MULTI-FUNCTIONAL BIOMASS SYSTEMS

Multifunctionele biomassasystemen
(met een samenvatting in het Nederlands)

Multifunktionale Biomassesysteme
(mit einer Zusammenfassung auf Deutsch)

PROEFSCHRIFT

TER VERKRIJGING VAN DE GRAAD VAN DOCTOR AAN DE UNIVERSITEIT UTRECHT
OP GEZAG VAN DE RECTOR MAGNIFICUS, PROF. DR. W.H. GISPEN
INGEVOLGE HET BESLUIT VAN HET COLLEGE VOOR PROMOTIES
IN HET OPENBAAR TE VERDEDIGEN
OP WOENSDAG 1 DECEMBER 2004 DES MIDDAGS OM 12.45 UUR

door

Veronika Dornburg

geboren op 21 december 1971 te Wülfrath (Duitsland)

Promoter: Prof. Dr. W.C. Turkenburg
Department of Science, Technology and Society, Faculty of Chemistry, Utrecht University

Co-promoter: Dr. A.P.C. Faaij
Department of Science, Technology and Society, Faculty of Chemistry, Utrecht University

This thesis is part of the research programme 'BioPush' sponsored by NWO (Netherlands Organisation for Scientific Research) and SenterNovem (The Netherlands Agency for Energy and the Environment). The research was executed at the Department of Science, Technology and Society and the Copernicus Institute for Sustainable Development and Innovation of Utrecht University.

Multifunctional biomass systems/Veronika Dornburg—Utrecht: Universiteit Utrecht, Faculteit Scheikunde, Proefschrift Universiteit Utrecht. Met literatuuropgave. —
Met en samenvatting in het Nederlands. Mit einer Zusammenfassung auf Deutsch.

ISBN: 90-393-3854-X

Кто сказал, что Земля не умерла?
Het, она затайлась время!
(Владимир Высоцкий)

Who said that the earth has died?
No, it holds its breath for a while!
(Vladimir Vysotsky)

Contents

Chapter 1:

Introduction

1 Biomass and greenhouse gas mitigation

In order to achieve sustainable development, many socio-economic and environmental issues of energy and material demand and supply still need to be solved (Goldemberg et al., 2000). Human-induced climate change is one of the most serious environmental concerns. Its negative long-term effects on human society and ecosystems are potentially large (IPCC, 2001c). Climate change can be the result of an increase of the greenhouse effect due to anthropogenic greenhouse gas (GHG) emissions (IPCC, 2001a). To limit the rate and level of climate change, a number of countries have formulated governmental policies to reduce these emissions. In the Kyoto Protocol of the Conference of Parties to the United Nations Framework Convention on Climate Change, quantitative objectives of greenhouse gas emissions reduction are specified. In this context, the European Union committed itself to an 8% reduction of 1990 greenhouse gas emissions in the period 2008 to 2012 (UNFCC, 1997)[1].

Mitigation of greenhouse gas emissions can be achieved by various options, i.e. increase of energy efficiency, increase of material efficiency, low carbon energy supplies, CO_2 capture and storage and enhanced carbon sequestration. Biomass can play a role in the mitigation process as well as in addressing other environmental, social and economic aspects of sustainable development[2] (Turkenburg et al., 2000). Biomass can be a substitute for conventional materials and can supply solid, liquid and gaseous fuels. Thus, biomass can lower the net CO_2 emissions of energy and material supplies (Johansson, 2000; IPCC, 2001b). In addition, biomass plantations can achieve net sequestration of carbon in plants and soils depending on the former use of the land.

---

[1] Greenhouse gas emissions in 1990 in the European Union were $4.2*10^9$ Mg CO_2 equivalent (UNFCC, 1997).

[2] For example, the production of biomass may improve access to modern energy carriers, create employment in rural areas, contribute to a more sustainable agriculture, reduce dependency on imported fuels and exploit new material markets.

Many uses of biomass for materials and energy are possible. Main uses for material are chemical products, pulp and paper and construction materials. Main uses for energy are the production of heat, electricity and transportation fuels. Figure 1.1 gives an overview of principal uses of biomass crops for energy and material purposes.

