th September).

Interannual variability in NEP followed the trend of NPP, but the high heterotrophic respiration determined the carbon source status of the plantation.

6.1. Introduction

During recent years, carbon (C) sequestration in terrestrial ecosystems received more attention than ever before, because of the enhanced greenhouse effect and possible consequences for human society. As forests are particularly important in the global carbon cycle (Bolin and Sukumar 2000), determining the net carbon balance of forest ecosystems became an important research topic during the last two decades. According to the Kyoto Protocol (Art. 3.3 and Art. 3.4), afforestation of former agricultural land can offer an opportunity to sequester carbon. Actually, extensive areas of agricultural land are being set aside in Europe, because of the agricultural production surplus (Rabbinge and van Diepen 2000). An expanding amount of these areas are planted with short-rotation forestry (SRF) plantations. In SRF plantations, carbon is stored in biomass and soil, while C emissions by the burning of fossil fuels are prevented when biomass is used as a substitute.

The balance between carbon uptake by photosynthesis and carbon loss during respiration controls the ecosystem carbon budget. Total ecosystem respiration (TER) is composed of aboveground and belowground respiration. Both above- and belowground respiration are the result of autotrophic and heterotrophic respiration processes. Net primary productivity (NPP), which is equivalent to above- and belowground growth, is the difference between gross primary productivity (GPP) and total above- and belowground autotrophic respiration (R_a) (Schulze 2000). NPP can also be described as the total amount of new organic matter produced during a specified time interval (Clark *et al.* 2001). Heterotrophic respiration (R_h) is associated with decomposition of dead organic material. Subtracting R_h from NPP yields net ecosystem productivity (NEP), which equals thus assimilation minus autotrophic plant respiration and minus heterotrophic respiration (Maier and Kress 2000, Curtis *et al.* 2002). As such, the ecosystem C balance, or NEP, can also be calculated by comparing gross primary productivity (GPP) with total ecosystem respiration (TER). NEP quantifies the change of carbon stock in the ecosystem, including growth of woody biomass and changes in soil carbon stocks (Schulze 2000). The overall C balance determines the carbon sink/source strength of an ecosystem (Maier and Kress 2000). It is important to note that all components of the carbon balance can show substantial interannual variability, and therefore, C balance studies should be executed over periods longer than one year preferably (Clark *et al.* 2001, Davidson *et al.* 2002).

Traditionally, the carbon balance of an ecosystem was assessed by the stock change method (Baldocchi 2003). This method is mainly based on recording the mass of carbon in above- and belowground biomass (leaves, stems and branches, roots), and in litter and soil pools (Bolin and Sukumar 2000). The change in pool sizes between sampling times gives an indication of the sink/source strength of the ecosystem (Williams *et al.* 2005). While measurements are relatively simple and cheap, they are labour-intensive too. A disadvantage of this approach is that especially belowground C pools (*e.g.*, fine roots, root litter) are difficult to determine (Smith 2004). Over the past two decades, the eddy covariance technique, which measures carbon fluxes between ecosystems and the atmosphere directly, became very popular. This technique determines the carbon balance at the ecosystem scale, and is as such an integration of photosynthetic and respiratory processes of large areas (Baldocchi 2003). High frequency automatic sampling offers the opportunity to study hourly or diurnal fluctuations in C fluxes (Ehman *et al.* 2002, Zha *et al.* 2004). The main problems of this system are related to stable nighttime conditions or advection occurring at steep slopes (Granier *et al.* 2000, Kolari *et al.* 2004), which have to be accounted for during the calculation process (Zha *et al.* 2004, Black *et al.* 2005, Williams *et al.* 2005). Moreover, the eddy covariance technique asks for a fetch that is large enough to eliminate interference of fluxes from surrounding areas with fluxes from the ecosystem under study (Carrara *et al.* 2003).

Most studies on short-rotation forestry plantations focus on the aboveground biomass production, and the possibility of preventing CO_2 emissions by substituting fossil fuels (*e.g.*, Bergkvist and Ledin 1998, Hofmann-Schielle *et al.* 1999, Telenius 1999, Hytönen and Issakainen 2001, Laureysens *et al.* 2004). Information on the overall C balance of SRF plantations is rather scarce, compared to studies in mature forests (Grigal and Berguson 1998, Kolari *et al.* 2004, Zha *et al.* 2004). As the area of SRF plantations is extending, there is a need to gain more insight in the carbon fluxes and the carbon balance of this type of ecosystems, to improve the assessment of regional or global carbon budgets. It is also important to understand how carbon fluxes vary with stand age, to perform landscape integration of these fluxes (Baldocchi 2003, Kolari *et al.* 2004). According to Carrara *et al.* (2003), information about the interannual variability of NEP is still rather poor. Therefore, the aims of this investigation were i) to study the different components of the carbon balance (NEP), being net primary productivity and heterotrophic respiration, in a short-rotation tree plantation; ii) to determine the sink/source status after the first five years of tree growth by comparing two alternative methods of NEP calculation (mass-balance and ecological approach) and iii) to analyse interannual changes in the carbon balance of this plantation.

6.2. Material and methods

6.2.1. Site description

The short-rotation plantation that served as study site for this investigation is extensively described in § 1.6.1. In Fig. 1.7, an overview is given of this short-rotation plantation, which initially existed of 24 individual plots. In this chapter, plots are numbered as indicated in Fig. 1.7.

6.2.2. Carbon balance or net ecosystem productivity (NEP)

Two approaches for assessing the carbon balance (NEP) of the short-rotation plantation at Zwijnaarde were applied. The ecological approach is based on the estimation of the net carbon balance from autotrophic and heterotrophic processes, while the mass-balance method estimates NEP on base of changes in carbon stocks (Curtis *et al.* 2002, Black *et al.* 2005). According to the ecological approach, NEP (t C ha^{-1} $year^{-1}$) is calculated as given by eq. 6.1 :

$$NEP_{eco} = NPP - R_h \tag{6.1}$$

where NPP (t C ha^{-1} $year^{-1}$) equals net primary productivity and R_h (t C ha^{-1} $year^{-1}$) is heterotrophic respiration. Eq. 6.1 can be rewritten in the form (Black *et al*. 2005) :

$$NEP_{eco} = L + AGD + D_a + D_b + VOC + H - R_{h(soil)} - R_{h(AGD)} - R_{h(consumers)} \tag{6.2}$$

where L is change in living biomass, AGD is annual change in dead material still attached to the trees, D_a is aboveground litter fall, D_b is belowground detritus production, VOC refers to carbon losses associated with the emission of volatile organic components (VOCs), H is related to herbivore removal of biomass, $R_{h(soil)}$, $R_{h(AGD)}$ and $R_{h(consumers)}$ are heterotrophic respiration related to microbial decomposition of soil organic matter, aboveground detritus and respiration by consumers, respectively. All fluxes are expressed in t C ha^{-1} $year^{-1}$. In young forests, AGD can generally be defined as all dead branches attached to the trees (Black *et al.* 2005). As our study concerned a very young plantation, we assumed that this component of the equation was negligible. Carbon losses from VOC emission are generally assumed to be small in forest ecosystems (Clark *et al.* 2001, Black *et al.* 2005, Trumbore 2006) and were neglected, as was the fraction of carbon lost because of herbivory (H). In general, $R_{h(soil)}$ is the main contributor to heterotrophic respiration R_h (Black *et al.* 2005), and is most often calculated as the difference between soil CO_2 efflux and root respiration (Curtis *et al.* 2002). As we assumed AGD to be negligible, $R_{h(AGD)}$ was neglected too. As in most studies (*e.g.*, Black *et al.* 2005), $R_{h(consumers)}$ was not accounted for. Concluding, NEP_{eco} (t C ha^{-1} $year^{-1}$) was simplified to :

$$NEP_{eco} = W + R + F - (SR - R_r) \tag{6.3}$$

where W is wood production (stems and branches), R is root production (both coarse and fine roots), F is the production of foliage, which is equivalent to D_a in a young deciduous forest, SR is soil respiration, determined by measuring soil CO_2 effluxes, and R_r is root respiration. All fluxes are expressed in t C ha^{-1} $year^{-1}$. It can be noted here that production of understorey vegetation is neglected in our study, in contrast to the studies of Curtis *et al.* (2002) and Harmon *et al.* (2004).

The more direct mass-balance approach calculates NEP (t C ha^{-1} $year^{-1}$) as :

$$NEP_{mass} = \Delta C_b + \Delta C_{dead} + \Delta C_{soil} \qquad \textbf{(6.4)}$$

where ΔC_b (t C ha^{-1} $year^{-1}$) is change in carbon stock in living biomass, ΔC_{dead} (t C ha^{-1} $year^{-1}$) is change in C stock in dead biomass and ΔC_{soil} (t C ha^{-1} $year^{-1}$) is change in soil carbon stock (Black *et al.* 2005). ΔC_b equals wood production W plus root production R (eq. 6.3). For both approaches, a positive NEP value represents a net flux from the atmosphere to the ecosystem, while a negative balance denotes the reverse. This means that the ecosystem can be considered as a sink or source for carbon, respectively. Short-term changes in ΔC_{soil} are generally very small and difficult to detect (Smith 2004, Black *et al.* 2005). Changes in ΔC_{soil} measured over longer time periods will be more accurate, and as such, it can be recommended to apply the mass-balance method over longer time scales (Giardina and Ryan 2002).

In this study, the ecological approach was used to determine NEP_{eco} during the first four years of tree growth in 8 plots of the plantation : two birch plots (B5 and B8), two poplar plots (P12 and P17), two willow plots (W19 and W23) and two maple plots (M13 and M24). Measurements were also performed in two reference plots, adjacent to the plantation. On these plots, called NP1 and NP2, no trees were planted. For these 10 plots, NEP_{eco} was calculated for the years 2001, 2002, 2003 and 2004, except for plots W23 and M24, which were destroyed during the experiment, due to building activities next to the plantation. NEP_{eco} was calculated for the willow plots W3, W15 and W19 and for the plots B8 and P12 in 2005 too. The mass-balance method on the other hand was used to calculate NEP_{mass} over the period 2001 till 2004 for the plots B5 and B8 (birch), P12 and P17 (poplar), W19 (willow), M13 (maple) and NP1 and NP2 (non-planted plots). As such, the two approaches could be compared for these eight plots. Finally, the information at plot level for the years 2001 to 2004 was averaged to determine the NEP of the whole plantation. Again, results of both NEP approaches were available, enabling a comparison of the two methods.

6.2.3. Net primary productivity (NPP)

6.2.3.1. Aboveground biomass

Dendrometric measurements allowed calculation of the amount of carbon stored annually in the aboveground woody biomass (stems and branches). At each sampling date, the diameter at 30 cm (d_{30}) of 20 randomly chosen trees per plot was measured with a mechanical calliper. In 2001, diameters were only measured once (end of December). In 2002 and 2003, monthly measurements were performed, while in 2004 and 2005, a bimonthly measurement scheme was followed. Tree diameters, and as such carbon sequestration in aboveground biomass, was studied in detail in the plots B5, B8, P12, P17, W19, W23, M13 and M24. Site- and species-specific allometric equations were applied to calculate aboveground biomass (stems and branches) from the tree diameter at 30 cm height (see Table 3.1). The total amount of carbon stored in the aboveground (AG) biomass was calculated by multiplying the mean AG biomass per tree (g AG DM tree^{-1}) with actual tree density (# trees ha^{-1}). A carbon content of 50 % was assumed for stems and branches. Actual tree density was determined in December 2003, 2004 and 2005 by counting all living trees per plot. Annual aboveground biomass production (W, see eq. 6.3) was calculated as the difference in standing aboveground biomass stock at the end of two consecutive years. For the year 2001, the aboveground biomass production was calculated by subtracting the planted biomass from the standing aboveground biomass determined in December 2001.

Besides measurements at plot level, a complete inventory of the plantation was executed in January 2005 and in January 2006, by measuring d_{30} for 20 randomly chosen trees of each plot. The same procedure as described above was used to determine the aboveground biomass production for all plots of the plantation.

6.2.3.2. Belowground biomass

In January 2002, the complete root systems of 5 trees, belonging to different diameter classes, were excavated, for each species. Roots were dried (60 °C) until constant weight, and the root-to-shoot ratio (R:S) was determined. The R:S ratio was 1.28 for birch, 0.91 for poplar, 1.54 for maple and 1.21 for willow. It should be mentioned here that both fine and coarse roots were included in this calculation. Based on R:S ratios, the total C stock in the roots can be calculated from the standing aboveground biomass. However, it can be expected that during the development of a forest ecosystem, R:S ratios are changing, and more specifically, are decreasing (Giardina and Ryan 2002, Pregitzer 2003). Root biomass was determined for the soil layers 0-15 and 15-30 cm in August 2004 (plot B8), January 2005 (plots B5, B8, P12, P17, W19 and M13) and January 2006 (plots W3, W15 and W19) by taking soil samples, washing them and separating roots (see Chapter 5). After drying (60 °C) until constant weight, dry roots mass was determined. Additional research at the plantation (B. Devos, pers. comm.) revealed that 80 % of all roots are found in the upper 30 cm of the soil. As such, total root biomass in the whole soil profile was calculated by multiplying the dry mass of roots found in the upper 30 cm by a factor 1.25. Comparison of the carbon stock in the roots calculated from aboveground biomass by applying the R:S ratio with the directly measured root biomass learned that higher root biomass was assessed with the R:S ratio method. Therefore, we assumed that the R:S ratio would decrease to 0.25 after 20 years of stand development. This value of 0.25 was comparable to results published by Shan *et al.* (2001) for 17 years old slash pine plantations, by Granier *et al.* (2000) for a 30 years old beech stand, by Longdoz *et al.* (2000) for a 60 years old beech stand

and by Vande Walle *et al.* (2001) for a 75 years old oak-beech and ash stand. For all species, R:S ratios for different years were assessed by linearly interpolating from the starting R:S value (see before) to a R:S of 0.25 after 20 years. For willow, this resulted in a R:S of 1.01 after 5 years of tree growth, which corresponded well with the mean R:S value found by the root biomass measurements in the willow plots W3, W15 and W19 (mean R:S of 1.01, determined in January 2006).

As was the case for aboveground biomass, a C content of 50 % was assumed for roots, and the difference in carbon stock in belowground biomass between two consecutive years was used as an estimate of the annual belowground biomass production R (eq. 6.3).

6.2.3.3. Foliage

Leaf area index (LAI) was estimated from optical measurements with a SunScan system (Delta-T Devices, Cambridge, UK). At each sampling date, 20 measurements were performed randomly in each of the individual plots. The SunScan system was held as close to the soil surface as possible, but above the herbaceous vegetation. LAI was measured 7 times during the growing season in 2001, 9 times in 2002, 8 times in 2003 and 5 times in 2004. In 2005, the maximum LAI was determined on 17th August for the plots B8, P12, W3, W15 and W19. Foliar mass was determined from LAI and specific leaf area (SLA, m^2 kg^{-1} DM). SLA was determined 2 times during the growing season in 2001, 3 times in 2002 and 8 times in 2003. A simple model relating SLA with total radiation sum (see Chapter 2) was used to assess SLA at each date of LAI measurements. Carbon stock in the leaves was calculated assuming a contribution of 50 % C to dry mass. The annual foliage production F (eq. 6.3) was considered as the amount of C stored in the leaves at the moment of maximum LAI.

6.2.4. Heterotrophic soil respiration

6.2.4.1. Soil CO_2 efflux

In April 2001, three PVC collars, with an 18.8 cm internal diameter, were inserted 6 cm in the soil of each of the ten study plots listed above (B5 and B8, P12 and P17, W19 and W23, M13 and M24, NP1 and NP2). Soil CO_2 efflux, often called soil respiration or SR, was measured fortnightly from May 2001 to January 2005 in these ten plots. From January 2005 to January 2006, fortnightly soil CO_2 efflux measurements were performed on eight rings in each of the plots W3, W15 and W19 (willow), and on four rings in the plots B8 (birch) and P12 (poplar). An EGM-1 analyser, coupled to a SRC-1 chamber (PP systems 1994), was used for the measurements. This system is based on the closed dynamic system principle. Detailed information on the system and measurement procedure can be found in Chapter 5. Simultaneously with soil CO_2 efflux, soil temperature was measured at a depth of 5 cm next to the PVC collar. Total soil CO_2 efflux was calculated for the leafed and the unleafed period of the year, by weighting soil CO_2 efflux values according to half the number of days between the former and the next measurement in relation to the actual sampling date. The leafed period, from 1st April to 30th September, referred to spring and summer, or can be considered as the 'warm' period (White and Nemani 2003). The unleafed period (1st January to 31st March and 1st October to 31st December) combined winter and autumn. Results for 2001 were obtained from measurements in the period from 1st May 2001 to 30th April 2002.

6.2.4.2. Autotrophic and heterotrophic components of soil CO_2 efflux

In the period May 2005 to April 2006, fortnightly root respiration measurements were executed in the laboratory on excised fine roots of birch (diameter < 2 mm). The exact methodology used for this study will be described in a subsequent manuscript, but is comparable to the technique used by Widén and Majdi (2001). Following root respiration function was established (n = 96, $R^2 = 0.67$) :

$$R_r = 0.00459 \cdot \exp(0.1433 \cdot T) \tag{6.5}$$

where R_r is root respiration (mg CO_2 g^{-1} DM h^{-1}) en T is cuvette air temperature (°C). Respiration of roots with a diameter between 2 and 5 mm and respiration of roots of the largest diameter class (> 5 mm) was 60 % and 50 % of the respiration observed for the finest roots (< 2 mm), expressed on a dry mass base and measured at a similar temperature, respectively.

So far, respiration measurements on poplar roots revealed no differences with respiration rates from roots of birches (unpublished data). Therefore, eq. 6.5 and the two ratios for the larger diameter classes were assumed to be valid for all tree species at the plantation in Zwijnaarde. Eq. 6.5 and the two ratios were used to calculate root respiration on a root biomass base from the soil temperature measured at the moment of soil CO_2 efflux measurements. Data on root biomass per m^2 soil for the soil layer from 0 to 30 cm were then used to convert root respiration on a root biomass base to root respiration on a soil area base. Root biomass information for the three root diameter classes was available for 12 rings in the birch plot B8 (sampling in August 2004) and for 8 rings in each of the willow plots W3, W15 and W19 (sampling in January 2006; see Chapter 5). As was the case for soil CO_2 efflux, total annual root respiration was assessed by weighting root respiration values according to half the number of days between the former and the next measurement in relation to the actual sampling date. Total annual root respiration appeared to account for only 5.1 % of total annual soil CO_2 efflux for these 36 rings on average, which is very low in comparison to values mentioned for other forest ecosystems (see § 6.4.1.2). However, based on our own measurements, heterotrophic respiration in the plantation at Zwijnaarde was considered to account for 95 % of soil CO_2 efflux.

6.2.5. Carbon stock changes

6.2.5.1. Biomass

To apply the mass-balance approach, ΔC_b, representing the change in carbon stock in living biomass, has to be determined (eq. 6.4). In our study, ΔC_b equalled the sum of W and R, being the net primary productivity of aboveground biomass and of roots, respectively. ΔC_b was calculated for the plots B5, B8, P12, P17, W19 and M13, and this over the period 2001 to 2004.

6.2.5.2. Litter

In January 2005, litter lying on top of the soil was gathered in 24 soil CO_2 efflux rings, 3 rings in each of the plots B5, B8, P12, P17, W19, M13, NP1 and NP2. After drying at 60 °C until constant weight, dry weight of the litter was determined. Taking into account a carbon content of 50 %, the mean carbon stock in the litter layer after four years of tree growth was assessed. This amount was considered as ΔC_{dead}.

6.2.5.3. Soil

In January 2005, soil samples were taken in the soil CO_2 efflux rings of the plots B5, B8, P12, P17, W19, M13, NP1 and NP2, as to determine the soil C stock to a depth of 30 cm. Therefore, the Walkley and Black method was applied on soil samples of the layers 0-5 cm, 5-15 cm and 15-30 cm for determination of the organic carbon content (%). Bulk density was determined for each soil layer. To this end, Kopecky rings were used to take undisturbed soil samples of a known volume. After drying the soil samples at 105 °C for 24 h, bulk density was determined as dry weight (g soil) divided by volume (cm^3). ΔC_{soil} was assessed by subtracting the soil carbon stock determined for each plot at the moment the plantation was established (May 2001, unpublished data) from the soil carbon stock in January 2005. As such, the integrated soil carbon stock change ΔC_{soil} (eq. 6.4) over four growing seasons (2001 to 2004) was estimated.

6.2.6. Meteorological information

As no meteorological information was gathered at the plantation, data from the Royal Meteorological Institute of Belgium (RMI 2001 to 2005) were used. Mean daily air temperature (°C) and total daily precipitation (mm) was available for the measuring station at Kruishoutem (20 km from Zwijnaarde), while daily incoming global radiation was only available for the station at Ukkel (51 km from Zwijnaarde). Data were available from 1st January 2001 to 31st December 2005.

6.2.7. Statistical analysis

Statistical analysis was performed by the programme SPSS 12.0. Normality of data was tested with the Shapiro-Wilk test, while t-tests, Duncan or Tamhane's T2-test were used to detect significant differences between means. The univariate GLM procedure was used to examine interactions between biomass production and soil

CO_2 efflux on the one hand, and mean air temperature, total precipitation and stand age on the other hand (see § 6.3.6). All tests were performed at the 95 % confidence level.

6.3. Results

6.3.1. Meteorological information

In Fig. 6.1, monthly means for incoming global radiation, mean daily air temperature and total precipitation are presented for the period 2001 to 2005. A clear seasonal trend in incoming global radiation (Fig. 6.1a) and in mean air temperature (Fig. 6.1b) was observed, while precipitation was more equally spread over the year (Fig. 6.1c). As can be seen from the error bars, interannual variability was much larger for precipitation than for global radiation and temperature (Fig. 6.1a-c). Fig. 6.1d and 6.1e illustrate that total radiation sum and mean air temperature were always higher in the leafed than in the unleafed period. On the contrary, total precipitation was found to be higher in the leafed period than in the unleafed period in 2001 and 2005, lower in 2002 and 2004, and equal in 2003. All five years appeared to have a mean air temperature well above the long-term average (1961-1990) of 9.8 °C (Fig. 6.1e). Total precipitation was much higher than the long-term average of 821 mm in 2001 and 2002, much lower in 2003 and 2004, and slightly lower in 2005 (Fig. 6.1f). The summer of 2003 was characterized by a European heat wave (Schonwiese *et al.* 2004, Poumadere *et al.* 2005, Gehrig 2006). This is reflected by the fact that the leafed period in 2003 had the highest total radiation sum, the highest mean air temperature and the lowest amount of rainfall of the five years considered in our study.

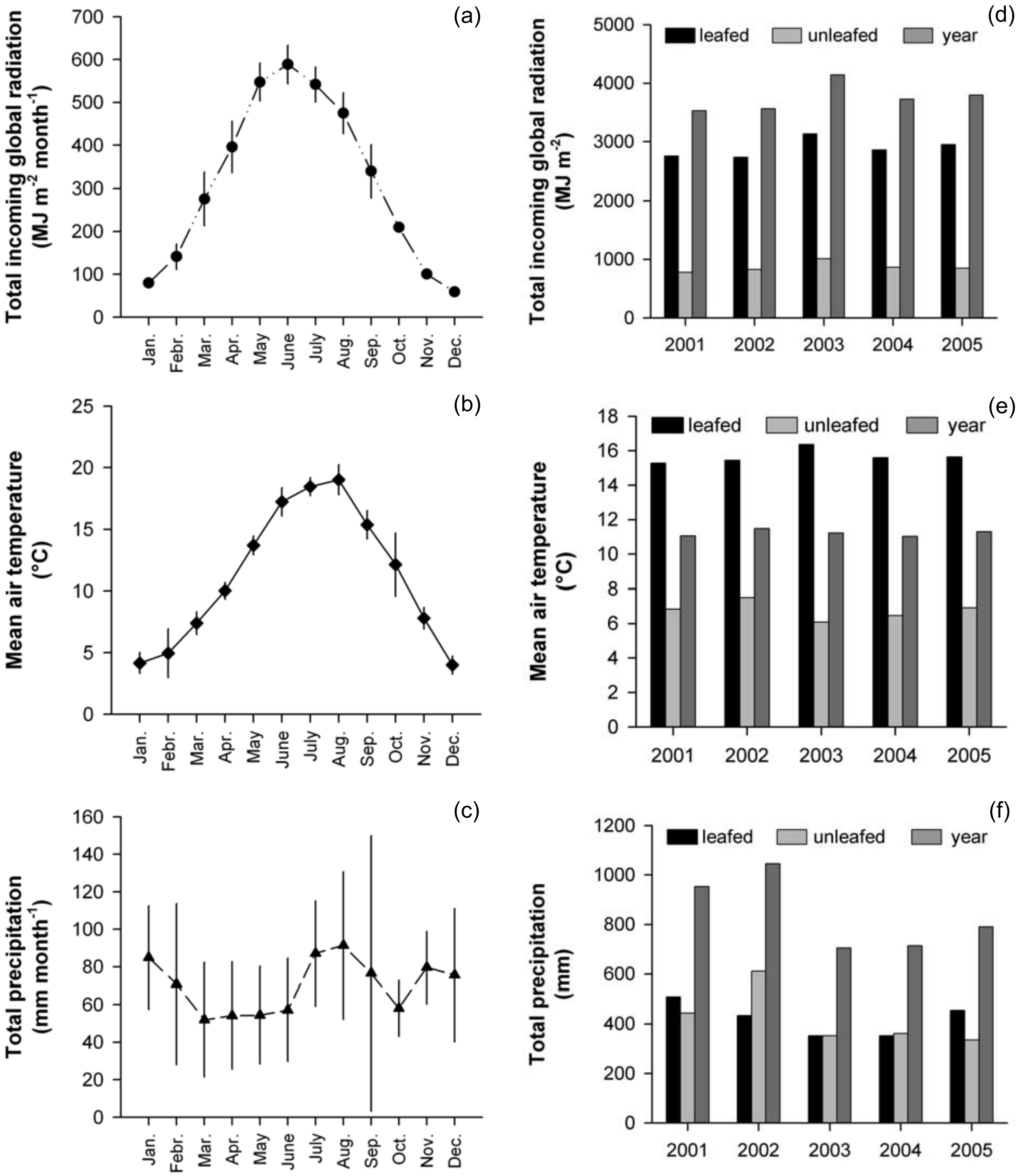

Figure 6.1. *Seasonal course of a) incoming global radiation, b) mean air temperature and c) precipitation; monthly means for 2001 to 2005, error bars indicate one standard deviation; in d), e) and f), total incoming global radiation, mean air temperature and total precipitation are given for the leafed (1st April to 30th September) and the unleafed period (1st January to 31st March and 1st October to 31st December), and for the whole year.*

6.3.2. Net primary productivity

The annual aboveground biomass production determined for different plots of the short-rotation plantation at Zwijnaarde is presented in Fig. 6.2. Aboveground biomass production was clearly lower for maple than for the other tree species. For birch and poplar, an increase in annual aboveground biomass production was seen from 2001 to 2004. In 2005, biomass production of poplar fell down to a very low level, while the production of birch was intermediate at plot B8, and high at plot B5, in comparison to the first four years. Willow grew very well during the first three years, but showed a decline in biomass production afterwards.

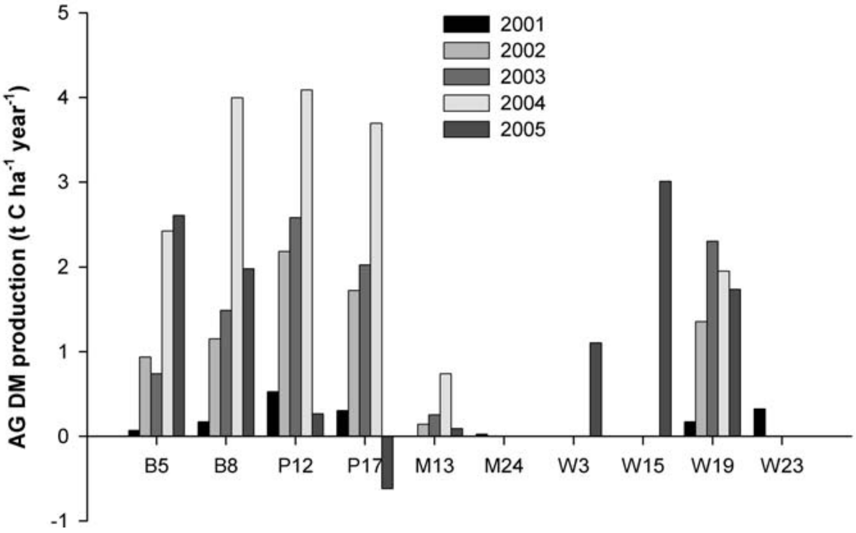

Figure 6.2. *Annual aboveground (AG) biomass (DM) production (t C ha^{-1} year^{-1}) for different plots of the short-rotation plantation at Zwijnaarde; B : birch, P : poplar, M : maple, W : willow*

To check if pooling of production data for plots B5, B8, P12, P17, M13 and W19 gave representative results for the whole plantation, t-tests were used to compare the mean aboveground biomass production of these 6 plots with the mean aboveground biomass production of all plots, determined from the complete inventory of the plantation, and this for January 2005 and January 2006. These tests revealed no significant differences between the mean AG productivity of the six plots which were studied in detail, and the mean of the whole plantation. Therefore, the mean of the six plots under study was further considered representative for the whole plantation.

The evolution of belowground production (Fig. 6.3) was very similar to the aboveground biomass productivity changes, as belowground biomass productivity was estimated by application of the R:S ratio. As R:S ratios were situated around 1 (minimum of 0.77, maximum of 1.54), absolute values of belowground productivity were in the same order of magnitude as results for aboveground productivity.

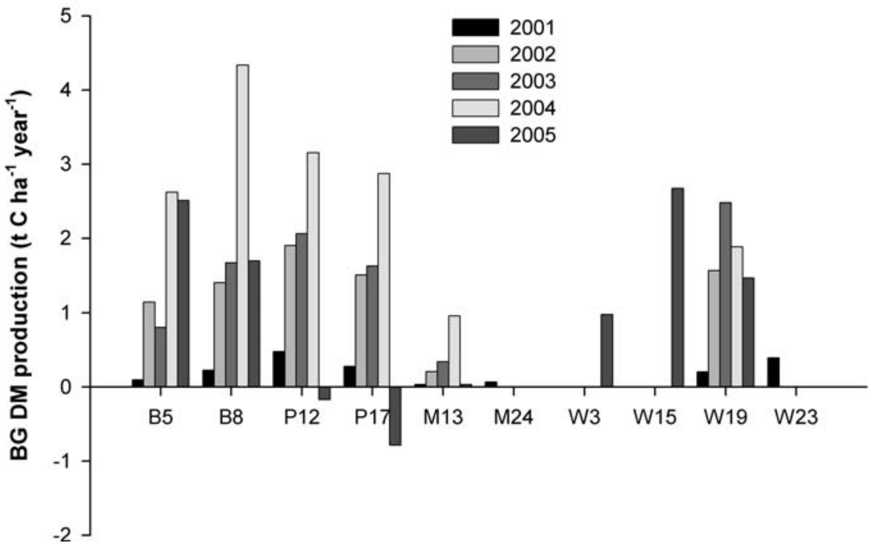

Figure 6.3. *Annual belowground (BG) biomass (DM) production (t C ha^{-1} year^{-1}) for different plots of the short-rotation plantation at Zwijnaarde; B : birch, P : poplar, M : maple, W : willow*

In Fig. 6.4, the evolution of the maximum LAI (LAI$_{max}$) is shown for eight plots of the short-rotation plantation at Zwijnaarde. For birch and maple, a continuously increasing trend in LAI$_{max}$ was observed. For poplar, LAI$_{max}$ was higher in 2004 than in 2005 (plot P12), while for willow, LAI$_{max}$ was highest in 2003, and diminished afterwards. As can be seen from Fig. 6.5, the amount of C stored in leaves ranged from 0.03 to 1.6 t C ha^{-1} year^{-1}.

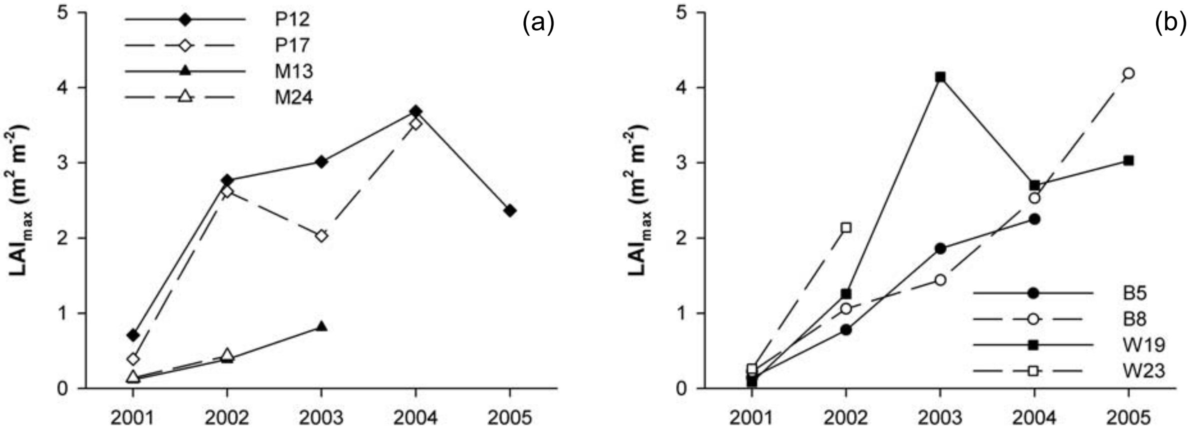

Figure 6.4. *Annual evolution of maximum LAI (LAI$_{max}$) in eight plots of the short-rotation plantation at Zwijnaarde during the period 2001-2005; a) poplar (P) and maple (M) plots, b) birch (B) and willow (W) plots*

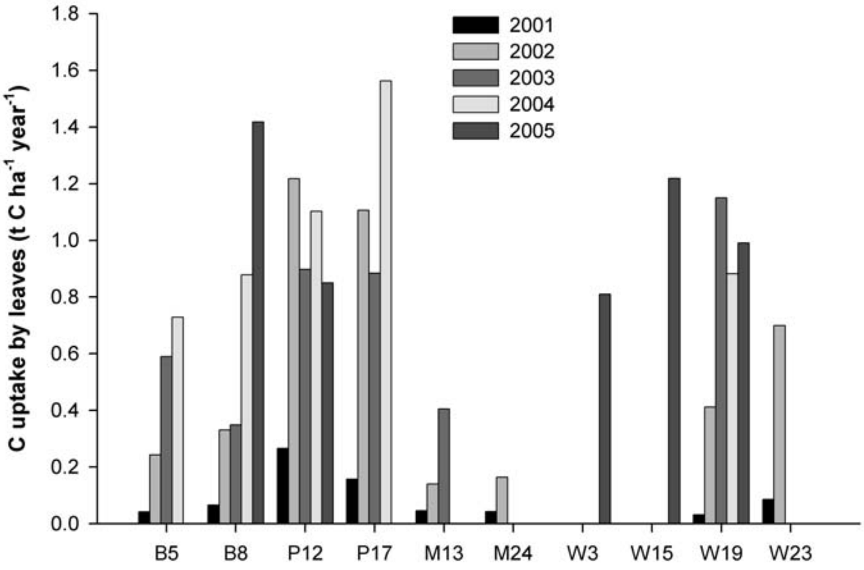

Figure 6.5. *Carbon stored in the leaves (t C ha^{-1} year^{-1}) during five consecutive growing seasons; B : birch, P : poplar, M : maple, W : willow*

6.3.3. Soil CO_2 efflux

In Table 6.1, mean total soil CO_2 effluxes during the leafed and unleafed periods of the years 2001 to 2005 are presented for the plantation as a whole. These values are obtained by averaging the soil CO_2 efflux totals determined at individual measurement points. After testing normality (Shapiro-Wilk test), Duncan or Tamhane's T2-tests were applied to test for differences between years. The total soil CO_2 efflux over an individual year was significantly lower in 2001 and 2003 than in 2004 and 2002, while 2005 showed a (significantly) higher soil CO_2 efflux than all other years. The same result was found for the soil CO_2 efflux observed during the leafed period. The year 2003 showed the significantly lowest soil CO_2 efflux in the unleafed period, while 2005 had a significantly higher soil CO_2 efflux in this period compared to 2001, 2004 and 2002. The (significantly) highest ratio between soil CO_2 efflux in the leafed period compared to the whole year was found for 2003, while the ratio was significantly lowest for 2001 (Table 6.1). As explained in § 6.2.4.1, heterotrophic respiration was calculated as 95 % of soil CO_2 efflux.

6.3.4. NEP : ecological approach

NEP_{eco} values, calculated by eq. 6.3, are presented in Fig. 6.6 for all plots where data were available during the period 2001 to 2005. It can be seen from this graph that the carbon balance on the non-planted plots (NP1 and NP2) was negative for all four years. For most years on most planted plots, a negative carbon balance was observed too. However, for the two birch plots (B5 and B8), the carbon balance became less negative during time (except for 2005). The two poplar plots P12 and P17, one of the birch (B8) and one of the willow plots (W19) appeared to have a positive balance after 4 years of tree growth. This means that the plots under study were mainly acting as a source for carbon during the first years of stand development, but after four years, carbon uptake was already larger than carbon release for some of the plots. In any case, the carbon balance was less negative for the planted than for the non-planted plots.

Table 6.1. *Total soil CO_2 efflux in the leafed period (1st April to 30th September), in the unleafed period (1st January to 31st March and 1st October to 31st December), and in the total year for the short-rotation plantation at Zwijnaarde; the ratio of soil CO_2 efflux in the leafed period compared to the total year and the number of sampling points (n) are also indicated. Standard deviation is given between brackets. Different letters within a column indicate significant differences (*p = *0.05).*

	leafed (t C ha^{-1})	**unleafed (t C ha^{-1})**	**total (t C ha^{-1} year^{-1})**	**ratio leafed/total**	**n**
2001	4.1 (0.5)^a	1.5 (0.4)^b	5.6 (0.8)^a	0.74 (0.05)^a	24
2002	5.5 (1.1)^b	1.6 (0.4)^b	7.1 (1.5)^b	0.77 (0.03)^b	18
2003	4.1 (0.8)^a	0.9 (0.4)^a	5.1 (1.1)^a	0.82 (0.06)^c	18
2004	5.2 (1.1)^b	1.5 (0.5)^b	6.7 (1.4)^b	0.77 (0.04)^b	18
2005	6.8 (1.2)^c	1.9 (0.3)^c	8.7 (1.4)^c	0.78 (0.03)^b	32

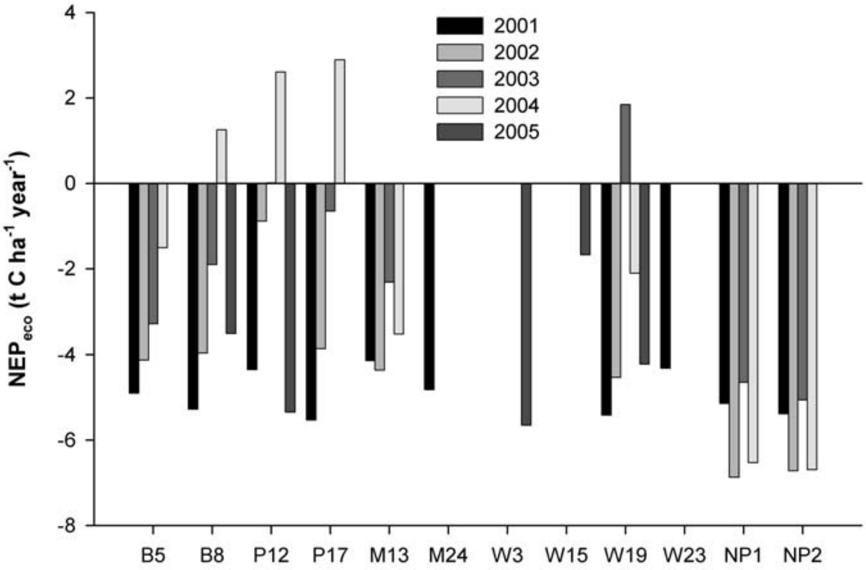

Figure 6.6. *Annual carbon balance determined by the ecological approach (NEP$_{eco}$), for different plots of the short-rotation plantation at Zwijnaarde during the first five years of tree growth; negative values indicate a net carbon loss, positive values a net carbon uptake; B : birch, P : poplar, M : maple, W : willow.*

Results of the six planted plots with data available for 2001 until 2004 were averaged and are presented in Table 6.2. This table is considered as representing the overall carbon balance of the plantation. It can be concluded that the overall carbon balance became less negative during the first years of tree growth. During the fourth year, the overall carbon balance almost equalled zero. The mean annual carbon balance of the plantation, assessed with the ecological approach, was -2.4 t C ha^{-1} year^{-1} during the first four years after the establishment (Table 6.2).