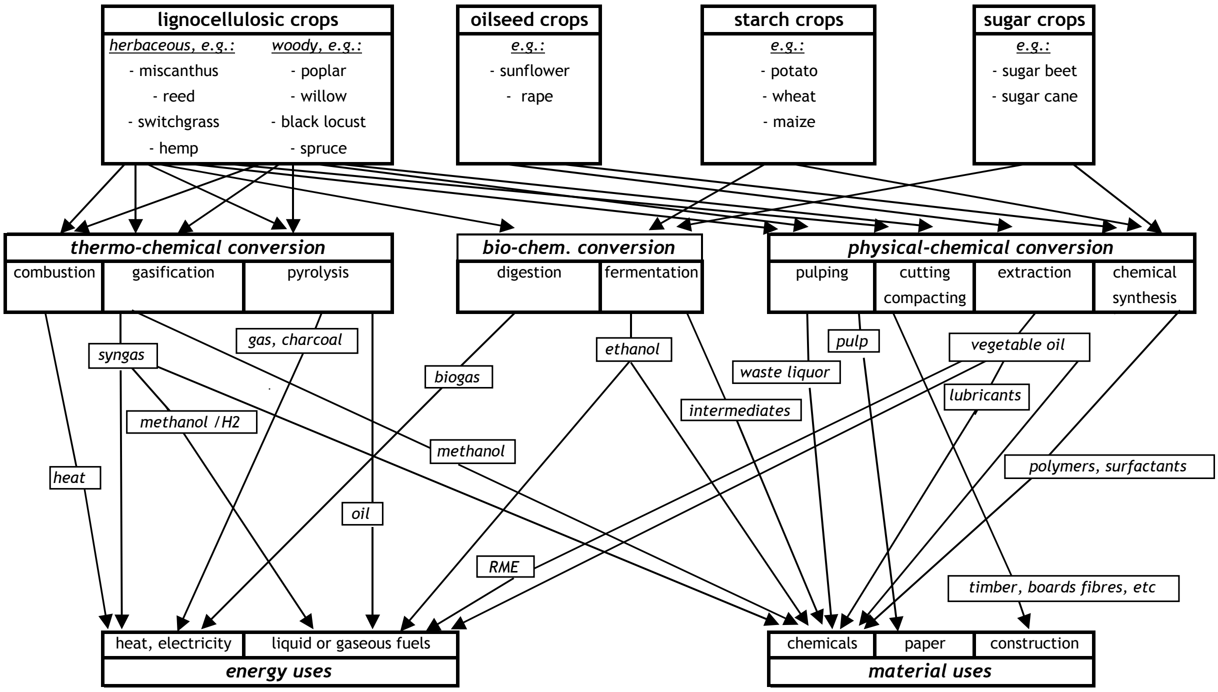


Figure 1.1: Principal uses of biomass for energy and material

In the scientific literature, the total global potential of biomass energy and material supplies is estimated to range between 80 to 1250 EJ/year in the long term (Hoogwijk et al., 2003), while the global primary energy consumption in the year 2001 was about 420 EJ/yr. (IEA, 2003a; IEA, 2003b). However, most studies expect a contribution of 50-200 EJ from biomass energy in 2050 with the largest part from dedicated energy crops and limited contributions from biomass residues (Berndes et al., 2003). Assuming that the global biomass potential of 50-200 EJ substitutes present primary energy supplies, GHG emission reductions of about $3*10^9$ to $12*10^9$ Mg CO_2/year could result (IEA, 2002c). For bio-materials the potential GHG emission reduction that can be achieved is more difficult to estimate, because various materials can be substituted. For example, Gielen et al. (2000) estimate that the technical potential of GHG emission reduction by bio-material use could be about $0.5*10^9$ Mg CO_{2eq}/year in Western Europe in 2030.

The potential amount of agricultural and forestry residues—available at low costs without an additional demand for land—is limited and estimated to be 30-90 EJ globally (Hoogwijk et al., 2004a). Therefore, dedicated energy and industrial crops also have to be used if biomass is to contribute substantially to the mitigation of greenhouse gas emissions. Currently, most alternatives for energy and material production from fossil fuels or the use of mineral resources are cheaper than energy or materials from dedicated crops, partly because agricultural land resource may be scarce and/or expensive. In scientific literature production costs of woody biomass range from 0.5 to 17.7 US$/GJ worldwide. In Europe this figure ranges from 2.5 to 16.4 US$/GJ (Hoogwijk et al., 2004b), while coal prices in Europe are at present about 2 US$/GJ (IEA, 2003c). (The potential of woody biomass production in Europe is estimated at 17-25 EJ (Johansson and Turkenburg, 2004).)

With regard to costs, the mitigation of GHG emissions by means of biomass use has to compete with other mitigation options. According to the IPCC, costs of selected GHG emissions reduction options are at present in a range of –110 to +60 US$/Mg CO_{2eq}[3], with GHG emission reduction costs of biomass substituting coal and gas ranging from –10 to +50 US$/Mg CO_{2eq} (IPCC, 2001b). However, most biomass uses from dedicated energy crops may at present be more expensive. For example, related to modern bio-fuels, Hamelinck (2004) estimates these costs to be in the range of 20 to 170 US$/Mg CO_2. Gielen et al. (2000) argue that most biomass material and energy uses will result in GHG emission reduction costs of above 50 US$/Mg CO_{2eq}. Consequently, to implement biomass for GHG emission mitigation on a large scale, GHG emission mitigation costs of many biomass material and energy options need to be reduced.

The availability of land for the production of biomass energy and materials is limited due to the competition with food production and the use of land for other purposes and functions (Hoogwijk et al., 2003). Consequently, several authors have indicated that an increasing demand for biomass for energy and material use will increase the market prices of agricultural land and subsequently the market prices of food, bioenergy and bio-materials; see e.g. Green (2000), Hoogwijk et al. (2004b) and Azar and Berndes (1999). This relation between demands, market prices and the amount of biomass supplied for food, materials and energy has been studied by e.g. De La Torre Ugarte et al. (2003), Gielen et al. (2000), Gielen et al. (2003) and Yamamoto et al. (2001)[4] and should be taken into account when analysing costs of biomass utilisation strategies.

[3] These GHG emission reduction options are divided in the following categories: Buildings and appliances (-110 to 10 US$/Mg CO2eq), transport, i.e. automobile efficiency improvements (-60 to 60 US$/Mg CO2eq), manufacturing (-70 to 60 US$/Mg CO2eq), agriculture (-20 to 60 US$/Mg CO2eq), wastes (-10 to 10 US$/Mg CO2eq) and energy supply (-10 to 60 US$/Mg CO2eq).

[4] De La Torre Ugarte et al. (2003) investigated the available land for energy crops in the U.S. depending on their market price using an agricultural policy model that includes demands and supplies in the agricultural sector. Gielen et al.

Currently, a modest amount of biomass is used for material and (non-traditional) energy supply in industrialised countries and most biomass used for energy are residues. In the European Union (EU-15) only about 2.5% of gross inland energy consumption or 1.5 EJ were derived from biomass in the year 2000 (Johansson and Turkenburg, 2004). On the material side biomass is used mainly for traditional application as pulp & paper, sawn wood and particleboard. In total wood products and pulp and paper accounted for about 25% of bulk material[5] production in the European Union in 1999, which is about 4 EJ. In contrast, advanced bio-materials are at present used to a much lesser degree. For example, bio-based plastics had a market share of below 0.1% (below about 3 PJ) in the European Union in 1998 (ECCP, 2001).

In conclusion, the availability of agricultural and forestry residues is limited, while the technical potentials for biomass production are quite high (Hoogwijk et al., 2004a). Focussing on Europe, one of the main reasons for the low share of biomass applications from dedicated crops are the often-high production costs due a.o. the relative low availability of agricultural land. Therefore, more competitive routes for the introduction of biomass are needed in the short to medium term.

2 Multi-functional biomass systems

Multi-functional biomass systems may contribute substantially to a more efficient use of biomass resources and agricultural land, resulting in low mitigation costs of greenhouse gas emissions. Therefore, in this thesis two concepts of multi-functional biomass systems—multi-product use and cascading—are investigated. In this section, the basic principle of multi-functional biomass systems, namely the multiple-use of resources, is discussed and the concepts of multi-product use and cascading are defined. Also, an overview of possible multi-functional biomass systems is given.

(2000) modelled the future use of biomass for energy and materials in Western Europe depending on the market price of GHG emissions. The authors use a linear optimisation model, which calculates optimal technological choices (e.g. kind of biomass production) given an exogenous demand of products. Yamamoto et al. (2001) estimated the global potential supply of bioenergy depending on the demand for material and food using a land use model that simulates competitions between the various uses. Finally, Gielen et al. (2003) analysed different biomass policies for CO2 emission reduction considering global demands for materials, energy and food modelling the competing land uses maximising consumer/producer surplus in an ideal market.

[5] Round wood, plastics, crude steel, aluminium, glass, paper & board, cement and bricks & tiles (Phylipsen et al., 2002; UN/ECE and FAO, 2001).

2.1 Multiple use of resources

Within the concept of multiple land use, land generates more than one type of product or service like the production of food, fodder, energy and materials, the protection of the soil, wastewater treatment, recreation, or nature protection; see e.g. (Börjesson, 1999; Londo, 2002; Lewandowski et al., 2003) In this thesis, however, we concentrate on multiple use of biomass resources. Hence, we exclude multiple land use from the analysis.

The multiple use of biomass resources can be achieved in several ways. First, different parts of a crop may each be used for a specific purpose. Second, processing of biomass may lead to various product and by-products. Finally, biomass may be used to produce materials and energy in succession, i.e. the recycling and cascading approach.

In scientific literature, the potential of using each part of a biomass resource for a specific purpose, thus increasing efficiency of biomass utilisation mainly with regard to costs, has often been discussed; see e.g. (DTO, 1997; Benjamin and Weenen, 2000).

Producing several products and by-products from biomass is also an often-examined concept. Wright and Cushman (1997) stress the importance of increasing the use of by-products, and Williams et al. (1995) and Turkenburg et al. (2000) discuss the co-production of fuels, heat and electricity. Another concept of multiple biomass resource use is the bio-refinery; see e.g. (Scott et al., 1997; Benjamin and Weenen, 2000; Elliot, 2004; Paster et al., 2003; Realff, 2003). A bio-refinery is an analogy to a petrochemical refinery where crude oil is completely reverted into many different value-added products maximising the economic benefits. In a bio-refinery, the biomass resource is converted into bio-materials and/or energy carriers. Often a main product of bio-refinery is a bulk chemical.

Increasing the efficiency of biomass utilisation by recycling of bio-materials and waste-to-energy conversion is discussed by several studies; see e.g. (Goverse et al., 2000; Fraanje, 1997; Boogardt, 2000). However, in these studies the cascading of wood products is investigated mostly in a qualitative way.

Hence, in scientific literature, the multiple use of biomass resources is seen as an important possibility to increase the efficient use of biomass for materials and energy, and reduces the costs of biomass options to mitigate GHG emissions. Nevertheless, only very few studies have investigated the potential of multi-functional biomass systems in a quantitative way.

2.2 Multi-product use and cascading

The principles of multiple biomass resource use can be summarised under the terms *'multi-product use'* and *'cascading'*. *Multi-product use* is defined as using biomass for different applications. *Cascading* is the subsequent use of biomass for a number of applications. In other words, biomass is used first for a material application; next, it may be recycled for several further material applications; and finally, energy is recovered from the bio-material waste.

In this thesis, multi-functional biomass systems are systems that combine in principle bio-material use(s) and bioenergy use(s). Figure 1.2 gives an overview of such a multi-functional biomass system.

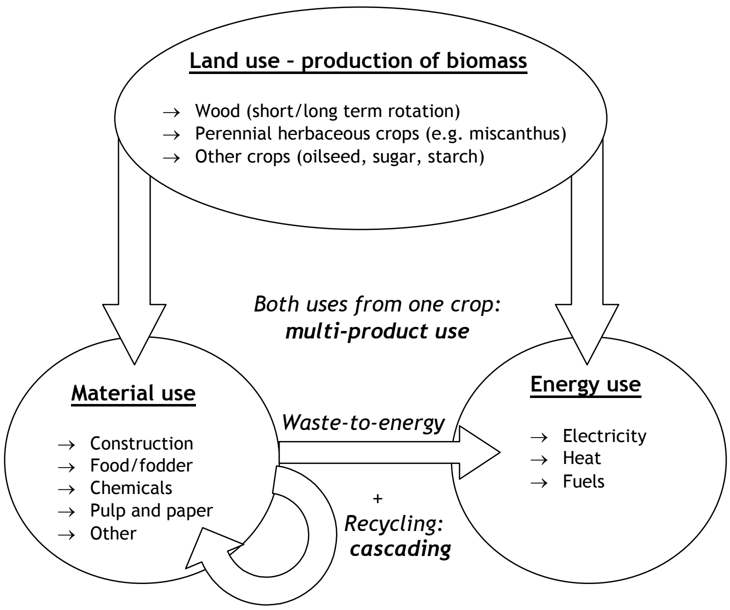


Figure 1.2: Schematic overview of multi-functional biomass systems

2.3 Multi-functional biomass systems

As many different biomass crops, bio-material applications and bioenergy conversion steps exist, obviously, a very large number of multi-functional biomass systems are possible. The main application of the biomass resource characterises the feasibility of a multi-functional biomass system. In the case of material production, the application determines the selection of crops, the availability of agricultural residues, the supply of by-products, potential recycling steps and waste-to-energy conversions.