Table 6.2. *Carbon storage in stems and branches (W), fine and coarse roots (R) and leaves (L) during the first four years of tree growth determined for the short-rotation plantation at Zwijnaarde; mean for 8 plots in 2001, 6 plots in 2002 and 2003 and 5 plots in 2004;* R_h *: heterotrophic soil respiration;* ΔC_b *: carbon storage in tree biomass (= W + R),* ΔC_{dead} *: carbon storage in the litter layer,* ΔC_{soil} *: change in soil carbon stock;* NEP_{eco} *: carbon balance, calculated as* $W + R + L - R_h$*; total values for the period 2001 to 2004 are given in the last column;* NEP_{mass} *: carbon balance, calculated as* $\Delta C_b + \Delta C_{dead} + \Delta C_{soil}$*; standard deviation is given between brackets.*

	2001 **(t C ha^{-1} year^{-1})**	**2002** **(t C ha^{-1} year^{-1})**	**2003** **(t C ha^{-1} year^{-1})**	**2004** **(t C ha^{-1} year^{-1})**	**Total for 2001 - 2004** **(t C ha^{-1} 4 years^{-1})**
W	0.20 (0.04)	1.3 (0.2)	1.6 (0.8)	2.8 (1.4)	5.8 (2.9)
R	0.22 (0.04)	1.3 (0.2)	1.5 (0.7)	2.6 (1.2)	5.6 (2.3)
L	0.09 (0.08)	0.5 (0.4)	0.7 (0.3)	1.0 (0.3)	2.3 (1.2)
R_h	5.4 (0.6)	6.7 (1.2)	4.8 (0.9)	6.4 (1.1)	23.4 (3.0)
ΔC_b					11.5 (5.3)
ΔC_{dead}					1.0 (0.4)
ΔC_{soil}					-17.8 (1.8)
NEP_{eco}	-4.8 (0.5)	-3.6 (1.4)	-1.1 (1.8)	-0.1 (2.7)	-9.7 (4.4)
NEP_{mass}					-5.4 (5.8)

6.3.5. NEP : mass-balance approach

Total ΔC_b, ΔC_{dead} and ΔC_{soil} during the period 2001 to 2004 for the plantation as a whole are given in Table 6.2. NEP_{mass} was -5.4 t C ha^{-1} 4 years^{-1}, or -1.4 t C ha^{-1} year^{-1} on average. In Fig. 6.7, NEP_{mass} results are compared with results of the ecological approach. NEP_{mass} for individual plots ranged from -3.3 t C ha^{-1} year^{-1} for M13 to +0.8 t C ha^{-1} year^{-1} for P12, while values for the non-planted plots were -2.4 and -4.0 t C ha^{-1} year^{-1} (Fig. 6.7). From this graph, it is clear that the ecological approach resulted in more negative estimates of NEP than the mass-balance method.

6.3.6. Interannual variability of biomass production and soil CO_2 efflux

GLM analysis did not reveal significant interactions between climatic conditions (mean annual air temperature and total annual precipitation) and total production of biomass (including stems, branches, roots and foliage). However, a significant increase (p = 0.025) of biomass production with increasing stand age was observed for the first four years after establishment of the plantation (Table 6.2). The univariate GLM procedure was further used to examine the interaction between total soil CO_2 efflux, mean air temperature and total precipitation at the SRF plantation at Zwijnaarde for the years 2001 to 2005. No significant interaction was found for the whole year and for the unleafed period. For the leafed period however, mean air temperature (p = 0.032), total precipitation (p = 0.033) as well as the interaction between these two variables (p = 0.033) had a significant influence on interannual differences in soil CO_2 efflux. For instance, the dry and warm leafed period in 2003 was characterized by a lower soil CO_2 efflux compared to the other summer periods.

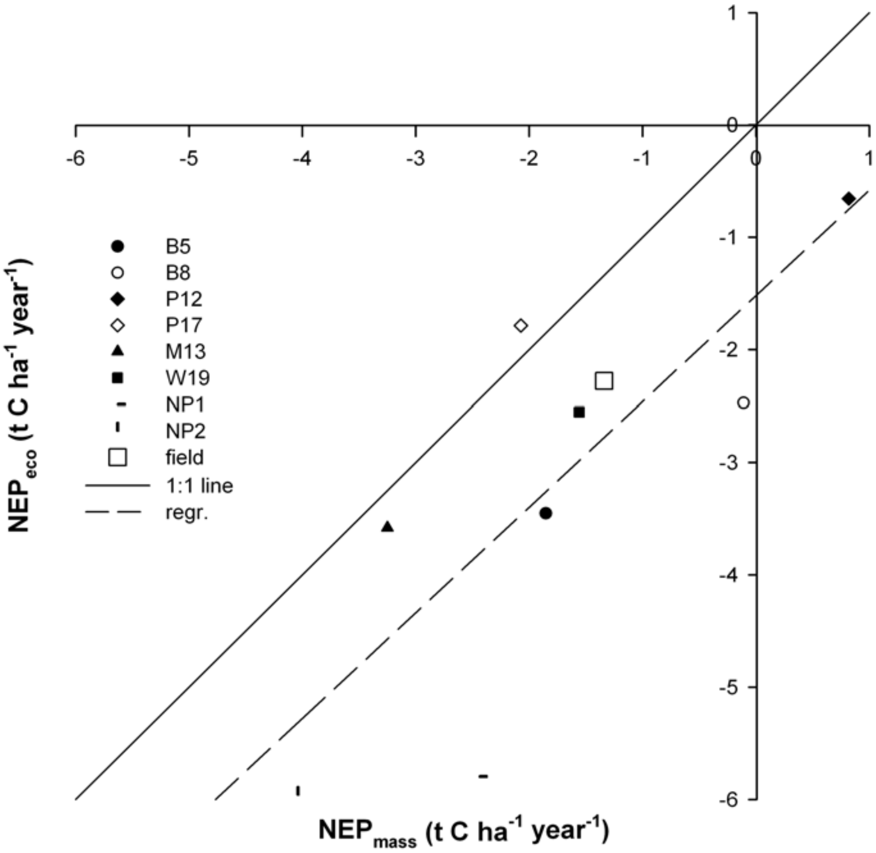

Figure 6.7. *Comparison of the carbon balance calculated with the mass-balance approach (NEP_{mass}) with the carbon balance determined according to the ecological approach (NEP_{eco}) for individual plots, and for the plantation as a whole ('field'); regression (regr.) equation is $NEP_{eco} = -1.52 + 0.94 \cdot NEP_{mass}$ ($R^2 = 0.62$); B : birch, P : poplar, M : maple, W : willow.*

6.4. Discussion

6.4.1. Components of the carbon balance

6.4.1.1. Net primary productivity

The aboveground biomass production values determined at Zwijnaarde (Fig. 6.2) were low compared to values reported for other SRF plantations of birch (Telenius 1999, Hytönen and Issakainen 2001), poplar (Armstrong *et al.* 1999, Hofmann-Schielle *et al.* 1999, Laureysens *et al.* 2004) or willow (Bergkvist and Ledin 1998, Hofmann-Schielle *et al.* 1999, Nordh and Verwijst 2004). However, as was extensively discussed in Chapter 3, no weed control, fertilization and/or irrigation was applied in our experiment, in contrast to most other studies. As was also stated by Deckmyn *et al.* (2004b), poplar is a tree species with a high water use. Therefore, the sandy soil of the plantation at Zwijnaarde is less suited for the cultivation of this species.

Based on results from the two inventories covering all plots of the plantation, the mean annual above- and belowground biomass increment (MAI) amounted to 2.9 t C ha^{-1} year^{-1} after 4 years of tree growth, and 3.0 t C ha^{-1} year^{-1} after 5 years, which equalled MAI after 5 years as observed by Elowson and Rytter (1993) on a gray alder plantation on a sandy soil. MAI values after 4 and 5 years were not significantly different. However, there were some remarkable differences between the four tree species. MAI of birch increased from 2.7 t C ha^{-1} year^{-1} after 4 years of tree growth to 3.2 t C ha^{-1} year^{-1} after 5 years, while for maple, an increase from 1.4 to 1.9 t C ha^{-1} year^{-1} was noted. On the other hand, MAI decreased from 3.2 to 2.9 t C ha^{-1} year^{-1} for poplar, and from 3.6 to 3.3 t C ha^{-1} year^{-1} for willow after 4 and 5 years, respectively. This change was statistically significant for willow only. The decrease of MAI for poplar can be due to the infection by *Colletotrichum gloeosporiodes*, while the decreasing MAI for both poplar and willow can also partly be caused by the dry summer periods in 2003 and 2004, as both are high water demanding species. As can be seen in Fig. 6.2, birch had a low biomass production during the first years. This was mainly due to severe root stress at the moment of planting. It took the birches some years to fully recover from this stress, but after three years, the biomass production was comparable to the one of poplar and willow. This indicates that birch can be an interesting alternative for the establishment of short-rotation plantations, as it can be expected that SRF plantations will most probably be planted on marginal agricultural soils (Ritter *et al.* 2003), which are less suitable for high-water demanding species as poplar and willow. Sowing birches instead of planting young saplings could be an option to overcome the problems with root damage during planting.

NPP increased from 0.5 to 6.4 t C ha^{-1} year^{-1} during the first four years of tree growth, averaged over the plantation (Table 6.2). This latter value is consistent with, although in the lower range of, NPP values of different forest ecosystems, reported in literature. NPP was 4.5 and 5.2 t C ha^{-1} year^{-1} in a 30 years old beech stand in France (Granier *et al.* 2000), while NPP amounted from 5.1 to 10.5 t C ha^{-1} year^{-1} in 5 North American deciduous forests studied by Curtis *et al*. (2002). Black *et al.* (2005) reported high NPP values of 12.5 and 12.7 t C ha^{-1} year^{-1} in a 14 years old Sitka spruce forest in Ireland. NPP of Himalayan poplar plantations was mentioned to increase from 9.5 t C ha^{-1} year^{-1} after the establishment year to 16.2 t C ha^{-1} year^{-1} after four years of tree growth (Lodhiyal and Lodhiyal 1997). However, it can be expected that the deep, fertile soils, with a pH between 5.5 and 7.5, on which these Himalayan plantations were situated were more suited for poplar cultivation than the sandy soil at Zwijnaarde.

Determination of litter fall is often done by placing baskets under the canopy (Shan *et al.* 2001, Curtis *et al.* 2002, Ehman *et al.* 2002, Giardina and Ryan 2002, Harmon *et al.* 2004, Thuille and Schulze 2006). Because of the young age of the trees, branches and leaves were hanging close to the forest floor at the SRF plantation at

Zwijnaarde. Therefore, LAI and SLA measurements were used to assess the total amount of C stored in the leaves. This foliar C stock increased from 0.1 t C ha^{-1} in the year of establishment to 1.0 t C ha^{-1} during the fourth growing season (Table 6.2). Increasing litter fall with stand age was also reported by Giardina and Ryan (2002) for Eucalyptus plantations at Hawaii and by Thuille and Schulze (2006) for a variety of spruce chronosequences in the Alps. Davidson *et al.* (2002) mentioned annual litter fall values between 1.0 and 2.7 t C ha^{-1} year^{-1} in different deciduous forests, with higher values for older stands. Annual leaf litter was 1.3 t C ha^{-1} year^{-1} in a 30 years old beech stand at Hesse (Granier *et al.* 2000), and ranged from 1.3 to 2.3 t C ha^{-1} year^{-1} in five deciduous forest stands studied by Curtis *et al.* (2002). The amount of carbon stored in the leaves (L, see Table 6.2) compared to the above- and belowground biomass (W + R) was fairly constant during the study period, and was 19 % at minimum and 23 % at maximum, in contrast to the clear decrease in this ratio after four years of tree growth in Himalayan poplar plantations, as reported by Lodhiyal and Lodhiyal (1997). In their study, leaf litter increased from 1.0 to 3.2 t C ha^{-1} year^{-1} from the first to the fourth year of tree growth.

At Zwijnaarde, high initial R:S ratios were observed (see § 6.2.3.2). This can be due to the absence of fertilization and weed control. Both management activities resulted in reduced R:S ratios in the study of Shan *et al.* (2001). Fertilization increases nutrient availability, and hence, trees presumably need relatively fewer fine roots to absorb nutrients and water. As weed control reduces competition for resources, trees will invest less carbon in the production of (fine) roots. The sandy soil on which the plantation is situated, can be another factor inducing a high R:S ratio, as C allocation to roots is higher on drier sites (Lambers *et al.* 1998, Larcher 2003).

6.4.1.2. Total and heterotrophic soil respiration

Mean annual soil CO_2 efflux rates determined at Zwijnaarde during the first 5 years of tree growth ranged from 5.1 to 8.7 t C ha^{-1} year^{-1} (Table 6.1). These values fell within the range from 4.0 to 12.0 t C ha^{-1} year^{-1} found in 16 European forests (Janssens *et al.* 2001b). They also accorded to values published by Raich and Schlesinger (1992), who mentioned annual soil CO_2 effluxes between 3.0 and 14.0 t C ha^{-1} year^{-1} for temperate deciduous forests. Longdoz *et al.* (2000) measured an annual soil CO_2 efflux of 8.7 t C ha^{-1} year^{-1} in a 60 years old beech stand at Vielsalm (Belgium), while Granier *et al.* (2000) reported a total annual soil CO_2 efflux of 5.8 t C ha^{-1} year^{-1} for 1996 and 6.6 t C ha^{-1} year^{-1} for 1997 in a 30 years old beech stand at Hesse (France). Although the plantation at Zwijnaarde was much younger and had a much higher planting density than most other studied forest ecosystems, total annual soil CO_2 effluxes appeared to be in the same order of magnitude as the release of carbon by the soil in older forests.

Measured root respiration rates varied between 0.02 and 0.19 mg CO_2 g^{-1} DM h^{-1}, which was in the lower range of results mentioned by Widén and Majdi (2001), who used a comparable method to measure root respiration as the one used in our study. From eq. 6.5, it can be calculated that the Q_{10} value of root respiration was 4.2. Q_{10} of root respiration was 4.6 and 5.0 in the studies of Boone *et al.* (1998) and Widén and Majdi (2001), respectively. In general, Q_{10} values for soil CO_2 efflux are much lower, and are mostly situated between 2.1 and 3.2, as mentioned by Fang *et al.* (1998), Buchmann (2000), Matteucci *et al.* (2000) and Widén and Majdi (2001), who all measured soil temperature at 5 cm depth. The Q_{10} of 4.2 determined for root respiration at Zwijnaarde corresponded very well with the mean Q_{10} value of 4.3 determined for soil CO_2 efflux in 24 sampling points in the three willow plots W3, W15 and W19 during the year 2005 (see Chapter 5). This could suggest that soil CO_2 efflux is mainly determined by root respiration processes. However, conversion of root respiration fluxes at a root mass base in fluxes at soil area base revealed only a minor contribution of root respiration to total soil CO_2 efflux (mean of 5.1 % for 36 sampling points). Widén and Majdi (2001) suggested that the calculated contribution of root respiration to total soil CO_2 efflux depends on the moment of root biomass determination, especially when root biomass is only determined once in the year. In our study, root biomass was determined in August for a birch plot (B8) and in January for three willow plots (W3, W15 and W19). The calculated root respiration contribution however was very low, for all four plots. Another possible reason for an underestimation of root respiration can be the use of an underestimated root biomass, when the wrong soil depth is considered. A test by Thierron and Laudelot (1996) in an oak-hornbeam forest at Chimay (Belgium) showed that most of the soil CO_2 efflux came from depths not much lower than 10 cm. Therefore, it is not probable that we underestimated root respiration because of an underestimation of root biomass, as we used root biomass up to a depth of 30 cm to express root respiration on a soil area base. As discussed in Chapter 5, Q_{10} values of soil CO_2 efflux appeared to vary annually at the plantation at Zwijnaarde, and are probably reflecting interannual variability in climatic conditions. For 2004, a Q_{10} of 2.7 was noted for plot W19, while Q_{10} was only 2.2 for birch plot B8 in the period August 2003 to July 2004. These low Q_{10} values also contradict the before mentioned possibility of a high contribution of root respiration to total soil CO_2 efflux.

The review by Hanson *et al.* (2000) revealed a mean contribution of root respiration to soil CO_2 efflux of 49 % in forests, but this contribution varies widely among investigations : 12 to 62 % in mixed pine and spruce forest stands (Widén and Majdi 2001), 30 to 50 % in a 8 years old loblolly pine plantation (Maier and Kress 2000), 54 % in Scots pine stands in Sweden (Högberg *et al.* 2001), 60 % in a French beech stand of 30 years old (Epron *et al.* 1999b), to as much as 90 % in an oak-hornbeam forest at Chimay (Thierron and Laudelot 1996). Root and heterotrophic respiration cannot be separated easily (Granier *et al.* 2000), and in some studies, a fixed ratio between root and soil respiration is assumed (*e.g.*, 50 % in the study of Ehman *et al.* 2002). We found a very low contribution of root respiration to total soil CO_2 efflux at the plantation at Zwijnaarde.

According to Raich and Schlesinger (1992), the contribution of root respiration to total soil CO_2 efflux depends on site characteristics as species, climate and stand age. Mean total root biomass in the top 30 cm of the soil amounted to 3.4 t DM ha^{-1} for the 12 sampling points in plot B8 and 12.6 t DM ha^{-1} for the 24 points in plot W3, W15 and W19. The higher amount of root biomass in the willow stands can be due to the higher tree density on these plots compared to the birch plot, according to the study of Giardina and Ryan (2002). Root biomass observed at Zwijnaarde was lower than the total root biomass in mature forests : 12 to 24 t DM ha^{-1} in three coniferous stands in Sweden (Widén and Majdi 2001), 25 t DM ha^{-1} in a 30 years old beech stand in France (Epron *et al.* 1999b), 57 t DM ha^{-1} for a 75 years old oak-beech and ash stand in Belgium (Vande Walle *et al.* 2001). The relatively low amount of root biomass at the plantation at Zwijnaarde, due to its young age, is the most probable reason for the low contribution of root respiration to total soil CO_2 efflux.

6.4.1.3. Soil carbon stock changes

A decrease of 17.8 t C ha^{-1} in soil carbon stock, or 16.8 t C ha^{-1} when litter is taken into account, was observed after four years of tree growth at the plantation (Table 6.2), which was a decrease of 42 % compared to the amount of carbon present in the soil at the moment of planting (mean of 42.8 t C ha^{-1} in the upper 30 cm of the six studied plots). This decrease was the result of a relatively small input of aboveground litter to the soil (Table 6.2), and a continuation of decomposition of residues from the preceding agriculture phase by heterotrophic organisms (Paul *et al.* 2002, Vesterdal *et al.* 2002, Paul *et al.* 2003). Our observation corresponded to results published by Thuille and Schulze (2006). These authors provided an overview of 21 studies of changing mineral soil carbon stocks after afforestation of pasture, grassland or arable land. Most investigations revealed a decrease in soil C stock during the initial 10 to 15 years. Thereafter, gains were reported for former arable land, while on converted grassland, carbon stocks did not always increase after initial losses. A German investigation (Jug *et al.* 1999) of changes in soil properties after afforestation with poplar and willow revealed an initial decline of soil carbon stock, due to enhanced mineralization. In a second phase, an enhanced litter input resulted in an increase in carbon stock in the upper soil layer after 10 years of tree growth. From an extensive overview of results on soil carbon stock changes after afforestation, Paul *et al.* (2002) concluded that soil carbon stocks decreased during the first five years of tree growth, but then recovered so that after about 30 years, there was little difference or slightly more C than in the preceding agricultural soil. The magnitude and rate of soil carbon stock changes depend on soil type, climate, species type and management practices (Paul *et al.* 2003). Grigal and Berguson (1998) also stated that a sharp initial loss of soil C can be expected under SRF plantations, with carbon accretion becoming positive in later years of the rotation, so that soil C reaches initial levels after 15 years of tree growth. Based on these results, it can be expected that soil carbon stocks will increase in the near future at the SRF plantation at Zwijnaarde. It can be doubted, however, whether very high soil carbon stocks will ever be reached. As stated by

Shan *et al.* (2001), soil carbon storage in sandy soils is limited, as conditions are favourable for decomposition of organic material, and because of the relatively low capacity of sandy soils to protect soil C from decomposition.

Noteworthy is the redistribution of carbon over the soil profile. Vesterdal *et al.* (2002) found an increase of the carbon stock in the upper soil layer (0-5 cm), while a decrease was observed for lower soil layers, after 30 years of afforestation with oak and Norway spruce. At our plantation, a similar redistribution pattern was already observed after 4 growing seasons (Fig. 6.8). For all three soil layers, a decrease in soil carbon stock was observed, but this decrease was only statistically significant in the two deeper layers (5-15 cm and 15-30 cm). Expressed in terms of percentage, this decrease amounted to -6.9, -29.9 and -58.7 % for the soil layers 0-5 cm, 5-15 cm and 15-30 cm, respectively. Expressed per depth unit, carbon loss from the soil amounted to 0.1, 0.4 and 0.9 t C ha^{-1} cm^{-1} depth for these three layers, respectively. Probably, the lower C loss in the upper layer is the result of the input from leaf litter.

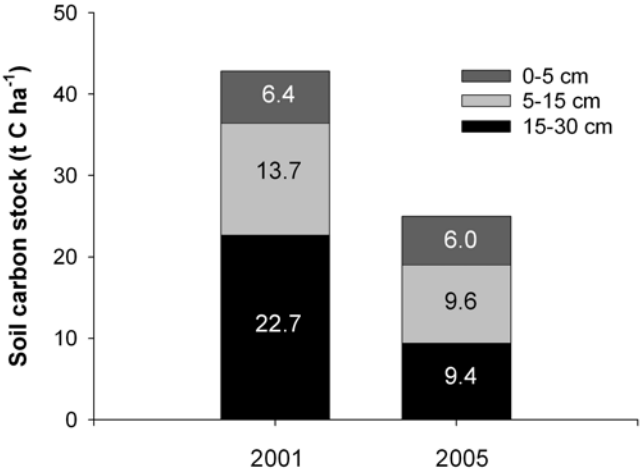

Figure 6.8. *Carbon stock in the soil layers from 0-5 cm, 5-15 cm and 15-30 cm depth, at the moment of establishment (2001) and after four years of tree growth (2005); mean for six plots of the short-rotation plantation at Zwijnaarde*

6.4.2. Comparison of approaches to assess NEP

As illustrated in Fig. 6.7, applying different approaches resulted in different estimates of NEP for the plantation at Zwijnaarde, as was reported in other studies too (Curtis *et al.* 2002, Harmon *et al.* 2004, Kolari *et al.* 2004, Williams *et al.* 2005). Differences between NEP determined with the eddy covariance technique or with an inventory approach were 51 % in the study of Black *et al.* (2005). In both approaches applied in this study, above- and belowground stock changes or fluxes have to be assessed. The ecological approach (eq. 6.3) almost always resulted in a more negative NEP value than the mass-balance approach (eq. 6.4), at individual plot level as well as at field level (Fig. 6.7, Table 6.2). As W plus R from eq. 6.3 was equal to ΔC_b from eq. 6.4, this cannot explain these differences. The amount of carbon stored in the leaves (eq. 6.3) was calculated from LAI and SLA measurements, which were performed several times during the growing season, with a well-known technique and a high number of repetitions. Therefore, there is no reason to assume that this value is largely over- or underestimated.

Curtis *et al.* (2002) stated that the simplistic approach of using a fixed ratio between total soil CO_2 efflux and soil heterotrophic respiration introduces a large uncertainty on NEP estimates. The contribution of roots to total soil CO_2 efflux was found to be only 5 % at the plantation at Zwijnaarde, which was much lower than values reported in literature, mostly for mature forest ecosystems. If the real contribution of root respiration was higher, heterotrophic respiration was overestimated, and the real difference in NEP between the two approaches would be lower. More research is needed to enhance the understanding of these belowground carbon fluxes.

Neglecting decomposition of coarse woody debris ($R_{h(AGD)}$ in eq. 6.2) seems less problematic, as this accounted for less than 10 % of total soil CO_2 efflux in mature forests (Curtis *et al.* 2002). In the young plantation at Zwijnaarde, almost no coarse woody debris was present at the time of the investigation.

In our study, we only determined soil carbon stocks to a depth of 30 cm, as was done by Black *et al.* (2005) too, and considered the change in this layer as ΔC_{soil} (eq. 6.4). Thirty centimetres is also the depth proposed by the IPCC for national carbon inventories (Houghton *et al.* 1997). Soil carbon stock changes of deeper layers could also be taken into account, although Jug *et al*. (1999) stated that main changes in soil carbon stocks will take place predominantly in the top soil horizons, and not in subsoils. Nevertheless, if the amount of carbon in these deeper layers also diminished, as was the case for the upper 30 cm (Table 6.2), NEP_{mass} would become more negative, and as such, would approximate better NEP_{eco}. However, as 20 % of the roots occurred below 30 cm, soil carbon stocks below this depth can also have increased over the first four years of tree growth, following the results of Hansen (1993), who found a significant soil carbon gain after afforestation in the 30 to 50 cm layer, which was attributed to tree root growth. An increase in C stock in deeper soil layers would result in a larger difference between the two NEP estimations.

Another remark that can be made is that NPP of the herbaceous vegetation has been neglected in this investigation. However, especially on the two non-planted plots, a well-developed herbaceous layer is present. When this would be taken into account too, NEP would be less negative (Fig. 6.7). This was previously mentioned by Clark *et al.* (2001), who stated that most studies up to now can be assumed to have underestimated NPP, because of omitting or neglecting several components of NPP.

It is difficult to conclude which of the two approaches gives the best estimate of NEP. Black *et al.* (2005) suggested that the NEP_{eco} method produces a more representative estimate of NEP than NEP_{mass}. However, the NEP_{eco} method is difficult to apply using conventional inventory data and in general, more assumptions have to be made than when using the mass-balance method. Especially fine root production and heterotrophic respiration processes have to be better understood. The main advantage of the ecological approach is that it can be used at relative short time scales (*e.g.*, one year), and gives an idea about interannual variation in NEP. The mass-balance method on the other hand, cannot be used at short time scales, especially because of difficulties to determine soil carbon stock changes with a high degree of certainty during short periods (Baldocchi 2003, Smith 2004, Black *et al.* 2005). In some cases, ΔC_{soil} is not even measured, is assumed to be zero (Curtis *et al*. 2002, Ehman *et al.* 2002, Kolari *et al.* 2004), or is derived from paired-sites comparisons or from measurements in a chronosequence, where space is considered as substitute for time (Paul *et al.* 2002, Ritter *et al.* 2003). In these two latter cases, inevitable differences in soil conditions among the investigated sites or stands can affect the conclusions (Vesterdal *et al.* 2002). Measurements needed to apply the mass-balance method are in general less complicated than for the ecological approach, where soil and root respiration measurements have to be performed. Anyhow, whatever method is applied, the belowground system is the most difficult compartment to evaluate, as was also stated by many other authors (Högberg *et al.* 2001, Curtis *et al.* 2002, Pregitzer 2003).

Based on the results of both approaches, it appeared that all individual plots were a source for carbon during the first four years of tree growth. Plot P12 was a small sink for carbon in this period, according to the mass-balance approach (Fig. 6.7). This was mainly due to the high biomass production at this plot. The plantation as a whole was a carbon source during the study period, but there was a clear trend towards a carbon balance equalling zero. Most other temperate forest ecosystems described in literature are mentioned to act as a sink for carbon (Ehman *et al.* 2002, Zha *et al.* 2004, Williams *et al.* 2005). The 73 years old mixed temperate forest studied by Carrara *et al.* (2004) was one of the exceptions on this general rule, with a mean C balance of -0.9 t C ha^{-1} $year^{-1}$ during the period 1997 to 2002. The carbon source status of the plantation at Zwijnaarde was not unexpected, as it is a young stand, with a low biomass production. Maier and Kress (2000) noted a negative C balance for unfertilized plots in an 8 years old loblolly pine plantation, while fertilized plots were obvious carbon sinks. This was mainly due to the much higher biomass production on these latter plots. The time needed for a regenerating forest to become a net sink for carbon depends on the degree of site disturbance, site microclimate, soil fertility (Scott *et al.* 1999), and initial soil carbon stocks and plant productivity. Estimates range from 15 years for north-eastern hardwood forests to as little as 3 years for slash pine plantations (Maier and Kress 2000).

6.4.3. Interannual variability of the carbon balance

Annual C balances of 15 European forests from the EUROFLUX project, determined with the eddy covariance technique, ranged from an uptake of 6.6 t C ha^{-1} $year^{-1}$ to a release of nearly 1 t C ha^{-1} $year^{-1}$ in the period 1996 to 1998, with a large variability between forests and years (Valentini *et al*. 2000). Carrara *et al.* (2003) gave an overview of average NEP values for different CARBOEUROFLUX sites. These average values ranged from -2.2 to 7.2 t C ha^{-1} $year^{-1}$, but varied largely between years. Measurements in a 60 to 80 years old forest have shown interannual variation in weather to affect annual NEP (Barford *et al.* 2001). More specifically, interannual variability can be caused by different factors, as differences in mean air temperature (Zha *et al.* 2004), total precipitation (Ehman *et al.* 2002, Flanagan *et al.* 2002), incoming radiation (Zha *et al.* 2004), vapour pressure deficit of the air (Carrara *et al.* 2004) or length of the growing season (Carrara *et al.* 2003). These climatic factors can affect GPP, NPP as well as respiration processes.

From the GLM analysis, it was concluded that mean air temperature, total precipitation as well as the interaction between these two variables affected soil CO_2 efflux during the leafed period significantly. Depending on the scale at which data were considered (regional or global), annual soil CO_2 effluxes correlated with annual precipitation or with air temperature in the analysis of Raich *et al.* (2002). As soil CO_2 efflux during the leafed period accounted for more than three fourth of total annual SR (Table 6.1), weather conditions in this period determined largely the interannual variability of soil CO_2 effluxes. Heterotrophic respiration is described to be more sensitive to periods of drought than root respiration (Widén and Majdi 2001, Borken *et al.* 2002). As heterotrophic respiration accounted for 95 % of total soil CO_2 effluxes, this can explain the observed sensitivity of SR to climatic conditions. Borken *et al.* (2002) and Irvine and Law (2002) also observed a reduction in annual soil CO_2 efflux due to summer droughts, while Scott-Denton *et al.* (2003) concluded from soil CO_2 efflux measurements in a mixed conifer subalpine forest that variation in soil temperature was the primary temporal control seasonally, whereas variation in soil moisture appeared to be the primary control interannually. Our results also followed the conclusion of Curiel Yuste *et al.* (2003), who stated that even in temperate maritime climates, total soil C losses can be significantly affected by drought during summer periods. Their study was performed in a Scots pine stand in the Belgian Campine region, situated on a sandy soil too.

No significant interaction between climatic conditions and production of biomass could be determined, but a significant increase of biomass production with increasing stand age was observed for the first four years after the establishment of the plantation (Table 6.2). This interannual increase in biomass production reflected the increase in LAI (Fig. 6.4), as was also observed in a regenerating Douglas fir stand during the first three years of tree growth by Humphreys *et al.* (2005). The lack of correlation between NPP and weather differed from patterns seen in mature forests, where biomass production is reported to dependent on climatic conditions (Barford *et al.* 2001, see also Chapter 8). The young age of the plantation we studied is the main reason for this difference. Our results confirmed the conclusion of Humphreys *et al.* (2005) that in recently harvested and replanted stands, interannual variations in NEP may be influenced more by the re-establishment of vegetation and decomposition of debris and underground biomass from the previous stand than by variations in weather.

During the first four years of stand development at Zwijnaarde, NEP was becoming less negative (Table 6.2). This trend was mainly due to the increase in biomass production, which was related to stand age, but not directly to climatic conditions. The interannual variability in soil respiration, and as such in heterotrophic respiration, was due to differences in temperature and precipitation during the leafed period, but was not large enough to level off the increasing trend in biomass production.

6.5. Conclusion

Low net primary productivity and high heterotrophic respiration fluxes resulted in a net carbon loss during the first four years of tree growth at the short-rotation plantation at Zwijnaarde. The carbon loss through decomposition outweighed the increase in NPP with time. While NPP was linked with stand age, climatic conditions, especially during the leafed period, were the primary drivers of interannual variability in soil CO_2 efflux. It can be expected, however, that the plantation will become a sink for carbon within the next years.

The assessed carbon loss from the plantation depended on the approach used for the calculations, with the ecological approach indicating a larger source strength. The assessment of the carbon balance of this short-rotation forestry plantation can be improved with addition of information on understorey contribution to carbon fluxes, even as herbivory losses. A better insight in the contribution of root respiration to total soil CO_2 efflux (ecological approach), or production of root litter (mass-balance approach) can also optimize the calculations. The mass-balance method could further be improved by taking more soil samples, in vertical as well as in horizontal directions. Whatever method is used, largest uncertainties are linked with belowground processes.

PART B.

BELGIAN FOREST ECOSYSTEMS

Chapter 7

Above- and belowground carbon stocks of two mixed deciduous forest stands in the Aelmoeseneie experimental forest

Adapted from : Vande Walle, I., S. Mussche, R. Samson, N. Lust and R. Lemeur. 2001. The above- and belowground carbon pools of two mixed deciduous forest stands located in East-Flanders (Belgium). Ann. Forest Sci. 58:507-517.

Abstract

Carbon (C) storage was studied in both an oak-beech and an ash stand located in the 80 years old Aelmoeseneie experimental forest (Gontrode, East-Flanders, Belgium). The total carbon stock amounted to 324.8 t C ha^{-1} in the oak-beech stand and 321.4 t C ha^{-1} in the ash stand. In the oak-beech stand, 41.5 % of the total C was found in the soil organic matter, 11.0 % in the litter layer and 47.5 % in the vegetation. In the ash stand, the soil organic matter contained 53.0 % of the total C stock, the litter layer only 1.0 % and the vegetation 46.0 %. Most vegetation carbon was found in the stems of the trees (51.1 % in the oak-beech and 58.7 % in the ash stand). Although total carbon storage appeared to be very similar in both forest stands, distribution of carbon over the different ecosystem compartments was related to species composition and site characteristics.

7.1. Introduction

Changes in land use and exploitation of fossil fuels caused an increase of the atmospheric CO_2 concentration from 280 ppm in the middle of the 19th century to 360 ppm at the moment (Foody *et al.* 1996, NOAA 2000). This increase, together with the rise of the global mean air temperature, will most probably continue in the 21st century. A more complete insight in the global carbon (C) cycle is indispensable to understand the causes and the consequences of the so-called greenhouse effect. The carbon cycle is strongly related to the carbon balance of terrestrial ecosystems. Forest ecosystems are the most important carbon pools on earth. Although only 28 % of the land surface is covered with forests, these forests contain more than 46 % of the carbon stored in the terrestrial biosphere (Bolin and Sukumar 2000). Moreover, forests store carbon for long time periods (Nabuurs and Mohren 1994).

The Ministerial Conference on the Protection of Forests in Europe (16th-17th June 1993, Helsinki, Finland) suggested to make an inventory of the biomass stored in the wood and forest stocks, in order to compare carbon stored in, and carbon taken up by, forests with the amount of CO_2 emitted by fossil fuel combustion. At the Conference of Kyoto (1997) most industrial countries agreed on the reduction of the CO_2 exhaust. On the other hand, more and more attention is given to carbon fixation in order to extract CO_2 from the atmosphere (Schimel 1995). A first step to assess the importance of forests in the global C cycle is to estimate the carbon stocks in these ecosystems.

Within forest ecosystems, the soil seems to be the largest carbon pool : approximately 60 to 70 % of the carbon in forests is stored as organic material in the soil (Zinke 1984, King *et al.* 1992, Harrison *et al.* 1995). The carbon content of forest soils increases with increasing longitude and altitude (Mellilo and Gosz 1983, Harrison *et al.* 1995, Brown 1996). Also climate, topography and texture are important factors related to the soil C content of forests (Post 1982, Schimel *et al.* 1994). In general, the accumulation of organic material in the soil increases with decreasing temperature, increasing precipitation, decreasing evapotranspiration-to-precipitation ratio and increasing clay content (Post 1982, Zinke 1984, Landsberg and Gower 1997).

Forests display a litter layer on top of the mineral soil. This litter layer is an important pool of nutrients and organic material (Gosz *et al.* 1976). The quantity and quality of the litter determine the decomposition rate. This decomposition defines the availability and mobility of essential elements, and as such, it influences the functional processes in the forest ecosystems (Vogt *et al.* 1986, Tietema *et al.* 1991). Different types of litter are distinguished (Jabiol *et al.* 1995) : mull, mor and moder. Mull humus is characterized by an intensive microbial activity : degradation of the organic material goes fast and this material is strongly mixed with the underlying mineral soil. Mull humus layers are usually very thin. Mor humus has a low microbial activity, which implements a slow degradation of the organic material and no mixture with the mineral soil. In the mor humus layer, three sublayers can be distinguished : an O^L-layer (litter layer) containing fresh, undegraded litter, an O^F-layer (fermentation layer) existing of fragmented, half degraded litter and an O^H-layer (humification layer) with humidified and compacted organic material. Moder humus has similar characteristics as mor humus, although there is some bioturbation. Both mor and moder humus types reduce the fertility of the ecosystem as many nutrients are immobilized in the accumulated litter (Remacle 1977, Delecour and Weissen 1981, Pastor and Bockheim 1984).

Dead wood is a structural and functional element in a forest ecosystem (Franklin 1981, Harmon and Hua 1991). Besides its functioning as a microhabitat for fauna and flora, it also influences water, carbon and nutrient cycles (Kimmins 1992, McCarthy and Bailey 1994). Stand age, location, tree species and management practices determine the amount of dead wood in a forest. In an undisturbed, old forest stand, the rate of dieback and the rate of decomposition are in steady state (Harmon *et al.* 1986, Tyrrell and Crow 1994). However, little information is available on the distribution and abundance of dead wood in forest ecosystems.

The carbon stocked in the tree layer varies widely, from 23 to 82 % of the total ecosystem carbon stock (Duvigneaud 1984, Nabuurs and Mohren 1994, Usol'tsev and Vanclay 1995), and depends highly on the tree species. The tree compartment itself can be split up in an above- and belowground part, and further in leaves, branches and stems and fine and coarse roots, respectively. Stand age and site characteristics seem to play an important role in the distribution of carbon over the different compartments (Vanninen *et al.* 1996). In forest stands on poor and dry soils, more carbon is allocated to the roots (Schulze 1982). The ratio fine roots-to-leaf biomass increases with the age of the stand, while the relative contribution of the leaves and fine roots to the total biomass decreases. The relative importance of the woody tissues on the other hand increases with stand age (Vanninen *et al.* 1996).

The objectives of this chapter were to synthesize and compare data on the carbon stocks in two mixed deciduous forest types in Belgium : an oak-beech and an ash stand. Both stands have a well-developed shrub layer. The age of the trees and the climate are equal for both stands. Main differences are the dominating tree species and the soil type.

7.2. Material and methods

7.2.1. Site description

This study was conducted in a mixed deciduous forest, called the Aelmoeseneie forest. This forest is property of Ghent University and is mainly used for educational and scientific purposes. It is located near the village of Gontrode (50°58' N, 3°49' E), which is situated 15 km south of Ghent (East-Flanders, Belgium). The oldest historical documents referring to this forest date from the year 864. After 4 years of overfelling during World War I (1914-1918), a replantation was necessary to compensate for the removed wood. Therefore, most of the mature trees are now about 80 years old. The total forested area covers 28 ha. The elevation of the forest soil surface varies between 11 and 21 m a.s.l. The area is gently sloping northwards. The main part of the forest is an individual mixture of mainly broad-leaved species (Samson *et al.* 1996, Janssens *et al.* 1998).

Since 1990, a zone of 1.83 ha was fenced and closed for the public. The fenced area is used for intensive scientific research. This experimental zone comprises two different forest types : an oak-beech stand (1.06 ha) and an ash stand (0.77 ha). As during the replantation of the forest the difference in soil type (Vandendriessche *et al.*

1993) was taken into account when choosing the main tree species, the ash stand is situated on the lower part of the forest. Both the species composition and the main stand inventory data are given in Table 7.1, as well as the maximum leaf area index (LAI) of the tree and the shrub layer, the humus and soil type. Differences in chemical soil characteristics of both stands are published by Vandendriessche *et al.* (1993). Mean annual air temperature (measured during the period 1984-1993) is 10.1 °C, with 2.8 °C in the coldest month (January) and 17.4 °C in the warmest month (August). Annual precipitation is 791 mm on average. Mean dates of first and latest frost are 10th November and 13th April, respectively, with a mean of 47 frost days per year (Samson *et al.* 1996).

Table 7.1. *Main stand characteristics of the two stands in the experimental zone of the Aelmoeseneie forest; BA : basal area, $DBH_{1.3}$: diameter at breast height (1.3 m) and LAI : leaf area index*

		Oak-beech stand	Ash stand
Species composition		% of BA	% of BA
Pedunculate oak (*Quercus robur* L.)		48.7	10.6
Common beech (*Fagus sylvatica* L.)		26.6	1.3
Common ash (*Fraxinus excelsior* L.)		4.0	59.5
Japanese larch (*Larix kaempferi* (Lambert) Carr.		12.5	4.5
Common maple (*Acer pseudoplatanus* L.)		3.0	15.8
Rowan (*Sorbus aucuparia* L.), hazel (*Corylus avellana* L.), Alder buckthorn *(Frangula alnus* Mill.*),* regeneration of maple (all together)		5.2	9.3
Stand inventory data(1)			
Density (trees ha^{-1})		345	403
Mean $DBH_{1.3}$ (cm)		26.1	26.9
BA (m^2 ha^{-1})		26.6	30.8
Standing wood volume (m^3 ha^{-1})		301	328
Maximum LAI (m^2 m^{-2}) (2)			
Tree layer		5.1	2.5
Shrub layer		0.4	2.0
Total		5.5	4.5
Humus type		Moder	Mull
Soil type	**(FAO classification)**	Dystric podzoluvisol	Dystric cambisol
	(USDA classification)	Haplic glossudalf	Thapto glossudalfic, aquic, dystric eutrochept

(1) *see Vande Walle* et al. *1998a;* (2) *leaf fall method, see Mussche 1997*

In 1993, a measuring tower was constructed in the middle of the scientific zone, at the common border of the two forest stands. This 35 m high tower, which contains five horizontal working platforms, gives direct access to the crown of the main tree species : oak, beech and ash. Both forest stands are continuously used for integrated scientific research, such as physiological, biogeochemical and soil science studies and modelling activities. Furthermore, two level II observation plots of the European

Programme for Intensive Monitoring of Forest Ecosystems are installed in the scientific zone. The results discussed in this chapter were obtained during the Belgian research programme BELFOR, which analysed the biogeochemical cycles in a series of Belgian model forests (Vande Walle and Lemeur 2001).