The bio-material application in a multi-functional biomass system should at least offer a good possibility for multi-product use or cascading. In the case of multi-product use, this

Table 1.1: Selected biomass material applications and their suitability for multi-functional biomass systems

Application	Biomass resource	Main substitution	By-products and residues	Cascading	Market price[a]	Market volume[b]	Spec. CO_2 emission reduction	Profitability	Add. market volume	Remarks
Pulp and paper										
Pulp and paper	Ligno-cellulose fibres	Pulp from forestry	Lignin, bark, twigs	Waste paper	430-440 $US\$_{2002}/Mg$	$34*10^6$ Mg_{2002} (0.1 EJ)	--/-	+	-/0	Well established, low additional potential
Chemicals										
Ethylene	Sugar, lignocellulose	Petrochem. ethylene	Agric. residues, fodder, lignin	Intermediate	550-620 $US\$_{2001}/Mg$	$20*10^6$ Mg_{2002} (1.0 EJ)	+/++	-/+	++	Better if ligno-cellulose route developed
Bio-based polymers	Sugar/starch ligno-cellulose	Petrochem. polymers	Agric. residues, fodder, lignin	Polymer recycling	770-1540 $US\$_{2002}/Mg$	$34*10^6$ Mg_{2002} (1.3 EJ)	0/+	-/+	0/++	Better if lingo-cellulose route developed
Fibre reinforced composites	Ligno-cellulose fibre	Glass fibre composites	Agric. residues	None	3180-6360 $US\$_{2001}/Mg$	$2*10^6$ Mg_{2002} (≈ 0 EJ)	-/0	+/++	-/0	Possible profitable, but only small market volume
Lactic, levulinic, succinic acid	Sugar/starch	Various petrochemicals	Agric. residues, food	Intermediate	See ethylene	See ethylene	-/+	-/+	0/+	Shift of chemical products necessary
Solvents	Vegetable oil	Petrochem. solvents	Agric. residues, fodder	Re-use	300-2000 $US\$_{2001}/Mg$	$4*10^6$ Mg_{1999} (0.2 EJ)	-/++	0	-/0	Small market
Lubricants	Vegetable oil	Petrochem. lubricants	Agric. residues, fodder	Re-use	≈3000 $US\$_{1998}/Mg$	$5*10^6$ Mg_{1999} (0.2 EJ)	+/++	0	-/0	Small market
Construction materials										
Sawn timber	Wood	Construction material	Small wood pieces	Wood products	190-210 $US\$_{2002}/m^3$	$80*10^6$ m^3_{2002} (0.8 EJ)	+/++	0/+	0/+	High requirements wood properties
Engineered wood prod.	Wood	Construction material	Small wood pieces	Sawn wood	≈420 $€_{2003}/m^3$	$1*10^6$ Mg_{2002} (0.0 EJ)	+/++	0/+	0/+	High requirements wood properties
Insulation	Ligno-cellulose fibres	Mineral insulation	Agricultural residues	Re-use fibres	40-50 ECU_{1995}/Mg	$52*10^6$ Mg_{1992} (0 EJ)	+/++	0	0/+	Recycling feasibility largely unknown
Fibre, particle board	Ligno-cellulose fibres	Board materials	Bark, twigs, agric. residues	Re-use fibres, part.	190-320 $US\$_{2002}/m^3$	$40*10^6$ m^3_{2002} (0.4 EJ)	0	0/+	0	Good cascading possibilities

--: not suited, -: poorly suited, 0: medium suited, +: well suited, ++: very well suited

[a] Values in this column are current market prices of wood products and non-biomass reference materials: pulp (FAO, 2004); ethylene (Anonymous, 2001) ; bulk plastics (Leaversuch, 2002), bulk fibre reinforced composites (DiGITIP, 2002); range solvents (Klass, 1998); synthetic lubricants (Technical Insights, 1999); sawn timber (FAO, 2004); engineered wood products (Finnforest, 2003); insulation material (EU, 1997); particle and fibreboard (FAO, 2004).

[b] Market volumes are in the case of wood products the currently produced amounts and in other cases the amounts of fossil or mineral alternative products in Western Europe. Thus, no estimation of future bio-based market potentials has been made, here. The potential in EJ is based on the heating value of the products. Market volumes: pulp (FAO, 2004); ethylene (CEFIC, 2003); plastics (APME, 2003); fibre reinforced composites (DiGITIP, 2002) – heating value of only glass fibres, partly also carbon fibres are used; solvents (ECCP, 2001); lubricants (ECCP, 2001); sawn timber (FAO, 2004) – for comparison density of wood and wood products is roughly 0.5 Mg/m^3 ; engineered wood products (UN/ECE and FAO, 2001); insulation material, i.e. mineral wool (EC, 2002); prices particle board and fibreboard (FAO, 2004).

means that residues or by-products from material production may be used and in the case of cascading, that the material may be recycled. Table 1.1 shows principle bio-material applications that fulfil this criterion.

Moreover, as discussed above, the bio-material should contribute to an efficient reduction of GHG emissions in terms of costs and land use. Three related parameters are used and evaluated qualitatively in Table 1.1. First, the *specific CO_2 emission reduction* describes the reduction of CO_2 emissions per unit of bio-material. This parameter depends mainly on the fossil fuel use during bio-material production and the production of the reference material. Second, the *profitability*, which is derived from, the difference between the market prices of bio-materials and their respective production costs. Third, the *additional market volume* describes the total technical potential minus current bio-material production. This parameter is an indicator for the total potential GHG emission reductions that can be achieved.

Main sectors of bio-material uses are *pulp and paper*, *chemicals* and *construction*. The *pulp and paper* industry currently uses a large part of woody biomass. The additional potential for pulp and paper production, to substitute the use of other materials—e.g. plastics—and, thus, to reduce GHG emissions is probably low. Contrary, the *chemical* industry is an interesting sector offering new opportunities for biomass utilisation (DTO, 1997). Especially bulk chemicals from biomass have a large potential to substitute fossil fuel feedstock in the chemical industry. Finally, bio-materials for *construction* have already a large market share. In this case, particularly a large cascading potential exists, based on a variety of recycling options.

Obviously, *biomass resources* should have a suitable quality for the application selected. Furthermore, biomass crops in a multi-functional biomass system should either have a high total biomass yield for bulk material and energy applications or have a high yield of a specific plant component required for a high-value added material application, e.g. fibres, sugar or starch. Finally, *bioenergy* conversion technologies in a multi-functional biomass system should be able to convert efficiently the available agricultural residues, processing residues and bio-material wastes.

Summarising, multiple biomass resource use may be a promising concept to decrease the costs and land use of GHG emission mitigation by biomass material and energy uses. In this thesis, we will focus on multi-functional biomass systems with multi-product use and cascading. Material application that seem promising for these systems are bulk chemical and construction materials.