7.2.2. Mineral soil

Soil samples were taken in both the oak-beech and the ash stand to determine the carbon content of the mineral soil (up to 1 m depth). In each stand, ten randomly chosen transects of 25 m length were sampled at six points, each 5 m separated from each other (n = 60). A soil core was used to take samples at different depths : 0-5 cm, 5-15 cm, 15-50 cm and 50-100 cm. After drying, sieving (mesh of 2 mm) and grinding, the method of Walkley and Black (Nelson and Sommers 1996) was used to determine the carbon concentration (g C g^{-1} dry soil). It has been reported that this method underestimates the real carbon concentration, and that the results have to be multiplied by 4/3, because only 75 % of the organic C in the soil is oxidized by this method (Nelson and Sommers 1996). Total carbon content (t C ha^{-1}) in each soil horizon was calculated from the carbon concentration, the bulk density (Vandendriessche *et al.* 1993) and the layer thickness. Normal distribution was checked for each soil layer (Kolmogorov-Smirnov test).

7.2.3. Litter layer

In both stands, the humus layer was collected at different spots of 0.25 m², at the same sampling points (n = 60) and at the same moment (May 1996) as used for the mineral soil sampling (see § 7.2.2). The O^L-, O^F- and O^H-layers were separated for the oak-beech stand. The material was weighed and dried (80 °C, 48 h). The carbon content of each sample was determined by loss-on-ignition (LOI). The results obtained this way were then used to calculate the mean C content of each layer.

In both stands of the Aelmoeseneie experimental forest, dead wood was collected on 5 randomly chosen plots of 100 m^2 (April 1996) following the methodology described by Janssens *et al.* (1998). As both stands have been managed for a long time, only a few dead trees are present. Therefore, all dead wood can be considered as lying on the forest floor. All dead wood with a diameter < 2.5 cm was sampled on one subplot (1 m^2) per plot. This subplot was extended to 25 m^2 for the diameter class 2.5-5.0 cm. The entire plot (100 m^2) was used for collecting dead wood with a diameter > 5.0 cm. The material collected was then weighed and dry weight (80 °C, until constant weight) was determined as well. The carbon concentration of the wood was detected by LOI. Based on total dry mass and C concentration, total C storage in the dead wood was calculated.

7.2.4. Carbon stocks in the vegetation

For all compartments of the vegetation, a carbon concentration of 50 % (on dry mass base) was assumed (Matthews 1993).

7.2.4.1. Aboveground carbon stocks

The shrub layer is a carbon pool that is neglected in many carbon sequestration studies. However, we wanted to calculate the amount of carbon in this layer too, in order to obtain a more complete insight in the total carbon stock in the two Aelmoeseneie stands. Ten square plots of 25 m^2 were randomly selected in each stand. In each plot, the complete aboveground shrub layer was removed (January 1996) and dried (80 °C, until constant weight). Total C storage in the shrub layer was then determined, assuming a carbon concentration of 50 % (see above).

In January 1997, all trees (diameter at breast height DBH > 7 cm) were numbered and circumferences at breast height (CBH) and tree heights were measured. Twelve oak trees and six ashes were cut down. For both species, a tree with the mean stem circumference (oak : 96.0 cm, ash : 111.0 cm), the model trees of Hohenadl (mean circumference ± st. dev.; st. dev. for oak : 26.2 cm, for ash : 32.4 cm) and some trees with an intermediate circumference were chosen. Stem volumes of these trees were calculated, based on mensuration data of stem discs of one meter length (Janssens *et al.* 1998). Following relationships between stem volume (V) and CBH were found :

$$V_{oak} = 0.000039 \cdot (CBH)^{2.200} \qquad (R^2 = 0.97) \qquad \mathbf{(7.1)}$$

$$V_{ash} = 0.000200 \cdot (CBH)^{1.853} \qquad (R^2 = 0.96) \qquad \mathbf{(7.2)}$$

with stem volume expressed in m^3 and CBH in cm. Stem volumes of beech, maple and larch were calculated based on the tables of Dagnelie *et al.* (1999) with stem circumference and tree height H as inputs :

$$\begin{aligned} V_{beech} = \; & -0.015572 + 0.0009231 \cdot CBH - 0.0000071407 \cdot CBH^2 - \\ & 0.000000077179 \cdot CBH^3 - 0.0013528 \cdot H + 0.0000040364 \cdot CBH^2 \cdot H \end{aligned} \qquad \mathbf{(7.3)}$$

$$\begin{aligned} V_{maple} = \; & 0.010343 - 0.0014341 \cdot CBH + 0.000034521 \cdot CBH^2 - \\ & 0.00000013053 \cdot CBH^3 + 0.00077115 \cdot H + 0.0000030231 \cdot CBH^2 \cdot H \end{aligned} \qquad \mathbf{(7.4)}$$

$$\begin{aligned} V_{larch} = \; & -0.03088 + 0.0014885 \cdot CBH - 0.0000049257 \cdot CBH^2 - \\ & 0.00000012313 \cdot CBH^3 - 0.0011638 \cdot H + 0.0000041134 \cdot CBH^2 \cdot H \end{aligned} \qquad \mathbf{(7.5)}$$

with V expressed in m^3, CBH in cm and H in m.

Total stem volume was multiplied by wood density of the respective species to calculate the total dry weight of the stems of all trees. Wood densities on a dry mass base were 500 kg m^{-3} for oak, 523 kg m^{-3} for ash, 566 kg m^{-3} for young beeches (DBH < 45 cm) and 550 kg m^{-3} for old beeches (CBH ≥ 45 cm) (Schauvliege 1995).

These values were based on the fresh volume. Wagenführ and Scheiber (1989) reported wood densities of 590 kg m^{-3} for maple and 550 kg m^{-3} for larch.

Regression equations between stem circumference and dry weight of the leaves on the one hand and dry weight of the branches on the other hand were established for oak, beech and ash (Janssens *et al.* 1998). These equations were used to calculate the dry weight of the leaves and the branches. As for maple and larch, no regression equations were established, stem biomass was considered as containing 75 % of the total biomass, 24 % was dedicated to the branches and 1 % to the foliage (Nabuurs and Mohren 1994). Multiplying the dry weight by 0.5 (see before) gave the total amount of carbon stored in the leaves and the branches.

7.2.4.2. Belowground carbon stocks

For two of the twelve oak trees (CBH of 86 and 97 cm) which were used to assess the aboveground carbon stocks, the coarse root systems were excavated in order to collect information on the belowground carbon stock. All coarse roots (diameter > 5 mm) were collected and weighed. Samples were dried (80 °C, until constant weight) to determine total dry weight of the root system. The coarse root system of the smallest tree amounted to 16.3 % of the total tree biomass, compared to 17.6 % for the larger tree. Duvigneaud (1984) found a similar root fraction of 17.0 % in a *Querceto-Coryletum* of 80 years. Literature values of root fractions were used to assess the carbon stored in the coarse roots of the other species : 16.8 % for beech, 16.3 % for ash and 17.0 % for maple and larch (Duvigneaud 1984).

During July and August 1997, soil samples were taken to study the vertical distribution of the fine roots. The used root auger had a total volume of 729 cm^3, and a length of 15 cm. Five depths were studied : 0-15, 15-30, 30-45, 45-60 and 60-75 cm. In the oak-beech stand, samples were taken at 7 locations, while in the ash stand 5 locations were sampled. Fine roots (diameter < 5 mm) were extracted, dried (60 °C, 48 h) and weighed. A more detailed description of the experimental set-up and the sampling strategy can be found in Vande Walle *et al.* (1998b).

7.3. Results and discussion

7.3.1. Mineral soil

Table 7.2 gives the mean carbon content (mg C cm^{-3} soil) of the mineral soil layers in both stands. In both stands, there was a clear decrease in carbon content with increasing soil depth. ANOVA analysis was applied to compare carbon contents in the different layers of both stands. No significant differences between the two stands could be found for the upper two layers (0-5 and 5-15 cm). For the lower layers (15-50 and 50-100 cm), the carbon content was always significantly higher ($p < 0.05$) in the ash stand than in the oak-beech stand. Previous studies have shown that in the ash stand, an extreme diversity of earthworms is present (Muys 1993). As those earthworms continuously mix the organic material with the mineral soil, the bioturbation of the soil is more intense in the ash stand, resulting in a more equally distribution of the organic material in this stand than in the oak-beech stand.

Table 7.2. *Mean carbon content (mg C cm^{-3} soil) of each mineral soil layer in the oak-beech and the ash stand (n = 60) with indication of significant differences between the stands (n.s. : not significant; * significant at* p *< 0.05)*

	Carbon content (mg C cm^{-3} soil)		
Depth (cm)	**Oak-beech stand**	**Ash stand**	
0-5	84.0	71.6	n.s.
5-15	34.7	38.3	n.s.
15-50	11.8	17.2	*
50-100	3.4	7.2	*

It seems that in both stands, large amounts of carbon are stored in the mineral soil (Table 7.3 : oak-beech : 135.0 t C ha^{-1}, ash : 170.5 t C ha^{-1}). Dutch investigators found similar, but slightly lower values ranging from 102 to 122 t C ha^{-1} for comparable forest ecosystems (Nabuurs and Mohren 1993b) while Janssens *et al.* (1999) found a carbon content of 114.7 t ha^{-1} over a depth of 1 m in a Belgian Scots pine forest. The forest they examined was, however, situated on a sandy soil. In such soils, carbon is less immobilized by the formation of organo-mineral-complexes than in loamy and clayey soils, as is the case in the Aelmoeseneie forest. Soil texture can partly explain the differences in carbon storage in the mineral soil.

7.3.2. Litter layer

In the holorganic horizon of the oak-beech stand, an O^{L}-, O^{F}- and O^{H}-layer could be distinguished. The amount of carbon stored in these layers were 0.6, 17.2 and 15.4 t C ha^{-1}, respectively. The O^{L}-layer in the ash stand only contained 0.1 t C ha^{-1}, and an O^{F}- and O^{H}-layer were lacking.

The litter formed in the ash stand decomposes very rapidly. The above-mentioned bioturbation causes the mixing of the organic material with the mineral soil. As such, almost no litter layer is found in the ash stand. The O^{F}- and O^{H}-layer of the oak-beech stand are well developed. Most of the carbon stored in the holorganic horizon is stored in these two layers. Janssens *et al.* (1999) found a storage of 25.5 t C ha^{-1} in the humus layer of a Belgian Scots pine forest. This is a value close to the 33.2 t C ha^{-1} which was found for the oak-beech stand. Micro-organisms, which have a C:N ratio of 6 to 16, prefer digestion of litter with a low C:N ratio (< 20) in order to satisfy their nitrogen (N) needs. The C:N ratio of the fresh ash litter in the Aelmoeseneie forest is 24, while the values for oak and beech are 29 and 42, respectively (Muys 1993). Due to its lower C:N ratio, the ash litter is faster degraded than the oak and the beech litter. The slow degradation of the dead biomass in the oak-beech stand causes therefore an accumulation of litter, which itself decreases the aeration, and, hence, has a negative effect on the speed of the litter degradation.

The mean C concentration of the dead wood was 48.9 % of dry weight. In Table 7.3, the C content (t C ha^{-1}) of the different diameter classes is presented for both stands. In the ash stand (3.0 t C ha^{-1}), more C was found in the dead wood than in the oak-beech stand (2.5 t C ha^{-1}). This difference is only due to the dead wood with a diameter > 5 cm. However, the difference was not statistically significant (t-test).

Table 7.3. *Carbon stock (t C ha^{-1}) of the soil, the litter and the vegetation compartment of the oak-beech and the ash stand of the Aelmoeseneie forest*

	Carbon stock (t C ha^{-1})	
Compartment	**Oak-beech stand**	**Ash stand**
Soil		
Organic material		
0-5 cm depth	*42.0*	*35.8*
5-15 cm depth	*34.7*	*38.3*
15-50 cm depth	*41.3*	*60.1*
50-100 cm depth	*16.8*	*35.8*
Total organic material	134.8	170.0
Dead roots	0.2	0.5
	135.0	**170.5**
Litter		
Holorganic horizon	33.2	0.1
Dead wood		
< 2.5 cm diameter	*1.6*	*1.6*
2.5 - 5 cm diameter	*0.6*	*0.6*
> 5 cm diameter	*0.3*	*0.8*
Total dead wood	2.5	3.0
	35.7	**3.1**
Vegetation		
Leaves		
Trees	*1.8*	*0.7*
Shrubs	*0.2*	*0.6*
Total leaves	2.0	1.3
Branches trees	42.5	26.9
Stems trees	78.7	86.9
Branches and stems shrubs	2.4	4.3
Coarse roots	25.1	22.8
Fine roots	3.4	5.8
	154.1	**148.0**
Total	**324.8**	**321.6**

Other investigators (Koop 1983, Buiting and ten Tuynte 1997) found dead wood stocks accounting for 10 to 30 % of the total aboveground biomass of forests. Values found here were much lower : 2.0 and 2.5 % for the oak-beech and the ash stand, respectively. This is caused by the removal of dead wood in the Aelmoeseneie forest for many decades. As, in view of a new forest management policy, dead wood is no longer removed since about 10 years, an increase of this dead wood carbon stock can be expected in the future.

7.3.3. Carbon stocks in the vegetation

7.3.3.1. Aboveground carbon stocks

Although the shrub layer showed a high diversity and was well developed in both stands (see Table 7.1), the total amount of carbon stored in this shrub layer was relatively small, *i.e.*, 2.6 t C ha^{-1} in the oak-beech stand and 4.9 t C ha^{-1} in the ash stand. In comparison with the total aboveground carbon stock, only 2.1 % was stored in the shrub layer of the oak-beech stand, and 4.1 % in the ash stand. These are small fractions, considering the important contribution of the shrub layer to the overall leaf area index (LAI), which was 7.3 % in the oak-beech stand and 44.4 % in the ash stand (Table 7.1). Although small, this stock should not be neglected. Indeed, the shrub layer in the ash stand contains even more carbon than the litter layer.

The amount of carbon stored in the aboveground tree biomass (leaves, branches and stems) totalled 123.0 t C ha^{-1} in the oak-beech and 114.5 t C ha^{-1} in the ash stand (Table 7.3). The partitioning over the different compartments was, however, different in the two stands. For the oak-beech stand, 1.5 %, 34.5 % and 64.0 % of the C is stored in the leaves, branches and stems, respectively. This is in contrast with the corresponding values of 0.6 %, 23.5 % and 75.9 % for the ash stand (Table 7.3). The larger relative amount of beeches present in the oak-beech stand explains the difference in carbon distribution, as beech trees contain as much carbon in their branches as in the stem wood. An interesting observation was the fact that beech accounted for 37.8 % of the carbon stored in the aboveground biomass of the oak-beech stand, while the beech trees only represented 26.6 % of the total basal area (Table 7.1).

Carbon stocks in the aboveground biomass of the Aelmoeseneie forest are comparable with values found in previous studies (Duvigneaud 1984, Nabuurs and Mohren 1994, Usol'tsev and Vanclay 1995, Janssens *et al.* 1999, Santa Regina and Tarazona 1999). Dutch investigators (Nabuurs and Mohren 1993a) showed that the carbon stock in living biomass is largest for beech forests, a conclusion comparable to results found here.

7.3.3.2. Belowground carbon stocks

The total amount of carbon stored in the coarse roots added up to 25.1 t C ha^{-1} in the oak-beech stand and 22.8 t C ha^{-1} in the ash stand, as is listed in Table 7.3. Fig. 7.1 illustrates clearly the different vertical distribution of fine roots (diameter < 5 mm) in the mineral soil of the two stands. In the upper two layers, much more fine roots were found in the ash stand than in the oak-beech stand : almost fourfold in the upper layer (3.0 compared to 0.8 t C ha^{-1}), and 85 % more in the second layer (1.3 compared to 0.7 t C ha^{-1}). This difference is mainly due to the well-developed shrub layer in the ash stand as these shrub species are mostly rooted in the upper layers of the forest soil. ANOVA analysis showed that the upper soil layer of the ash stand contained significantly more fine roots than all other layers.

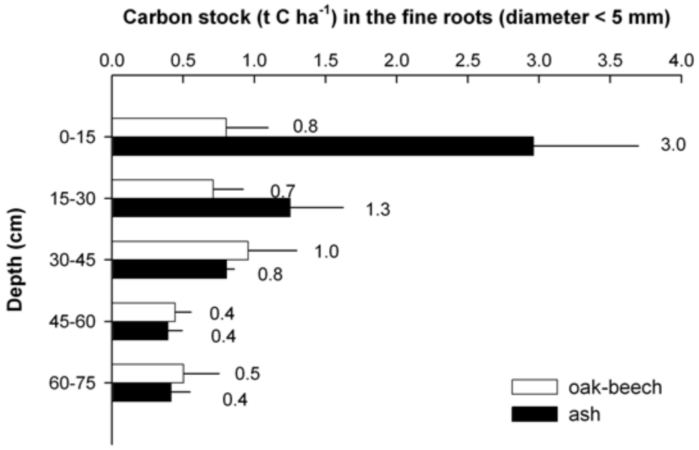

Figure 7.1. *Vertical distribution of the carbon stock (t C ha^{-1}) in the fine roots (diameter < 5 mm) in the oak-beech and the ash stand of the Aelmoeseneie forest; error bars indicate one standard error of the mean.*

Total carbon stocks in the living fine roots amounted to 3.4 t C ha^{-1} in the oak-beech stand, compared with 5.8 t C ha^{-1} in the ash stand (Fig. 7.1 and Table 7.3). Much less dead roots were found, *i.e.*, 0.2 t C ha^{-1} and 0.5 t C ha^{-1}, for the oak-beech and the ash stand, respectively (Table 7.3).

The ratio of fine roots to leaves (both expressed in t C ha^{-1}) was 1.7 in the oak-beech stand, and 4.5 in the ash stand. It was shown that the LAI in the oak-beech stand was 22 % higher than in the ash stand (Table 7.1). When expressed as biomass (t C ha^{-1} in the leaves), the oak-beech stand contained 54 % more carbon in the leaves than the ash stand (Table 7.3). This means that the mean specific leaf area (SLA) was lower (13.7 m^2 leaf kg^{-1} DM) in the oak-beech than in the ash stand (17.2 m^2 leaf kg^{-1} DM). This higher SLA in the ash stand increases the relative importance of the carbon storage in the fine roots compared to the leaves. Janssens *et al.* (1999) found a value of 0.6 for the ratio of fine roots to needles. In the Scots pine forest they studied there was, however, no shrub layer present, causing a lower amount of fine roots. On the other hand, they found 3.0 t C ha^{-1} to be stored in the needles, which is far more than the values found here.

7.3.4. Overview of the carbon stocks

The total carbon stock present in both stands (Table 7.3) was rather similar, *i.e.*, 324.8 t C ha^{-1} in the oak-beech stand, and 321.6 t C ha^{-1} in the ash stand. The distribution of carbon over the different compartments (Fig. 7.2) was less comparable. The most striking difference was found in the litter layer : while in the oak-beech stand, this layer contained 11.0 % of the total carbon, it only accounted for 1.0 % of the total C stock in the ash stand. On the other hand, the fraction of carbon stored in the mineral soil was much higher in the ash stand (53.0 %) than in the oak-beech stand (41.6 %). The contribution of the living phytomass was comparable : 47.4 % in the oak-beech and 46.0 % in the ash stand. Less than one fifth of all carbon stored in the vegetation was found in the belowground organs (fine and coarse roots) : 18.5 % in the oak-beech stand, and 19.3 % in the ash stand. Partitioning of carbon over living biomass, litter layer and mineral soil in the Aelmoeseneie forest was in agreement with the results reported by Nabuurs and Mohren (1994).

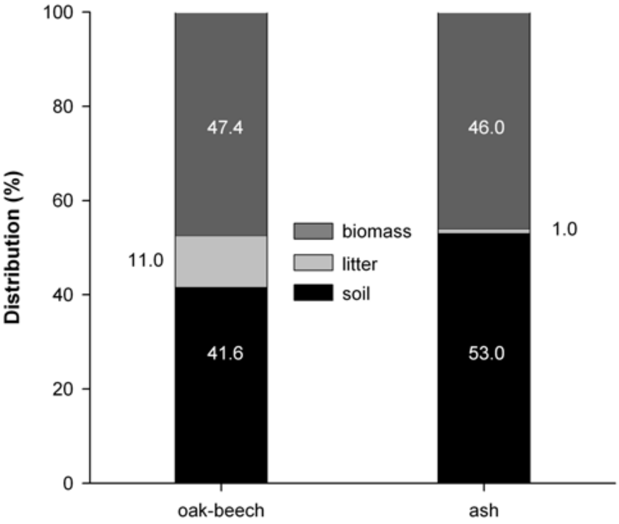

Figure 7.2. *Carbon in the biomass, the litter and the soil compartment of the oak-beech and the ash stand as a percentage of the total amount of C stored in these stands*

The contributions of the living (47.4 % in the oak-beech and 46.0 % in the ash stand) and the non-living compartments (52.6 % and 54.0 %) were very similar in both stands. As such, one can conclude that although the species composition of the forest stands and the soil characteristics are different, the total amount of carbon stored in the ecosystem was very similar. This was also true for the distribution between living and non-living compartments. It seems that for forest ecosystems of different composition but situated in identical climatic regions, their carbon storage will not change very much. This conclusion is confirmed by the results of Janssens *et al.* (1999), obtained for a Scots pine forest, situated in the same climatic region. Although a different main tree species and another soil type, the pine forest yielded comparable values of 58.0 % for the total carbon in the non-living compartment and 42.0 % for the living carbon stock.

7.4. Conclusion

The study revealed that both the oak-beech and the ash stand have important carbon stocks. The total amount of carbon stored (324.8 and 321.6 t C ha^{-1}, respectively) and the distribution between living and non-living compartments seemed to be very similar. The partitioning of carbon over the different compartments of the ecosystem is highly related to the tree species and the site characteristics. Leaves and branches were proportionally more important in the oak-beech stand than in the ash stand. Due to rapid degradation of fresh litter, the holorganic horizon represented a much smaller carbon stock in the ash stand than in the oak-beech stand. On the other hand, more intense bioturbation caused a better mixture of the organic material with the mineral soil, which, therefore, contained more carbon in the ash stand than in the oak-beech stand. The results presented in this chapter form the basis for the understanding of the carbon cycle in the experimental forest Aelmoeseneie. Eventually, these data are also valuable for the validation of dynamic vegetation models used to assess carbon storage in temperate forest ecosystems.

Chapter 8

Carbon sequestration in the Aelmoeseneie experimental forest from 1991 to 2005

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Abstract

As forests are important components of the global carbon (C) cycle, getting more insight in the carbon uptake capacity of these ecosystems is needed to assess the role they can play in mitigating climate change. In this study, the net ecosystem productivity (NEP) was assessed for two mixed deciduous stands, an oak-beech (OB) and an ash (AS) stand, of the 80 years old Aelmoeseneie forest (Gontrode, Belgium). As three consecutive full inventories of the OB and the AS stand were available (end of 1990, 1997 and 2005), carbon sequestration in the living biomass and in the dead wood pool was calculated by tracking individual trees, for two multiyear periods (1991-1997 and 1998-2005). Data on litter layer and soil carbon stock changes were also available.

Increments of total solid wood (stems and branches with a circumference > 22 cm) amounted to 6.32 and 5.71 m^3 ha^{-1} $year^{-1}$ on average in the period 1991-1997, for OB and AS, respectively. The increment was slightly higher in the period from 1998 to 2005, amounting to 6.81 and 6.89 m^3 ha^{-1} $year^{-1}$, for OB and AS, respectively. In the first period (1991-1997), the living biomass sequestered on average 2.43 ± 0.27 t C ha^{-1} $year^{-1}$ in the oak-beech stand, and 2.19 ± 0.21 t C ha^{-1} $year^{-1}$ in the ash stand. Carbon sequestration in living biomass was assessed at 2.63 ± 0.27 t C ha^{-1} $year^{-1}$ for OB and 2.64 ± 0.21 t C ha^{-1} $year^{-1}$ for AS in the period from 1998 to 2005. This higher carbon sequestration was most probably due to higher mean annual air temperatures combined with a higher total annual precipitation in the second period. Differences in atmospheric elemental deposition were not significant for the two periods. Expressed as relative growth rates, the oak-beech stand seemed to be slightly more productive than the ash stand.

Mean annual NEP equalled 3.33 ± 2.30 and 3.57 ± 2.30 t C ha^{-1} $year^{-1}$ in the oak-beech stand for the periods 1991-1997 and 1998-2005, respectively. For the ash

stand, NEP values were 2.25 ± 1.81 t C ha^{-1} $year^{-1}$ (1991-1997) and 2.97 ± 1.81 t C ha^{-1} $year^{-1}$ (1998-2005). From this study, it was obvious that both stands of this mid-successional mixed deciduous forest were considerable carbon sinks during the period 1991 to 2005.

8.1. Introduction

Increasing awareness of rising atmospheric CO_2 concentration and its major influence on climate and vegetation functioning has strengthened the interest in forest carbon (C) balance studies. As the Kyoto Protocol asks strategies to balance industrial CO_2 emissions, detailed knowledge of the carbon cycle in terrestrial ecosystems is needed to reveal the role these ecosystems can play in climate change mitigation. The net carbon exchange by a forest ecosystem over a given time period is termed net ecosystem productivity (NEP), and equals the difference between gross primary productivity (GPP) and total ecosystem respiration (TER) (Curtis *et al.* 2002, Ehman *et al.* 2002). As such, NEP captures a variety of processes and feedbacks associated with carbon cycling between the atmosphere, vegetation and soil pools (Black *et al.* 2005). NEP, being the result of a delicate balance between C uptake (photosynthesis) and C loss (respiration), shows a strong diurnal, seasonal and interannual variability (Valentini *et al.* 2000, Zha *et al.* 2004). Annual NEP values vary widely between forest ecosystems, due to differences in climatic conditions, soil fertility, management practices, species composition, stand age and site history (Janssens *et al.* 2001b).

Traditionally, inventories of carbon stocks were used to assess the carbon balance of a forest stand (Black *et al.* 2005). This mass-balance approach (NEP_{mass}) is based on the quantification of changes in C stocks in living (ΔC_b) and dead (ΔC_{dead}) biomass and in the soil (ΔC_{soil}) compartment between sampling times :

$$NEP_{mass} = \Delta C_b + \Delta C_{dead} + \Delta C_{soil} \qquad \textbf{(8.1)}.$$

As ΔC_{soil} is generally very small, it is difficult to detect against the considerably higher background soil carbon stock. An additional difficulty arises from the high degree of spatial heterogeneity in soil C stocks (Baldocchi 2003). In some cases, ΔC_{soil} is not even measured or is assumed to be zero (Kolari *et al.* 2004). A second method for assessing ecosystem balances is based on the estimation of the net C balance from heterotrophic and autotrophic processes (Ehman *et al.* 2002, Black *et al.* 2005). This ecological approach (NEP_{eco}) includes an assessment of annual CO_2 losses from the ecosystem, as well as the net gain from autotrophic processes, and is mainly based on flux measurements :

$$NEP_{eco} = NPP - R_h \qquad \textbf{(8.2)}$$

where NPP is net primary productivity and R_h is heterotrophic respiration due to microbial decomposition of soil organic matter, litter, aboveground detritus and respiration by consumers. In recent years, the eddy covariance (EC) technique has emerged as an alternative way to assess ecosystem carbon exchange due to improvements of instrument techniques (Curtis *et al.* 2002). This method is based on knowledge in the fields of fluid dynamics and micrometeorology, and relies on the sampling of turbulent air motions to determine the net difference of material moving across the canopy-atmosphere interface (Baldocchi *et al.* 2003). As measurements are made directly at the ecosystem scale, the EC method integrates numerous biological and physical processes affecting the carbon cycle.

National carbon accounting methodologies for reporting to the Intergovernmental Panel on Climate Change (IPCC) and the Kyoto Protocol are mostly based on forest inventory data and biometric approaches (Black *et al.* 2005). Forest biomass inventory studies rely on allometric relationships and biomass expansion factors *s.l.* (*i.e.*, wood density, biomass expansion factors *s.s.* and carbon content) to scale incremental changes in diameter at breast height (DBH) to changes in carbon stocks at plot and landscape scales. Ideally, carbon stock changes in the shrub layer, the dead wood compartment and the forest soil should be taken into account to assess NEP.

In this study, the net ecosystem productivity was determined for two stands of the experimental zone in the deciduous forest Aelmoeseneie, situated at Gontrode (Belgium). These stands, an oak-beech (OB) and an ash (AS) stand, were both planted in 1920 and were growing under identical meteorological conditions. Moreover, detailed information on atmospheric elemental deposition has been gathered since 1992 in both stands, in the framework of the European level II network. The mass-balance approach was used to determine NEP_{mass} (eq. 8.1). Three full inventories of the experimental zone were conducted at the end of 1990, 1997 and 2005. As such, ΔC_b during the periods 1991 to 1997 and 1998 to 2005 was calculated for the oak-beech and for the ash stand. Data on changes in dead wood stocks and in litter and soil carbon stocks were also available. Error propagation was applied to assess the uncertainty on the NEP_{mass} values obtained for both periods (1991-1997 and 1998-2005) and for both stands. Results from this investigation can help to optimize carbon stock (change) estimates of Flemish forest ecosystems, as these estimates suffer from a lack of wood increment data for Flanders (see Chapter 10).

8.2. Material and methods

8.2.1. Site description

This study was conducted in the mixed deciduous forest Aelmoeseneie, situated in Gontrode, 15 km south of the city of Ghent (50°58' N, 3°49' E). The first time the forest Aelmoeseneie was mentioned in written documents was in 864 (Samson *et al.* 1996). On the map of Ferraris, dating back to 1775, the region was still indicated as being forested. Due to heavy fellings during World War I, the Aelmoeseneie forest was replanted in 1920. Since then, slight thinnings were performed at irregular time intervals. Actually, the Aelmoeseneie forest covers 28 ha. In 1990, an area of 1.83 ha was fenced in order to reserve it for scientific research. Since then, a zero management policy was applied. The main characteristics of the site and of the experimental zone have been extensively described in Chapter 7. Briefly, there is a temperate maritime climate, with a mean annual air temperature and precipitation of 10.1 °C and 791 mm (measured in the period 1984-1993), respectively (Samson *et al.* 1996). The experimental zone comprises two different stand types, an oak-beech stand (OB, 1.061 ha), with pedunculate oak (*Quercus robur* L.) and beech (*Fagus sylvatica* L.) as dominant species, and an ash stand (AS, 0.766 ha), where common ash (*Fraxinus excelsior* L.) is the dominant species. Other abundant species in the experimental zone are maple (*Acer pseudoplatanus* L.), larch (*Larix kaempferi* (Lambert) Carr.), hazel (*Corylus avellana* L.) and rowan (*Sorbus aucuparia* L.). The boundary between the two stands was fixed based on visual observations of the vegetation in combination with a permanent grid of aluminium bars, which divide the experimental zone in squares of 100 m^2 (Fig. 8.1). Since 1992, a level II observation plot was delineated in each of the two stands of the experimental zone of the Aelmoeseneie forest, in the framework of the UN/ECE Forest Intensive Monitoring Programme (EU/ICP Forests). In 1993, a 35 m high measuring tower was constructed at the transition of the oak-beech and the ash stand, as indicated on Fig. 8.1.

8.2.2. Forest inventory, diameter at breast height and basal area

In December 1990, December 1997 and January 2006, a full inventory of the experimental zone was performed. During these inventories, circumference at breast height (1.3 m) ($CBH_{1.3}$, in cm) was determined for all trees ($CBH_{1.3}$ > 22 cm) with a measuring tape, the species was noted, and the (X,Y)-coordinates within the zone were registered. Moreover, numbered metal plates were attached to trees that reached the minimum requirement of $CBH_{1.3}$ (*i.e.*, $CBH_{1.3} \geq 22$ cm).

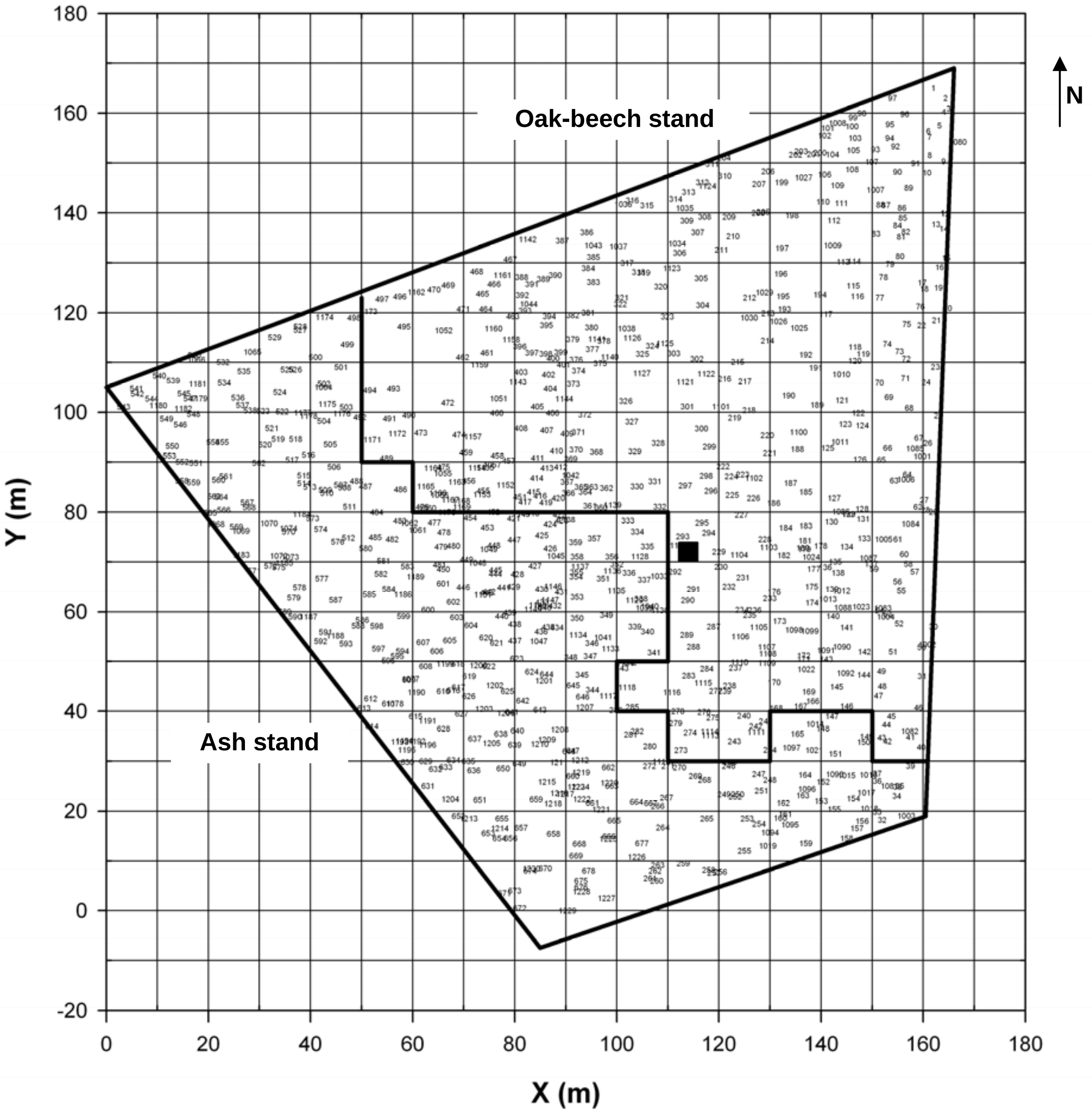

Figure 8.1. *Schematic overview of the experimental zone of the Aelmoeseneie forest with indication of the measuring tower (■) and all tree numbers*

Under the assumption of circular tree cross sections, tree diameter at 1.3 m height ($DBH_{1.3}$, in cm) was calculated from $CBH_{1.3}$ for all trees. From a preliminary analysis, it appeared that the beech population could be split up in two subpopulations : a young ($DBH_{1.3} < 45$ cm) and an old population ($DBH_{1.3} \geq 45$ cm). Where relevant, this terminology will be used in the remainder of the text. Stem basal area at breast height ($BA_{1.3}$, in cm^2) was calculated as :

$$BA_{1.3}=\frac{CBH_{1.3}^{2}}{4\cdot\Pi} \qquad \textbf{(8.3)}.$$

Stem basal area increment was calculated according to the method proposed by Clark *et al.* (2001), and was based on tracking $BA_{1.3}$ changes of individual trees. As such, the increment of a surviving tree was calculated as the difference between its $BA_{1.3}$ at the end and at the beginning of the time interval. As the time interval in this study was longer than 2 years, $BA_{1.3}$ increment of trees that died during this interval, but were still standing, was calculated in the same way (Clark *et al.* 2001). Total stand increment had to be adjusted for ingrowth. This was done by calculating $BA_{1.3}$ increment of each new tree as the difference between its $BA_{1.3}$ at the end of the interval, and $BA_{1.3}$ of a tree with minimum dimension (*i.e.*, $CBH_{1.3}$ of 22 cm). Summing $BA_{1.3}$ increments of all trees gave the total stand $BA_{1.3}$ increment. Relative growth rates (RGR) were calculated as the change in total basal area (TBA) divided by the mean TBA during the considered time period.

8.2.3. Tree height, total solid wood volume and stem biomass

Tree height was calculated by using site- and species-specific regression equations. These equations were calculated from measurements performed in 1997 in the experimental forest, on a number of model trees per species (see Chapter 7 for details), and had an S-shaped form :

$$H = exp\left(b_0 + \left(\frac{b_1}{CBH_{1.3}}\right)\right) \tag{8.4}$$

where H is tree height (m), $CBH_{1.3}$ is circumference at breast height (cm) and b_0 and b_1 are species-specific model coefficients. These equations were available for oak, beech, ash, larch and maple. Species-specific coefficients b_0 and b_1 are given in Table 8.1.

Table 8.1. *Coefficients (and standard deviation) of the regression equation between tree height (H, in m) and circumference at breast height ($CBH_{1.3}$, in cm) (see eq. 8.4);* R^2 *and the number of sampled trees (n) are also indicated.*

	b_0		b_1		R^2	n
Oak (*Quercus robur* L.)	3.5464	(0.0573)	-31.5825	(5.2528)	0.79	12
Beech (*Fagus sylvatica* L.)	3.5749	(0.0641)	-36.9552	(4.7662)	0.96	6
Ash (*Fraxinus excelsior* L.)	3.4335	(0.0687)	-9.1024	(6.8343)	0.30	6
Larch (*Larix kaempferi* (Lambert) Carr.)	3.4606	(0.0656)	-18.5249	(9.2468)	0.37	9
Maple (*Acer pseudoplatanus* L.)	3.5371	(0.0627)	-27.3370	(3.3489)	0.92	9

Furthermore, species-specific allometric equations were used to determine total solid wood volume (TSW) of each tree. The expression 'total solid wood' refers to the combination of stem and branches with a circumference exceeding 22 cm. Relationships for oak and ash were established with data from the experimental forest (eq. 7.1 and 7.2), while for beech, larch and maple, regressions established by

Dagnelie *et al.* (1999) were used, with $CBH_{1.3}$ and tree height as input parameters (eq. 7.3, 7.4 and 7.5).

Stem wood biomass was calculated for each tree by multiplying TSW with specific wood density (WD) of the stem wood. Schauvliege (1995) determined a mean wood density of 500 ± 28 and 523 ± 34 kg dry mass (DM) m^{-3} for oak and ash, respectively. Young beeches ($DBH_{1.3}$ < 45 cm) appeared to have a specific density of 566 ± 39 kg DM m^{-3}, while WD was 550 ± 12 kg DM m^{-3} for old beeches (DBH1.3 ≥ 45 cm) (Table 8.2). For maple and larch, no site-specific WD values were available. Therefore, a literature study was performed to determine mean WD for these two species. As is extensively described in Vande Walle *et al.* (2005), data selection was limited to data reported for Belgium, Denmark, France, Germany, Great Britain, Ireland and the Netherlands (see Chapter 9). Literature sources from which data were extracted were Baritz and Strich (2000), Bartelink (1996 and 1997), COST-E21 (2001), Dieter and Elsasser (2002), Grote (2002), Guilley *et al.* (1999), Houghton *et al.* (1997), IPCC (2003), Janssens *et al.* (1999), Joosten and Schulte (2002), Joosten *et al.* (2004), Lebaube *et al.* (2000), Lefèvre *et al.* (2000), Levy *et al.* (2004), Löwe *et al.* (2000), Milne and Brown (1997), Mund *et al.* (2002), Nabuurs *et al.* (2000), Pignard *et al.* (2000), Ponette *et al.* (2001), Pontailler *et al.* (1997), Schalck *et al.* (1978), Vande Walle and Lemeur (2001) and Vande Walle *et al.* (2001). Mean wood density for maple was 557 ± 35 kg DM m^{-3}, while for larch, a mean WD of 474 ± 51 kg DM m^{-3} was found (Table 8.2).