3 Efficiency of biomass systems

Concluding that an efficient reduction of GHG emissions at low costs and with minimal land use is an important criterion for the evaluation of (multi-functional) biomass systems, leads to the question of how to quantify this efficiency. Important parameters are savings of non-renewable energy consumption (GJ), GHG emission reduction (Mg CO_{2eq}), (agricultural) land use (ha) and costs (€). From, these parameters the efficiency of GHG emission reduction, i.e. costs (€/Mg CO_{2eq}) and land use (ha/Mg CO_{2eq}) can be derived.[6]

3.1 Savings of non-renewable energy consumption

Investigating energy balances of different biomass systems is a common way to compare especially bioenergy systems. Furthermore, energy balances can be an important indicator of GHG emission balances as most GHG emissions in biomass systems are related to fossil energy uses and savings. In order to set up an energy balance, generally energy inputs and outputs of a biomass system have to be accounted for. These inputs are direct energy uses, e.g. diesel use for tractor in biomass production, or indirect energy uses, e.g. energy uses for the production of a biomass combustion plant. In scientific literature, the energy inputs are compared often to the energy content of the bioenergy carrier resulting in net energy outputs or net energy ratios; see e.g. the overview of bioenergy balances of bio-ethanol in Shapouri et al. (2002) or the comparison of energy crops in Venturi and Venturi (2003).

Taken into account that a material or energy carrier produced from biomass replaces a reference product with the same function, energy balances can also be used to investigate primary or non-renewable energy savings. This approach has been applied to compare energy balances of bio-based materials with each other (see e.g. Patel et al., 2003) and to compare bio-materials to bioenergy production (see e.g. Kaenzig et al., 2004). Because evaluating biomass systems by savings of non-renewable energy consumption allows comparing different types of products and systems, this parameter is used for the analysis of multi-functional biomass systems in this thesis.

3.2 GHG emission reductions

Most important greenhouse gases in the context of biomass systems are carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). Bioenergy and -material have a 'short car-

[6] Note, that many other criteria may play a role in the assessment of biomass systems, e.g. eutrophication, health impacts, contribution to rural development, employment or public perception. Given the central research question of this thesis, however, these aspects are not considered here.

bon cycle', i.e. carbon is removed from the atmosphere during biomass growth and then released during biomass use. Thus, in comparison to the 'long' carbon cycle of fossil energy carriers CO_2 emissions are reduced when fossil fuel use is replaced by the use of biomass resources. Nitrous oxide is emitted in biomass production systems during N-fertilisation. N_2O may be an important fraction of the total GHG emissions of biomass production systems depending on the agricultural management scheme; see e.g. IPCC (2001b) and Heller et al. (2003). Finally, methane may be emitted if (part of) the biomass is decomposed anaerobically, e.g. stumps left during forest harvesting or land filled bio-materials. However, in most agricultural biomass system—apart from rice cultivation—methane emissions play a minor role (IPCC, 2001b). All GHG emissions can be expressed by their global warming potential (IPCC, 2001a). In this thesis, we use the expression CO_2 equivalents when evaluating GHG emissions based on the global warming potential of each greenhouse gas during 100 years. (Consequently, 1 kg CH_4 is equivalent to 23 kg CO_2 and 1kg N_2O is equivalent to 296 kg CO_2 (IPCC, 2001a).)

Accounting of GHG emission balances follows the same principle as accounting for energy balances, i.e. determining all direct and indirect GHG emissions of biomass systems. GHG emission reductions are also calculated by comparing the emissions of the biomass system to the emissions of a reference system that produced comparable materials and/or energy carrier; see e.g. Börjesson and Gustavsson (2000). An important methodological aspect in this context is the choice of the reference system as very different materials and energy carriers may be substituted by biomass utilisation (Vikman et al., 2004; Schlamadinger et al., 1997). Waste treatment plays an important role, too, because the type of waste treatment influences the release of carbon embodied in materials; see e.g. Finnveden and Ekvall (1998) and Patel et al. (2003). For example, incineration releases all embodied carbon but might recover energy and, thus, reduce GHG emissions from fossil energy, while land filling may store part of this carbon.[7] Consequently in this thesis, special attention is paid to the choice of the reference system and waste treatment of bio-materials and reference materials are included in the GHG emission balances.

Time aspects may also play an important role in GHG emission reduction accounting. Mitigation of carbon dioxide emissions can be achieved by the reduction of carbon emissions from fossil resources or the storage of carbon outside the atmosphere, e.g. in forests or bio-materials. However, if carbon is stored in forest or bio-material, carbon dioxide

[7] Gardner et al. (2002) show that large part of wood remains intact in a 20 to 30 year old Australian landfills. However, IPCC (2001b) argues that even though some of the carbon might be sequestered infinitely, land filling is a disadvantageous waste treatment technology for organic wastes compared with composting and digestion.

emissions are not avoided for an indefinite period. Methods to account for such carbon storage are widely discussed, especially for carbon sequestration in forests; see e.g. OECD and IEA (2001), Kirschbaum (2003) and Richards and Stokes (2004).

For bioenergy production, Schlamadinger et al. (1997) developed a methodology to account for changes in carbon stocks in the whole system. This methodology includes the definition of a reference system including for example land use and energy production. Moreover, it is argued that rather the dynamic change of carbon stocks in the different carbon pools—e.g. fossil fuels, standing forest—should be measured over a longer time period than CO_2 emission reduction at a certain moment in time. However, studies that compare the temporarily storage of carbon in bio-material systems with the substitution of fossil energy carriers and aggregate the results as total GHG emission reduction are rare. Methodological work in this field is still going on; see e.g. IEA (2004a). In this thesis, we attempt to describe carbon stock changes in biomass material and energy systems in terms GHG emission reductions.

3.3 Agricultural land use

Land use in different biomass systems per unit of non-renewable energy savings, GHG emission reduction or costs can vary considerable. This is due to different yields of crops, materials and energy carriers and to different values of biomass products in terms of energy savings, GHG emission reductions and costs. In early studies of biomass system land use played a minor role. However, the comparison on an agricultural area basis becomes more and more an important issue; see e.g. IPCC (1995), van den Broek et al. (2001), Keith (2001) and Kaenzig et al. (2004). Also, Dinkel et al. (1996), who carry out an analysis of bio-based polymer production, conclude from their result that a comparison of bio-materials and bioenergy on an agricultural land use basis would be desirable.

The impact of land use can also be described by other criteria, e.g. influence on soil quality, erosion prevention or bio-diversity; see e.g. Baitz et al. (2000) and Lindeijer (2000). The implementation of biomass systems on a large-scale in order to reduce a significant amount of GHG emissions, however, is mainly limited by the costs and scarcity of agri cultural land. Therefore, in this thesis we concentrate on the amount of land that is used for bio-material and energy production and the reduction of GHG emissions as important parameters.

3.4 Costs of biomass systems

Finally, costs of biomass systems are an often-studied criterion; see e.g. JRC (2003), Hamelinck (2004) and Hallam et al. (2003). Like GHG emission and energy balances, net costs can be calculated over the whole biomass systems accounting for all costs and revenues. Cost calculations from an industrial perspective, using typically high discount rates and short lifetimes, lead to higher cost estimates than calculations from a societal perspective, using typically lower discount rates and longer lifetimes. Moreover, analyses from a societal perspective often consider the whole biomass systems, while in an industrial perspective often single processing steps, e.g. bio-electricity production or bio-material production, are regarded. This may also lead to different cost estimates.

Moreover, costs can be calculated using either a bottom-up or top-down approach. In the bottom-up or 'engineering' approach, costs of every part of the biomass system, e.g. agricultural land rent, transport costs or equipment for biomass combustion, are estimated and added up. In contrast, the top-down approach uses aggregated data from economic sectors and estimates the induced changes in the economy by the introduction of a biomass system. Therefore, in a top-down approach, cost estimates on a technology level are much less detailed. On the other hand, interactions between sectors—resulting for example in market prices changes—can be investigated much better.

Consequently, results of bottom-up and top-down studies are principally different but can supplement each other. Richards and Stokes (2004) compare the results of these different approaches in several studies for carbon sequestration. For bio-material and bioenergy systems it is found that the change of market prices of agricultural land and biomass products can play an important role, if these systems are introduced on a large scale. For example, Azar and Berndes (1999) show that food and land prices increase due to higher bioenergy prices and Otto and Gallagher (2004) studied price effects in the feed market due to an increased production of ethanol. However, bottom-up cost estimates of bioenergy and material systems typically do not include such market price changes.

As governments initiate GHG emission reduction policies, cost estimates from a societal point of view using a bottom-up approach, seem an appropriate approach to determine specific benefits of different multi-functional biomass systems. To evaluate costs and effects of large-scale implementation of multi-functional biomass systems, however, market price changes caused by this implementation need to be investigated, too.

3.5 Efficiency of multi-functional biomass systems

Summarising, in the literature, many different biomass systems have been compared in view of their energy savings, GHG emission reduction, costs and to a lesser degree land use efficiencies. Methods for this comparison are available or in the case of GHG emission reductions still in development.

However, only very few studies have explicitly addressed multi-functional biomass systems, i.e. multi-product use and cascading, in a quantitative way. In the context of multi-product use, studies of bio-material production often take into account the use of processing residues for energy (see e.g. Hekkert and Worrell, 1998) but usually do not include the utilisation of agricultural residues. Furthermore, few studies investigate the benefits of producing several products from a biomass resource quantitatively. As an exception Wyman (2003) investigated potential economic synergies of bio-refineries.

In view of cascading, several studies have compared recycling and reuse of a material to virgin production or other waste treatment strategies; see e.g. Finnveden and Ekvall (1998). However, cascading chains of biomass including several successive applications have not been compared quantitatively.

Summing up, the potential benefits of multi-functional biomass systems still need to be proven in quantitative analyses. For this purpose, adaptation or further development of methodologies are sometimes needed. Particularly, problems of allocation or system expansion to account for different products and land uses in multi-functional biomass systems have to be solved; see e.g. Overend (2004) and Edwards (2004). Moreover, the issue of accounting for the time dimension in especially long-life cascading applications needs to be solved. Finally, the integration of market price changes of bio-materials, bioenergy and land due to the large-scale introduction of biomass systems in bottom-up costs analysis deserves special attention.

4 Objective and scope of this thesis

Summarising the possibilities of multi-functional biomass use to contribute to a necessary reduction of costs and an efficient use of land of bioenergy system need to be better understood in a quantitative way. Therefore, the central research question of this thesis is:

What is the potential of multi-functional biomass systems to improve the costs and the land use efficiency of saving non-renewable energy consumption and reducing GHG emissions in quantitative terms?

Two main aspects play an important role in answering this central question. First, methodologies to account for costs, land use, GHG emissions and non-renewable energy consumptions need to be adapted for the evaluation of such multi-functional biomass systems. Second, the potential benefits depend on the kind of biomass system regarded, i.e. the materials and energy carriers produced, the scale of the system and the multi-functional resource uses applied. The mechanism of this dependence have to be studied in order to identify promising multi-functional biomass systems.

Therefore, in the following chapters the performance of multi-functional biomass systems is quantified. In this thesis, we investigate biomass system costs from a societal perspective using e.g. low discount rates. A main focus will be on the review of methodologies for accounting GHG emissions, non-renewable energy consumption, agricultural land use and costs as well as the adaptation of these methodologies to special aspects of multi-functional biomass use. The analysis of the potential benefits of multi-functional biomass systems is carried out by several case studies of biomass systems including various waste treatment technologies for the short term —present until 2010[8]—that appeared promising after a first review.

Because at present the shift of biomass production to more favourable areas seems to be an alternative for more efficient biomass systems,[9] these case studies are situated in Europe and concentrate on Poland in order to investigate the potential of biomass production in the new EU-member states of Central Eastern Europe.[10]

In *Chapter 2* of this thesis, the concept of multi-product use and its potential impacts on fuel costs of bioenergy and GHG emission reduction per area of agricultural land use are investigated. Especially, the relation between the economic value and the specific GHG emission reduction of a possible material application and the potential benefits of multi-product use is analysed. As for different parts of a crop more or less valuable material ap-

[8] Because of short timeframe, increase of agricultural productivity is not considered in this thesis.
[9] In the longer term, also an increase of productivity of biomass production is expected.
[10] Within Europe the new accession states of the European Union in Central Eastern Europe seem to be a good option, because these states have at present large areas of available agricultural land, potentially high to medium crop yields and comparably low costs of land and labour; see e.g. van Dam et al. (2004).

plications are possible, the benefits of multi-product use vary with the percentage of crop used for material applications, which is investigated in this chapter.