Table 8.2. *Mean values (and standard deviation) for specific wood density (WD), carbon content (CC) and biomass expansion factor* s.s. *(BEF) of the main tree species of the experimental zone in the Aelmoeseneie forest; DM : dry mass, TSW : total solid wood, ABGC : above- and belowground carbon; m indicates the number of measurements, while n stands for the number of values that were available from literature; oak :* Quercus robur *L., beech :* Fagus sylvatica *L., ash :* Fraxinus excelsior *L., larch :* Larix kaempferi *(Lambert) Carr., maple :* Acer pseudoplatanus *L.*

	WD (kg DM m^{-3} TSW)		**CC (kg C kg^{-1} DM)**		**BEF (kg ABGC kg^{-1} TSW C)**	
Oak	500 (28)	(m = 51)	0.48 (0.03)	(n = 3)	1.50 (0.00)	(n = 1)
Beech young	566 (39)	(m = 18)	0.48 (0.03)	(n = 10)	1.64 (0.13)	(n = 3)
Beech old	550 (12)	(m = 20)	0.48 (0.03)	(n = 10)	1.64 (0.13)	(n = 3)
Ash	523 (34)	(m = 43)	0.49 (0.02)	(n = 6)	1.50 (0.00)	(n = 1)
Maple	557 (35)	(n = 3)	0.49 (0.02)	(n = 6)	1.50 (0.00)	(n = 1)
Larch	474 (51)	(n = 8)	0.47 (0.06)	(n = 3)	1.75 (0.35)	(n = 2)

Stand TSW and biomass increments were calculated as described above for $BA_{1.3}$ increment.

8.2.4. Net ecosystem productivity

We applied the mass-balance method (Black *et al.* 2005) to assess NEP of the two forest stands, for the periods 1991-1997 and 1998-2005. As can be seen from eq. 8.1, changes in living (ΔC_b) and dead (ΔC_{dead}) biomass, and in soil carbon stocks (ΔC_{soil}) were summed to obtain an estimate of NEP.

To assess ΔC_b, the total amount of carbon in the above- and belowground biomass of each standing tree was calculated. Therefore, individual stem biomass (kg TSW DM tree^{-1}) was multiplied by carbon content (CC, kg C kg^{-1} DM) and by a species-specific biomass expansion factor *s.s.* (BEF, kg above- and belowground C kg^{-1} TSW C). Carbon increment per tree was calculated as the difference in total above- and belowground carbon (ABGC) at the end and at the beginning of the interval, for surviving trees as well as for trees dying during the considered time interval. ABGC increment of new trees was assessed as the difference between ABGC at the end of the interval, and ABGC of a tree with minimum dimension. In Table 8.2, species-specific CC and BEF values are listed. These values represent the mean values, and the standard deviation on these values, obtained by the extensive literature review, which was described by Vande Walle *et al.* (2005). Literature sources from which data were extracted were listed in the previous paragraph. Summing ABGC increments of all standing trees gave an estimate of ΔC_b for the whole stand.

The change in carbon stock in dead biomass (ΔC_{dead}) was split up in two parts, one referring to the change in carbon stock in the litter layer ($\Delta C_{dead\ LL}$), and one indicating the carbon stock change in lying dead trees ($\Delta C_{dead\ trees}$). Measurements of carbon stocks in the litter layer were performed in the level II plot of the oak-beech stand in 2004 (G. Genouw, pers. comm.). In total, 24 samples (0.25 m x 0.25 m) of the litter layer were taken. Three composite samples for chemical analysis were obtained by combining three times eight single samples. Results for 2004 were compared to litter layer carbon stocks available for 1996 (33.2 ± 5.6 t C ha^{-1}, Table 7.3) to determine the mean annual change in $C_{dead\ LL}$ in the oak-beech stand. This value was used for both considered time periods (1991-1997 and 1998-2005). As the litter layer in the ash stand only contained 0.1 ± 0.01 t C ha^{-1} in 1996 (Table 7.3), it was assumed that $\Delta C_{dead\ LL}$ was equal to zero for this stand. The total amount of carbon in the above- and belowground biomass of fallen trees was assessed by multiplying the total solid wood volume of these trees with species-specific WD, BEF and CC (Table 8.2). Summation of ABGC of all trees that fell during one of the two periods was considered as an estimate of $\Delta C_{dead\ trees}$ for that period (1991-1997 or 1998-2005).

In 2004, measurements were performed in the level II plot of the oak-beech stand to determine the soil carbon stock to a depth of 1 m (G. Genouw, pers. comm.). Bulk density was determined for 24 single samples, for different soil layers (0-5 cm, 5-10 cm, 10-20 cm, 20-40 cm and 40-100 cm). Three composite samples, formed by combining eight singles samples, were used for chemical analysis. Comparing the soil carbon stock determined for 2004 with the soil carbon stock known for 1996 (Table 7.3) enabled the calculation of the mean ΔC_{soil} for the period 1998-2005 in the oak-beech stand. The same mean annual soil carbon stock change was assumed for the period from 1991 to 1997. For the ash stand, no soil carbon stock data were available for 2004. Therefore, it was assumed that ΔC_{soil} for this stand was equal to ΔC_{soil} for OB.

8.2.5. Statistical analysis

Differences between mean annual air temperature, precipitation, radiation and elemental deposition in the periods 1991-1997 and 1998-2005 or between characteristics of both stands were checked for significance with SPSS 12.0. When data were normally distributed, independent samples t-tests were used, after testing equality of variances. For non-normally distributed data, the Mann-Whitney test was used to compare means. All tests were performed at the 95 % significance level.

Uncertainty on the assessed NEP values was calculated by error propagation (NIST 2006), starting at individual tree level. Schauvliege (1995) found a good agreement between measured stem volumes and volumes determined by the allometric relationships proposed by Dagnelie *et al.* (1999). Therefore, we assumed that uncertainty on the conversion of $CBH_{1.3}$ to total solid wood volume was zero, for all tree species. Standard deviations on wood density values, biomass expansion factors and carbon content (Table 8.2) were then used to determine the uncertainty on ΔC_b and $\Delta C_{dead\ trees}$. Relative uncertainty was expressed in terms of percentage by dividing absolute uncertainty values by total values and multiplying by 100.

8.3. Results

8.3.1. Stem number distribution and evolution

In Fig. 8.1, a schematic overview of the experimental zone is given, with indication of all tree numbers. In total, 957 trees were measured during the inventory of 2006, and 17 tree species were observed in the tree layer of the experimental zone (Vande Walle *et al.* 2006). Besides pedunculate oak, red oak (*Quercus rubra* L.) was found too. The term 'oak' is used in the remainder of the text to indicate pedunculate oak, while red oak is included in the 'other species' category. When a tree consisted of more than one stem, each stem was considered as an individual tree.

In January 2006, the total stem number (living and dead standing trees together) amounted to 452 and 576 trees ha^{-1} in the oak-beech and the ash stand, respectively. In OB, 26 dead trees ha^{-1} were noted (18 oaks, 2 beeches, 5 rowans and 1 other), compared to 7 dead trees ha^{-1} in AS (3 ashes, 1 maple and 3 others). In Table 8.3, the evolution of the stem number from 1990 to 2005 is given, for living and dead trees together. From this table, it can be seen that especially the number of maples and hazels increased in both stands, while the number of ashes in the ash stand diminished during the second time period, mainly due to storm damage.

Table 8.3. *Evolution of stem number (# trees ha^{-1}) in the oak-beech and the ash stand of the experimental zone in the Aelmoeseneie forest; data for the end of 1990, 1997 and 2005*

	Oak-beech stand			**Ash stand**		
	1990	**1997**	**2005**	**1990**	**1997**	**2005**
Oak	141	139	141	35	38	37
Beech	62	82	101	4	4	7
Ash	7	8	8	145	158	150
Larch	18	18	18	8	8	8
Maple	23	56	102	85	121	247
Hazel	5	13	46	10	39	98
Rowan	1	15	21	1	3	3
Others	10	15	16	13	29	27
Total	267	345	452	302	399	576

8.3.2. Stem diameter and basal area

When only the three main species oak, beech and ash were considered, mean diameter at breast height (DBH$_{1.3}$) was considerably, although not significantly, lower in the oak-beech stand (31.0 ± 17.4 cm) than in the ash stand (37.6 ± 13.4 cm) in January 2006. The overall mean DBH$_{1.3}$, however, was much more similar : 23.2 ± 18.1 cm in OB and 22.5 ± 16.7 cm in AS, respectively. This was mainly due to the larger contribution of maple and hazel to the total tree number in the ash stand (59.5 %) compared to the oak-beech stand (32.7 %). As trees of these two species had in general smaller diameters, the overall mean diameter largely decreased when these two species were taken into account.

Mean DBH$_{1.3}$ for all species together determined in the winter of 1997 amounted to 26.1 ± 17.4 cm for the oak-beech stand, and 26.9 ± 15.8 cm for the ash stand (Vande Walle *et al.* 1998a). Because of the large number of young trees in the inventory of 2005 compared to 1997, mean DBH$_{1.3}$ decreased for both forest stands from 1997 to 2005. This is illustrated in Fig. 8.2, from which it can be noted that in both stands, the number of trees in the smallest DBH$_{1.3}$ class increased considerably in the period 1997 to 2005, indicating a strong ingrowth.

Besides a larger tree number, the ash stand had a higher total basal area (TBA, 35.4 m^2 ha^{-1}) than the oak-beech stand (31.5 m^2 ha^{-1}) at the end of 2005 (Fig. 8.3 and Table 8.4). In the oak-beech stand, the contribution of oak, beech and ash to the total basal area slightly diminished from 79.4 % in 1997 (Vande Walle *et al.* 1998a) to 78.8 % in 2005. In 2005, these three species contributed only 68.4 % to the total basal area of the ash stand, a decrease of 2.5 % compared to 1997. In AS, the contribution of maple increased from 16.6 to 18.9 % of total basal area, because of the strong ingrowth of this species. The contribution of maple to the total stem number increased from 30.3 to 42.9 % in the same period.

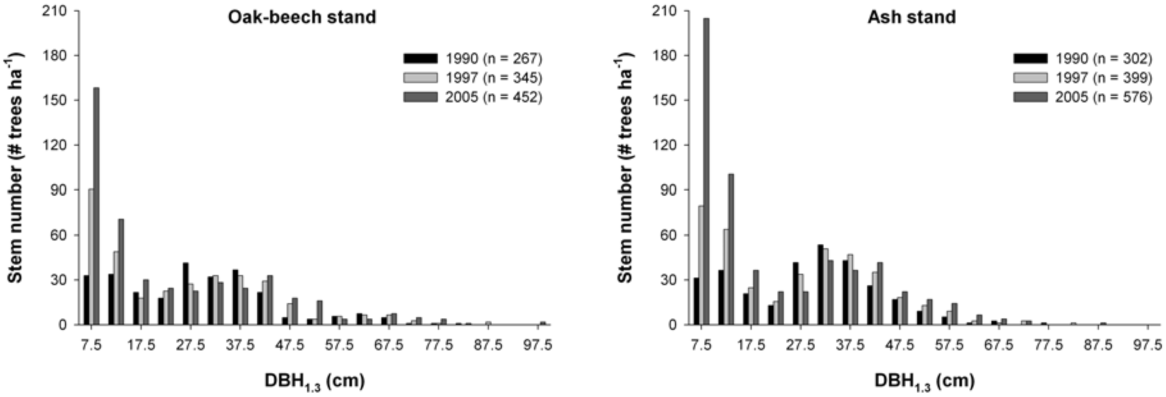

Figure 8.2. *Frequency distribution of diameter at breast height (DBH$_{1.3}$) of the oak-beech stand (left) and the ash stand (right) of the experimental zone in the Aelmoeseneie forest; n indicates total stem number (# trees ha^{-1}); data for the end of 1990, 1997 and 2005.*

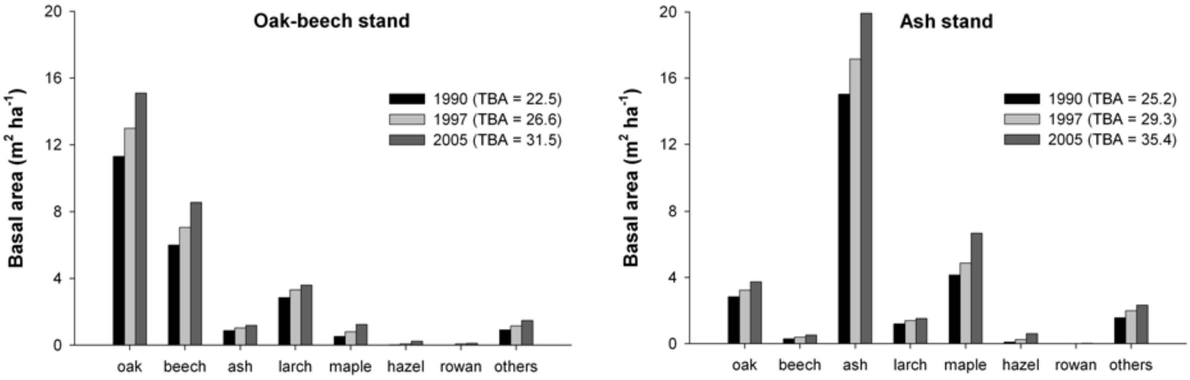

Figure 8.3. *Contribution of different tree species to the total basal area (TBA, m^2 ha^{-1}) in the two stands of the experimental zone in the Aelmoeseneie forest; data for the end of 1990, 1997 and 2005*

Table 8.4. *Total basal area (TBA), total solid wood volume (TTSW), stem biomass (TSB) and above- and belowground carbon (TABGC) in the two stands of the experimental zone in the Aelmoeseneie forest; for TSB and TABGC, uncertainty is given between brackets; data for the end of 1990, 1997 and 2005.*

	Oak-beech stand			**Ash stand**		
	1990	**1997**	**2005**	**1990**	**1997**	**2005**
TBA (m^2 ha^{-1})	22.5	26.6	31.5	25.2	29.3	35.4
TTSW (m^3 ha^{-1})	264.1	309.0	364.3	285.7	326.3	383.7
TSB (t DM ha^{-1})	135.4 (0.7)	158.5 (0.8)	187.3 (0.9)	149.5 (0.9)	170.8 (1.0)	201.2 (1.1)
TABGC (t C ha^{-1})	102.0 (0.7)	119.4 (0.8)	140.9 (1.0)	110.4 (1.0)	126.1 (1.1)	148.5 (1.3)

In absolute terms, the mean annual TBA increase was larger in the ash stand than in the oak-beech stand in both periods (Table 8.5). Expressed as relative growth rate (RGR_{TBA}), the oak-beech stand was more productive in the period 1991-1997 than the ash stand, while the opposite was true for the period 1998-2005.

Table 8.5. *Mean annual increment in total basal area (ΔTBA), total total solid wood volume (ΔTTSW) and total stem biomass (ΔTSB) in the two stands of the experimental zone in the Aelmoeseneie forest; data for the periods 1991-1997 and 1998-2005; uncertainty on ΔTSB is given between brackets; relative growth rates (RGR, expressed in %) were calculated by dividing the mean annual increment by the mean TBA, TTSW or TSB during the considered period.*

		Oak-beech stand		**Ash stand**	
		1991-1997	**1998-2005**	**1991-1997**	**1998-2005**
ΔTBA	$m^2\ ha^{-1}\ year^{-1}$	0.53	0.57	0.55	0.66
ΔTTSW	$m^3\ ha^{-1}\ year^{-1}$	6.32	6.81	5.71	6.89
ΔTSB	$t\ DM\ ha^{-1}\ year^{-1}$	3.25 (0.14)	3.53 (0.14)	2.99 (0.19)	3.64 (0.18)
RGR_{TBA}	%	2.16	1.96	2.02	2.04
RGR_{TTSW}	%	2.21	2.02	1.86	1.94
RGR_{TSB}	%	2.21	2.04	1.86	1.96

8.3.3. Stem wood volume and biomass

Total total stem wood volume (TTSW) in the ash stand (383.7 $m^3\ ha^{-1}$) exceeded total TSW in the oak-beech stand (364.3 $m^3\ ha^{-1}$) by 5.3 % at the end of 2005 (Fig. 8.4). The contribution of standing dead trees was only 1.1 % in AS and 4.0 % in OB. Based on absolute values of stem wood volume increment, both stands were most productive in the period 1998-2005, while relative growth rate RGR_{TTSW} was highest in the period 1991-1997 for OB, and in the period 1998-2005 for AS (Table 8.5).

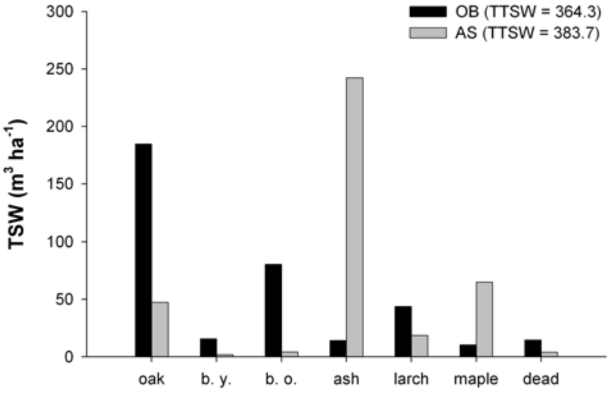

Figure 8.4. *Total solid wood (TSW) per species in the oak-beech (OB) and the ash (AS) stand of the experimental zone in the Aelmoeseneie forest at the end of 2005; TTSW : total TSW (in $m^3\ ha^{-1}$); b. : beech, y. : young, o. : old*

Total stem biomass at the end of 2005 was 13.9 t DM ha^{-1} higher in the ash stand (201.2 ± 1.1 t DM ha^{-1}) than in the oak-beech stand (187.3 ± 0.9 t DM ha^{-1}) (Table 8.4). In this latter stand, oak and beech accounted for 52.0 and 29.8 % of total stem biomass, respectively. Whereas ash only contributed 4.0 % to the total stem biomass in the oak-beech stand, this species accounted for 63.0 % of total stem biomass in the ash stand. In AS, maple was the second most important species concerning stem biomass, with a contribution of 18.8 %. Oak (12.0 %) and beech (1.7 %) were much less important in this plot.

A visual representation of cumulative stem biomass with increasing $DBH_{1.3}$ is given in Fig. 8.5 for both stands, for the end of 1990, 1997 and 2005. Mean stem biomass increment was 3.53 ± 0.14 and 3.64 ± 0.18 t DM ha^{-1} $year^{-1}$ in the period 1998 to 2005 for the oak-beech and the ash stand, respectively (Table 8.5). In the period 1991-1997, this increment was only 3.25 ± 0.14 t DM ha^{-1} $year^{-1}$ in OB and 2.99 ± 0.19 t DM ha^{-1} $year^{-1}$ in AS. As appeared from the relative growth rates RGR_{TSB} (Table 8.5), the OB stand was more productive than the ash stand in both periods.

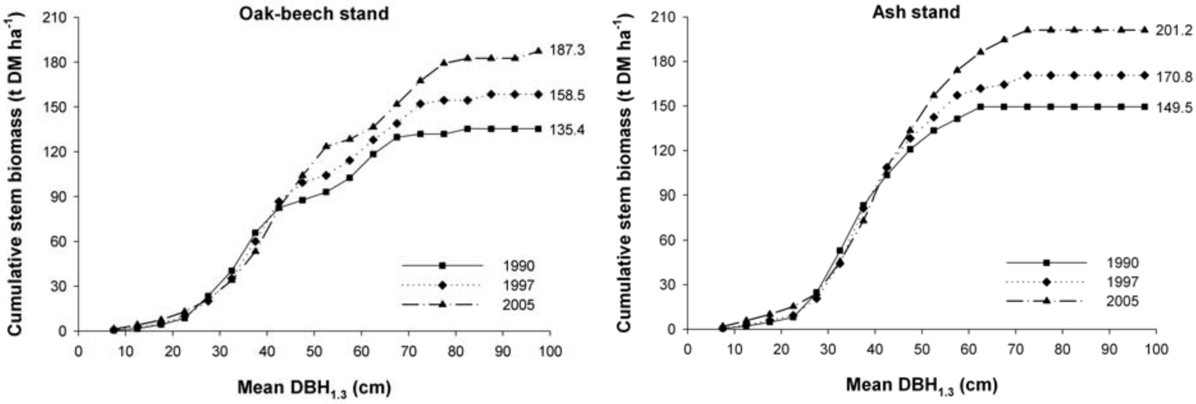

Figure 8.5. *Cumulative stem biomass (t DM ha^{-1}) in function of diameter at breast height ($DBH_{1.3}$) classes of 5 cm for the oak-beech stand (left) and the ash stand (right) of the experimental zone in the Aelmoeseneie forest; total stem biomass values (t DM ha^{-1}) for the end of 1990, 1997 and 2005 are indicated at the right side of the curves.*

8.3.4. Net ecosystem productivity

In the period 1991-1997, net ecosystem productivity amounted to 3.33 ± 2.30 and 2.25 ± 1.81 t C ha^{-1} $year^{-1}$ in the oak-beech and the ash stand, respectively (Table 8.6). Higher values (3.57 ± 2.30 t C ha^{-1} $year^{-1}$ for OB and 2.97 ± 1.81 t C ha^{-1} $year^{-1}$ for AS) were noted for the period 1998-2005. The change in lying dead wood carbon stock $\Delta C_{dead\ trees}$ was small compared to the change in living biomass ΔC_b (Table 8.6). Measurements in the level II plot of the oak-beech stand revealed an increase of the carbon stock in the litter layer from 33.2 ± 5.6 t C ha^{-1} in 1996 (Table 7.3) to 40.4 ± 9.7 t C ha^{-1} in 2004. As such, $\Delta C_{dead\ LL}$ was 0.90 t C ha^{-1} $year^{-1}$ on average (Table 8.6). Mean soil carbon stock to a depth of 1 m determined for the level II observation plot of the OB stand in 2004 amounted to 134.6 ± 12.0 t C ha^{-1}. As this C stock was assessed at 134.8 ± 7.2 t C ha^{-1} in 1996 (Table 7.3), ΔC_{soil} was assumed to equal zero, for both time periods, and both stands (Table 8.6). From the calculated NEP values, it was obvious that both the oak-beech and the ash stand acted as a sink for carbon over the period 1991 to 2005.

Table 8.6. *Mean annual change in living biomass (ΔC_b), lying dead trees ($\Delta C_{dead\ trees}$), litter layer ($\Delta C_{dead\ LL}$) and soil (ΔC_{soil}) carbon stocks for two stands of the experimental zone in the Aelmoeseneie forest; NEP$_{mass}$ is determined as the sum of ΔC_b, $\Delta C_{dead\ trees}$, $\Delta C_{dead\ LL}$ and ΔC_{soil}; uncertainty is given between brackets; all values are expressed in t C ha^{-1} year^{-1}.*

	Oak-beech stand		**Ash stand**	
	1991-1997	**1998-2005**	**1991-1997**	**1998-2005**
ΔC_b	2.43 (0.27)	2.63 (0.27)	2.19 (0.21)	2.64 (0.21)
$\Delta C_{dead\ trees}$	0.00 (0.00)	0.04 (0.00)	0.06 (0.00)	0.33 (0.01)
$\Delta C_{dead\ LL}$	0.90 (1.40)	0.90 (1.40)	0.00 (0.00)	0.00 (0.00)
ΔC_{soil}	0.00 (1.80)	0.00 (1.80)	0.00 (1.80)	0.00 (1.80)
NEP$_{mass}$	3.33 (2.30)	3.57 (2.30)	2.25 (1.81)	2.97 (1.81)

8.4. Discussion

8.4.1. Development of the two forest stands

As can be seen in Table 8.3, the stem number of beech increased considerably from 1991 to 2005 in the oak-beech stand. The total number of maple trees increased in both forest stands, while especially in the ash stand, hazel was characterized by a large ingrowth. For none of the species, a large dieback was observed during the 15 years of the study. Differences in stand development between the oak-beech and the ash stand are mainly due to differences in species composition and soil characteristics, as site history, management and climatic conditions are identical. The large ashes appeared to be sensitive to storm damage, and the disappearance of a number of these large trees during the period 1998-2005 due to storm events certainly lowered biomass increment in the ash stand. Although there was a large ingrowth of maple trees in this stand, total biomass increment of these trees was rather limited, because of the relative small dimensions of these young trees. Ingrowth of maple was more pronounced in the ash stand (162 trees ha^{-1} in the period 1991-2005) than in the oak-beech stand (79 trees ha^{-1}).

Stem wood increment in the Aelmoeseneie forest, ranging from 5.71 to 6.89 m^3 ha^{-1} year^{-1} (Table 8.5), was comparable to the annual wood volume increment of 6 to 7 m^3 ha^{-1} year^{-1} in the period 1995-1998 mentioned by Janssens *et al.* (1999) for a Flemish Scots pine stand of 68 years old. Löwe *et al.* (2000) provided an overview of TSW increment rates for 13 European countries. Increment data for Gontrode (Table 8.5) fell within the lower range of values from this European study (4.0 to 16.0 m^3 ha^{-1}

year^{-1}), but were still considerably higher than minimum values reported for Flemish case studies, as summarized in Table 10.3.

8.4.2. Comparison of forest evolution during two multi-year periods

Ecosystem productivity is limited by canopy photosynthesis, which determines the upper limit of the gross primary productivity of an ecosystem (Lebaube *et al.* 2000, Schulze 2000). The second most important component of an ecosystem carbon balance is soil respiration, the release of CO_2 from the soil to the atmosphere (Pregitzer 2003). As such, interannual differences in NEP are associated with climatic conditions as radiation, temperature, precipitation, …, as these factors influence photosynthesis as well as decomposition rates (Curtis *et al.* 2002, Ehman *et al.* 2002, Pregitzer 2003, Smith *et al.* 2003). The study of White and Nemani (2003) showed that midsummer drought can cause a reduction in total carbon uptake, while Zha *et al.* (2004) concluded that especially the weather conditions in spring and summer were important to explain interannual variations in NEP of a boreal Scots pine stand. Changes in temperature and precipitation can also change growing season length, which is an important determinant of NPP in temperate forests (Carrara *et al.* 2003, Boisvenue and Running 2006). The influence of climate and growing season length can in some cases shift a terrestrial ecosystem from a sink to a source of carbon (Valentini *et al.* 2000). In literature, it is also suggested that increased inputs of atmospheric N deposition may combine with elevated CO_2 to increase forest productivity and carbon sequestration (Brown 2002, Mund *et al.* 2002, Hom 2003). Forest inventory methods are based on actual measured growth rates, which are subject to factors such as CO_2 fertilization, N deposition and climate change (Hom 2003).

To search for differences in climatic conditions in the periods 1991-1997 and 1998-2005, data for the period 1991 to 2005 were obtained from the Royal Meteorological Institute of Belgium. Three parameters were selected : monthly total horizontal radiation (MJ m^{-2} month^{-1}) measured in the meteorological station of Melle (1 km away from Gontrode), and monthly mean air temperature (°C) and monthly total precipitation (mm month^{-1}) measured in Kruishoutem (20 km from Gontrode). From Table 8.7, it can be seen that the annual mean of the monthly mean air temperature was significantly higher in the period 1998-2005 (11.2 °C) than in the period 1991-1997 (10.3 °C). This observation was in accordance to the study of Schwartz *et al.* (2006), which proved that average annual temperatures are getting warmer across nearly all temperate land areas in the northern hemisphere since 1950. Total radiation and total precipitation were also higher in the period 1998-2005 than in 1991-1997, although these differences were not statistically significant (Table 8.7). As can be concluded from the coefficient of variation (CV), given in the same table, the period 1991-1997 was more variable concerning mean annual air temperature and total annual precipitation than the period 1998-2005. As is depicted in Fig. 8.6, mean annual temperature was higher than the mean for the period 1991 to 2005 from 1998 onwards. The same trend, although not so obvious, was seen for total radiation. Both periods 1991-1997 and 1998-2005 were characterized by a number of dry and a number of wet years (Fig. 8.6). For all years together, there was a significant positive relationship between mean annual air temperature and total precipitation (r = 0.581). When only the growing season from May to September was considered, conclusions were the same as for the whole year, with a significantly higher mean air temperature in the period 1998-2005, and higher, although not significantly, values for total precipitation and total horizontal radiation for this period too.

Table 8.7. *Mean annual (ann.) monthly mean air temperature (temp.) (°C), mean total annual precipitation (mm year^{-1}) and mean total annual radiation (MJ m^{-2} year^{-1}) for the periods 1991-1997 and 1998-2005; data from the RMI measuring stations at Melle (radiation) and Kruishoutem (temperature and precipitation); for bulk deposition and throughfall, the amount of total nitrogen (NO_3-N + NH_4-N, kg N ha^{-1} year^{-1}), sulphate sulphur (SO_4-S, kg S ha^{-1} year^{-1}) and potentially acidifying (Pot. acid.) deposition (eq. ha^{-1} year^{-1}) are given; standard deviation (St. dev.), coefficient of variation (CV) and total number of years for which data were available (n) are also indicated. Different letters within one line indicate significant differences between the two periods.*

	1991-1997				**1998-2005**			
	Mean	**St. dev.**	**CV**	**n**	**Mean**	**St. dev.**	**CV**	**n**
Climatic variable								
Ann. mean air temp.	10.3^a	0.7	6.5	7	11.2^b	0.2	2.1	8
Total ann. precipitation	731.9^a	147.2	20.1	7	870.6^a	123.7	14.2	8
Total ann. radiation	3526.1^a	214.6	6.1	7	3686.1^a	258.0	7.0	8
Bulk deposition								
Total N	20.1^a	5.6	28.0	5	14.8^a	3.7	25.2	8
SO_4-S	10.6^a	3.4	31.7	5	15.6^b	2.3	14.9	8
Pot. acid. deposition	1355.4^a	397.2	29.3	5	1298.1^a	238.1	18.3	8
Throughfall oak-beech								
Total N	29.6^b	7.3	24.7	6	22.9^a	3.1	13.7	8
SO_4-S	27.0^a	8.5	31.4	6	28.5^a	7.4	25.8	8
Pot. acid. deposition	1594.7^a	721.8	45.3	6	1120.6^a	391.6	34.9	8
Throughfall ash								
Total N	23.0^a	4.8	20.6	6	18.5^a	1.7	9.0	5
SO_4-S	30.3^a	6.7	22.2	6	29.5^a	3.4	11.7	5
Pot. acid. deposition	358.0^a	781.3	218.2	6	-456.1^a	489.7	-107.4	5

Since 1992, deposition of chemical elements is monitored in precipitation (bulk deposition), throughfall and stem flow water in the two level II observation plots. Data from 1992 to 2004 were derived from the reports published by De Schrijver and Lust (1999, 2000), De Schrijver *et al.* (2001), Genouw *et al.* (2003, 2004, 2005), Nachtergale *et al.* (2002) and Neirynck and Lust (2000), while data for 2005 were put at our disposal by Genouw (pers. comm.). In Table 8.7, deposition data for the periods 1991-1997 and 1998-2005 are given. Total nitrogen (NO_3-N + NH_4-N) in bulk deposition was lower, although not significantly, in the period 1991-1997 than from 1998 to 2005, while significantly more sulphur deposition (SO_4-S) was observed in the second period (1998-2005). Total potentially acidifying deposition $AC(pot)_{dep}$, expressed in eq. ha^{-1} $year^{-1}$, was calculated as the difference between acidifying ($N_{dep} + S_{dep} + Cl_{dep}$) and basic ($Ca_{dep} + Mg_{dep} + K_{dep} + Na_{dep}$) deposition, as defined by ICP (ICP MM 2004) :

$$AC(pot)_{dep} = N_{dep} + S_{dep} + Cl_{dep} - (Ca_{dep} + Mg_{dep} + K_{dep} + Na_{dep}) \quad \textbf{(8.5)}.$$

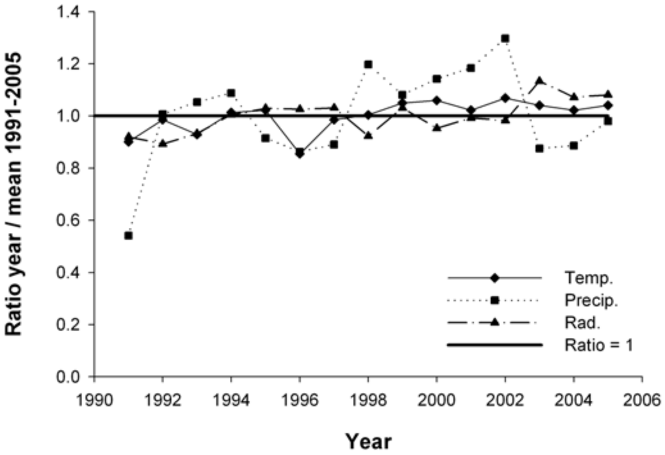

Figure 8.6. *Ratio of annual mean monthly air temperature (Temp.), total annual precipitation (Precip.) and total annual radiation (Rad.) compared to the mean values over the period 1991-2005*

Total potentially acidifying deposition was not significantly different between the two periods (Table 8.7). No significant relationship could be found between total annual precipitation and nitrogen, sulphur or potentially acidifying deposition. For throughfall deposition, no significant differences were found between the two periods within one forest stand, except for total N, which was higher in the first than in the second period in the oak-beech stand (Table 8.7). Within both periods, however, OB appeared to have a much higher total potentially acidifying deposition in throughfall than AS. This was mainly due to a much higher acidifying deposition and less basic deposition in OB compared to AS (data not shown). For all years together, the ratios of elemental amounts in throughfall in the ash stand compared to the oak-beech stand were 0.89, 0.74, 1.22, 1.39, 2.24, 1.72, 1.35 and 1.00 for NO_3-N, NH_4-N, SO_4-S, Cl, Ca, Mg, K and Na, respectively. Differences in throughfall deposition in the two forest stands can be attributed to species-specific differences in canopy exchange processes (Genouw *et al*. 2004) and to the difference in the degree of base saturation of the soil, which is higher in the ash than in the oak-beech stand (De Schrijver *et al*. 2001).

The analysis of climatic and deposition data revealed that climatic differences between the two time periods were most pronounced. Therefore, the higher temperature in combination with more precipitation was probably the main reason for the enhanced tree growth in the period 1998-2005 in comparison to the period 1991-

1997 (Table 8.5). Moreover, nitrate leaching is observed in the soil profile (De Schrijver and Lust 1999), indicating that the forest in Gontrode is nitrogen saturated. As such, it can be doubted whether nitrogen deposition actually has a large impact on forest growth (Boisvenue and Running 2006). As indicated in Table 8.5, the ash stand seemed to profit more from the higher temperature and precipitation than the oak-beech stand, as RGR_{TSB} of AS was enhanced from 1.86 % in the first period to 1.97 % in the second period, while for OB, RGR_{TSB} decreased from 2.21 to 2.03 %. However, it should also be stated that differences in carbon sequestration in the two periods were not very high (Table 8.6). This observation follows the study of Barford *et al.* (2001), who noted that rates of long-term carbon sequestration in a northern hardwood forest changed slowly, because they are driven by ecosystem properties that evolve slowly, *i.e.,* stand composition, biomass and mortality, soil fertility and coarse dead wood pool size.

Our conclusion was in line with the results of the European Recognition project (RECOGNITION 2006). Based on analysis of data from long-term experiments as well as from current and recent growth investigations, and from modelling exercises, Recognition scientists concluded that increased nitrogen deposition from the atmosphere was the main cause of the observed increases in growth of Norway spruce, Scots pine and European beech until the late 1990's. Model results, however, suggested that in the future, nitrogen effects will diminish and atmospheric CO_2 concentrations and climate change will be the major causes of increased forest growth rate expected in the coming decades.

8.4.3. Mass-balance approach to determine NEP

Intercomparisons of different methods to assess NEP of forest ecosystems showed that there may be discrepancies between the different NEP estimates. The magnitude and the sign of the difference between NEP assessments varied depending on the type, age and topography of forest stands (see examples given in Black *et al.* 2005). According to Barford *et al.* (2001), C budgets determined by the mass-balance approach should not be expected to reconcile with C sequestration rates assessed by the eddy covariance method in a single year, due to annual shifts in C fluxes. These authors state that several years are required to determine mean rates of C sequestration using either mass-balance or eddy covariance methods.

Due to practical constraints, it is almost impossible to obtain reliable data on carbon fluxes in the experimental zone of the Aelmoeseneie forest with the eddy covariance technique, because of a limited fetch. Using the ecological method (eq. 8.2), carbon fluxes have to be measured frequently in the field, which makes this method impractical for long-term assessment of NEP. We applied the mass-balance method to determine NEP of the two stands in the experimental zone. This approach quantifies the change in carbon stocks of the ecosystem, including growth of woody biomass and changes in soil carbon stocks (Schulze 2000). This method, however, is time-consuming, as it asks for a large number of measurements. Repeated measurements, if they occur at all, are generally limited to 10 year intervals (Houghton 2005). Here, we discuss a number of factors contributing to the uncertainty on NEP assessments based on the mass-balance method.

Coomes *et al.* (2002) stated that the precision of total C stock (change) estimates can be improved by increasing the sample size of the component with largest variance (biomass, dead wood, soil), either by increasing the number of plots, or the size of the plots. Brown (2002) mentioned that the sampling error accounted for 90 to 99 % of the overall error in C stock change calculations, while the measurement and regression error were much less important. The sampling error related to the determination of changes in biomass carbon stocks was reduced to a minimum in our study, as we performed a full inventory, and all trees were measured. Error propagation revealed a relative uncertainty on total stem biomass of only 0.5 and 0.6 % in OB and AS, respectively, as can be calculated from results presented in Table 8.4. Relative uncertainty on total above- and belowground biomass carbon stocks (TABGC) equalled 0.7 % for OB and 0.9 % for AS (Table 8.4). Errors on total stem biomass changes ranged from 4.0 to 6.4 % (Table 8.5). Higher errors, ranging from 8.0 to 11.1 % (Table 8.6), were found for living biomass carbon stock changes.

Carbon is accruing in forests not only in standing living wood, but also in other above- and belowground C pools, including standing dead wood, forest floor litter, coarse woody debris and soil organic matter (Curtis *et al.* 2002). In our study, standing dead trees were taken into account, as was the change in forest floor litter. The change in coarse woody debris was also determined for both stands of the experimental zone in the Aelmoeseneie forest. We applied wood density values, biomass expansion factors *s.s.* and carbon content values of living trees for (standing and lying) dead trees too. Using identical WD, BEF and CC values for dead and living wood introduces an additional error in NEP assessments, as, for example, wood density of dead wood differs from WD of living wood (Harmon *et al.* 2004). However, BEF *s.l.* values of dead wood are hardly known (Brown 2002), and the error introduced by this calculation procedure was not taken into account.

Carbon sequestration in the shrub layer was neglected in our study. As discussed in Chapter 7, this compartment only represented a very small carbon stock compared to the tree layer (< 5 %), and it can be doubted whether the change in carbon stock in this shrub compartment is of large significance in the assessment of NEP_{mass}. Clark *et al.* (2001) also stated that aboveground biomass increment in forests is strongly dominated by overstorey trees, and that in most closed forests, aboveground biomass production can be reliably based on the biomass increment by trees above a carefully chosen minimum size, as was done in this study. This means that neglecting the shrub and herb layer is not so important in the light of determining forest NEP.

It is very difficult to decide whether soils are C sinks or sources as changes in soil C pools are difficult to quantify due to small fluxes and large stocks in soils (Schulze 2000, Valentini *et al.* 2000). Soil C stocks are generally considered to be relatively stable in late-successional forests (Curtis *et al.* 2002). Hom (2003) stated that the largest impact on forest soil C stocks over decades to centuries comes from changes in land use and vegetation cover (species composition). From the study of Smith (2004), it appeared that it can be expected that it takes at least 10 years before a significant change in soil organic carbon (SOC) can be observed, even with a minimum detectable difference of 3 % of background SOC, which asks a large number of samples to be taken (> 100), even for fields characterized by a low CV (< 25 %). Even then, carbon inputs to the soil have to be enhanced by at least 15 % compared to the reference situation, implying serious changes in land use or management. No change in soil carbon stock was observed for the oak-beech stand of the Aelmoeseneie forest (Table 8.6). This result was not unexpected, as no land-use change, change in vegetation cover or seriously enhanced carbon input to the soil has occurred in the period 1991-2005 in the Aelmoeseneie forest. Lettens *et al.* (2005) reported a mean annual C stock change of 0.675 t C ha^{-1} year^{-1} in the soil layer from 0 to 30 cm in Belgian forests for the period 1960 to 2000. These authors assumed that soil carbon stocks at deeper soil layers were constant during this period. However, no data on mean forest soil carbon stock changes in Belgian forests for the period 1990-2005 are available. As such, it is difficult to explain the difference between results from both studies, although the fact that the Aelmoeseneie forest is situated on an old forest soil will certainly be one of the main causes of this difference.