For this analysis, a case study of hemp, poplar and wheat is carried out. Material uses regarded for multi-product use are the use of wheat grains for food, wheat straw for animal litter, hemp bark fibres for reinforced composites, hemp core fibres for animal litter, hemp seeds for food and cosmetics and poplar wood chips for pulp. For energy uses parts of the crops are used as solid fuel for electricity production. This case study compares potential benefits in a Western European and a Central Eastern European Country, i.e. the Netherlands and Poland.

Chapter 3 examines the concept of cascading more closely. In this chapter, the main focus is on the development of methodology for accounting CO_2 emissions and costs of cascading, as methodological aspects play a large role if several materials and energy carriers are produced subsequently from a given biomass resource. In this context, the boundaries of the biomass cascading system and the respective reference system, the inclusion of land use, the time dimension and the definition of reference applications are important issues that are discussed in detail.

Moreover in this chapter, key parameters and issues that influence the efficiency of biomass cascading chains are identified like it is done in Chapter 2 for multi-product use. Parameters to investigate the performance of cascading chains are their CO_2 emission reduction, their total costs per area of agricultural land use and their CO_2 mitigation costs. Special attention is paid to the influence of time and discount rates, to the combination of different material and energy applications in cascading chains and to the historical variations of markets prices on the performance parameters. This is done by a case study of cascading chains of short rotation (SR) poplar. Different material and energy applications of SR poplar are regarded within these cascading chains, i.e. particle lumber, MDF board, transportation pallets, pulp, ethylene, viscose, methanol and electricity. Geographical scope of this case study is Poland.

Chapter 4 deepens the investigation of the importance of land use efficiency as criteria to evaluate non-renewable energy consumption and GHG emission reduction of biomass uses and to compare the benefits of bio-materials and bioenergy. Here, the difference between a mass-based evaluation of bio-materials and an area-based evaluation is analysed. Moreover, the possible improvement of savings of non-renewable energy consumption and GHG emission reduction of bio-material by the use of agricultural residues is investigated. The influence of allocation methods on the results is particularly studied.

Subject of this investigation are different bio-based polymers including natural fibre composites. These polymers are compared to each other, using different evaluation criteria and to some options of bioenergy production. This analysis is based on an extension of existing Life Cycle Assessment (LCA) studies.

Following *Chapter 5* studies a more complex multi-functional biomass system, combining multi-product use and cascading. The influence of key factors in the set-up of biomass systems—including the kind and amount of multi-functional biomass resource use—on the savings of non-renewable energy consumption, the reduction of GHG emissions and costs of the biomass system are studied. In Chapter 2 and 3, it turned out that market prices of materials are crucially important for the economic success of multi-functional biomass use, but that these market prices can be quite variable and may depend on the market size. Therefore, in this chapter a market analysis is added to the chain analysis that takes into account increasing land prices and decreasing prices of material products and by-products if biomass systems are introduced on a large scale. This market analysis is combined with economies of scale to determine impact of production volume on the costs of multi-functional biomass systems.

In this study, the case of a poly lactic acid (PLA) bio-refinery system in Poland is analysed. PLA is a commercially produced bio-based polymer that appeared promising from the assessment of bio-based polymers in Chapter 3. The bio-refinery system includes the production of biomass, i.e. wheat or short rotation wood, the use of residues for bioenergy, the use of material by-products, the substitution of petrochemical polymers by PLA products, back-to monomer recycling of parts of the PLA produced and waste treatment of the remaining PLA.

Finally, in *Chapter 6* the introduction of multi-functional biomass systems on a large scale is studied. Main focus of the analysis is the effect of increasing scale of biomass use on the mitigation costs of GHG emissions. Increasing agricultural land prices, decreasing biomaterial and bioenergy prices due to a larger demand or supply of these commodities are taken into account as well as increasing biomass supply costs.

For this market analysis of GHG mitigation costs, four biomass systems from the previous chapters are compared to each other, i.e. the production of MDF board, PLA, methanol and electricity. The case study is again situated in Poland.

This thesis finishes with a summary, the conclusions of the previous chapters and a reflection on the results of the overall thesis related to the central research question.

Chapter 2:

Economic and greenhouse gas emission analysis of bioenergy production using multi-product crops – Case studies for the Netherlands and Poland*

Abstract

In the face of climate change that may result from greenhouse gas emissions, the scarcity of agricultural land and limited competitiveness of biomass energy on the market, it is desirable to increase the performance of bioenergy systems. Multi-product crops, i.e. using a crop partially for energy and partially for material purposes can possibly create additional incomes as well as additional GHG emission reductions. In this study, the performance of several multi-product crop systems is compared to energy crop systems, focussed on the costs of primary biomass fuel costs and GHG emission reductions per hectare of biomass production. The sensitivity of the results is studied by means of a Monte-Carlo analysis. The multi-product crops studied are wheat, hemp and poplar in the Netherlands and Poland. GHG emission reductions of these multi-product crop systems are found to be between 0.2 and 2.4 Mg CO_{2eq}/(ha*yr) in Poland and 0.9 and 7.8 Mg CO_{2eq}/(ha*yr) in the Netherlands, while primary biomass fuel costs range from –4.1 to –1.7 €/GJ in the Netherlands and from 0.1 to 9.8 €/GJ in Poland. Results show that the economic attractiveness of multi-product crops depends strongly on material market prices, crop production costs and crop yields. Net annual GHG emission reductions per ha are influenced strongly by the specific GHG emission reduction of material use, reference energy systems and GHG emissions of crop production. Multi-product use of crops can significantly decrease primary biomass fuel costs. However, this does not apply in general, but depends on kind of crops and material uses. For the examples analysed here, net annual GHG emission reductions per ha are not lowered by multi-product use of crops. Consequently, multi-product crops are not a priori an option to increase the performance of bioenergy systems. Further research on the feasibility of large-scale multi-product crop systems and their impact on land and material markets is desirable.