8.4.4. Carbon sink strength of the Aelmoeseneie experimental forest

Carbon stocks and carbon sequestration capacities of various forest ecosystems have been reported to vary largely, depending on climate, species, site fertility and sylvicultural regime (Granier *et al.* 2000). Field data indicate that most temperate and boreal forests are significant sinks for C, with soil respiration being the main determinant of the carbon balance in European forests (Valentini *et al.* 2000). Measurements conducted in the framework of the CARBOEUROFLUX network in the period 1995 to 2001 resulted in NEP assessments ranging from 7.20 t C ha^{-1} year^{-1} to -2.20 t C ha^{-1} year^{-1} for a whole range of European forests, as listed by Carrara *et al.* (2003). These authors dedicated the carbon source status of a Scots pine stand in Brasschaat (Belgium) to recent forest management activities, which influenced carbon fluxes in the considered forest. In Table 8.8, results of a number of case studies are given. As can be seen from this overview, NEP values depend on the applied method (mass-balance or ecological approach, eddy covariance technique). As indicated in Table 8.6, NEP values assessed for Gontrode ranged from 2.25 to 3.57 t C ha^{-1} year^{-1}. Uncertainty on litter layer and soil carbon stock changes were much higher than uncertainty on (dead and living) biomass carbon stocks, as can be seen from the same table.

Table 8.8. *Net ecosystem productivity (NEP) reported for different forest ecosystems; N : latitude; EC : eddy covariance method, NEP_{eco} : ecological approach, NEP_{mass} : mass-balance approach; values between brackets indicate the range of NEP values for a specific site.*

Site description	N	Period	Method	NEP (t C ha^{-1} year^{-1})	Reference
Scots pine, 50 years old	62°52'	1999-2002	EC	1.52, 1.01, 1.72, 2.05	Zha *et al.* 2004
Sitka spruce, 14 years old	52°57'	2002-2003	EC	8.90 (8.39-9.44)	Black *et al.* 2005
			NEP_{eco}	9.52 (7.89-11.16)	
			NEP_{mass}	13.02 (11.30-14.75)	
		2003-2004	EC	8.31 (7.69-8.91)	
			NEP_{eco}	9.38 (7.30-11.44)	
Scots pine stand, 70 years old	51°18'	1997-2001	EC	-1.10	Carrara *et al.* 2003
Beech, 30 years old	48°40'	1996	EC	2.18	Granier *et al.* 2000
		1997	EC	2.57	
5 mixed deciduous forests, 50 to 120 years old	35°37' to 45°47'		EC	1.67 to 5.77	Curtis *et al.* 2002
			NEP_{eco}	0.73 to 3.54	
			NEP_{mass}	1.75 to 3.20	
Northern hardwood forest	42°30'	9 years mean	EC	2.00	Barford *et al.* 2001
		8 years mean	NEP_{mass}	1.60	
Mixed deciduous forest, 60 to 80 years old	39°19'	1998	EC	2.37	Ehman *et al.* 2002
			NEP_{eco}	2.71	
		1999	EC	2.71	
			NEP_{eco}	3.77	

NEP values obtained for Gontrode were close to the mean C sequestration strength of 2.8 t C ha^{-1} year^{-1} calculated for all Flemish forests in the period 1990-2000 (see § 10.4.3). Nabuurs and Mohren (1993a) assessed mean stem wood increment of Dutch forests at 9 m^{3} ha^{-1} year^{-1} in the period 1984-1989 on average, which is approximately 50 % higher than TSW increment values assessed for Gontrode (Table 8.5). However, mean C sequestration in Dutch forests was only 0.97 t C ha^{-1} year^{-1}, due to an assumed harvest of 50 % of the annual TSW increment (Nabuurs and Mohren 1993a). As no wood was harvested in Gontrode, carbon sequestration was higher than the value given for the Dutch forests (Table 8.6). Results for Gontrode confirmed the conclusion of Barford *et al.* (2001) and Kolari *et al.* (2004) that mid-successional forests are mainly acting as a carbon sink, especially due to carbon fixation in the woody compartment of the forest ecosystem.

8.5. Conclusion

The two mixed deciduous stands of the mid-successional forest Aelmoeseneie studied in this investigation appeared to act as a carbon sink during the period 1991 to 2005. This result added to the accumulating evidence that many terrestrial ecosystems are not at biomass equilibrium (Coomes *et al.* 2002). It was also concluded that changing climatic conditions (air temperature, precipitation) had a larger impact on forest growth than differences in elemental deposition.

Net ecosystem productivity assessments were based on the mass-balance approach in this study. This approach offers a good opportunity for long-term carbon balance research. As three full inventories of the experimental zone were available, sampling errors were minimized, and tree biomass increment could be calculated by tracking all individual trees. Changes in dead wood carbon stocks were also determined, while the shrub layer was neglected. It is doubtful whether this latter pool has a large impact on the overall NEP; however, additional measurements could be executed to confirm this statement. Measurements revealed that soil carbon stocks did not change during the studied period. This result was not unexpected, as the Aelmoeseneie forest is situated on a site which has been covered by forest for several centuries, and no drastic changes in forest management were applied during the 15 years study period.

Chapter 9

Growing stock-based assessment of the carbon stock in the Belgian forest biomass

Adapted from : Vande Walle, I., N. Van Camp, D. Perrin, R. Lemeur, K. Verheyen, B. van Wesemael and E. Laitat. 2005. Growing stock-based assessment of the carbon stock in the Belgian forest biomass. Ann. Forest Sci. 62:853-864.

Abstract

Belgian forests covered 693181 ha in 2000, representing 22.7 % of the total land area. As no biomass or carbon (C) stock data are included in the Flemish and Walloon regional forest inventories, species-specific wood densities, biomass expansion factors *s.s.* and carbon content values were critically selected from literature. Based on these conversion and expansion factors, and on data from the forest inventories, the total C stock in the living biomass of productive Belgian forests was assessed at 60.9 Mt C in the year 2000. 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}). The conversion from wood volume to wood biomass based on wood density values reported in literature appeared to introduce the largest variability in the assessment of the carbon stocks. 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 of the determination of biomass expansion factors *s.s.* (BEFs), the establishment of new BEFs does not have the highest priority in the framework of improving the assessment of the biomass carbon stock in the Belgian forests. 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.

9.1. Introduction

All over the world, countries try to fulfil their commitments under the United Nations Framework Convention on Climate Change (UNFCCC), and they seek to achieve the national engagements of the Kyoto Protocol (KP). Countries are allowed to offset their emission reduction targets by increasing biological carbon (C) sequestration in terrestrial ecosystems, as indicated in Art. 3.3 (afforestation, reforestation and deforestation) and Art. 3.4 (additional human-induced activities) of the KP (Schulze *et al.* 2002). Consequently, an increasing interest exists in the accurate measurement of forest carbon stocks (Bolin and Sukumar 2000, Brown 2002, Jalkanen *et al.* 2005). Globally, forests represent important carbon stocks : while only occupying 28 % of the world's area covered by terrestrial ecosystems, they contain 77 % of the carbon stored in the biomass and 39 % of all soil C (Bolin and Sukumar 2000, IPCC 2001).

The Intergovernmental Panel on Climate Change (IPCC) foresees through the so-called 'bottom-up approach' for the calculation of the stock changes over the first commitment period (2008-2012). This approach is based on the use of data available in national or regional forest inventories. Most forest inventories are firstly meant for sound forest management practices and are intended to monitor the wood volumes of economical interest. Biomass measurements are usually not included in the sampling procedure (Coomes *et al.* 2002). An exhaustive review of the various forest inventories conducted by the member states in the European Union and an analysis of their potentials to meet the requirements under the KP were developed in 2000, as represented in a synoptic table by Laitat *et al*. (2000). However, if the inventory-based carbon budgeting method has to be applied, then calculation techniques are needed to convert and expand stem wood volume into total aboveground and belowground carbon stocks (Schroeder *et al*. 1997). If tree-wise data are available, biomass equations can be applied (Van Camp *et al.* 2004, Jalkanen *et al.* 2005). Otherwise, the use of a conversion procedure based on so-called biomass expansion factors *s.l.* (BEFs *s.l.*) is proposed by the IPCC guidelines for the cases where no biomass information is readily available (Houghton *et al.* 1997). Within this context, the question remains which of the conversion and expansion factors reported in literature represent the most suitable values to apply for a specific region or country.

In Belgium, climate policy is formulated at the federal level, requiring cooperation between regional and federal administrations. Belgian forests covered 693181 ha in 2000 (Perrin *et al.* 2000, AB&G 2005a). This represented 22.7 % of the total land surface. The Flemish and the Walloon region used a similar sampling strategy for their forest inventory (Lecomte and Rondeux 1994, AB&G 2001). As is the case for other countries, the Belgian forest inventories are not considering tree biomass or carbon stocks. Therefore, values were selected from literature for : i) wood density per tree species, ii) species-specific expansion factors to calculate total aboveground and belowground biomass starting from stem biomass, and iii) carbon content value for conversion of biomass into carbon stock.

The objectives of this chapter were : i) to critically select biomass expansion factors *s.l.* applicable for the most important tree species in the Belgian forests, ii) to calculate the total carbon stock in the living biomass of the Belgian forests for the year 2000 and iii) to identify the biomass expansion factors *s.l.* which introduce the largest variability in the carbon stock calculations.

9.2. Material and methods

Belgium has a temperate maritime climate, with moderate temperature variability, prevailing westerly winds, heavy cloud cover and regular rain. The definitions of 'forest' in the Belgian inventories are based on minimum requirements : an area of 0.5 ha and 0.3 ha, a width of 25 m and 9 m and a canopy closure of 20 % and 10 % in Flanders and the Walloon provinces (= Wallonia), respectively (Lecomte and Rondeux 1994, AB&G 2001). These slight differences are due to specific aspects of the two regional policies on land-use management. The consequences on the total wood volume, however, are negligible. The distribution of forests in Belgium is shown in Table 9.1. The total forest area in Flanders amounted to 146381 ha in 2000, based on the regional forest mapping (AB&G 2005a), while Walloon forests covered 544800 ha (Perrin *et al.* 2000). The data presented hereafter do not include the forests in the Brussels-Capital Region. Moreover, the study was focussed on productive forests only, and as such, data on non-productive or so-called 'Forests not available for wood supply' or FNAWS (FAO 2000) were excluded from the analysis.

Table 9.1. *Distribution of forests over the three Belgian regions : Flanders, Brussels-Capital and Wallonia; forest cover gives the ratio of the regional forest area to the total regional area.*

Region	Total area (km^2)	Forest area (km^2)	Forest cover (%)	% of the total Belgian forest area
Flanders	13521	1463	10.8	21.1
Brussels Capital	162	20	12.3	0.3
Wallonia	16845	5448	32.3	78.6
Belgium	**30528**	**6931**	**22.7**	**100.0**

9.2.1. The regional forest inventories of Belgium

The sampling points of the regional forest inventories were selected according to a 1.0 km x 0.5 km grid oriented from the east to the west on the National Geographic Institute (NGI) maps at a scale of 1/25000. The rectangular grid had the advantage of going against the orientation of the relief elements oriented along a southwest-northeast axis and against ecological and geological gradients predominant in the N-S orientation. Each grid intersection, located in a forest, represented the centre of a sampling plot. For plots at edges or borders, the plot centre was moved towards the inside of the forest (Lecomte and Rondeux 1994, AB&G 2001).

Sampling plots are circular and of 1000 m^2 each. The following information was collected : category of property (private or public : state, region or province), municipality, forest type, stand structure and development stage, commercial quality for broadleaf species with a section exceeding 22 cm circumference, evidence of damage caused by game and the health and condition for harvest (these two last categories are only available for the Walloon forests). Topography (exposition and slope), soil texture and drainage class, age (class), canopy closure, tree species, circumference at 1.5 m and total and dominant heights were also collected. Basic information in the Flemish and the Walloon inventories was therefore very similar. Moreover, the same volume tables were applied to calculate the total solid wood (TSW) volume from tree circumference and tree height. The terminology 'total solid wood' refers to the combination of stem and branches with a circumference exceeding 22 cm (Dagnelie *et al.* 1999).

In Flanders, 2665 plots were sampled in the framework of the first forest inventory, which was constituted in the period 1997-1999 (AB&G 2001). This regional inventory is intended to be repeated every 10 years, to allow, *e.g.*, the calculation of growth rates in the Flemish forests. The first Walloon forest inventory was completed in 1984. For this study, the current permanent systematic sampling was used. This second inventory started in 1994 and covers each year 10 % of the approximately 11000 sampling points (Lecomte and Rondeux 1994). In 2000 (reference year for this study), 50 % of the sampling points of the second inventory were measured.

With more than 13000 plots over a territory of 30528 km^2, forest inventories in Belgium have one of the highest sampling rates in Europe. Compared to other countries or regions, the Belgian sampling grid, with each sampling point representing 50 ha of forest, is very dense (Laitat *et al.* 2000, Dieter and Elsasser 2002). In comparison, one plot represents 2400 ha of forest land in the U.S. (Brown 2002).

Based on the information of the regional forest inventories, the total area (ha) and the total solid wood volumes (m^3) of different species (groups) were calculated for the Flemish and Walloon productive forests, or so-called forests available for wood supply (FAWS) (FAO 2000). Summation of these values, given in Table 9.2, gives the total area and volume for Belgium. Flanders represented 24.0 % of the Belgian productive forest area, and Wallonia 76.0 %. In total, 75.9 % of the coniferous forest area and 76.2 % of the deciduous forest area were found in Wallonia, compared to 24.1 % and 23.8 %, respectively, for Flanders.

9.2.2. *Biomass expansion factors* s.l.

Calculation of the amount of carbon stored in the biomass of trees is usually based on biomass expansion factors *s.l.* (Nabuurs *et al.* 2000, Brown 2002, Gracia and Sabaté 2002, Bascietto and Scarascia-Mugnozza 2004). Conversion factors are used to calculate amounts of dry mass (t DM) from information on the volume (m^3), or to convert dry mass (t DM) into carbon stock (t C). Expansion factors on the other hand give the possibility to scale up information from a smaller to a higher level, *e.g.*, from stem volume to total aboveground and belowground volume. In this study, 'biomass expansion factors *s.l.*' (BEFs *s.l.*) is used as the collective name for both conversion factors and expansion factors. Three categories of BEFs *s.l.* were distinguished here. In the first place, this concerned wood density (WD), in order to convert fresh wood volume to wood dry mass. Secondly, biomass expansion factors *s.s.* (BEFs) were used to calculate i) aboveground (AG) dry mass from solid wood dry mass, ii) belowground (BG) dry mass from aboveground dry mass, or iii) total aboveground and belowground dry mass from solid wood dry mass. Finally, the carbon content (CC) enabled the conversion from total dry mass to total carbon stock.

Table 9.2. *Area and total solid wood volume for different tree species in Flanders and Wallonia, for the year 2000; information deduced from the Flemish and Walloon forest inventories (AB&G 2001, AB&G 2004, H. Lecomte, pers. comm.)*

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

As the aim of our study was to improve the methodology for the calculation of the total carbon stock in the living biomass of Belgian forests, some explicit conditions were applied for the selection of biomass expansion factors *s.l.* from literature. For the expansion factors *s.s.*, foliage had to be included, in accordance with the IPCC methodology (IPCC 2003). The analysis was limited to data reported for Belgium, Denmark, France, Germany, Great Britain, Ireland and the Netherlands. These countries all belong to the Central-Western European or North-Western European group of countries as indicated in the TBFRA report of the FAO (2000). Values were selected for ten (groups of) species occurring in the Belgian forests : pines (*Pinus* sp.), Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), larches (*Larix* sp.), Norway spruce (*Picea abies* (L.) Karst.), beech (*Fagus sylvatica* L.), oaks (*Quercus robur* L. and *Q. petraea* L.), mixed 'noble' species (including maple (*Acer pseudoplatanus* L.), elms (*Ulmus* sp.), ash (*Fraxinus excelsior* L.) and red oak (*Quercus rubra* L.)) and poplars (*Populus* sp.). Other species were grouped in the 'other coniferous' or 'other deciduous' species class. The intention was to select BEFs *s.l.* for three age classes : 1-20 years, 21-40 years and more than 40 years, and to define a minimum, a maximum and a median value for each category of BEF *s.l.*

9.2.3. Total carbon stock in Belgian forests

The total aboveground and belowground carbon stock for a specific species can be calculated by eq. 9.1 or 9.2 :

$$\text{total (AG + BG) C} = \text{TSW-volume} \cdot \text{WD} \cdot \text{BEF1} \cdot (1 + \text{BEF2}) \cdot \text{CC} \qquad \mathbf{(9.1)}$$

or $$\text{total (AG + BG) C} = \text{TSW-volume} \cdot \text{WD} \cdot \text{BEF3} \cdot \text{CC} \qquad \mathbf{(9.2)}$$

where total (AG + BG) C is total C stock in the aboveground and belowground biomass (t C), TSW-volume is total volume of the solid wood (m^3), WD is wood density (t DM m^{-3}), BEF1 is ratio aboveground dry mass to total solid wood dry mass (t AG DM t^{-1} TSW DM), BEF2 is ratio belowground dry mass to aboveground dry mass (t BG DM t^{-1} AG DM), BEF3 is ratio aboveground and belowground dry mass to total solid wood dry mass (t (AG + BG) DM t^{-1} TSW DM) and CC is carbon content (t C t^{-1} DM).

For all species, species-specific BEF3 values were found in literature. Therefore, the total carbon stock in the Belgian forest biomass was calculated by applying eq. 9.2. Carbon stocks were calculated by species, for Flanders and for Wallonia, and were then summed to give the total C stock for Belgium. To acknowledge the uncertainty linked with the use of biomass expansion factors *s.l.*, C stocks were calculated based on minimum, median and maximum values of the expansion factors *s.l.* found in literature.

9.2.4. Predominant expansion factor category

In a first test, the difference between eq. 9.1 and 9.2 was assessed. For pine, Douglas fir, 'other coniferous', beech and 'other deciduous', species-specific values for BEF1, BEF2 and BEF3 were available (see Table 9.3). For these five (groups of) species, the total carbon stock was calculated by using both eq. 9.1 (with BEF1 and BEF2) and eq. 9.2 (with BEF3), and this for the minimum, median and maximum BEFs *s.l.* The ratio of the result of eq. 9.1 to the result of eq. 9.2, expressed in terms of percentage, represented the difference between both equations.

In a second analysis, eight scenarios were established, to analyse all possible combinations of the minimum and maximum values of WD, BEF and CC (see Table 9.4). For each scenario, the total carbon stock was calculated for each species. The resulting carbon stocks are indicated as Resx, with x referring to the scenario applied. As for all species species-specific values for BEF3 were found in literature, eq. 9.2 was used for this second test.

Table 9.3. *Minimum (min.), maximum (max.) and median (med.) values of biomass expansion factors (BEFs)* s.l. *of different tree species (groups); DM : dry mass; TSW : total solid wood (stems and branches with a diameter > 7 cm); AG : aboveground biomass, foliage included; BG : belowground biomass; 'noble species' : maple, ash, elm and red oak; # gives the number of values found for a particular tree species (group) and expansion factor category; see text for references.*

	Wood density (t DM m^{-3})				**Carbon content (t C t^{-1} DM)**			
Species	***min.***	***max.***	***med.***	**#**	***min.***	***max.***	***med.***	**#**
Pine	0.39	0.60	0.48	13	0.40	0.55	0.50	9
Douglas fir	0.37	0.54	0.45	7	0.50	0.50	0.50	1
Larch	0.41	0.55	0.47	8	0.40	0.50	0.50	3
Spruce	0.34	0.45	0.38	15	0.40	0.51	0.50	5
Other coniferous	0.35	0.50	0.40	20	0.40	0.50	0.50	7
Beech	0.55	0.72	0.56	11	0.44	0.51	0.49	10
Oak	0.50	0.72	0.60	9	0.45	0.50	0.50	3
Mixed noble	0.52	0.69	0.59	9	0.50	0.50	0.50	1
Poplar	0.34	0.55	0.41	48	0.50	0.50	0.50	1
Other deciduous	0.38	0.77	0.55	34	0.45	0.50	0.50	6

	AG / TSW or BEF1 (t DM t^{-1} DM)				**BG / AG or BEF2 (t DM t^{-1} DM)**				**AG + BG / TSW or BEF3 (t DM t^{-1} DM)**			
Species	***min.***	***max.***	***med.***	**#**	***min.***	***max.***	***med.***	**#**	***min.***	***max.***	***med.***	**#**
Pine	1.14	1.40	1.32	5	0.16	0.16	0.16	1	1.43	2.00	1.50	7
Douglas fir	1.18	2.24	1.28	10	0.17	0.17	0.17	1	1.50	2.00	1.71	3
Larch	1.14	1.36	1.30	3					1.50	2.00	1.75	4
Spruce	1.14	1.71	1.29	9					1.50	2.00	1.75	2
Other coniferous	1.14	1.71	1.33	5	0.18	0.25	0.20	3	1.50	2.00	1.75	4
Beech	1.16	2.04	1.34	9	0.23	0.25	0.24	2	1.50	1.75	1.67	3
Oak	1.24	1.39	1.32	2					1.50	1.50	1.50	1
Mixed noble	1.29	1.29	1.29	1					1.50	1.50	1.50	3
Poplar									1.50	1.50	1.50	1
Other deciduous	1.24	1.40	1.32	2	0.20	0.22	0.21	2	1.50	1.50	1.50	1

Table 9.4. *Eight scenarios applied for the impact study of three categories of biomass expansion factors* s.l. *: wood density (WD), biomass expansion factor* s.s. *(BEF) and carbon content (CC)*

Scenario	WD ($t\ DM\ m^{-3}$)	BEF ($t\ DM\ t^{-1}\ DM$)	CC ($t\ C\ t^{-1}\ DM$)
1	Minimum	Minimum	Minimum
2	Minimum	Minimum	Maximum
3	Minimum	Maximum	Minimum
4	Minimum	Maximum	Maximum
5	Maximum	Minimum	Minimum
6	Maximum	Minimum	Maximum
7	Maximum	Maximum	Minimum
8	Maximum	Maximum	Maximum

Impact factors of wood density (IM-WD), biomass expansion factor *s.s.* (IM-BEF) and carbon content (IM-CC) on the total carbon stock for Belgium as a whole, for all conifers and all deciduous species, and for a specific species were calculated as follows :

$$IM - WD = \left[\frac{\frac{Res5}{Res1} + \frac{Res6}{Res2} + \frac{Res7}{Res3} + \frac{Res8}{Res4}}{4} \cdot 100 \right] - 100 \qquad \textbf{(9.3)}$$

$$IM - BEF = \left[\frac{\frac{Res3}{Res1} + \frac{Res4}{Res2} + \frac{Res7}{Res5} + \frac{Res8}{Res6}}{4} \cdot 100 \right] - 100 \qquad \textbf{(9.4)}$$

$$IM - CC = \left[\frac{\frac{Res2}{Res1} + \frac{Res4}{Res3} + \frac{Res6}{Res5} + \frac{Res8}{Res7}}{4} \cdot 100 \right] - 100 \qquad \textbf{(9.5)}.$$

Impact factors calculated with eq. 9.3, 9.4 and 9.5 gave the mean ratio between the results obtained with the maximum and minimum values of the expansion factor category under consideration, expressed in terms of percentage. These impact factors were then used to find the expansion factor category introducing the largest variability in the calculation of the total C stock.

9.3. Results

9.3.1. Biomass expansion factors s.l.

Table 9.3 gives an overview of the selected minimum, maximum and median biomass expansion factors *s.l.* for each species (group). In addition, the number of available values for each species and each expansion factor category are indicated. These BEFs *s.l.* fulfilled the conditions stated above, and were reported by (at least one of) the following authors : Baritz and Strich (2000), Bartelink (1996 and 1997), COST-E21 (2001), Dieter and Elsasser (2002), Grote (2002), Guilley *et al*. (1999), Houghton *et al*. (1997), IPCC (2003), Janssens *et al.* (1999), Joosten and Schulte (2002), Joosten *et al*. (2004), Lebaube *et al.* (2000), Lefèvre *et al*. (2000), Levy *et al*. (2004), Löwe *et al.* (2000), Milne and Brown (1997), Mund *et al*. (2002), Nabuurs *et al.* (2000), Pignard *et al.* (2000), Ponette *et al*. (2001), Pontailler *et al.* (1997), Schalck *et al*. (1978), Vande Walle and Lemeur (2001) and Vande Walle *et al*. (2001). BEFs *s.l.* from other studies that were inconsistent with (one of) the above-mentioned selection conditions (*e.g.*, Brown 2002, Lehtonen *et al.* 2004), were not taken into consideration. As can be seen from Table 9.3, no age classes were distinguished in the end, although it is widely known (*e.g.*, Mund *et al.* 2002, Porté *et al.* 2002, Lehtonen *et al.* 2004, Jalkanen *et al.* 2005) that BEFs *s.l.* are age-dependent. However, we couldn't find enough species-specific BEF *s.l.* values with a clear indication of tree age to make the distinction of three age classes meaningful in the framework of this study.

From Fig. 9.1, it can be seen that wood density (WD) values were in general significantly lower for coniferous than for deciduous tree species, except for poplar. Minimum values ranged from 0.34 t DM m^{-3} for spruce and poplar to 0.55 t DM m^{-3} for beech (Table 9.3). The range of the maximum values spanned from 0.45 t DM m^{-3} for spruce to 0.77 t DM m^{-3} for 'other deciduous' species. Median values varied between 0.38 t DM m^{-3} for spruce and 0.60 t DM m^{-3} for oak.

For all species except poplar, species-specific values were found for the first biomass expansion factor *s.s.*, the ratio of the aboveground biomass to total solid wood biomass (BEF1). Boxplots of the BEF1 values are shown in Fig. 9.2 for those species for which at least 5 values were available. Minimum values ranged from 1.14 t DM t^{-1} DM for pine, larch, spruce and other conifers, to 1.29 t DM t^{-1} DM for noble species (Table 9.3). While the range of the maximum BEF1 values was rather large, from 1.29 t DM t^{-1} DM for mixed noble species to 2.24 DM t^{-1} DM for Douglas fir, the median values for all species ranged only from 1.28 t DM t^{-1} DM for Douglas fir to 1.34 t DM t^{-1} DM for beech. For the ratio belowground biomass to aboveground biomass (BEF2), values were found only for 5 of the 10 species categories considered. A review study for all major biomes (Cairns *et al.* 1997) showed that most values for BEF2 are found to be between 0.20 and 0.30. Values found here fitted the lower half of this range, or were lower. For each of the selected species, values were found for the ratio total (aboveground and belowground) biomass to solid wood biomass (BEF3). Median values for BEF3 were in general higher for coniferous than for deciduous species (Table 9.3), except for pine and beech.

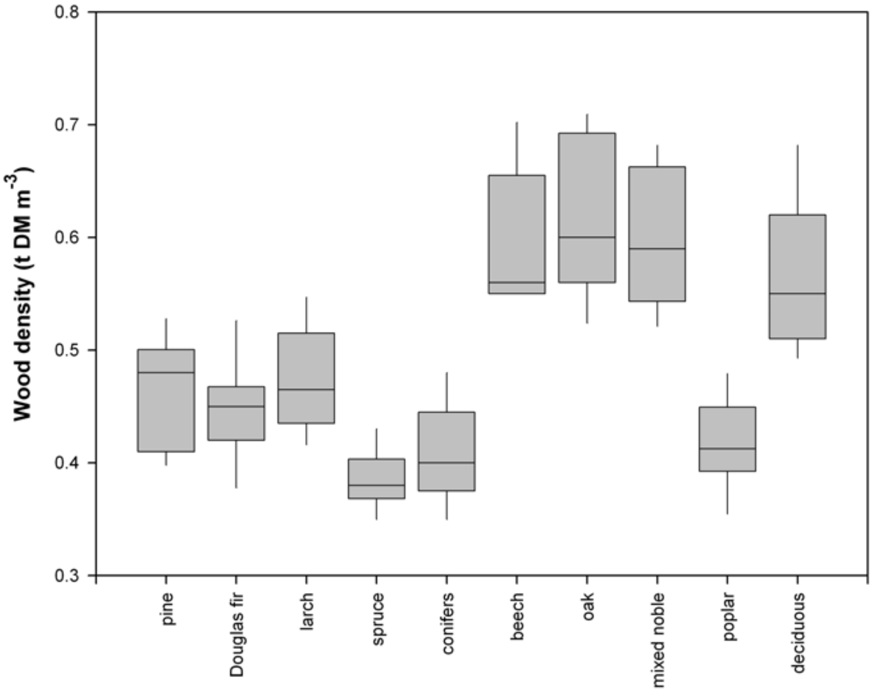

Figure 9.1. *Boxplots of wood density values for different tree species (groups); the horizontal line within each box represents the median value for each species (group). Each box contains 50 % of the observed values within the limits of the first and the third quartile. Error bars indicate 10 and 90 % quartiles.*

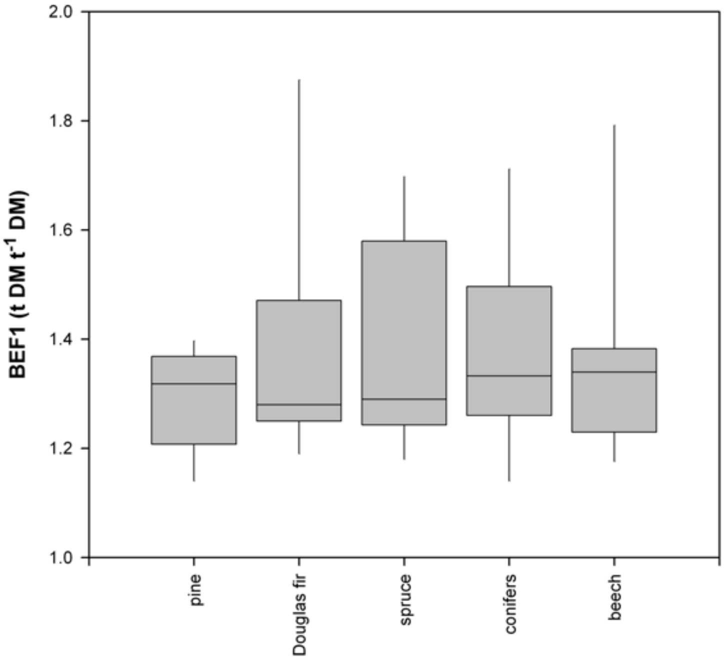

Figure 9.2. *Boxplots (see Fig. 9.1 for detailed description) of BEF1 values for different tree species (groups); only species (groups) for which more than 5 BEF1 values were available are presented.*

The median values for carbon content, graphically presented in Fig. 9.3, were 0.50 t C t^{-1} DM for all species, except beech, which had a median value of 0.49 t C t^{-1} DM. Remarkable is the high range of CC values reported for coniferous tree species (Table 9.3).

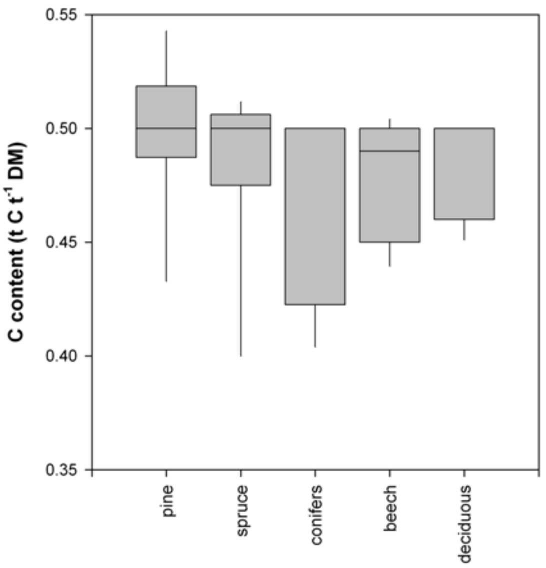

Figure 9.3. *Boxplots (see Fig. 9.1 for detailed description) of carbon content values for different tree species (groups); only species (groups) for which more than 5 CC values were available are presented.*

9.3.2. Total carbon stocks in the Belgian forest biomass

The total carbon stocks in the living biomass of the Flemish and the Walloon productive forests are given in Table 9.5. Summation of the Flemish and Walloon stocks gave the total carbon stock in the productive forests of Belgium for the year 2000, presented in the same table. Based on the use of median expansion factors for all species, the total carbon stock in Belgian forests amounted to 60.9 Mt C. In total, 20.2 % of the Belgian living forest biomass carbon stock was located in the Flemish forests, while the Walloon forests contained 79.8 % of the forest carbon of Belgium. The minimum BEF *s.l.* scenario resulted in a value of 42.8 Mt C for the total carbon stock in Belgium, while the maximum scenario result amounted to 83.5 Mt C.

Based on the total carbon stocks resulting from the median expansion factor scenario on the one hand (Table 9.5), and the total forest area in both Flanders and Wallonia on the other hand (Table 9.2), the mean carbon stock per area unit was calculated. In Flanders, productive forests contained 85.2 t C ha^{-1} on average. The mean value for the Walloon forests amounted to 105.9 t C ha^{-1}. When all Belgian productive forests were considered together, a mean carbon stock of 101.0 t C ha^{-1} was found.

Table 9.5. *Total carbon stock (Mt C) in the living biomass of Belgian productive forests in the year 2000; calculations based on minimum, median and maximum biomass expansion factors* s.l.

	Minimum	**Median**	**Maximum**
Flanders			
Coniferous	3.3	5.3	9.5
Deciduous	5.3	6.9	9.0
Coppice	0.06	0.09	0.13
Total Flanders	**8.7**	**12.3**	**18.6**
Wallonia			
Coniferous	13.8	22.3	31.5
Deciduous	18.5	23.9	30.4
Coppice	1.8	2.4	3.0
Total Wallonia	**34.1**	**48.6**	**64.9**
Belgium			
Coniferous	17.1	27.6	41.0
Deciduous	23.9	30.8	39.4
Coppice	1.9	2.5	3.1
Total Belgium	**42.8**	**60.9**	**83.5**

9.3.3. Predominant expansion factor category

The results of the first test, comparing the application of eq. 9.1 and eq. 9.2 for five (groups of) species, are given in Table 9.6. The ratios were lower than 100 % for the minimum BEF *s.l.* scenario and higher than 100 % for the maximum scenario (except for pine). When the median BEF *s.l.* values were used, both ratios lower (Douglas fir, other coniferous species and beech) and higher (pine and other deciduous species) than 100 % were found.

Table 9.6. *Comparison of the carbon stock calculated by eq. 9.1 (using BEF1 and BEF2) and eq. 9.2 (based on BEF3); ratios in %, giving the outcome of (result eq. 9.1 / result eq. 9.2) . 100*

	Minimum	**Median**	**Maximum**
Pine	92.5	101.9	81.0
Douglas fir	92.0	87.6	131.0
Other coniferous	89.7	91.4	107.0
Beech	95.1	99.5	145.7
Other deciduous	99.2	106.5	113.9

The impact factors for wood density, biomass expansion factor *s.s.* and carbon content calculated with eq. 9.3, 9.4 and 9.5 are given in Table 9.7. Wood density had the highest impact factor regarding the calculation of the total C stock in Belgium (42.9 %), while the impact factors of the carbon content (17.4 %) and the biomass expansion factors *s.s.* (17.1 %) were comparable. When coniferous and deciduous trees were considered separately, it became clear that for both species types, WD had the highest IM : 38.5 % for coniferous trees, and 46.8 % for deciduous species. The IM of BEF3 was almost ten times higher for conifers than for deciduous species (34.9 % and 3.7 %, respectively). Moreover, the IM of the carbon content for coniferous species (28.6 %) was more than threefold the IM found for deciduous species (8.5 %).

The impact factors of wood density, biomass expansion factors *s.s.* and carbon content were also calculated for the tree species (groups) separately (Table 9.7). In some cases, the minimum and maximum values of the parameters were the same (see Table 9.3), which resulted in an impact factor equal to zero. Wood density had the highest impact factor regarding the C stock assessment of all species (groups) except spruce. For spruce, the IM of BEF3 was slightly higher (33.3 %) than the IM of wood density (32.4 %). From Table 9.7, it can also be seen that the impact factor of BEF3 was higher for the coniferous species than for the deciduous species. However, for three of the five groups of deciduous species, only one species-specific BEF3 value was found.

Table 9.7. *Impact factor (IM) of wood density (WD), biomass expansion factor* s.s. *(BEF) and carbon content (CC) on the total carbon stock of Belgian forests; [1] indicates that only one value was available for this BEF* s.l. *category.*

	IM-WD	**IM-BEF**	**IM-CC**
Belgium	42.9	17.1	17.4
Coniferous	38.6	34.9	28.6
Deciduous	46.9	3.7	8.5
Spruce	32.5	33.3	27.7
Douglas fir	46.0	33.3	0.0[1]
Larch	34.2	33.3	25.0
Pine	53.9	40.0	38.0
Other coniferous	42.9	33.3	25.0
Beech	30.9	16.7	15.9
Oak	44.0	0.0[1]	11.1
'Noble' species	32.7	0.0	0.0[1]
Poplar	61.8	0.0[1]	0.0[1]
Other deciduous	102.6	0.0[1]	11.1

9.4. Discussion

For the calculation of the total carbon stock in the Belgian forest biomass, eq. 9.2 was used. In this study, our strict selection conditions for biomass expansion factors *s.l.* should sustain the assumption that the carbon stock assessed by applying these BEFs *s.l.* is reasonable for Belgium. This means that the carbon stock in the living biomass of the Belgian forests in the year 2000 amounted to 60.9 Mt C, or was at least expected to fall within the range between 42.8 Mt C and 83.5 Mt C. Shrubs and very small trees were not included in this value. Schroeder *et al*. (1997) reported that small trees contained as much as 75 % of the biomass in trees with a diameter at breast height greater than 10 cm, in stands with a low aboveground biomass stock. However, it can be assumed that in most Belgian forests, this shrub and small tree pool represents only a very small carbon stock (Lecomte and Rondeux 1994, AB&G 2001). Another pool that was neglected in this study, was the dead wood in the forests. In the Flemish and Walloon forest inventories, information on dimensions of standing and lying dead wood is available. However, the decomposition phase of the dead wood is not noted. This information is crucial to assess the dead wood density from literature values (Brown 2002, Coomes *et al.* 2002). A specific study on the determination of the C stock in this dead wood compartment is therefore needed.

In Flanders, 49.3 % of the forests are younger than 40 years, while in Wallonia, only 26.1 % have an age lower than 40 years. This explains the lower mean carbon stock per area unit in Flanders (85.2 t C ha^{-1}) compared to Wallonia (105.9 t C ha^{-1}). This lower carbon stock is also linked with a lower mean standing volume in the Flemish forests (222.8 m^3 ha^{-1}), compared to the Walloon forests (274.1 m^3 ha^{-1}). Mean standing volumes and mean carbon stocks for neighbouring countries are listed in Table 9.8. The mean carbon stock in the biomass of Belgian forests, 101.0 t C ha^{-1}, was considerably higher than the values reported for Great Britain (36.8 t C ha^{-1}), Ireland (38.3 t C ha^{-1}), Denmark (56.6 t C ha^{-1}), the Netherlands and France (both 59.0 t C ha^{-1}). German forests on the other hand had a carbon stock of 105.7 t C ha^{-1}, which is slightly higher than the value for Belgium. There are several explanations for the differences in mean carbon stock. The main reason is the difference in mean standing volume, which can on its turn be due to various causes. Forests in Ireland for example are mainly planted since 1950. This results in a predominance of young forests, mainly on peat soils, with a related low standing stock (Byrne and Perks 2000). The lower standing stock in Dutch forests compared to Belgium can be attributed to the fact that forests in the Netherlands are mainly concentrated on poor, dry sandy soils (Nabuurs and Mohren 1993a), while in Belgium, forests are located on richer sandy-loam and loamy soils too. The close agreement between the standing stock in German productive forests (276.0 m^3 ha^{-1}) and in Wallonia (274.1 m^3 ha^{-1}) is reflected in an almost identical mean C stock (105.7 t C ha^{-1} for Germany, 105.9 t C ha^{-1} for Wallonia).

Table 9.8. *Total solid wood volume (TSW) and mean carbon stock per ha forest area in Belgium and neighbouring countries*

	TSW (m^3 ha^{-1})	**C stock (t C ha^{-1})**	**Reference**
Great Britain (G.B.)		36.8	Milne and Brown (1997)
conifers		21.1	Milne and Brown (1997)
deciduous		61.9	Milne and Brown (1997)
Ireland	152.4	38.3	Gallagher *et al.* (2004)
Denmark	132.4	56.6	Vesterdal (2000)
France	151.0	59.0	IFN (2004) and Pignard *et al*. (2000)
The Netherlands	170.0	59.0	Nabuurs and Mohren (1993a)
Belgium	261.9	101.0	this study
Flanders	*222.8*	*85.2*	this study
Wallonia	*274.1*	*105.9*	this study
Germany	276.0	105.7	Baritz and Strich (2000)

Besides the difference in standing stock, the choice of the BEFs *s.l.* used for the carbon stock calculation explains the differences in mean carbon stock. The solid line in Fig. 9.4 indicates the relationship between standing stock and mean carbon content based on the overall Belgian BEF of 0.39 t C m^{-3} TSW. The overall BEF for the Netherlands (0.35 t C m^{-3} TSW) and Ireland (0.25 t C m^{-3} TSW) were lower than the ones used for the carbon stock assessment in Belgium, Germany and France, which were all close to 0.39 t C m^{-3} TSW. Wood density values used by Nabuurs and Mohren (1993a) were comparable to the minimum values used in our study. The mean carbon stock in Belgium based on the minimum BEF *s.l.* parameter scenario amounted to 70.9 t C ha^{-1}, which is much closer to the Dutch C stock of 59.0 t C ha^{-1}. The overall BEF for Denmark was 0.43 t C m^{-3} TSW, which is 10 % higher than the Belgian one. This high Danish BEF confirms the statement of Vesterdal (2000) that Danish wood densities differ from IPCC default factors because of a lower volume to dry weight ratio.

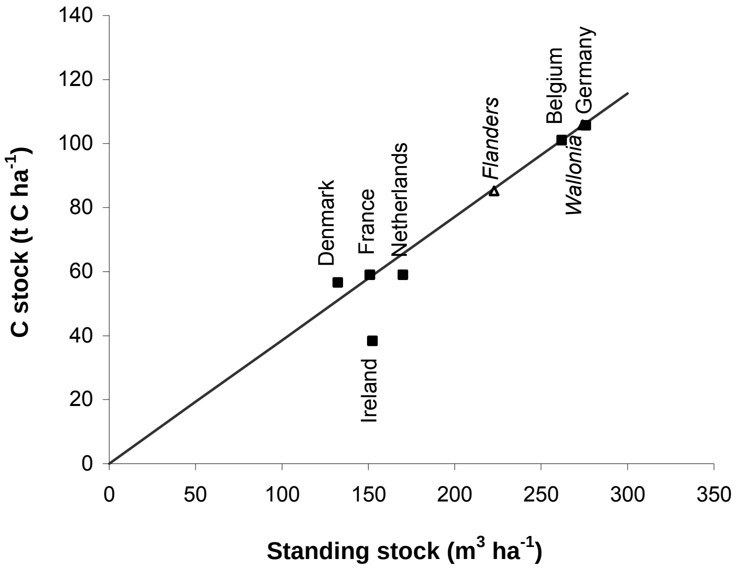

Figure 9.4. *Relationship between standing stock (m^3 ha^{-1}) and carbon stock (t C ha^{-1}) for Belgium and neighbouring countries; the solid line gives the C stock when the overall Belgian BEF* s.l. *is used.*

In our study, shrubs, forest floor vegetation and dead wood are not considered. In the German study of Baritz and Strich (2000), however, these carbon pools were taken into account. Nabuurs and Mohren (1993a) only refer to living biomass, while it is not clear if foliage and fine roots were included in the work of Milne and Brown (1997). The fact that some studies consider more compartments of the forest ecosystems than other studies, can also contribute to differences in mean carbon stocks when starting from the same basic information, being total standing solid wood volume. And even this last concept should be handled with some caution, as the definition of 'total solid wood' is not always the same (Dagnelie *et al.* 1999, FAO 2000).

From our study, it cannot be concluded unambiguous if using eq. 9.1 (with BEF1 and BEF2) always gives lower or higher C stock values than eq. 9.2 (with only BEF3), as ratios were both higher and lower than 100 % for the five (groups of) species studied (Table 9.6). As it is advised by the IPCC (2003) to use species-specific BEF values, it seems appropriate to apply eq. 9.2, as for each species, species-specific BEF3 values are available. However, if one really wants to know the division between aboveground and belowground biomass, eq. 9.1, and as such, BEF1 and BEF2, has to be used. From Table 9.3, it can also be seen that in total, 31 species-specific values for BEF1 were found, compared to only 18 BEF3 values. This could be a reason to prefer to use the first equation, where the 'other coniferous' and the 'other deciduous' species values for BEF2 can be used in these cases where no species-specific values for this parameter were found. However, one should always keep in mind that the choice of the equation influences the final result of the carbon stock assessment.

In general, wood density appeared to introduce the largest variability in the calculation of the carbon stock. This result confirmed the study of Bascietto and Scarascia-Mugnozza (2004), who found that the major contributor to carbon increment error was the variability of the wood density measures, and not the variation in carbon content. In total, we selected 174 wood density values from literature. The range of these values was large, which is due to the geographical dependence of this parameter (Ilic *et al.* 2000, Ketterings *et al.* 2001, Coomes *et al.* 2002, Porté *et al.* 2002). This large range could possibly also be the consequence of the fact that different methods were applied to measure wood density. Information on the measurement method, however, was only rarely available.

The lower impact factors of the biomass expansion factors *s.s.* compared to wood density can be due to the low number of values found for this category, or to the fact that this expansion factor category is less related to the geographical location within Belgium than wood density. This last assumption is confirmed by the study of Cairns *et al*. (1997), who found no apparent relationship between BEF2 and soil texture, precipitation, temperature or latitudinal class. Species-specific values of BEF3 were found for all species, which was not the case for BEF1 and BEF2. Although some sources mention the age-related character of these BEFs (*e.g.*, Schroeder *et al.* 1997, Ponette *et al.* 2001, Wirth *et al.* 2004b), it was not possible to find BEF1, BEF2 or BEF3 values for the three age classes of the distinguished species, because of the explicit conditions applied in our study. Moreover, the volume and age class for which the BEFs were established were not communicated in most cases, a problem also signalled by Van Camp *et al.* (2004).

Generally, the impact factor of the carbon content was only slightly lower than the IM of the BEFs (Table 9.7). For this carbon content category, 46 values were available, compared to only 29 values for BEF3. So, the range of carbon content seemed to be less pronounced than the range of BEFs. As it appeared (Table 9.3) that the median CC value for all species except beech was equal to the default IPCC value of 50 % carbon in dry matter (Houghton *et al.* 1997, Watson *et al.* 2000), it seems appropriate to use this value for all species (groups).

Results of the second test indicated that improving wood density assessment by additional measurements is certainly the most efficient way to improve the C stock assessment in Belgian forests. This means that in an ideal situation, wood density should be measured at the sampling plot level, as this would eliminate the uncertainty due to the geographical variation of this parameter. Our results also confirmed the IPCC guidelines (Houghton *et al.* 1997), which stated clearly that average default wood density values can only be used for initial calculations, and that it is much better to use actual measured average values. However, the method used to determine wood density (X-ray densitometry, volumetric-gravimetric method) should be well-described and harmonized, to eliminate this factor of uncertainty as well. Simpson (1993) emphasized that an exact description of the procedure followed to determine wood density is indispensable for a correct application of published values. A similar nationwide study is going on in Australia, where the aim is to establish a database with wood density values for all relevant tree species (Ilic *et al.* 2000). In the Australian study, the geographic distribution of wood densities is taken into account. The relation between wood density and age of the tree is not so clear, as results from studies are sometimes contradictory to each other (Ilic *et al.* 2000, Porté *et al.* 2002).

Determination of biomass expansion factors *s.s.* is a time-consuming, laborious and destructive activity (Grote 2002, Zianis and Mencuccini 2004). Therefore, the establishment of new BEFs should not have the highest priority for future forest inventories in Belgium. However, more detailed investigations on biomass expansion factors *s.s.* can undoubtedly contribute to the improvement of the forest carbon stock assessment for Belgium, as also mentioned by Van Camp *et al.* (2004).

9.5. Conclusion

In Belgium, data of two regional forest inventories are available. However, these inventories do not provide biomass or carbon stock information. The IPCC guidelines advise to use species-specific BEFs *s.l.* to convert information from forest inventories into carbon stocks. For Belgium, no country-specific biomass expansion factors *s.l.* are available. Therefore, we selected wood densities, biomass expansion factors *s.s.* and carbon content values from literature, applying specific selection criteria. Using the selected species-specific expansion factors, the total carbon stock in the living biomass of the Belgian productive forests was calculated for the year 2000. In a final analysis, the BEF *s.l.* category that introduces the largest variability in the calculation of the C stocks in the Belgian forest biomass was detected.

According to our methodology, the total carbon stock in the living biomass of the Belgian forests amounted to 60.9 Mt C in the year 2000, with a minimum value of 42.8 Mt C and a maximum value of 83.5 Mt C. Shrubs and dead wood were excluded from this calculation. The mean value of 101.1 t C ha^{-1} for the Belgian forests was only slightly lower than the value reported for Germany, and was considerably higher than values for other neighbouring countries as the Netherlands, France, Ireland and Great Britain. These differences were on the one hand due to a different standing stock volume, and on the other hand to another overall BEF *s.l.*

From our analysis, it was not possible to conclude if using two biomass expansion factors *s.s.* (BEF1 and BEF2) results systematically in a higher or lower C stock value than when only one expansion factor *s.s.* (BEF3) is applied. The IPCC guidelines recommend the use of species-specific values instead of more general ones when possible. As such, it is recommended to use eq. 9.2. When one specifically wants to know the partitioning of the carbon stock over the above- and the belowground biomass compartments in detail, two separate BEFs should of course be used.

From a second test, it became clear that wood density introduces the largest variability in the C stock calculations. Therefore, more research is needed concerning this BEF *s.l.* category. Ideally, the wood density should be measured at plot level in the regional forest inventories, by a well-described and harmonized method.

The exact definition of reported biomass expansion factors *s.s.* was sometimes missing, or it was not clear if the foliage was included in the BEFs. These values were not selected for our calculations. A more exact and complete description of biomass expansion factors *s.s.* is therefore strongly recommended. The determination of biomass expansion factors *s.s.* is a laborious and time-consuming activity. Therefore, the establishment of new BEF values is not of the highest priority for improving the assessment of the C stock in Belgian forests. Detailed studies however, could help to reduce the uncertainty related to this type of biomass expansion factor.

The impact factor of the carbon content on the C stock assessment was higher for coniferous than for deciduous species. As the median value for all species (except beech) was equal to the IPCC default value of 50 % carbon in dry mass, we propose to use this value for future carbon stock calculations.

Chapter 10

Carbon sequestration in Belgian forests in the period 1990 to 2000

Adapted from : Vande Walle, I., R. Samson, N. Van Camp, K. Verheyen and R. Lemeur. Carbon sink strength of Belgian forests in the period 1990 to 2000. *Submitted to Annals of Forest Science.*

Abstract

In this chapter, the carbon (C) sequestration in Belgian forest ecosystems in the period 1990 to 2000 was determined. Therefore, a linear back-calculation was performed, starting from the year 2000. Uncertainty analysis on data from Flemish forests revealed that species-specific annual increment rates largely influenced the value of the biomass carbon stock change, while biomass expansion factors had the largest impact on the calculation of the biomass carbon stock. Although data on wood harvest were scarce, this appeared to be of minor importance regarding carbon stock (change) calculations in Flemish forests during the considered decade. The total carbon stock in the biomass of Belgian forests increased from 52.2 Mt C in 1990 to 60.9 Mt C in 2000. Including forest soils, the mean annual carbon sequestration during this decade amounted to 1282.4 kt C year^{-1}. When the loss of carbon due to deforestation was taken into account, the net carbon sequestration by Belgian forests was assessed at 1133.5 kt C year^{-1}, which equalled 3.2 % of total Belgian greenhouse gas emissions in the base year 1990. Despite the fact that Belgian forests were a clear carbon sink in the period 1990 to 2000, and can be expected to act as a sink in the next decades, the Belgian government decided not to use forest carbon sinks under the Art. 3.4 as a means to fulfil its commitments under the Kyoto Protocol. This decision was mainly based on the high verification costs and the small cap assigned to activities under Art. 3.4.

10.1. Introduction

The rising concentration of atmospheric CO_2 and its possible greenhouse effect have heightened interest in carbon (C) cycling at the global level (FAO 2000). Forest ecosystems play an important role in the global carbon cycle, because of their large carbon stocks and the large amounts of carbon exchanged between forests and the atmosphere (Bolin and Sukumar 2000). Forests can help to mitigate climate change by sequestering carbon from the atmosphere in both soil and biomass carbon pools

(Mellilo *et al.* 1995, Winjum and Schroeder 1997). The carbon sequestration capacity of forest ecosystems was acknowledged in the Kyoto Protocol (KP), as direct reference was made to forests in Art. 3.3 (afforestation, reforestation and deforestation) as well as in Art. 3.4 (additional human-induced activities related to changes in greenhouse gas emissions by sources and removals by sinks in the agricultural and the land-use change and forestry categories) (Schulze *et al.* 2002). In Art. 3.4, it is stated that each country that wants to use these activities to meet the commitments under the Kyoto Protocol should establish its level of carbon stocks in 1990, and should estimate the changes in carbon stocks in subsequent years (KP 1997, Schlamadinger and Karjalainen 2000).

Belgium, as one of the countries that ratified the Kyoto Protocol, had to decide whether it would implement activities under Art. 3.4 to reach its commitment of a greenhouse gas emission reduction of 7.5 % in the first commitment period (2008-2012). The first aim of this chapter was to describe in detail the calculation of the carbon stock change in the biomass of Flemish forests in the period 1990-2000, and to discuss the causes of uncertainty on this assessment. Afterwards, results for Flanders were combined with carbon stock changes in the biomass of Walloon forests and in forest soils, in order to assess the total change in carbon stock in the Belgian forests from 1990 to 2000. This change was then compared with the cap on Art. 3.4 activities as fixed in the Kyoto Protocol, as to help policy makers to decide on implementing this article or not. It should be stated here that only productive forests were considered in this study. Non-productive forest areas were excluded from the analysis. Non-productive forest areas refer to water, open spaces, roads etc, which occur within forested areas (BGHGI 2005).

10.2. Methods

In the Methods and the Results section, the main focus was set on the calculation of the carbon stock (change) in the biomass of Flemish forests. A comparable calculation method was applied on data of the Walloon forests (see § 10.2.8). The results for the Walloon forests will not be dealt with in detail, but are included in the Discussion section (§ 10.4.3 and § 10.4.4). The calculation procedure applied basically followed the procedure proposed by the IPCC (Penman *et al.* 2003).

10.2.1. Evolution of total solid wood volume in Flemish forests in the period 1990 to 2000

The total carbon stock in the biomass of Belgian forests was calculated for the year 2000 by Vande Walle *et al.* (2005; see Chapter 9). That study was based on information from the forest mapping of 2000 and the two Belgian regional forest inventories, one for Flanders and one for Wallonia. As for Flanders no data concerning total solid wood (TSW) were directly available for 1990, a linear back-calculation from 2000 onwards was applied to assess the total solid wood volume in the Flemish forests in 1990. The expression 'total solid wood' refers to the combination of stem and branches with a circumference exceeding 22 cm (Dagnelie *et al.* 1999). As was described in Chapter 9, we distinguished ten species (groups) occurring in the Flemish forests : pines (*Pinus* sp.), Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), larches (*Larix* sp.), Norway spruce (*Picea abies* (L.) Karst), beech (*Fagus sylvatica* L.), oaks (*Quercus robur* L. and *Q. petraea* L.), mixed 'noble' species (including maple (*Acer pseudoplatanus* L.), elms (*Ulmus* sp.), ash (*Fraxinus excelsior* L.) and red oak (*Quercus rubra* L.)) and poplars (*Populus* sp.). Other species were grouped in the 'other coniferous' or 'other deciduous' species class. Starting from the year 2000, total solid wood volume of the previous year (n-1) was calculated according to eq. 10.1 :

$$TSW_{n-1} = TSW_n - Incr_n + Harv_n \tag{10.1}$$

where TSW_{n-1} is total solid wood volume at the end of the previous year n-1 (m^3), TSW_n is total solid wood volume at the end of the present year n (m^3), $Incr_n$ is wood increment in the present year n (m^3) and $Harv_n$ is wood harvest in the present year n (m^3). Wood increment was calculated as the total area (ha) of a tree species multiplied by the species-specific mean annual increment (m^3 ha^{-1} $year^{-1}$). This method basically accords to the IPCC Guidelines (Houghton *et al.* 1997). The evolution of the total solid wood volume was calculated for each species (group) separately. Eq. 10.1 was applied until n-1 equalled 1990. As such, summation of the species-specific results gave an assessment of the total solid wood volume in all Flemish forests, for all years from 1990 to 2000.

10.2.2. Total solid wood volume in the year 2000

In Chapter 9 (Table 9.2), an overview is given of the total solid wood volume of the main tree species (groups) in the Flemish productive forests for the year 2000. In total, $32.2 \cdot 10^6$ m^3 of total solid wood was present in these forests. The contributions of the individual tree species (groups) to this volume are listed in Table 10.1.

Table 10.1. *Contribution (%) of tree species (groups) to the total solid wood volume ($32.2 \cdot 10^6$ m^3) of Flemish productive forests in the year 2000*

Species	Contribution	Species	Contribution
Pine	39.9	Beech	7.8
Douglas fir	1.2	Oak	11.5
Larch	2.4	Mixed noble deciduous[1]	7.3
Spruce	1.6	Poplar	16.2
Other coniferous	0.5	Other deciduous	11.6
Total coniferous	45.7	Total deciduous	54.3

[1] *comprising maple (*Acer pseudoplatanus *L.), elms (*Ulmus *sp.), ash (*Fraxinus excelsior *L.) and red oak (*Quercus rubra *L.)*

10.2.3. Total forest area in Flanders

In 1990 as well as in 2000, a forest mapping was executed in Flanders (AB&G 2005a). The mapping of 1990 was based on visual interpretation of colour infrared aerial photographs from the period 1978-1990. The mapping of 2000 was realized by comparing the mapping of 1990 with black-and-white aerial photographs from 1995, in combination with complementary verification on field sites in the period July 1999 to July 2000. To be considered as forest, areas had to satisfy the legal definition of a forest, which implies a minimum area of 0.5 ha, a minimum width of 25 m and a crown closure of at least 20 % (AB&G 2005a). From the two forest mappings, it appeared that the total forest area in Flanders diminished from 152490 ha in 1990 to 146380 ha in 2000. Productive forests accounted for 99 % of the total forest area (Vande Walle *et al.* 2005, Weyembergh *et al.* 2005). Almost no information on forest area was available for years in between 1990 to 2000, except for some scarce data on afforestation and deforestation activities in Flanders in the period 1994 to 2000 (Van Gossum 2005, Van Gossum and Serbruyns 2005). Therefore, a linear decrease of the total forest area in Flanders in the period 1990 to 2000 was assumed.

10.2.4. Area per tree species (group)

In the framework of the calculation of the total carbon stock in the Belgian forest biomass in 2000 (Chapter 9), the area of productive forests per tree species (group) was determined from the contribution of the basal area of a species (group) to the total basal area. The total area of productive forests amounted to 144730 ha in 2000 in Flanders, and was distributed among species (groups) as given in Table 10.2. No detailed information on the contributions of individual species (groups) to the total forest area was available before the first regional forest inventory of 2000 (AB&G 2001). Therefore, we supposed that species' contributions as calculated for the year 2000 (Table 10.2) were valid for all years in the period 1990 to 1999 too. Consequently, a linear decrease in forest area was assumed for all species (groups) in the period 1990 to 2000, according to the overall decrease in productive forest area of 604 ha year^{-1}. As such, the ratios listed in Table 10.2 were applied to calculate the annual evolution of the forest area per tree species (group) from the total forest area in Flanders in the period 1990 to 2000.

10.2.5. Species-specific mean annual increment

As the regional forest inventory has only been executed once in Flanders, no increment data are actually available for this region (Van der Aa *et al.* 2003). Therefore, we used increment data published by Löwe *et al.* (2000) for our back-calculation. These authors provided a comprehensive overview of stem wood increment data from 13 European countries. We selected data from Belgium, Denmark, Germany, the United Kingdom, Ireland and the Netherlands, which all belong to the Central-Western European or North-Western European group of countries as indicated in the TBFRA report of the FAO (2000). The mean annual increment per tree species (group) for these countries is given in Table 10.3. This data set was called 'Europe'. Subsequently, we extracted minimum and maximum increment data from a number of Flemish case studies (Van Den Berge *et al.* 1992, Janssens *et al.* 1999, De Schrijver and Lust 2000, Deckmyn *et al.* 2004b, Vande Walle *et al.* 2006). These Flemish minimum and maximum data sets are also listed in Table 10.3. In general, it is assumed that the mean annual increment in Flemish forests is at least 5 m^3 ha^{-1} (Van der Aa *et al.* 2003 and 2005). Therefore, we also considered a fixed Flemish data set, which means that the annual increment of all species (groups) was considered to equal 5 m^3 ha^{-1} $year^{-1}$ (Table 10.3).

Table 10.2. *Contribution (%) of tree species (groups) to the area of Flemish productive forests in the year 2000*

Species	Contribution	Species	Contribution
Pine	43.9	Beech	5.4
Douglas fir	0.9	Oak	9.9
Larch	2.1	Mixed noble deciduous[1]	7.1
Spruce	2.0	Poplar	15.0
Other coniferous	0.6	Other deciduous	13.2
Total coniferous	49.5	Total deciduous	50.5

[1] *comprising maple (*Acer pseudoplatanus *L.), elms (*Ulmus *sp.), ash (*Fraxinus excelsior *L.) and red oak (*Quercus rubra *L.)*

Table 10.3. *Species-specific mean annual increment (m^3 ha^{-1} $year^{-1}$); Fl. : Flanders, min. : minimum, max. : maximum, n indicates the number of case studies on which minimum and maximum values are based; Wal. : Walloon forests; see text for references.*

Species	Europe	Fl. min.	Fl. max.	n	Fl. fixed	Wal.
Pine	7.0	6.0	14.9	7	5.0	8.8
Douglas fir	12.5	5.9	16.0	2	5.0	23.3
Larch	8.5	3.4	13.6	4	5.0	11.4
Spruce	11.0	10.5	10.5	1	5.0	17.4
Other coniferous	8.3	10.0	10.0	1	5.0	15.5
Beech	7.1	5.2	17.2	10	5.0	6.3
Oak	6.1	3.4	10.0	7	5.0	3.8
Mixed noble deciduous	5.4	5.4	7.9	4	5.0	6.4
Poplar	9.8	13.0	13.0	1	5.0	6.5
Other deciduous	5.4	3.5	5.9	2	5.0	6.4

10.2.6. Wood harvest

For the period 1974 to 1998, only the total annual volume of harvested wood (m^3 $year^{-1}$) is known, for the public forests of Flanders (AB&G 2005b). From 1999 onwards, the contribution of coniferous and deciduous wood is also known. Harvest data for the period 1990 to 2000 are listed in Table 10.4.

In Flanders, 70 % of the forests are private properties (AB&G 2005a). As no information on wood harvest in private forests was available for the period 1990 to 2000, we assumed that the mean harvest per area unit in private forests was similar to the harvest in public forests. As such, total harvest in the Flemish forests was calculated by multiplying the values given in Table 10.4 by a factor 10/3. To apply eq. 10.1, the total harvest was divided over the various tree species (groups) according to their contribution to the total solid wood volume, as listed in Table 10.1.

Table 10.4. *Wood harvest in Flemish public forests in the period 1990 to 2000 (m^3 $year^{-1}$) (AB&G 2005b)*

Year	Coniferous	Deciduous	Total
1990			45600
1991			104210
1992			34475
1993			47047
1994			38613
1995			72447
1996			58607
1997			51256
1998			73514
1999	47733	35381	83114
2000	45617	42879	88496

10.2.7. Annual carbon stock change

Wood volume information in the regional forest inventory as well as increment and harvest data are expressed in m^3 of total solid wood. In the framework of the Kyoto Protocol, results are expected to be expressed in t C of total biomass, comprising total aboveground (stems, branches, foliage) and belowground (roots) compartments (Löwe *et al.* 2000). This means that biomass expansion factors *s.s.* (BEF) have to be applied to scale up the information on total solid wood to the whole tree level, including foliage, branches and roots. Moreover, conversion factors are needed to calculate the amount of carbon stored in total above- and belowground biomass. Therefore, both wood density (WD, t DM m^{-3} wood) and carbon content (CC, t C t^{-1} DM) have to be assessed. For this study, we used results of the extensive literature review performed by Vande Walle *et al.* (2005; see Chapter 9). In that review, biomass expansion factors *s.l.*, comprising wood density, biomass expansion factors *s.s.* and carbon content, for the main tree species (groups) in Belgian forests were selected from reported data for Belgium and other European countries. Species-specific minimum, median and maximum values of wood density (WD), biomass expansion factors *s.s.* (BEF) and carbon content (CC) were available. Multiplying WD by BEF and by CC resulted in an overall BEF. Species-specific overall BEF values based on median, minimum and maximum WD, BEF and CC values are listed in Table 10.5. Multiplying annual changes in total solid wood volume (m^3 $year^{-1}$) by the overall BEF (t C m^{-3}) gave annual changes in carbon stock contained in the total above- and belowground biomass (t C $year^{-1}$). The overall BEF values were also used to convert total solid wood volume (m^3) to total above- and belowground biomass carbon stock (t C).

10.2.8. Calculation procedure and data sources for Walloon forests

A comparable calculation procedure as applied for Flemish productive forests was used to calculate the total C stock in the biomass of Walloon forests in 1990, and to assess the mean annual carbon stock change in these forests between 1990 and 2000. Data on forest area were derived from Perrin *et al.* (2000) and Vande Walle *et al.* (2005). The total area of Walloon productive forests was assumed to decrease from 469090 ha in 1990 to 458700 ha in 2000. Total solid wood volume and species' contributions to TSW were taken from Vande Walle *et al.* (2005), while wood harvest values mentioned by Perrin *et al.* (2000) were used. Species-specific mean annual increments were available for Walloon forests, as two forest inventories have already been executed (Vande Walle *et al*. 2005). These increments were mentioned in the Belgian Greenhouse Gas Inventory for the period 1990 to 2002 (BGHGI 2004), and are listed in Table 10.3. Finally, the median overall BEF factors (Table 10.5) were used to calculate the carbon stock (change) in Walloon forests.

Table 10.5. *Species-specific overall biomass expansion factors (t C m^{-3}) for the considered tree species (groups), used for scaling up total solid wood volume (m^{3}) to total above- and belowground carbon stock (t C)*

Species	Median	Minimum	Maximum
Pine	0.360	0.223	0.665
Douglas fir	0.385	0.278	0.540
Larch	0.411	0.246	0.550
Spruce	0.333	0.204	0.461
Other coniferous	0.350	0.210	0.500
Beech	0.458	0.361	0.639
Oak	0.450	0.338	0.540
Mixed noble deciduous	0.443	0.390	0.518
Poplar	0.308	0.255	0.413
Other deciduous	0.413	0.257	0.578

10.3. Results

10.3.1. Change in area and total solid wood of Flemish forests in the period 1990 to 2000

In Fig. 10.1, the evolution of the total productive forest area in Flanders is depicted. The area decreased at a mean rate of 604 ha year^{-1}, from 150770 ha in 1990 to 144730 ha in 2000. Despite the decline in forest area, the total solid wood volume increased from 23.9 . 10^{6} m^{3} in 1990 to 32.2 . 10^{6} m^{3} in 2000. This estimation is based on the 'European' scenario, which means that the mean annual increment of all species (groups) was set at the European mean value (see Table 10.3). The increase in total solid wood volume corresponded with an increase of the mean TSW volume from 158.8 m^{3} ha^{-1} in 1990 to 222.8 m^{3} ha^{-1} in 2000.

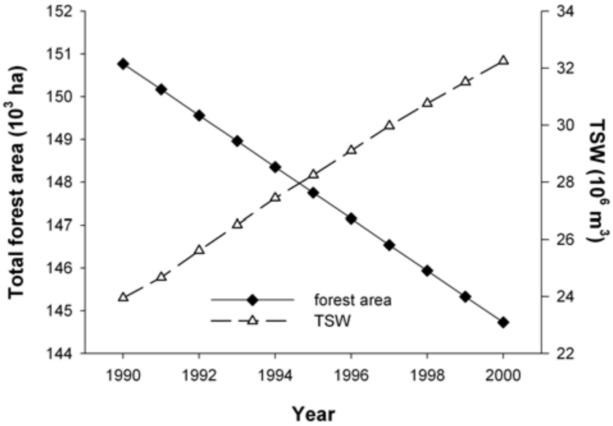

Figure 10.1. *Evolution of total productive forest area and total solid wood volume (TSW) in Flemish forests in the period 1990 to 2000*

10.3.2. Carbon sequestration in Flemish forests in the period 1990 to 2000

Based on the median overall BEF values (Table 10.5), total solid wood volume per tree species (group) was converted to total carbon stock in the above- and belowground biomass. For all species, a net increase in carbon stock was calculated for the period 1990 to 2000, as illustrated in Table 10.6. The total carbon stock in the biomass of Flemish productive forests amounted to 12.34 Mt C in the year 2000 (Table 9.5). Starting from this value, and taking into account the European increment rates (Table 10.3) and the median overall BEF values (Table 10.5), the total biomass carbon stock was found to be 9.25 Mt C in the year 1990. The mean annual carbon stock increase of the Flemish forest biomass was therefore 309.2 kt C year^{-1} in the period 1990 to 2000, almost equally divided over coniferous and deciduous tree species (Table 10.6). As such, the mean biomass carbon stock increased from 61.3 t C ha^{-1} in 1990 to 85.2 t C ha^{-1} in 2000.

Table 10.6. *Carbon stock in the above- and belowground biomass of different species (groups) in 1990 and 2000, and mean annual carbon stock change during this period; results for the Flemish forests*

Species	**C stock 1990 (Mt C)**	**C stock 2000 (Mt C)**	**Mean C stock change (kt C year^{-1})**
Pine	3.33	4.63	130.4
Douglas fir	0.09	0.14	5.3
Larch	0.23	0.32	8.6
Spruce	0.08	0.18	9.4
Other coniferous	0.04	0.06	2.3
Beech	0.96	1.15	18.4
Oak	1.37	1.66	29.3
Mixed noble deciduous	0.86	1.04	18.2
Poplar	1.12	1.60	48.2
Other deciduous	1.16	1.55	39.1
Total	**9.25**	**12.34**	**309.2**

10.4. Discussion

10.4.1. Forest area in Flanders

The Long-term Plan Forestry for Flanders, formulated in 1993 (Vitse 2004), aimed at a forest cover of 12 % in 2010. Afterwards, an increase of 1 % each 5 years would result in a forest index of 30 % in 2100. In 2000, forests covered only 10.8 % of the total area of Flanders (Table 9.1). Within the European Union, only Ireland (8.6 %), the Netherlands (10.0 %) and the United Kingdom (10.3 %) had a lower forest index (AB&G 2005a). The mean forest area per inhabitant in Flanders was only 0.025 ha in 2000 (Dumortier *et al.* 2003). This value is much lower than the mean of 1.3 ha per inhabitant reported for the whole temperate and boreal forest zone (FAO 2000). In the Spatial Structure Plan of the regional government of Flanders, it was stated that 10000 ha of ecologically sound forest expansion was foreseen in the period 1994 to 2007 (RSV 2004). In the period 1990 to 2000, a net decrease of total forest area of 6110 ha was noted (AB&G 2005a). In the period 1994 to 2003, 2051 ha forest has been established (mean of 205 ha year^{-1}), but the deforestation rate amounted to 126 ha year^{-1} in this period. This means that from 1994 to 2003, the net increase in forest area was only 79 ha year^{-1} (Dumortier *et al*. 2005). At this rate, it will take 127 years before the aim of 10000 ha of afforestation will be reached. There are several reasons for this slow afforestation rate. Each deforestation activity should be compensated by an effective afforestation, or by a financial contribution to a so-called forest compensation fund (Dumortier *et al.* 2003). Until now, this fund has not been used to establish new forests, as it is almost impossible to find areas where these forests can be created (Dumortier *et al.* 2005), because of the large pressure on land use in the densely populated area of Flanders. Moreover, the afforestation rate only amounts to 205 ha year^{-1}, in spite of financial subsidies (Vitse 2004). A decrease in value of afforested land restrains landowners to establish forests. It also appears that farmers are not easily inclined to afforest their agricultural land, and if they do so, afforestation is mainly realized on marginal soils (Vitse 2004).

10.4.2. Uncertainties on calculations of biomass C stock (change) in Flemish forests

In order to calculate total biomass carbon stock for the year 1990, and mean annual change in C stock for the period 1990 to 2000, several data sources were used. Each of these sources contributed to the uncertainty on the final result to a different extent. Here, we give an overview of the impact of the different data sources on the carbon stock calculation. We did not aim to perform a quantitative analysis, as data were too scarce to do so. However, we wanted to highlight the most important sources of uncertainty, as this can help to decide on future actions that should be taken to improve the assessment of the carbon stock (change) in Flemish forests.

10.4.2.1. Change in forest area

For 1990 as well as for 2000, a forest mapping was performed, which covered the whole Flemish territory. Therefore, the total forest area can be assumed to be accurately determined. As almost no information on forest area was available for the years in between 1990 and 2000, we assumed a linear decrease in forest area in this period. Actually, this annual decrease was the net result of afforestation and deforestation. More detailed information on the area which is annually afforested or deforested could improve the carbon stock change assessment. Deforestation of mature forest areas results in large carbon losses when the biomass is removed, while the biomass carbon stock on newly afforested land is much lower than in mature forests. This difference could be taken into account when actual afforestation and deforestation rates would be known.

The species' contributions to the total forest area in 1990 were also unknown, as no regional forest inventory was performed before 2000. As such, we assumed these contributions to be equal to the contributions in 2000. Because of the actual forest policy of forest conversion, more deciduous species than coniferous species are used for afforestation (Van der Aa *et al.* 2003, Dumortier *et al.* 2005). As such, the species composition of Flemish forests will have (slightly) changed in the period 1990 to 2000, in favour of deciduous species. However, as forest conversion is a rather slow process (Dumortier *et al.* 2005), the change in species' contributions to the overall forest area can be expected to have only a minor impact on the carbon stock calculations.

10.4.2.2. Species-specific mean annual increment

As mean annual increment values were lacking for Flemish forests (Van der Aa *et al.* 2003), we selected four increment scenarios from published values (Table 10.3). In Fig. 10.2, total biomass C stocks resulting from the European, the minimum, the maximum and the fixed Flemish scenario are given. For all four scenarios, the median overall BEF values (Table 10.5) were applied to convert total solid wood volume in total above- and belowground carbon stock.

It appeared from Fig. 10.2 that the choice of annual increment rates had a considerable impact on the determination of the total C stock in 1990. The minimum and the fixed Flemish scenario resulted in a 6.0 % and 12.0 % higher C stock estimate compared to the European scenario, respectively. When the maximum annual increment rates were taken into account (Flemish maximum scenario), a 31.4 % lower C stock was assessed for 1990 than with the European scenario.

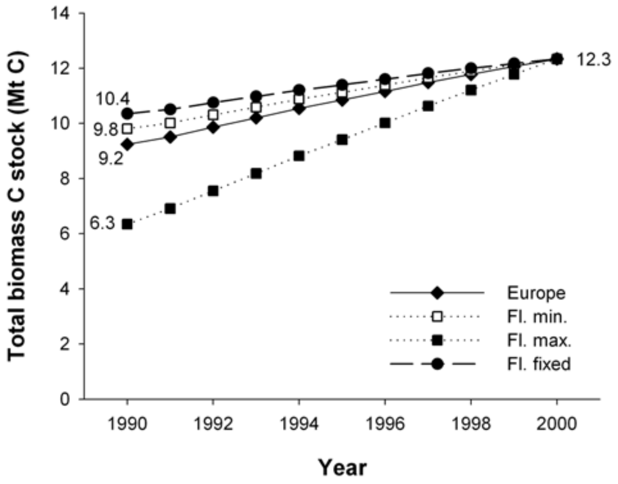

Figure 10.2. *Change in total carbon stock in Flemish forest biomass in the period 1990 to 2000 according to the European, the minimum (min.), maximum (max.) and fixed Flemish (Fl.) scenario; values of total biomass C stock (Mt C) in 1990 and 2000 are indicated on the graph.*

Mean annual carbon stock changes for the period 1990 to 2000 are listed in Table 10.7. Compared to the European scenario, the minimum and fixed increment rate scenarios gave a mean C stock change which was 19.1 % and 35.8 % lower, respectively. On the other hand, using the maximum species-specific increment rates led to a 93.8 % higher C stock change rate than when the mean European increment rates were applied. From these results, it was obvious that more detailed information on the actual increment rates in the Flemish forests will help to optimize calculations of C stock change rates. Mean increment rates will become available for Flanders once the second regional forest inventory (RFI) will be completed. This second RFI is planned for 2007.

Table 10.7. *Mean carbon stock change in Flemish forest biomass in the period 1990 to 2000, according to eight different scenarios; the ratio (in terms of percentage) compared to the European scenario is also given; see text for detailed information on the scenarios.*

Scenario	Increment	Overall BEF	Harvest	Mean C stock change (kt C year^{-1})	Ratio (%)
Europe	Europe	Median	Equal	309.2	100.0
Fl. min.	Min. Flanders	Median	Equal	253.3	81.9
Fl. max.	Max. Flanders	Median	Equal	599.2	193.8
Fl. fixed	Fixed Flanders	Median	Equal	198.4	64.2
Low harv.	Europe	Median	Low	338.2	109.4
High harv.	Europe	Median	High	251.1	81.2
Min. BEF	Europe	Minimum	Equal	213.7	69.1
Max. BEF	Europe	Maximum	Equal	477.5	154.4

10.4.2.3. Wood harvest

In the back-calculation procedure (eq. 10.1), annual wood harvest is taken into account. Data of wood harvest for the period 1990 to 2000 were only available for public forests, as was the case in other European countries too (Dieter and Elsasser 2002, Gallagher *et al.* 2004). In the European scenario, we assumed that the mean harvest per area unit for private forests was equal to the harvest in public forests. Most probably, harvest in Flemish private forests is overestimated in this way (Van Gossum *et al.* 2005). To test the impact of an over- or underestimation of the harvest in private forests, we calculated the total biomass carbon stock supposing that the harvest in private forests was only half of the harvest in public forests ('low harvest scenario'), expressed on an area base. We also tested the consequences of a harvest per area unit which was twice that for public forests ('high harvest scenario'). Results of these calculations are given in Fig. 10.3 and Table 10.7.

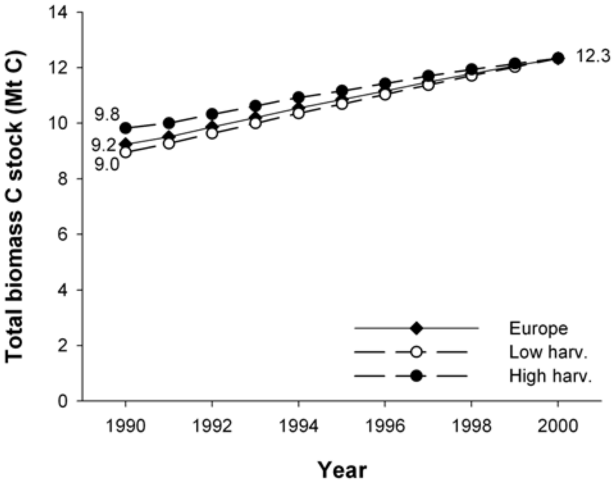

Figure 10.3. *Change in total carbon stock in Flemish forest biomass in the period 1990 to 2000 according to the European, the low harvest (Low harv.) and the high harvest (High harv.) scenario; values of total biomass C stock (Mt C) in 1990 and 2000 are indicated on the graph.*

From Fig. 10.3, it can be seen that the low harvest and the high harvest scenario resulted in a total carbon stock in 1990 which was 3.1 % lower and 6.3 % higher than for the European scenario, respectively. The mean carbon stock change amounted to 338.2 kt C year^{-1} for the low harvest scenario, and 251.1 kt C year^{-1} for the high harvest scenario, which was 9.4 % higher and 18.8 % lower than the European scenario C stock change rate (Table 10.7), respectively. From this test, it appeared that the impact of wood harvest data is not as high as the impact of annual increment rates, although it is not negligible. Therefore, it would be interesting to obtain more detailed information on wood harvest in private forests too. By preference, the contribution of the different species (groups) to the harvested wood volume should be known, as to optimize carbon stock (change) calculations. Actually, a computerized system (IVANHO) is being set up, in order to enable the processing of wood harvest data of public forests in Flanders (Van der Aa *et al.* 2003). Ideally, data on private forests should be gathered in a similar database.

10.4.2.4. Biomass expansion factors s.l.

To investigate the impact of the choice of biomass expansion factors *s.l.* to scale up total solid wood volume to total above- and belowground biomass carbon stock, we used the minimum and maximum overall BEF values (Table 10.5) to convert TSW in total biomass C, while the European annual increment rates were applied (Table 10.3). Fig. 10.4 shows the evolution of the total C stock in the biomass of Flemish forests according to the minimum and maximum BEF scenario, and the European scenario. When minimum BEF values were applied, the total C stock in 1990 was 29.3 % lower in comparison to the European scenario. A 50.0 % higher C stock was found for the maximum BEF scenario. It should be remarked here that the total C stock in 2000 was assessed at 8.7 Mt C and 18.6 Mt C according to the minimum and the maximum BEF scenario, respectively. These values were 29.7 % lower and 51.1 % higher than the value of 12.3 Mt C which was found when the median BEF values were applied (*i.e.*, as was the case in the European scenario).

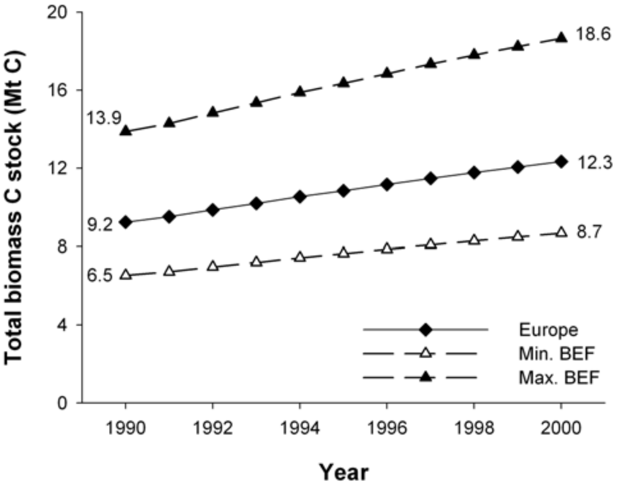

Figure 10.4. *Change in total carbon stock in Flemish forest biomass in the period 1990 to 2000 according to the European, the minimum overall BEF (Min. BEF) and the maximum overall BEF (Max. BEF) scenario; values of total biomass C stock (Mt C) in 1990 and 2000 are indicated on the graph.*

The mean carbon stock change in the period 1990-2000 amounted to 213.7 kt C year^{-1} for the minimum BEF scenario and 477.5 t C year^{-1} for the maximum BEF scenario (Table 10.7). These rates were 30.9 % lower and 54.4 % higher than the mean rate of 309.2 kt C year^{-1} found for the European scenario, respectively. This test showed that the choice of a BEF value had a very high impact on the calculation of the total carbon stock. The impact of BEF on the C stock change rate was also important, but appeared to be lower than the impact of the uncertainty on the increment rates (Table 10.7). The study described in Chapter 9 revealed that especially wood density had a large impact on the calculation of the total forest biomass carbon stock in Belgian forests in the year 2000. Therefore, a more detailed inventory of wood density values, and when possible, of biomass expansion factors *s.s.*, at the Flemish regional level could certainly help to improve the assessment of carbon stocks and carbon stock changes in Flemish forest ecosystems.

Biomass expansion factors for harvested wood volumes differ from those applicable to growing stock, since the age distribution of harvested trees may deviate from the age distribution of all trees in the forest, and because stumps and root systems remain in the forest (Schoene 2006). Moreover, due to new harvesting techniques and new forest management, more harvest residues are now left in the forest than in earlier times (Van der Aa *et al.* 2005). As such, using the overall BEF values to convert harvested total solid wood to carbon stock overestimates the removal of carbon from the forest. BEF values specific for harvested wood could therefore improve the assessment of the real harvested carbon stock. However, BEFs for harvested wood volumes are rarely known (Schoene 2006).

10.4.2.5. Neglected carbon fluxes

In this study, we only took change in forest area, stem increment and annual harvest into account to calculate the change in forest carbon stock (eq. 10.1). Carbon can also be lost naturally, mainly by forest fires (FAO 2000). As forest fires are not common in Flanders, neglecting these carbon losses will have no appreciable effect on the carbon stock (change) assessment.

10.4.3. Carbon stock change in Belgian forest ecosystems

Total C stocks in the forest biomass in 1990 and 2000 are given in Table 10.8 for both Flemish and Walloon productive forests. Results for 1990 were obtained by the back-calculation procedure. According to the European scenario for Flanders, the total C stock in Belgian forest biomass increased from 52.2 Mt C in 1990 to 60.9 Mt C in 2000. The mean annual change in biomass carbon stock was 870.1 kt C year^{-1} for Belgium as a whole (Table 10.8). This value represented 1.5 % of the mean biomass carbon stock in this period (56.6 Mt C).

Table 10.8. *Total biomass carbon stock in 1990 and 2000 and mean annual biomass C stock change during this period for Flemish and Walloon forests; values for 1990 were obtained by the back-calculation procedure; for Flanders, the European scenario was applied (see Table 10.7).*

	1990 (Mt C)	**2000 (Mt C)**	**Mean annual change (kt C year^{-1})**
Flanders	9.2	12.3	309.2
Wallonia	43.0	48.6	560.9
Belgium	52.2	60.9	870.1

In the national greenhouse gas inventory for the period 1990 to 2002 (BGHGI 2004), it was mentioned that the contribution of sinks in Flemish forests to the total carbon absorption capacity of Belgian forests was very limited. However, although Flemish forests only represented 20.4 % of the TSW in Belgian forests and 24.0 % of the total forest area in Belgium (Vande Walle *et al.* 2005), they accounted for 35.5 % of the annual carbon sequestration in forest biomass (Table 10.8). Expressed on an area base, carbon sequestration in forest biomass was much higher in Flanders (2.1 t C ha^{-1} $year^{-1}$) than in Wallonia (1.2 t C ha^{-1} $year^{-1}$). In Flanders, mean TSW increased from 158.8 m^3 ha^{-1} in 1990 to 222.8 m^3 ha^{-1} in 2000, while mean standing TSW increased from 230.8 m^3 ha^{-1} to 274.1 m^3 ha^{-1} in Walloon forests in the same period. This higher increase in TSW in Flanders compared to Wallonia is mainly the result of a much higher wood harvest in Walloon forests. In these latter forests, total wood harvest equalled 66 % of the total growth in the period 1990 to 2000, while in Flanders, harvest was assessed at only 20 % of the total growth. This large difference can be the consequence of the different ages of Flemish and Walloon forests. As stated in Chapter 9, 49.3 % of Flemish forests were younger than 40 years (in 2000), while in Wallonia, only 26.1 % of the forests had an age lower than 40 years. It seems likely that more wood is harvested when forests get older. Nabuurs and Mohren (1993a) assessed the mean carbon sequestration at only 1.0 t C ha^{-1} $year^{-1}$ in Dutch forests, for the period 1984 to 1989. These authors applied species- and site-specific stem volume increment values, which fell within the range of minimum and maximum values considered here (Table 10.3). They also considered stem wood density for the main tree species separately. However, they assumed that branches and roots accumulated no carbon at all, and due to a lack of data, they estimated that half of the actual increment was harvested each year. When this was an overestimation of the real harvest, carbon sequestration was underestimated.

Biomass, dead wood, litter pool as well as soil carbon stocks have to be considered in the framework of the Kyoto Protocol (Noble *et al.* 2000). Lettens *et al.* (2005) assessed the mean change in soil carbon stock from 0 to 30 cm in Belgian forests at 0.675 t C ha^{-1} $year^{-1}$ for the period 1960 to 2000. These authors assumed that soil carbon stocks at deeper soil layers were constant during this period. This value was higher than the 0.5 t C ha^{-1} $year^{-1}$ calculated by Liski *et al.* (2002) as mean change in forest soil carbon stock from 1950 to 1990, for countries from NW-Europe, including Belgium. However, these authors only considered the top 20 cm of the mineral soil, and used a modelling approach instead of measurements. In this study, it was assumed that the carbon stocks in dead wood and litter remained constant during the study period, as was mentioned in the national inventory report of Belgium (BGHGI 2006).

Combining changes in biomass stocks (870.1 kt C $year^{-1}$, Table 10.8) with the change in soil carbon (412.3 kt C $year^{-1}$), the total amount of carbon sequestered by Belgian forests was assessed at 1282.4 kt C $year^{-1}$ in the period 1990 to 2000. This means that in total, 12.8 Mt C was taken up by the Belgian forests during these ten years. As such, it is clear that Belgian forests acted as a sink for carbon in the period 1990 to 2000. Expressed on an area base, mean carbon sequestration in the period 1990 to 2000 amounted to 2.8 t C ha^{-1} $year^{-1}$ and 1.9 t C ha^{-1} $year^{-1}$ for Flemish and Walloon forests, respectively. Janssens *et al.* (2003) mentioned an average sink strength of 0.7 t C ha^{-1} $year^{-1}$ for forests from several European countries. This average sink strength is considerably lower than the value found for Belgian forests. The sandy-loam and loamy soils on which part of the Belgian forests are situated (Vande Walle *et al.* 2005), the low harvest rate and the temperate maritime climate can explain higher annual increments in Belgian forests compared to other European countries. Moreover, the biomass expansion factors *s.l.* used to translate changes in total solid wood volume into carbon sequestration can have caused part of the discrepancy between these Belgian and European carbon sink strengths. About 70 % of the sequestration reported by Janssens *et al.* (2003) was attributed to tree biomass, and 30 % to forest soils. This corresponded well to the situation for Belgium, where the contribution of biomass was 76 % and 64 %, for Flanders and Wallonia, respectively. As such, soils contributed 24 % and 36 % to the total forest carbon sequestration in Flemish and Walloon forests, respectively. Houghton (2005) also stated that carbon sequestration by forest biomass was far more important than the sequestration in forest soils, while Liski *et al*. (2002) found a mean contribution of soils (0-20 cm) to the total carbon sequestration of 40 % for 16 European countries.

10.4.4. Forest carbon sequestration in relation to the Kyoto Protocol

In the period 1990 to 2000, the productive forest area in Flanders as well as in Wallonia decreased (Perrin *et al.* 2000, AB&G 2005a). The annual change in forest area (ha $year^{-1}$) was multiplied by the mean carbon stock per ha (t C ha^{-1}), determined for each year during this period, and for Flemish and Walloon forests separately. This calculation gave the carbon loss occurring because of the removal of tree biomass due to deforestation. For Flanders, this carbon loss was 454.1 kt C in the period 1990-2000, while for Wallonia, 1034.7 kt C was lost in the same period. Soil carbon stock changes after deforestation were not directly accounted for. It was also assumed that aboveground as well as belowground biomass was completely removed from the field during deforestation, which may not always be the case. This latter assumption can level off the neglect of soil carbon stock changes. This means that for Belgium as a whole, a mean carbon loss of 148.9 kt C $year^{-1}$ occurred in the period 1990 to 2000 because of deforestation.

In the period 1990 to 2000, Flemish as well as Walloon forests were strong carbon sinks due to a stem wood increment which largely exceeded wood harvest. As discussed previously (§ 10.4.3), the total amount of C fixed in the Belgian forests, comprising both biomass and soil, amounted to 1282.4 kt C $year^{-1}$. Flemish forests sequestered 408.7 kt C $year^{-1}$ in biomass and soil, while mean C sequestration in Walloon forests was 873.7 kt C $year^{-1}$. Consequently, the net carbon fixation by the Belgian forests, calculated as the difference between C uptake by these forests and

C loss through deforestation, amounted to 11335.2 kt C or 1133.5 kt C year^{-1} from 1990 to 2000. Total Belgian greenhouse gas emissions excluding forests amounted to 145661 kt CO_2 eq. year^{-1} in the base year 1990 (EMIS 2006). Carbon fixation by forests (1282.4 kt C year^{-1}) amounted thus to 3.2 % of these emissions.

Reporting and accounting under the Art. 3.3 of the Kyoto Protocol (afforestation, reforestation and deforestation) is mandatory. On the contrary, each country can decide on using the terrestrial carbon sinks under Art. 3.4 to reach its greenhouse gas emission reduction target. However, only 15 % of the actual carbon sequestration in forests may be declared under the Art. 3.4 of the Kyoto Protocol (Schulze *et al.* 2002). The other 85 % is supposed to represent the effects of elevated CO_2 and nitrogen deposition, and is as such not the consequence of human-induced activities. As discussed in the previous paragraph, Belgian forests sequestered 1282.4 kt C year^{-1}. Under Art. 3.4, only 15 % of this value, or 192.4 kt C year^{-1} may be declared. Moreover, each country was assigned a cap, *i.e.*, maximum amount allowed, on activities under Art. 3.4. For Belgium, this cap amounts to 30 kt C year^{-1} (Schulze *et al*. 2002). This means that Belgian forests were a much larger carbon sink than may be accounted for under Art. 3.4. Despite a positive intention towards the use of forest (and agriculture) management practices under Art. 3.4 of the protocol noted in earlier years (Amano and Sedjo 2003), the Belgian government will not use the possibility of Art. 3.4 activities (OVK 2006), although Belgian forest ecosystems were a clear carbon sink in the period 1990 to 2000, and will most probably be a sink for carbon in the coming decades (Nabuurs *et al.* 2003). The main reasons for this decision are the high costs related to verification of carbon sequestration under Art. 3.4 and the relatively small cap on this C sequestration. As inventories are the basis for accounting, they cannot be used for verification. By preference, an independent top-down approach is needed for verification. A possibility is to measure directly atmospheric CO_2 concentrations (Schulze *et al.* 2002). In the study of Dieter and Elsasser (2004), costs associated with the verification of carbon sink effects of forestry projects turned out to be prohibitive for the use of this type of projects as a means to fulfil commitments under the Kyoto Protocol, unless a simplified verification procedure could be used.

10.5. Conclusion

A lack of species-specific mean annual increment rates for the Flemish forests caused the largest uncertainty on the assessment of the biomass carbon stock change in the period from 1990 to 2000, while the absence of biomass expansion factors *s.l.* for forests in this region was the main cause of uncertainty on the calculation of the biomass carbon stock present in the year 1990. Other possible causes of uncertainty appeared to be of minor importance, although more detailed information on afforestation and deforestation rates and on wood harvest in private forests could help to optimize the assessment of the carbon stock (change) in Flemish forests.

Flemish as well as Belgian forests acted as a carbon sink in the period 1990 to 2000, and there is no reason to assume that the forests will become a source of carbon in the near future, given the fact that Belgian forest policy aims at a sustainable forest management. This includes that wood harvest is not allowed to exceed wood increment (Van der Aa *et al.* 2005). Tree growth accounted for more than two thirds of the total carbon sequestration in Belgian forests, while forest soils were a smaller carbon sink.

Chapter 11

General discussion and conclusions

In the framework of this PhD, carbon (C) sequestration in both short-rotation forestry (SRF) plantations (Chapter 2 to 6) and in Belgian forest ecosystems (Chapter 7 to 10) was studied. Results of the present research have already been discussed extensively in the preceding chapters. Therefore, it is the aim of this concluding chapter to provide a more general discussion of the findings and the implications of the study. As was already mentioned in Chapter 1, results of different chapters are integrated according to five topics. For each topic, final conclusions are formulated, and suggestions for future research are given.

11.1. Establishment and biomass production of short-rotation forestry plantations in Flanders

Poplar and willow are generally considered as the most convenient tree species to be used for short-rotation forestry (Scarascia-Mugnozza *et al.* 1997, Fang *et al.* 1999, Verwijst and Telenius 1999, Proe *et al.* 2002, Laureysens *et al.* 2005). At the plantation in Zwijnaarde, birch and maple were also chosen to be grown under SRF, to test whether these species can offer an alternative for the use of poplar and willow.

Specific leaf area and photosynthetic leaf characteristics, which are important in the upscaling of photosynthesis from leaf to canopy level, were in the same range for birch and poplar (Table 2.1 and 2.3). Leaf senescence started much earlier for poplar than for birch (§ 2.4.1), suggesting that growing season length for these species is quite different. However, more research is needed to test the influence of growing season length on biomass production of the different tree species. Mean actual biomass production after 4 years of tree growth was 1.2, 2.6, 3.4 and 3.5 t dry mass (DM) ha^{-1} year^{-1} for maple, birch, willow and poplar, respectively (Table 3.3). Although the survival rate of maple was very high (Table 3.3), biomass production of this species was very low in this study. Therefore, it can be questioned whether this is a good species for short-rotation forestry. Poplar and willow on the other hand appeared to have the highest biomass production after 4 years of tree growth. However, poplar suffered severely from an infection by *Colletotrichum gloeosporioides*. Deraedt and Ceulemans (1998) also mentioned an infection of poplar with rust and other pathogens in a Flemish SRF plantation. From the regional forest vitality inventory, which has been executed annually in Flemish forests since 1987, it appeared that the percentage of damaged trees is higher for poplar than for any other species (Sioen and Roskams 2003). Mean annual biomass production of poplar and willow decreased after the fourth year, as was discussed in § 6.4.1.1. This reduction can be due to the dry summers in 2003 and 2004. Both poplar and willow are known to be species with a high water use, and optimal pH for growth of these species ranges from 5 to 7.5 (Ledin 1996b, Deraedt and Ceulemans 1998, Mitchell *et al.* 1999, Tahvanainen and Rytkönen 1999). Therefore, the low water availability and the low pH of the sandy soil on which the plantation was situated (§ 1.6.1) can be expected to be suboptimal for the growth of poplar and willow.

Mean annual increment (MAI) of birch increased from 2.7 t C ha^{-1} year^{-1} after 4 years to 3.2 t C ha^{-1} year^{-1} after 5 years (§ 6.4.1.1). This confirmed the suggestion made in Chapter 3 (§ 3.4.4.1), where it was stated that a rotation length of 4 years is too short for birch to reach the maximum biomass production rate. Optimal rotation lengths for birch suggested in literature range from 8 to 12 years (Ledin 1996a, Hytönen and Issakainen 2001). A further follow-up of the biomass production at the plantation is needed to see if MAI of birch will continue to increase during the next years. Birch also appeared to have the highest mean calorific value of the wood (§ 4.3.1), which makes this species interesting towards energy production. Moreover, birches are only moderately high consumers of water, and according to Ferm (1993), this species has a higher drought stress resistance than willow.

Based on the results from this investigation, it was concluded that marginal agricultural soils are suboptimal for the growth of poplar and willow. Birch can be considered as an interesting alternative for the establishment of SRF plantations in Flanders, especially on dry soils. However, the successful application of a tree species for SRF plantation establishment also depends on the easiness to harvest the trees, and on the regeneration capacity of the species. These two characteristics have not been studied in this PhD project, but certainly deserve further investigation.

Biomass production values at Zwijnaarde fell in the lower range of values reported in literature (Table 3.4, 3.5 and 3.6). However, the study site was situated on a sandy soil and individual plots were 400 m^2, whereas in most other studies, smaller plots were used, which can lead to an overestimation of biomass production potential (Pontailler *et al.* 1999). Furthermore, in contrast to most other studies, no weed control, fertilization and/or irrigation was applied at Zwijnaarde. Possibly, applying weed control or fertilization could enhance biomass production. However, this assumption needs more investigation, as results from other studies are sometimes contradictory to each other (Ledin 1996b, Hofmann-Schielle *et al.* 1999, Telenius 1999, Kopp *et al.* 2001, Deckmyn *et al.* 2004a). Using herbicides and fertilizers also increases greenhouse gas emissions during the production and application processes, and it enhances the risk for environmental problems as leaching of nitrogen to the groundwater. Moreover, to gain optimal profit from all functions of SRF plantations, including soil water protection and provision of a (temporary) habitat, it is important to restrict the use of fertilizers and herbicides or pesticides as much as possible.

Until 2006, the legal status of short-rotation forestry plantations in Flanders was unclear (Meiresonne 2005). In the Flemish Forest Decree of 13th June 1990 (FD 1990), it was written that plantations mainly aimed for wood production, such as poplar or willow plantations, were considered as forest, but no explicit reference was made to short-rotation forestry plantations. This lack of judicial clarity seemed to restrict farmers from planting trees under a short-rotation forestry management system, as they feared that these agricultural soils would then be considered as forest. The judicial status of SRF plantations was clarified by the extension of the original Forest Decree by the Flemish government on 19th May 2006 (DFG 2006). Since that date, it is explicitly mentioned that short-rotation forestry plantations are not considered as forest, and as such, do not fall under this decree. Moreover, crop production systems where the aboveground biomass is completely harvested after a certain period, with a maximum of 3 years after planting or after the previous harvest, are also excluded from the Forest Decree since May 2006. Additionally, it can be remarked that maximum crop rotation length is restricted to 8 years by the decree from 19th May 2006 (DFG 2006). By this adaptation of the Forest Decree, there came an end to the lack of clarity on the judicial status of SRF plantations in Flanders. However, the future will have to show to what extent this will help to convince farmers to establish SRF plantations on their (marginal) agricultural soils.

11.2. Biomass carbon stocks in different forest ecosystems

In this research project, a lot of effort was dedicated to the determination of biomass carbon stocks in different forest ecosystems. Results of this assessment were reported in Table 6.2 for the short-rotation plantation at Zwijnaarde, Table 8.4 for the oak-beech (OB) and the ash (AS) stand of the mixed deciduous forest Aelmoeseneie, and in § 9.3.2 for Flemish and Walloon forests. These data are compiled in Table 11.1. It can be remarked here that biomass carbon stocks for the oak-beech and the ash stand of the Aelmoeseneie forest were calculated in Chapter 7 too. Total biomass carbon stocks found in Chapter 7 (Table 7.3) were higher than values obtained in Chapter 8 for the same forest stands (Table 8.4). The carbon stock in the shrub layer was included in calculations of Chapter 7 and not in calculations of Chapter 8, but was far too small (2.6 and 4.9 t C ha^{-1} for OB and AS, respectively) to explain the large differences between the results from the two chapters (34.7 t C ha^{-1} and 21.9 t C ha^{-1} for OB and AS, respectively). These differences can be attributed to the different approaches applied in the two chapters. Conversion of diameter at breast height to total solid wood volume (TSW) was identical (§ 7.2.4.1 and § 8.2.3). However, equations to calculate tree height from circumference at breast height (CBH), which were needed for TSW calculation of beech, maple and larch, were recalculated for Chapter 8, and were of an S-shaped form (eq. 8.4) instead of the cubic function used in Chapter 7 (Vande Walle *et al.* 1998a). This resulted in lower tree heights for calculations in Chapter 8, and consequently, in lower total solid wood volumes, especially for trees with a large CBH. Furthermore, species- and site-specific regression equations were used in Chapter 7 to calculate dry mass of foliage and branches of oak, beech and ash (§ 7.2.4.1), and literature values of root fractions were applied to calculate root biomass (§ 7.2.4.2). In Chapter 8 on the contrary, European biomass expansion factors obtained from an extensive literature study were used to calculate above- and belowground biomass from total solid wood volume for the main tree species of the two stands (§ 8.2.3). Wood density values were identical in Chapter 7 and Chapter 8 for oak, beech and ash, but were lower in Chapter 8 than in Chapter 7 for larch and maple (§ 7.2.4.1 and Table 8.2). Moreover, carbon content values (CC) used in Chapter 8 (Table 8.2) were slightly lower than the CC value of 0.50 used in Chapter 7. These differences in conversion and expansion of identical basic data (inventory of the two forest stands) caused the lower total biomass carbon stocks found in Chapter 8 compared to Chapter 7.

Table 11.1. *Carbon stocks (t C ha^{-1}) in the above- and belowground biomass of different forest ecosystems; SRF : short-rotation forestry plantation at Zwijnaarde, with indication of the number of years of tree growth; OB and AS : oak-beech and ash stand of the Aelmoeseneie experimental forest; total biomass C stocks for Flemish, Walloon and Belgian forests are also given. Carbon stocks in shrubs are not included in these values.*

	Foliage	Stems and branches	Roots	Total
SRF, 1 year	0.09	0.20	0.22	0.51
SRF, 2 years	0.5	1.5	1.5	3.5
SRF, 3 years	0.7	3.1	3.0	6.8
SRF, 4 years	1.0	5.8	5.6	12.4
OB				119.4
AS				126.1
Flanders				85.2
Wallonia				105.9
Belgium				101.0

Data reported in Table 11.1 illustrate the diminishing relative importance of the foliage C stock in the total biomass C stock, starting from 17.6 % after 1 year of tree growth to 8.1 % after 4 years. Roots represented 43.8 % of total biomass C stocks during the first four years of tree growth on average. Biomass expansion factors for mature forests (Table 9.3) showed that root biomass carbon stocks represent between 14 and 20 % of total biomass carbon stocks in mature forests. This indicated that the aboveground biomass C stock becomes more and more important during forest evolution, in contrast to the belowground and the foliage C stocks, which are much more important in the initial phase of forest development.

The mean total biomass C stock in the Aelmoeseneie forest, 122.8 t C ha^{-1}, was considerably higher than the mean value for Flanders, which was only 85.2 t C ha^{-1}. Main causes of the higher carbon stock in the Aelmoeseneie forest are the age of the forest at Gontrode (80 years, whereas approximately half of the Flemish forests are younger than 40 years (§ 9.4)), the sand loamy soil on which the forest is situated, and the absence of wood harvest in the two studied stands of the Aelmoeseneie forest.

Belgian forests have a high average biomass carbon stock compared to forests in neighbouring countries (Fig. 9.4). Differences in biomass carbon stocks can be due to mean forest age, to climatic conditions, or to soil characteristics, as was discussed in § 9.4. However, determination of biomass C stocks relies on the conversion of easily measurable parameters as stem diameter or stem circumference and tree height to amount of carbon. This conversion introduces uncertainty on the assessed C stock value. In this study, both allometric relationships and biomass expansion factors *s.l.* were used for the conversion process.

Allometric relationships were used to convert stem diameter at 30 cm height to aboveground dry mass of birch, maple, poplar and willow trees growing on the SRF plantation at Zwijnaarde (Table 3.1). Harvesting and processing of trees to establish allometric relationships is time-consuming, but it is important to use site-specific as well as species-specific relationships to make the conversion of diameter to biomass as accurate as possible (Wirth *et al.* 2004b). However, it is known that these relationships may change during stand development (Telenius 1999, Porté *et al.* 2002). Therefore, taking new samples to extend the range of tree diameters will be needed when these allometric relationships will be used in the future. As circumference was available for each individual tree, determination of the carbon stocks in trees of the two stands of the Aelmoeseneie experimental forest (§ 7.2.4.1) was based on site-specific allometric relationships (for oak and beech), and on allometric relationships published in literature (beech, larch and maple; Dagnelie *et al.* 1999). The lack of site- or region-specific allometric relationships may cause large difficulties on carbon stock assessments of forested areas, as these relationships may change from one site to another, due to differences in climate or soil characteristics (Nelson *et al.* 1999, Ketterings *et al.* 2001). This can introduce large errors on C stock calculations when relationships of a different region have to be applied. The harvest and processing of mature trees, needed to establish allometric relationships, is not evident. As such, the choice of allometric relationships from literature becomes an important challenge in view of C stock calculations. Even when site- or region-specific allometric relationships are available, it can be doubted whether these can still be applied when they have been established decades ago. These relationships may have changed through time, because of changes in forest growth, caused by increases in site productivity, nitrogen deposition, or changes in silvicultural practices or tree species mixture (Joosten and Schulte 2002).

Calculation of root biomass at the short-rotation plantation in Zwijnaarde was based on root-to-shoot ratios (R:S). Determination of this ratio is even more laborious than the determination of allometric relationships, as roots have to be excavated from the soil before they can be processed. A decrease in R:S with increasing tree age was found for the willow trees at Zwijnaarde (§ 6.2.3.2), in accordance to literature (Giardina and Ryan 2002, Pregitzer 2003). Additional measurements on birch, maple and poplar plots are needed to confirm the assumed decrease in R:S for these species. As excavating root systems of mature trees is a tremendous work, only two root systems were excavated in Gontrode (§ 7.2.4.2). From this work, it was concluded that literature values for R:S ratios could be used for this forest ecosystem. It can be questioned whether taking new measurements are worth the effort to acquire additional information on carbon stored in the root systems of the trees in the Aelmoeseneie forest.

Determination of the total C stock in the Belgian forest biomass was based on data from the two regional forest inventories (§ 9.2.1). As is the case in most countries, no tree-wise data are enclosed in these inventories, and as such, biomass expansion factors *s.l.* (BEFs *s.l.*) were used to convert total solid wood volume (m^3 ha^{-1}) to total above- and belowground carbon stock (t C ha^{-1}), as proposed by the IPCC guidelines (Houghton *et al.* 1997). As no country-specific BEFs *s.l.* data were available for Belgium, an extensive literature study was performed (§ 9.2.2). This resulted in the selection of species-specific wood density, biomass expansion factors *s.s.* (BEFs) and carbon content values which were assumed to be applicable for Belgian forests (Table 9.3). Based on the median conversion and expansion factors, the total carbon stock in the Belgian forest biomass amounted to 60.9 Mt C in the year 2000, of which 12.3 and 48.6 Mt C was stored in the Flemish and the Walloon forests, respectively. Based on minimum and maximum BEFs *s.l.*, this stock amounted to 42.8 and 83.5 Mt C, respectively. This result indicated the importance of the BEF *s.l.* choice for C stock calculations. Each additional research on BEFs *s.l.* can undoubtedly contribute to the improvement of the forest biomass C stock assessment for Belgium. Impact analysis, however, revealed that wood density introduced the largest uncertainty on the C stock calculations (Table 9.7), and, therefore, more research on this characteristic deserves the highest priority for improving carbon stock assessments in Belgian forest biomass. It should also be stressed that a clear description of the applied method to determine wood density is of the utmost importance for the correct application of reported values (Simpson 1993). The demand for an extensive method description also holds for published BEF values, as the lack of this information was the reason for exclusion of many data in the literature review which was performed in Chapter 9. As the median value for carbon content was 50 % for all species (except beech), it seems reasonable to use this IPCC default value (Watson *et al.* 2000) for C stock calculations in Belgian forest ecosystems.

Although it is known that BEFs *s.l.* are age-dependent (Ponette *et al.* 2001, Lehtonen *et al.* 2004, Wirth *et al.* 2004b, Jalkanen *et al.* 2005), no age classes were considered after the extensive literature study (§ 9.3.1). Future research on BEFs *s.l.* should take this age-dependence into account. In Chapter 8, BEFs *s.l.* from the extensive literature study were used for determination of C stocks in dead wood. The lack of BEFs *s.l.* values for dead wood can only be solved by new research programmes, as this information is almost absent in literature (Brown 2002, Harmon *et al.* 2004).

11.3. Evaluation of methodologies to assess the carbon balance of a forest ecosystem

Several methods can be used to determine the carbon balance or net ecosystem productivity (NEP) of a forest ecosystem. Traditionally, the stock change method, or so-called mass-balance method, was used (Baldocchi 2003). This method is based on the determination of changes in carbon stocks in biomass, litter and soil compartments over time (Bolin and Sukumar 2000, Williams *et al.* 2005), and provides as such insight in which forest compartments are storing or losing carbon (Barford *et al.* 2001). More recently, the ecological approach, based on measuring autotrophic and heterotrophic carbon fluxes, is being used for the assessment of the C balance of forest ecosystems (Curtis *et al.* 2002, Black *et al.* 2005). To apply this method, the amount of carbon taken up by the vegetation as well as soil CO_2 effluxes and root respiration have to be assessed. Thirdly, the micrometeorological eddy covariance technique offers the opportunity to assess NEP immediately at the ecosystem level (Baldocchi 2003), and enables interannual comparison of NEP fluxes of a forest ecosystem (Grace 2004). Ecosystem modelling is an alternative way to determine carbon balances of forest ecosystems (Rasse *et al.* 2001, Williams *et al.* 2005, Verbeeck *et al.* 2006). It is difficult to decide which method is the best to be used for NEP calculation of a forest ecosystem. As each methodology has its specific prerequisites to be applicable, the choice of the method differs from study to study, depending on the technical, instrumental and time limitations of the investigation.

The mass-balance method was applied for the determination of NEP at the SRF plantation in Zwijnaarde (eq. 6.4) as well as for NEP calculation of the two stands at Gontrode (eq. 8.1). The simplicity of the measurements makes this method applicable in many situations. However, taking samples is labour-intensive, and as such, repeated measures are often performed with large time intervals (Houghton 2005). Calculation of stock changes in aboveground biomass relies on the determination of easily measurable parameters as stem diameter or circumference, and the choice of appropriate allometric relationships or biomass expansion factors *s.l.* This choice can introduce large uncertainty on the results, as discussed in § 11.2. Difficulties may also emerge when changes in belowground compartments have to be detected. Carbon stocks in root systems are most often calculated from aboveground biomass carbon stocks by using root-to-shoot ratios; however, these R:S ratios differ according to stand age and site characteristics. Determination of changes in soil carbon stocks over short time periods is also problematic, as an enormous number of samples have to be analysed before significant changes can be detected, when no differences in management were applied (Smith 2004). At the plantation in Zwijnaarde, large soil CO_2 effluxes were not compensated by litter inputs from plants, which induced a serious decrease of soil C stocks, even after 4 years (§ 6.4.1.3). On the contrary, as was illustrated for the oak-beech stand at Gontrode, soil carbon stocks did not alter during a period of 8 years in this mature forest (§ 8.3.4).

The ecological approach was used for NEP assessment during the first 5 years of tree growth at the short-rotation plantation at Zwijnaarde (eq. 6.3). This method can be used on shorter time scales than the mass-balance approach. As such, the ecological approach is more suited to study interannual variability of net primary productivity (NPP), soil CO_2 effluxes and NEP. As shown in Table 6.2 and discussed in § 6.4.3, NPP of the short-rotation forestry plantation at Zwijnaarde appeared to increase with stand age, while temperature and precipitation during the leafed period (1st April to 30th September) were the main factors determining interannual variability in soil CO_2 effluxes. The ecological method, however, also has a number of restrictions or disadvantages. As is the case for the mass-balance approach, appropriate allometric equations or biomass expansion factors *s.l.* have to be chosen when the change in biomass C stock is based on easily measurable parameters as stem diameter or circumference. Moreover, soil CO_2 effluxes vary largely both in space and time, as was extensively discussed in Chapter 5. This implies that soil CO_2 efflux measurements should be spread well in time as well as in space to obtain reliable flux estimates. The contribution of root respiration to total soil CO_2 effluxes is another important factor in the calculation of NEP according to the ecological approach. Measurements at the plantation in Zwijnaarde revealed that root respiration only contributed 5 % to total soil CO_2 effluxes. This contribution was much lower than values mentioned in literature for mature forests (Hanson *et al.* 2000). Root respiration fluxes determine the assessment of heterotrophic fluxes when the ecological approach is used. Therefore, more investigation on root respiration fluxes is certainly needed to improve NEP assessments. This investigation should include temperature sensitivity, root dimension as well as species dependence of root respiration. Because of the large number of measurements needed to apply the ecological method, this approach is impractical for long-term assessment of NEP.

Using different methods for NEP calculation of the short-rotation forestry plantation at Zwijnaarde resulted in different NEP values (Fig. 6.7), as was mentioned by other authors too (Curtis *et al.* 2002, Harmon *et al.* 2004, Kolari *et al.* 2004, Williams *et al.* 2005). For instance, NEP_{mass} was assessed at -1.4 t C ha^{-1} $year^{-1}$ after 4 years of tree growth at the plantation in Zwijnaarde, while NEP_{eco} amounted to -2.4 t C ha^{-1} $year^{-1}$

(Table 6.2). Possible causes of this difference were discussed in § 6.4.2. Main uncertainties were linked to root respiration fluxes, and to carbon stock changes in deeper soil layers. It would be an interesting exercise to compare mass-balance and ecological NEP calculations with results from an eddy covariance measurement system. However, the limited fetch of the SRF plantation as well as the Aelmoeseneie forest forms a prohibitive objection for application of the eddy covariance technique on these sites.

Inevitably, calculation of the carbon balances at Zwijnaarde and Gontrode were incomplete, due to technical and time restrictions (§ 6.2.2 and § 8.4.3). Additional research on the role of herbaceous and shrub vegetation, decomposition of dead wood, emission of volatile organic compounds, and eventually herbivory, can contribute to an improved assessment of the carbon balances of the short-rotation plantation and of the Aelmoeseneie experimental forest.

As only one regional inventory was available for the Flemish forests, a simple modelling approach was used to determine the carbon sink or source strength of these forests in the period 1990 to 2000 (Chapter 10). The absence of region-specific increment data appeared to induce the largest uncertainty on carbon stock change calculations (§ 10.4.2). This problem will be solved when data from the second regional inventory, which is planned for 2007, will become available. Forest soil carbon stock changes were calculated by Lettens *et al.* (2005) for the period 1960 to 2000. New measurements are needed to confirm the trend of an increase in soil carbon stocks of 0.675 t C ha^{-1} year^{-1} (0-30 cm depth) calculated for the period 1960-2000. However, this implies a tremendous work to be done.

Concluding, it can be stated that whatever method is used, the belowground carbon fluxes are the most difficult to evaluate.

11.4. Carbon balance of different forest ecosystems

Net ecosystem productivity was calculated for a short-rotation forestry ecosystem during the first 4 years of tree growth (Chapter 6), for two mature forest stands (Chapter 8), and for all Belgian forests (Chapter 10). Results from these calculations are presented in Fig. 11.1. Whereas the mean carbon balance was negative for the short-rotation plantation, both stands of the Aelmoeseneie forest, as well as Flemish and Walloon forests, obviously acted as carbon sinks.

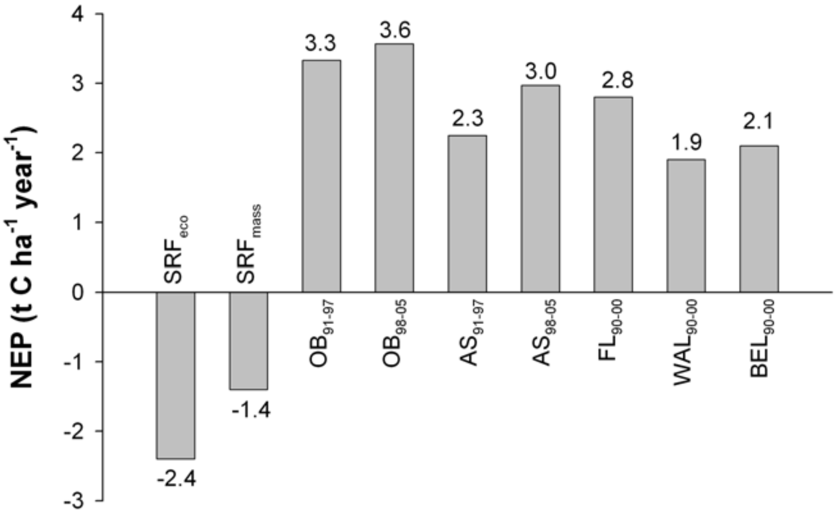

Figure 11.1. *Net ecosystem productivity (NEP) of different forest ecosystems; SRF : short-rotation plantation at Zwijnaarde; eco : ecological approach, mass : mass-balance approach (Chapter 6); OB and AS : oak-beech and ash stand of the Aelmoeseneie experimental forest, respectively; for OB and AS, mean annual NEP is given for the periods 1991-1997 (91-97) and 1998-2005 (98-05), calculated according to the mass-balance approach (Chapter 8); FL : mean for Flemish forests; WAL : mean for Walloon forests; BEL : mean for Belgian forests; for FL, WAL and BEL, mean annual NEP is given for the period 1990-2000 (90-00), according to the back-calculation procedure described in Chapter 10.*

Studies on short-rotation forestry plantations are most often focussed on aboveground biomass production, and the possibility of preventing greenhouse gas emissions by substituting fossil fuels (*e.g.*, Hofmann-Schielle *et al.* 1999, Telenius 1999, Hytönen and Issakainen 2001, Laureysens *et al.* 2004). Almost no information on the overall carbon balance of this type of ecosystem is available. From the research at Zwijnaarde, it became clear that the plantation was a source for carbon during the first four years of tree growth, with a mean C loss of 2.4 t C ha^{-1} year^{-1} according to the ecological approach, or 1.4 t C ha^{-1} year^{-1} based on the mass-balance calculation procedure. As indicated in Table 6.2, NEP became less negative throughout the 4 years study period. In the fourth year, NEP only amounted to -0.1 t C ha^{-1} year^{-1}. Therefore, it can be expected that the plantation will act as a sink for carbon in the near future. As discussed in § 6.4.2, the carbon source status of the plantation was not unexpected, and was mainly due to the combination of a relative low biomass production with large CO_2 effluxes from the soil, caused by high soil heterotrophic activity (§ 6.4.1.3). Increasing biomass production combined with a higher litter input to the soil, and as such, a build-up of the soil C stock, can be expected to result in a positive carbon balance during the coming years. The exact carbon sink strength that will be reached is difficult to predict, and can only be assessed by new measurements of carbon stocks or carbon fluxes. So far, the impact of aboveground biomass harvesting or the application of fertilizers or herbicides on the carbon balance of a short-rotation plantation remains unknown. Therefore, more research is needed to get more insight in this impact.

A unique data set of three full inventories of the oak-beech and the ash stand of the Aelmoeseneie forest enabled the calculation of NEP for these two forest stands (§ 8.2.2). As illustrated in Fig. 11.1, both stands of the Aelmoeseneie forest were a stronger carbon sink in the period 1998-2005 than from 1991 to 1997. This difference appeared to depend more on differences in climatic conditions in these two periods than on differences in elemental deposition (§ 8.4.2). The back-calculation procedure applied in Chapter 10 (§ 10.2.1) revealed a mean NEP value of 2.8 and 1.9 t C ha^{-1} year^{-1} for Flemish and Walloon forests in the period 1990 to 2000, respectively. This difference was mainly attributable to the low wood harvest in the (younger) Flemish forests (§ 10.4.3).

NEP values calculated for the different ecosystems fell within the range of C balance values published for other temperate forest ecosystems (Nabuurs and Mohren 1993a, Valentini *et al.* 2000, Ehman *et al.* 2002, Carrara *et al.* 2003, Black *et al.* 2005). As was observed for the plantation at Zwijnaarde (Table 6.2), carbon fluxes may display appreciable interannual variability. This implies that measurements should be performed over a time period which is long enough to cover interannual variability of NEP, if one wants to assess the mean carbon sink or source strength of a forest ecosystem. Comparing carbon balances of forest ecosystems is therefore most reasonable when this comparison is based on long-term measurements of carbon stock changes or carbon fluxes, as NEP values determined for a single year at some arbitrary time in the life cycle, or even for a few years, may give a misleading picture of ongoing C sequestration potential of the vegetation (Bolin and Sukumar 2000).

11.5. Role of forest ecosystems in reaching the Kyoto Protocol target for Flanders and Belgium

As short-rotation forestry plantations cannot be categorized as forests, they do not fall under Art. 3.3 of the Kyoto Protocol. The Belgian government decided to make no use of possible carbon sequestration means under Art. 3.4 (OVK 2006). Therefore, possible carbon sequestration in SRF plantations will not be directly accountable under the Kyoto Protocol. However, biomass produced at SRF plantations can be used for energy production. As such, SRF plantations can indirectly contribute to reach the Kyoto Protocol target for Flanders, when fossil fuels are substituted by biomass produced at these plantations, reducing as such CO_2 emissions.

Bioenergy production processes appeared to be more energy efficient than energy production processes based on fossil fuels (Table 4.5, § 4.4.1). Despite this high energy efficiency, the maximum amount of electricity that can be provided by SRF biomass is only 0.16 % of the total electricity production in Flanders. The maximum CO_2 emission reduction potential of SRF plantations was estimated at only 0.09 % of the total annual CO_2 emissions in this region. These low results were mainly attributable to the low biomass production at the plantation in Zwijnaarde, and to the land scarcity in Flanders, resulting from large claims on available land for all types of land use, due to the high population density.

A number of assumptions were made in order to calculate the CO_2 emission reduction potential of SRF plantations in Flanders (§ 4.2.6). On-site drying of biomass was assumed, which asks less energy input than off-site storage. A total area of 10000 ha of SRF plantations was assumed, which can be considered as the maximum area that will ever become available for establishment of SRF plantations in Flanders. Moreover, burning of biomass was considered as a CO_2 neutral process. However, the carbon balance of the SRF plantation at Zwijnaarde appeared to be negative during the first years of tree growth (Table 6.2). It is expected that this balance will become positive in the next years, but it is questionable whether high amounts of carbon will ever be fixed in the sandy soil on which the plantation is situated. It is generally known that poor sandy soils, which are the most likely to be put down to trees, are the least efficient at retaining organic matter (Poulton *et al.* 2003). These elements indicate that the CO_2 emission reduction potential calculated in Chapter 4 can be considered as an optimistic assessment of this reduction potential for Flanders. Ideally, measurements of soil carbon stock changes should be continued for a much longer time period, as to see to what extent the initial soil carbon losses as observed at the plantation in Zwijnaarde will be compensated. Continued measurements of biomass production are needed to assess the carbon balance during the next years. Alternatively, models could be used to simulate the NEP evolution over a complete rotation length.

In the calculation procedure of the CO_2 emission reduction potential of SRF plantations in Flanders, only green electricity production was taken into account. As combined heat and power plants are much more efficient, the use of this type of plants seems to be the most interesting application of SRF in Flanders, linked to the establishment of small-scale plantations in the close neighbourhood. These plantations could be established on marginal arable soils or on polluted sites. They could also be important because of other environmental benefits, amongst which prevention of soil erosion or their function as (temporary) habitat for many species.

There are two key policy-related reasons for measuring carbon in forests : (1) commitments under the United Nations Framework Convention on Climate Change (UNFCCC), and (2) for implementation of the Kyoto Protocol (Brown 2002). The UNFCCC requires that all Parties to the Convention commit themselves to develop, periodically update, publish, and make available to the Conference of Parties (COP) their national inventories of emissions by sources and removals by sinks of all greenhouse gases using comparable methods (Houghton *et al.* 1997). Land-use change and forestry is one sector for which a national inventory of sources and sinks of greenhouse gases must be developed. With reference to forests, the inventory must include estimates of carbon emissions and removals caused by changes in forest biomass stocks due to forest management, harvesting, plantation establishment, abandonment of lands that regrow to forests, and forest conversion to non-forest use. All these changes imply that measurements of carbon in biomass stocks must be made. The recognition that forestry activities could be both sources and sinks of carbon led to their inclusion in the Kyoto Protocol. There are several articles in the KP that refer to forests (*e.g.*, Art. 3.3 and 3.4); all of them refer to the emissions and removals as being real, measurable and verifiable changes in carbon stocks.

National carbon accounting methodologies for reporting to the Intergovernmental Panel on Climate Change and the Kyoto Protocol are mostly based on forest inventory data and biometric approaches (Black *et al.* 2005). Applying this methodology, Belgian forest ecosystems acted clearly as a sink for carbon in the period 1990-2000 (Table 10.8). The case study in the Aelmoeseneie forest confirmed this result (Table 8.6). Reporting under Art. 3.3 (afforestation, reforestation and deforestation) of the Kyoto Protocol is mandatory. The loss in forest area in the period 1990 to 2000 implied a mean annual carbon loss of 148.9 kt C year^{-1} for Belgium as a whole (§ 10.4.4). This loss was largely compensated by the annual C uptake by the remaining forests, which averaged 1282.4 kt C year^{-1} in the same period. This carbon fixation equalled 3.2 % of total annual Belgian greenhouse gas emissions in the year 1990. However, when a government wants to take forest growth into account as a measure to fulfil the Kyoto Protocol commitments, this growth has to be reported under Art. 3.4. Reporting under this article is not obliged, but can be done on voluntary basis. When forest carbon uptake is included in the value reported under Art. 3.4, only 15 % of the actual carbon sequestration in forests may be declared (Schulze *et al.* 2002). Moreover, the country-specific cap on Art. 3.4 is only 30 kt C year^{-1} for Belgium. Because of this small cap, and the high costs related to the verification of the carbon sequestration, the Belgian government recently decided (OVK 2006) that activities under Art. 3.4 will not be used to reach the Kyoto Protocol commitment of reducing greenhouse gas emissions in Belgium with 7.5 % compared to 1990 levels during the first commitment period (2008-2012).

In the framework of the federal research programme CASTEC, carbon sequestration in Belgian terrestrial ecosystems was studied. Besides forests, cropland and grassland ecosystems were investigated (CASTEC 2006). In Table 11.2, a comprehensive overview of mean soil carbon stocks in different Flemish and Belgian ecosystem types is given. From this table, it is clear that soil carbon stocks in forests were much higher than under cropland. The mean carbon stock under Flemish agricultural grassland, amounting to 143.0 t C ha^{-1}, was more comparable to values found in the Aelmoeseneie forest. As was illustrated in Table 7.3, the litter layer in a forest can represent a large carbon stock too (oak-beech stand : 35.7 t C ha^{-1}). It was calculated in Chapter 9 that the mean C stock in the biomass of Flemish forests amounted to 85.2 t C ha^{-1} in the year 2000. Given the absence of (permanent) vegetation on cropland, and the limited amount of carbon in the biomass of grassland, it is obvious that forest ecosystems represent much larger total carbon stocks than cropland and grassland, expressed per area unit.

Table 11.2. *Mean soil carbon stock for the year 2000 for different ecosystem types; Fl. : Flemish; Bel. : Belgian; OB and AS : oak-beech and ash stand of the Aelmoeseneie experimental forest, respectively*

Ecosystem type	**Soil C stock (t C ha^{-1})**	**Depth**	**Reference**
Fl. cropland	75.0	0-1 m	Sleutel *et al.* 2003
Fl. cropland	52.0	0-0.3 m	Lettens *et al.* 2005
Bel. cropland	50.0	0-0.3 m	Lettens *et al.* 2005
Fl. agricultural grassland	143.0	0-1 m	Mestdagh 2005
Fl. natural grassland	170.8	0-1 m	Mestdagh 2005
Fl. grass-covered verges	195.3	0-1 m	Mestdagh 2005
Fl. urban grassland	146.2	0-1 m	Mestdagh 2005
Fl. grassland	86.0	0-0.3 m	Lettens *et al.* 2005
Bel. grassland	79.0	0-0.3 m	Lettens *et al.* 2005
Fl. forests	79.0^{a}	0-0.3 m	Lettens *et al.* 2005
Bel. forests	91.0^{a}	0-0.3 m	Lettens *et al.* 2005
Aelmoeseneie OB	94.4^{a}	0-0.3 m	Table 7.3
	134.8^{a}	0-1 m	Table 7.3
	170.7^{b}	0-1 m	Table 7.3
Aelmoeseneie AS	99.9^{a}	0-0.3 m	Table 7.3
	170.0^{a}	0-1 m	Table 7.3
	173.6^{b}	0-1 m	Table 7.3

^{a} *litter layer excluded,* ^{b} *litter layer included*

As can be seen in Table 11.3, analysis of data on cropland soil carbon stocks revealed a mean decline of 0.87 t C ha^{-1} year^{-1} in the period 1990 to 1999, while for grassland, soil carbon stocks diminished at a mean rate of 1.4 t C ha^{-1} year^{-1} in the period 1990 to 2000 (CASTEC 2006). This means that these two types of ecosystems were clear sources for carbon, which was mainly attributed to shifts in management. Experimental and modelling results also indicated that these ecosystem types can be expected to act as a source for carbon in the near future (CASTEC 2006). As discussed in Chapter 10, Flemish forests acted as a clear sink for carbon in the period 1990 to 2000, with a mean strength of 2.8 t C ha^{-1} year^{-1} (§ 10.4.3). Total areas for cropland and grassland in the year 2000 were available from the National Institute for Statistics (FOD-EC 2006), and are given in Table 11.3. Combining mean NEP values with the total area per vegetation class gave an assessment of the annual carbon balance of each vegetation class, which is listed in Table 11.3 too.

Table 11.3. *Total area in 2000, mean NEP in the period 1990-2000, and total annual NEP of Flemish cropland, grassland and forests*

	Total area (ha)	**Mean NEP (t C ha^{-1} year^{-1})**	**Total annual NEP (kt C year^{-1})**
cropland	406677	-0.87	-353.8
grassland	262229	-1.4	-367.1
forests	144730	+2.8	405.2

From these results, it can be seen that the total area of forests only amounted to 22 % of the total area of cropland and grassland together. However, this forested area was fixing more than half of the total carbon release from cropland and grassland together, expressed on an annual base. Moreover, carbon sequestration in the Flemish forests was higher than the total carbon release from cropland or from grassland, when these vegetation types were considered separately.

These results all indicate that forest ecosystems are the most important vegetation type regarding carbon sequestration potential in Flanders, and Belgium. Therefore, whether or not they are taken into account to reach the Kyoto Protocol target, forest ecosystems should certainly be protected, as they represent the vegetation type with the largest carbon stocks per area unit, and they have an important carbon absorbing capacity.

With this final conclusion, the author sincerely hopes that this thesis may have contributed to a better understanding of forest carbon dynamics in Belgian forest ecosystems, and that this enhanced insight in the role of forests regarding carbon sequestration will further be applied to develop improved policy decisions related to forest management or conservation.

Chapter 12

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Summary

Increased emissions of greenhouse gases by human activities are enhancing the natural greenhouse effect, and are expected to have severe consequences for human society. Terrestrial ecosystems are believed to offer an opportunity to mitigate climate changes by fixing carbon dioxide (CO_2), the main greenhouse gas. Because of their large carbon (C) stocks and the large amounts of carbon exchanged with the atmosphere through the processes of photosynthesis and respiration, forest ecosystems play an important role in the global carbon cycle. In this study, the carbon sequestration capacity of short-rotation forestry (SRF) plantations and forests in Belgium were examined. Both types of ecosystems are possible means to fulfil the commitments to the Kyoto Protocol which were made by the Belgian government.

To investigate the carbon fluxes in a short-rotation forestry ecosystem, a plantation of 9600 m^2 was established on former agricultural land at Zwijnaarde (Belgium). The plantation was composed of 24 plots of 400 m^2. Birch (*Betula pendula* Roth) and maple (*Acer pseudoplatanus* L. - Tintigny) were planted as 2 year old saplings on 4 plots each, with a tree density of 6667 trees ha^{-1} (spacing distance : 1.0 m x 1.5 m). Twenty cm unrooted cuttings of poplar (*Populus trichocarpa* x *deltoides* - Hoogvorst) and willow (*Salix viminalis* - Orm) were planted on eight plots each, with an initial density of 20000 trees ha^{-1} (spacing distance : 1.0 m x 0.5 m).

Specific leaf area and leaf photosynthetic characteristics are important in the light of upscaling photosynthesis from leaf to canopy level. Values for these characteristics were in the same range for birch and poplar growing on the short-rotation plantation at Zwijnaarde. Mean actual aboveground biomass production after 4 years of tree growth amounted to 1.2, 2.6, 3.4 and 3.5 t dry mass ha^{-1} $year^{-1}$ for maple, birch, willow and poplar, respectively. Poplar suffered severely from an infection by *Colletotrichum gloeosporioides*. Because of the dry summer periods in 2003 and 2004, the mean biomass production of poplar and willow was decreasing after the first 4 years of tree growth. For birch, however, mean annual increment (MAI) was increasing with time, and it can be expected that the maximum biomass production will only be reached after 8 to 10 years of tree growth. Moreover, birch wood appeared to have the highest calorific value of the four studied species, which is an interesting characteristic in the view of energy production. From these results, it was concluded that birch could be considered as an interesting alternative for establishing short-rotation plantations in Flanders, especially on dry marginal soils.

The carbon balance, or net ecosystem productivity (NEP), of an ecosystem is the result of two large fluxes : CO_2 fixation by photosynthesis and release of carbon as CO_2 by ecosystem respiration. Soil CO_2 efflux, which is the result of both autotrophic and heterotrophic processes, is the main respiration flux. Extensive research at the

plantation revealed that temporal variations in soil CO_2 efflux were mainly caused by changes in soil temperature. Soil C content, soil pH, leaf area index and root biomass appeared to be the main driving factors of short-distance spatial variations in soil CO_2 effluxes. Two methods were applied to determine the carbon balance of the plantation. The ecological approach is based on measurements of carbon fluxes, while the mass-balance method relies on the determination of (changes in) carbon stocks. According to the first approach, the short-rotation plantation was a source for carbon during the first 4 years of tree growth, with a mean loss of 2.4 t C ha^{-1} year^{-1}. Whereas annual carbon losses through heterotrophic respiration processes mainly depended on the temperature and precipitation during the leafed period, carbon uptake by the trees was increasing with stand age. Using the mass-balance method, the mean annual carbon loss was assessed at 1.4 t C ha^{-1} year^{-1} during the first four years of tree growth.

A case study was performed in the oak-beech (OB) and the ash stand (AS) of the experimental zone of the 80 years old Aelmoeseneie forest (Gontrode, Belgium). Total carbon stocks, including the vegetation, litter and soil compartment, amounted to 324.8 t C ha^{-1} in OB, and 321.4 t C ha^{-1} in AS. Approximately half of this carbon stock was found in the vegetation of the two stands. Based on data from three full inventories of the two stands, NEP was assessed at 3.3 ± 2.3 and 2.3 ± 1.8 t C ha^{-1} year^{-1} in the period 1991-1997 for OB and AS, respectively. The carbon balance was slightly higher in the period 1998-2005, and amounted to 3.6 ± 2.3 and 3.0 ± 1.8 t C ha^{-1} year^{-1} for OB and AS, respectively. The increase in NEP appeared to depend on higher air temperatures and increased precipitation in the period 1998-2005 compared to 1991-1997, while differences in elemental deposition between these two periods were not significant.

Belgian regional forest inventories contain no data on carbon stocks, as is the case in many countries. An extensive literature study was performed to select wood density, biomass expansion factors *s.s.* and carbon content values applicable for Belgian forests. These values were then used to convert total solid wood volume data from the forest inventories to total biomass carbon stocks. As such, the total carbon stock in Belgian forest biomass in the year 2000 was assessed at 60.9 Mt C, with a mean of 101.1 t C ha^{-1}. The choice of the wood density value had the highest impact on the calculated total C stock. Starting from the year 2000, a back-calculation was performed to assess the carbon sequestration by Belgian forests in the period 1990 to 2000. Mean carbon uptake by Belgian forests amounted to 2.1 t C ha^{-1} year^{-1} in this period, including changes in biomass and soil carbon stocks. For this calculation, it was assumed that litter and dead wood carbon stocks did not change during the considered time period. The calculation of the carbon sink strength in Flemish forests will be improved when region-specific increment data will become available.

Due to low biomass production results at Zwijnaarde and land scarcity, short-rotation forestry plantations appeared to have a limited capacity for energy production and CO_2 emission reductions in Flanders. However, SRF could be implemented at smaller scales, and could combine carbon sequestration with other functions as prevention of soil erosion and provision of a (temporary) habitat for different species. Belgian forests were obvious carbon sinks in the period 1990-2000, in contrast to Flemish cropland and grassland ecosystems. Therefore, these forests should be protected, whether or not the Belgian government wants to make use of the means offered by Art. 3.4 of the Kyoto Protocol to reach the greenhouse gas emission reduction target.

Samenvatting

Menselijke activiteiten veroorzaken stijgende broeikasgasemissies die op hun beurt het natuurlijke broeikaseffect versterken. Dit zal vermoedelijk ernstige gevolgen hebben voor de menselijke samenleving. Terrestrische ecosystemen bieden een mogelijkheid om klimaatsverandering tegen te gaan door het vastleggen van koolstofdioxide (CO_2), het belangrijkste broeikasgas. Bossen spelen een belangrijke rol in de globale koolstofcyclus, vanwege hun grote koolstofvoorraden en de grote hoeveelheden koolstof (C) die uitgewisseld worden met de atmosfeer via fotosynthese en respiratie. In dit onderzoek werd de koolstofvastleggingscapaciteit van plantages met korte omloopstijd en van Belgische bossen onderzocht. Beide ecosysteemtypes vormen een middel om de verplichtingen na te komen die de Belgische overheid aangegaan is onder het Kyoto Protocol.

Om de koolstofcyclus in een systeem onder korte-rotatie-bosbouw (KRB) te onderzoeken werd een plantage aangelegd op voormalig akkerland in Zwijnaarde (België). De totale oppervlakte bedroeg 9600 m^2 en was samengesteld uit 24 plots van 400 m^2. Berk (*Betula pendula* Roth) en esdoorn (*Acer pseudoplatanus* L. - Tintigny) werden als tweejarige bomen geplant op 4 plots, met een dichtheid van 6667 bomen ha^{-1} (plantafstand : 1.0 m x 1.5 m). Stekken van populier (*Populus trichocarpa* x *deltoides* - Hoogvorst) en wilg (*Salix viminalis* - Orm) werden geplant op 8 plots, met een initiële dichtheid van 20000 bomen ha^{-1} (plantafstand : 1.0 m x 0.5 m).

Specifieke bladoppervlakte en fotosynthesekenmerken van bladeren zijn belangrijke gegevens voor het opschalen van fotosynthesefluxen van blad- naar bestandsniveau. Waarden voor deze karakteristieken waren vergelijkbaar voor berk en populier die groeiden op de plantage te Zwijnaarde. De gemiddelde actuele bovengrondse biomassaproductie na 4 jaar bedroeg 1.2, 2.6, 3.4 en 3.5 t droge stof ha^{-1} $jaar^{-1}$ voor respectievelijk esdoorn, berk, wilg en populier. Populier had sterk te lijden onder een aantasting door *Colletotrichum gloeosporioides*. De droge zomers van 2003 en 2004 waren er de oorzaak van dat de gemiddelde biomassatoename van populier en wilg verminderde na 4 jaar. Voor berk echter steeg de gemiddelde jaarlijkse biomassaproductie met de tijd. Bovendien kan verwacht worden dat de maximale biomassaproductie voor deze soort slechts zal bereikt worden na 8 tot 10 jaar. Het hout van berk had ook een hogere calorische waarde dan het hout van de andere boomsoorten, wat een belangrijk gegeven is naar energieproductie toe. Op basis van deze resultaten kon geconcludeerd worden dat berk een interessant alternatief biedt voor populier en wilg voor de aanleg van plantages met korte omloopstijd in Vlaanderen, vooral op droge, marginale gronden.

De koolstofbalans, of netto-ecosysteemproductiviteit (NEP), van een ecosysteem is het resultaat van twee grote koolstoffluxen : de opname van CO_2 door fotosynthese en de vrijstelling van koolstof als CO_2 door respiratie. Bodem-CO_2-efflux, of de zogenoemde bodemrespiratie, is het resultaat van autotrofe en heterotrofe processen, en vormt de belangrijkste respiratieflux van een ecosysteem. Uitgebreid onderzoek op de plantage toonde aan dat temporele variatie in bodemrespiratie vooral veroorzaakt werd door veranderingen in bodemtemperatuur. Bodemkoolstofgehalte, bodem-pH, bladoppervlakte-index en wortelbiomassa waren de belangrijkste oorzaken van ruimtelijke variaties in bodemrespiratie op korte afstand. Er werden twee methodes gebruikt om de koolstofbalans van de plantage te berekenen. De ecologische methode is gebaseerd op het meten van koolstoffluxen, terwijl de massa-balansbenadering steunt op het bepalen van (veranderingen in) koolstofvoorraden. Volgens de eerste benadering was de plantage een bron van koolstof tijdens de eerste 4 jaar, met een gemiddeld verlies van 2.4 t C ha^{-1} jaar^{-1}. Jaarlijkse verliezen door heterotrofe respiratieprocessen waren vooral afhankelijk van de temperatuur en neerslag tijdens de bebladerde periode, terwijl koolstofopname door de bomen toenam met stijgende leeftijd. Als de massa-balansbenadering gebruikt werd, werd het koolstofverlies geschat op gemiddeld 1.4 t C ha^{-1} jaar^{-1} tijdens de eerste 4 jaar na de aanleg van de plantage.

Er werd een casestudy uitgevoerd in het eiken-beuken- (EB) en het essenbestand (ES) van de experimentele proefzone in het 80-jaar oude proefbos Aelmoeseneie (Gontrode, België). De totale koolstofvoorraad, met inbegrip van de vegetatie, de strooisellaag en de bodem, bedroeg 324.8 t C ha^{-1} in EB en 321.4 t C ha^{-1} in ES. Ongeveer de helft van deze totale koolstofvoorraad was opgeslagen in de vegetatie van de twee bosbestanden. Op basis van drie volledige inventarissen werd een gemiddelde NEP van 3.3 ± 2.3 (EB) en 2.3 ± 1.8 (ES) t C ha^{-1} jaar^{-1} berekend voor de periode 1991-1997. De koolstofbalans was iets hoger in de periode 1998-2005, met waarden van 3.6 ± 2.3 en 3.0 ± 1.8 t C ha^{-1} jaar^{-1} voor respectievelijk EB en ES. De stijging in NEP bleek te wijten te zijn aan hogere luchttemperaturen en verhoogde neerslag in de periode 1998-2005 ten opzichte van 1991-1997, terwijl deposities van elementen niet significant verschilden tussen deze twee periodes.

Belgische regionale bosinventarissen bevatten geen gegevens over koolstofvoorraden. Via een uitgebreide literatuurstudie werden waarden voor houtdichtheid, biomassa-expansie-factoren en koolstofinhoud geselecteerd die bruikbaar waren voor Belgische bossen. Deze data werden gebruikt voor de omrekening van stamhoutvolumegegevens uit de bosinventarissen naar totale koolstofvoorraden in de volledige boven- en ondergrondse biomassa. Op die manier werd berekend dat de biomassa van de Belgische bossen in het jaar 2000 een totale voorraad van 60.9 Mt C bevatte, met een gemiddelde van 101.1 t C ha^{-1}. De keuze van de waarde voor houtdichtheid bleek de grootste impact te hebben op het bekomen resultaat. Startend van het jaar 2000 werd een terugrekening uitgevoerd om de koolstofvastlegging in Belgische bossen in de periode 1990 tot 2000 in te schatten. De gemiddelde koolstofvastlegging in de biomassa en de bodem van Belgische bossen bedroeg 2.1 t C ha^{-1} $jaar^{-1}$ in deze periode. Voor deze berekening werd verondersteld dat er geen veranderingen plaatsvonden in de koolstofvoorraden van dood hout en in de strooisellaag. Deze berekeningen zullen geoptimaliseerd kunnen worden op het moment dat aanwasgegevens voor Vlaanderen beschikbaar zullen komen.

Vanwege de lage biomassaproductie in Zwijnaarde en gebrek aan beschikbare ruimte hebben plantages met korte omloopstijd een beperkte capaciteit voor energieproductie en CO_2-emissiereductie in Vlaanderen. Korte-rotatiebosbouwsystemen zouden wel op kleine schaal kunnen toegepast worden. Hierbij zou de koolstofvastlegging kunnen gecombineerd worden met andere functies als het voorkomen van bodemerosie of het vormen van een (tijdelijke) habitat voor allerlei dier- en plantensoorten. Belgische bossen waren duidelijke sinks voor koolstof in de periode 1990-2000, in tegenstelling tot Vlaamse akker- en graslanden. Daarom moeten deze bossen zeker beschermd worden, ongeacht of de Belgische overheid gebruik wenst te maken van de mogelijkheden geboden door Art. 3.4 van het Kyoto Protocol om de broeikasgasemissiereductiedoelstelling te halen.

Curriculum vitae

A. Personal data

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B. Education

1998 - 2005	Ph.D. training in Applied Biological Sciences, Faculty of Bioscience Engineering, Ghent University, Gent
1991 - 1996	Bio-engineer land and forestry management, Faculty of Bioscience Engineering, Ghent University, Gent
1985 - 1991	Secondary school (Latin-Mathematics), Lyceum Hemelsdaele, Brugge

C. Professional experience

2003 - now	Assistant at the Laboratory of Plant Ecology (Ghent University)
2001 - 2003	Research assistant at the Laboratory of Plant Ecology (Ghent University), CASTEC project (CArbon Sequestration potential in different Belgian Terrestrial ECosystems : quantification and strategic exploration)
1997 - 2001	Research assistant at the Laboratory of Plant Ecology (Ghent University), BELFOR project (Biogeochemical cycles of BELgian FORest ecosystems related to global change and sustainable development)

D. Publications

D.1. Publications in international journals with peer review

Vande Walle, I., A. De Schrijver, R. Samson, K. Verheyen and R. Lemeur. Carbon sequestration in the Aelmoeseneie experimental forest (Gontrode, Belgium) from 1991 to 2005. Submitted to European Journal of Forest Science.

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Sleutel, S., S. De Neve, G. Hofman, P. Boeckx, D. Beheydt, O. Van Cleemput, I. Mestdagh, P. Lootens, L. Carlier, N. Van Camp, H. Verbeeck, **I. Vande Walle**, R. Samson, N. Lust and R. Lemeur. 2003. Carbon stock changes and carbon sequestration potential of Flemish cropland soils. Global Change Biol. 9:1193-1203.

Rasse, D.P., L. François, M. Aubinet, A.S. Kowalski, **I. Vande Walle**, E. Laitat and J.-C. Gérard. 2001. Modelling short-term CO_2 fluxes and long-term tree growth in temperate forests with ASPECTS. Ecol. Model. 141:35-52.

Vande Walle, I., S. Mussche, R. Samson, N. Lust and R. Lemeur. 2001. The above- and belowground carbon pools of two mixed deciduous forest stands located in East-Flanders (Belgium). Ann. Forest Sci. 58:507-517.

D.2. Publications in proceedings of scientific congresses

Vande Walle, I., B. Looman and R. Lemeur. 2005. Fine root biomass in a 4-years old birch stand (Zwijnaarde (B)). *In* Woody root processes - Impact of different tree species. Eds. K. Lõhmus and I. Ostonen. Proceedings of the COST E38-workshop, Tartu (EE), 5 - 9 June 2005, p 56.

Vande Walle, I., N. Van Camp, D. Perrin, R. Lemeur, N. Lust, B. van Wesemael and E. Laitat. 2004. Koolstofvoorraden in de biomassa van Belgische bossen : sensitiviteits-analyse van biomassa-expansiefactoren *s.l.* Proceedings of the winter symposium of NecoV, Gent (BE), 14 - 15 January 2004, p 45.

Vande Walle, I., N. Van Camp, R. Lemeur and N. Lust. 2003. Koolstofpools en -fluxen in bossen en plantages met een korte rotatie. Proceedings of the workshop 'Koolstofopslag in terrestrische ecosystemen : een opportuniteit in het Kyotoprotocol ?', organized by TI-KVIV, Melle (BE), 23 October 2003, pp 69-91.

Vande Walle, I., N. Van Camp, R. Lemeur and N. Lust. 2003. Carbon balance of a two years old short-rotation tree plantation in Zwijnaarde. Proceedings of the SEB Annual Main Meeting, Southampton (UK), 31 March - 4 April 2003, Comparative Biochemistry and Physiology 134, p S187.

Van Camp, N., **I. Vande Walle**, N. Lust and R. Lemeur. 2003. Above- and belowground carbon stocks in Flemish forests - quantification and uncertainty. Proceedings of the SEB Annual Main Meeting, Southampton (UK), 31 March - 4 April 2003, Comparative Biochemistry and Physiology 134, p S194.

Vande Walle, I., N. Van Camp, R. Lemeur and N. Lust. 2001. Soil respiration in a bio-energy plantation. Mededelingen Faculteit Landbouwwetenschappen Universiteit Gent 66(4):129-133.

Vande Walle, I., N. Van Camp, R. Lemeur and N. Lust. 2001. Carbon sequestration potential in Belgian forest ecosystems. Proceedings of the Global Change Open Science Conference 'Challenges of a changing earth', Amsterdam (NL), 10 - 13 July 2001.

Vande Walle, I., K. Janssen and R. Lemeur. 2000. Photosynthetic assimilation in a mixed deciduous forest : the Farquhar approach. Mededelingen Faculteit Landbouwwetenschappen Universiteit Gent 65(4):51-56.

Vande Walle, I., K. Janssen and R. Lemeur. 1999. Photosynthetic characteristics of some deciduous tree species : the Farquhar approach. Mededelingen Faculteit Landbouwwetenschappen Universiteit Gent 64(4):205-210.

D.3. Abstracts of presentations at scientific congresses

Vande Walle, I., N. Van Camp, R. Lemeur and K.. Carbon balance of a young tree plantation. Book of abstracts of the AFFORNORD Conference - 'Effects of afforestation on ecosystems, landscape and rural development', Reykholt (IS), 18 - 22 June 2005, p 40.

D.4. Other publications

Vande Walle, I., K. Verheyen and R. Lemeur. 2006. Inventaris proefbos Aelmoeseneie (Gontrode), januari 2006. Internal publication, Ghent University, Laboratory of Plant Ecology and Laboratory of Forestry, Ghent, 38 p.

Beheydt, D., A. Goossens, P. Boeckx, O. Van Cleemput, S. Sleutel, S. De Neve, G. Hofman, I. Mestdagh, P. Lootens, L. Carlier, **I. Vande Walle**, H. Verbeeck, R. Lemeur, N. Van Camp, N. Lust and K. Verheyen. 2005. Carbon sequestration potential in different Belgian terrestrial ecosystems : quantification and strategic exploration (CASTEC). Final report, Belgian Science Policy, Brussels, 84 p.

Lewyckyj, N., F. Veroustraete, L. Bertels, M. Ruts, W. Debruyn, R. Samson and **I. Vande Walle**. 2003. Retrieval of biophysical properties from CHRIS/PROBA-observed BRDF's. VITO, study realised for ESA, the European Space Agency, 42 p.

Vande Walle, I. and R. Lemeur (eds.). 2001. Biogeochemical cycles of Belgian forest ecosystems related to Global Change and sustainable development. Final report to the OSTC/DWTC, 274 p.

Vande Walle, I. and R. Lemeur (eds.). 1999. Biogeochemical cycles of Belgian forest ecosystems related to Global Change and sustainable development. Annual report 3 to the OSTC/DWTC, 118 p.

Vande Walle, I. and R. Lemeur (eds.). 1998. Biogeochemical cycles of Belgian forest ecosystems related to Global Change and sustainable development. Annual report 2 to the OSTC/DWTC, 114 p.

Vande Walle, I., S. Mussche, M. Schauvliege, R. Lemeur and N. Lust. 1998. Inventaris proefbos Aelmoeseneie (Gontrode) 1997. Internal publication, Ghent University, Laboratory of Plant Ecology and Laboratory of Forestry, Ghent, 33 p.

Vande Walle, I., S. Willems and R. Lemeur. 1998. Root length and distribution in the mineral soil of a mixed deciduous forest (experimental forest Aelmoeseneie). Silva gandavensis 63:1-15.

Vande Walle, I. and R. Lemeur. (eds.). 1997. Biogeochemical cycles of Belgian forest ecosystems related to Global Change and sustainable development. Annual report 1 to the OSTC/DWTC, 81 p.

Vande Walle, I., M. Schauvliege, R. Samson, S. Mussche, R. Lemeur and N. Lust. 1997. Photosynthesis measurements on adult trees : a comparison between field and laboratory measurements. Silva gandavensis 62:36-50.

E. Participation in congresses, symposia or workshops

E.1. Participation in international congresses with oral presentation

AFFORNORD Conference 'Effects of afforestation on ecosystems, landscape and rural development', Reykholt (IS), 18 - 22 June 2005
'Carbon balance of a young tree plantation'

Fifth Plenary meeting of COST E21 'The role of forest for mitigating greenhouse gas emissions', Thessaloniki (GR), 27 - 29 November 2003
'Carbon stock in the Belgian forest biomass : sensitivity analysis of Biomass Expansion Factors'

E.2. Participation in international congresses with poster presentation

'Biomass for Energy - Challenges for Agriculture', conference organized within the framework of the North Sea Bio Energy project, Brugge (BE), 25 - 26 September 2006
'Carbon sequestration potential of bio-energy plantations in Flanders'

COST Action E38 'Woody Root Processes - Impact of Different Tree Species', Tartu (EE), 5 - 9 June 2005
'Fine root biomass in a 4-years old birch stand (Zwijnaarde (B))'

COST Action E21 'Contribution of forest and forestry to the mitigation of the greenhouse effect' - Final plenary meeting, Dublin (IE), 7 - 9 October 2004
'Identifying and quantifying uncertainties associated with estimates of carbon stocks in Belgian forest biomass and harvested wood products' and
'Temporal changes in allometric relationships during the first years of tree growth. Case-study for birch, maple, poplar and willow (Zwijnaarde – Belgium)'

COST Action E38 'Woody root processes', Fine root dynamics workshop, Jådraäs (SE), 7 - 11 June 2004
'Distribution of root length and root mass in an oak-beech and an ash stand (Aelmoeseneie forest, Gontrode (B))'

Wintermeeting NecoV (Nederlands-Vlaamse vereniging voor ecologie) 'Ecotechniek en natuurbouw', Gent (BE), 14 - 15 January 2004
'Koolstofvoorraden in de biomassa van Belgische bossen : sensitiviteitsanalyse van biomassa-expansiefactoren'

Annual Main Meeting of the Society for Experimental Biology, Southampton (UK), 31 March - 4 April 2003
'Carbon balance of a two years old short-rotation tree plantation (Zwijnaarde – Belgium)' and

'Above- and belowground carbon stocks in Flemish forests – quantification and uncertainty'

COST E21 4th Whole Action Meeting 'Contribution of forest and forestry to mitigate greenhouse effects', Valencia (ES), 7 - 8 October 2002

'C stocks in Flemish forests, grassland and arable soils'

International Conference under the Belgian Presidency of the European Union 'Carbon sinks and biodiversity', Liège, 24 - 26 October 2001

'Carbon sequestration potential in different Belgian terrestrial ecosystems : quantification and strategic exploration' and

'Carbon sequestration potential in Belgian forest ecosystems'

Global Change Open Science Conference 'Challenges of a Changing Earth', Amsterdam (NL), 10 - 13 July 2001

'Carbon sequestration potential in Belgian forest ecosystems'

2nd International Workshop on Forest Ecosystem Modelling, Upscaling and remote sensing (INFORMUS), Antwerpen (BE), 21 - 25 September 1998

'The photosynthetic conversion efficiency of a Belgian deciduous forest ecosystem'

GCTE-LUCC Open Science Conference on Global Change 'The Earth's Changing Land', Barcelona (ES), 14 - 18 March 1998

'Influence of Global Change on the photosynthetic conversion efficiency of a Belgian deciduous forest ecosystem'

E.3. Participation in national congresses with oral or poster presentation

Seminarie 'Energie uit houtige biomassa', seminarie Bio-ingenieur Land- en Bosbeheer, academiejaar 2006-2007, Faculty of Bioscience Engineering, Ghent University, Gent, 1 March 2007

'Potentiële koolstofvastlegging in aanplantingen met korte omloopstijd in Vlaanderen'

Studiedag 'De rol van bos in het klimaatbeleid in Noord en Zuid', organized by Groenhart vzw, Brussel, 31 May 2006

'Koolstofopslag in bodem en biomassa van Belgische bos- en landbouwecosystemen'

Studiedag 'Koolstofopslagpotentieel in verschillende Belgische terrestrische ecosystemen : kwantificering en strategisch onderzoek / Potentiel du stockage de carbone dans les écosystèmes terrestres belges : quantification et exploration stratégique', organized by CASTEC, Brussel, 18 February 2005

'Energy production and carbon balance of a young tree plantation'

Studie- en vervolmakingsdag 'Koolstofopslag in terrestrische ecosystemen : een opportuniteit in het Kyotoprotocol ?', organized by TI-KVIV, Melle, 23 October 2003

'Koolstofpools en -fluxen in bossen en plantages met een korte rotatie'

7th FLTBW PhD symposium, Gent, 10 October 2001

'Soil respiration in a short rotation plantation'

6th FLTBW PhD symposium, Gent, 11 October 2000

'Photosynthetic assimilation in a mixed deciduous forest : the Farquhar approach'

5th FLTBW PhD symposium, Gent, 13 October 1999

'Photosynthetic characteristics of some deciduous tree species : the Farquhar approach'

4th FLTBW PhD symposium, Gent, 7 October 1998

'Photosynthetic conversion efficiency of three forest tree species'

E.4. Participation in congresses without presentation

Workshop organized by the British Society of Soil Science, 'Soils as carbon sinks : opportunities and limitations', London (UK), 28 June 2002

Open joint seminar between the RECOGNITION-project, IUFRO working group 4.1.08, EFI, the Swedish University of Agricultural Sciences and the Institute of Forest and Landscape, Hörsholm, Denmark, 'European forests in a changing environment', Alnarp (SE), 16 May 2001

International Workshop on Prospects for Co-ordinated Activities in Core Projects of GCTE, BAHC and LUCC, Wageningen (NL), 16 - 19 November 1997

E.5. Participation in workshops

Studiedag 'Starters in het natuuronderzoek', organized by Aminal and Inverde, Brussel (BE), 22 March 2006

Docententraining 'Multiple Choice', organized by the afdeling Onderwijsondersteuning of the Directie Onderwijsaangelegenheden (UGent), Gent (BE), 9 November 2005

Summerschool 'Terrestrial Carbon Cycle and Earth Observation', organized by the Centre for Terrestrial Carbon Dynamics, Malham Tarn (UK), 18 - 26 August 2005

Studiedag 'Starters in het bosonderzoek', organized by Aminal and Inverde, Brussel (BE), 22 March 2005

Summerschool 'Integrated methodology on soil carbon flux measurements', organized by the European Science Foundation, Trento (IT), 12 - 24 September 2004

Natuurontwikkeling in Vlaanderen : tips voor de toekomst, organized by the Instituut voor Natuurbehoud, in collaboration with AMINAL Afdeling Natuur, de Vlaamse Landmaatschappij and AMINAL Afdeling Bos en Groen, Brussel (BE), 26 March 2004

Beleidssymposium 'Is koolstofvastlegging in ecosystemen een optie voor het Vlaamse klimaatbeleid ? Landgebruiksscenario's voor broeikasgasreductie', organized by the Ministerie van de Vlaamse Gemeenschap – AMINAL, Grimminge (BE), 29 May 2002

Symposium 'Op zoek naar een duurzame dialoog tussen onderzoek en beleid', organized by the Federale Diensten voor Wetenschappelijke, Technische en Culturele Aangelegenheden, Brussel (BE), 24 - 25 November 1999

Li-Cor LI-6400 end-user workshop, seminar organized by CaTec, Wageningen (NL), 2 June 1999

Course on linear regression analysis, organized by the Centrum voor Statistiek in collaboration with Open Universiteit Gent, Gent (BE), March - May 1999 (12 hours)

Changement climatique et politiques de reduction des emissions de gaz à effet de serre, seminar organized by NETRAM, Liège (BE), 3 March 1999

'Klimaatverandering. Het Protocol van Kyoto : bedreiging of uitdaging ?', symposium organized by Federale Raad voor Duurzame Ontwikkeling, Brussel (BE), 19 May 1998

Studiedag V.V.P.F., Limburgs Universitair Centrum, Diepenbeek (BE), 21 March 1997

F. Teaching activities

Practical exercises at Ghent University :

2005 - now	Ecologie, 2^{nd} Bachelor Bioscience Engineering (15 h per year)

2005 - now	Meteorology and Climatology, Complementary Studies in Physical Land Resources (30 h per year)
2002 - now	Toegepaste vegetatie-ecologie, Bio-engineer in Land Management and Forestry (15 h per year)
2000 - 2004	Ecologie en duurzame productiesystemen, 2nd year Bio-engineer (15 h per year)
1997 - now	Ecofysiologie (30 h per year)
1997 - 1998	Numerische vegetatie-ecologie (15 h per year)

G. Supervision of M.Sc. thesis students

2004 - 2005	Marc Bosmans. Evaluatie van de TRAC-sensor ter karakterisering van kruinarchitectuur en bladoppervlakte-index.
2004 - 2005	Brecht Looman. Temporele en spatiale variatie in de bodemrespiratie van een jonge plantage van berk (*Betula pendula* Roth), populier (*Populus trichocarpa* x *deltoides* - Hoogvorst), wilg (*Salix viminalis* - Orm) en esdoorn (*Acer pseudoplatanus* L. - Tintigny).
2001 - 2002	Mari Paz Montoro Pérez. Soil respiration and carbon balance in a short-rotation tree plantation.
1999 - 2000	Katrien Janssen. De Farquhar-benadering van de fotosynthetische assimilatie in een gemengd loofbos.
1998 - 1999	Eva Wollaert. Simulatie van de waterbalans in een gemengd loofbos (proefbos Aelmoeseneie).
1997 - 1998	Sofie Willems. Invloed van de biotische en abiotische bodemkarakteristieken op de waterbalans van een gemengd loofbos (proefbos Aelmoeseneie).

H. Reviews for international journals and reports

Forest Ecology and Management (1), Journal of Environmental Management (1)

Good Practice Guidance for Land Use, Land-Use Change and Forestry (published by the Institute for Global Environmental Strategies (IGES) for the IPCC, 2003)

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