* Accepted for publication in *Biomass & Bioenergy*; Co-Authors: G. Termeer and A.P.C. Faaij.

1 Introduction

The utilisation of biomass instead of fossil fuels to fulfil our energy needs can contribute to the reduction of greenhouse gas (GHG) emissions and the dependence on fossil fuel energy carriers. Therefore, nowadays policy objectives often aim at a relatively large increase of the share of bioenergy to the total energy supply. For example, the European Union aims at a share of 8.5% of the total supply in 2010 whereas in 2000 it was 3% (EC, 1997). However, a large contribution of bioenergy can only be obtained by the production and utilisation of energy crops. Two major barriers to achieve that are the scarcity of land and the relative high costs of the bioenergy carriers produced.

Land and especially good quality land that can be used to produce bioenergy is scarce and could limit the supply and competitiveness of bioenergy significantly (Green, 2000; Keith, 2001). This scarcity of land is caused by the competition between the production of food, bio-material and bioenergy on available agricultural and forestry areas and other competing land uses, e.g. urbanisation and nature development (Turkenburg et al., 2000; Hoogwijk et al., 2003).

In addition, many authors have identified the costs of bioenergy as a major problem for the introduction of bioenergy on a large scale; see e.g. Green (2000) and IPCC (2001b). For example, Dinkelbach et al. (1999) estimate energy crop production costs at about 3-6 €/GJ in the Netherlands. Hallam et al. (2001) estimate herbaceous crop production costs at about 2-5 €/GJ in the U.S. At the same time, fossil fuels can be obtained on the market for about 2-3 €/GJ for coal and 2-5 €/GJ for natural gas (IEA, 2002d).

To maximise GHG emission reduction per area of land and to reduce the costs of biomass for bioenergy, using a crop only partially for energy and partially for material purposes may be a solution. This is because the material component of the crop may create (1) additional incomes and (2) additional GHG emissions reductions. Various 'material' applications - in this context defined as non-energy applications- are possible such as food, fodder, pulp & paper, construction material and chemicals. A crop that is split into two or more parts that are used for material and energy applications is a *multi-product crop*. In the rest of this chapter we will regard this concept of multi-product crops more closely.

In general, all agricultural and forest crops are suitable as multi-product crop. Typically, the main component of a crop, i.e. the component that is the primary objective of production, is used for material applications including food. The rest of the plant, i.e. the 'residue', is often left on the field or in the forest, but can be used for energy purposes. Examples of main crop components are seeds, ligno-cellulosic fibres, lumber, vegetable oils, starch and sugar. Examples of typical residues are bark, small twigs, plant stalk and leaves. Even though the utilisation of the main component for material purposes and the residue for energy production is most common, other combinations are possible. Residues can be used for material applications, e.g. grain straw as construction material, and main components for bioenergy, e.g. combustion of lumber.

As a consequence, a broad variety of multi-product crop systems are possible. Well-known examples of multi-product crop systems in the context of 'traditional' agriculture are the production of cereals and sugar cane and the combustion of straw and bagasse for heat or power production. Less traditional multi-product crop system are possible, too. An example is the production of maize, where starch is used for polymers production, by-products from starch production for fodder and the stalks and leaves for energy purposes.

As discussed above, multi-product crops systems have the potential to improve the competitiveness of bioenergy systems and the impact of these systems on GHG emission reductions. However, these benefits are not taken to be granted. Many parameters influence the performance of multi-product crop systems. First, characteristics of the biomass production system, e.g. crop yields, production costs and energy use, play an important role. These characteristics in turn depend on the intensity of agricultural practice (mechanisation, fertiliser input, etc.) and local circumstances (soil, climate, etc). Second, material uses determine the performance of multi-product crop systems. Important variables are the kind of substituted material, material market prices and energy inputs of material production. Finally, the kind of energy use (e.g. end use energy carrier, conversion technology) and performance of the reference energy system are influential, too.

Several authors have studied relations between land availability, material markets and bioenergy supply. For example, Hoogwijk et al. (2003) determined bioenergy costs depending on food and energy demands on a global scale. Gielen et al. (2000) calculated the reduction of GHG emissions by biomass in relation to energy and material demands in Europe. Marrison and Larson (1996) investigated the relation between bioenergy supplies and food production in Africa and Leemans et al. (1996) investigated this relation on a global scale. De La Torre Ugarte and Ray (2000) modelled bioenergy supplies in relation to food and material demands in the U.S. Even though biomass material production and residue utilisation for energy are typically included in this model, possible benefits of multi-product crop utilisation (in relation to the various parameters discussed above) have not been quantified, yet.

Therefore, we *determine primary biomass fuel costs and GHG emission reductions per area of land of multi-product crop systems*. The results are compared to utilising the whole crops for either energy or material only. In that way benefits of multi-product crops are determined. Moreover, because many different variables influence the results, the sensitivity of the results on various parameters is studied by means of a Monte-Carlo analysis.

In our analysis a *case study approach* is followed focussed on three multi-product crops: *wheat, hemp* and *short rotation poplar*. Wheat is a 'traditional' annual food crop that is cultivated for its grain, while wheat straw can be used for energy generation. Hemp is a promising industrial crop for material uses. Fibres and seeds of hemp are used for e.g. textiles, composites pulp, food and chemicals. The remaining hemp straw can be used for energy generation. Finally, short rotation poplar is a promising perennial energy crop, because of its potentially high biomass yields. Moreover, besides for energy production short rotation wood can be used for material purposes like pulp and particleboard.

In this study, we restrict ourselves to multi-product crop systems in Europe focussed on two countries, *the Netherlands* and *Poland,* characterised by different types of agricultural production systems and economic conditions. The Netherlands is a Western European Country with a highly intensified agricultural production system. For example, wheat yields in the Netherlands are the highest within Europe. Furthermore, costs of land and labour are high. The Netherlands are EU-member receiving EU agricultural subsidies. Poland is a Central Eastern European country in transition. Agricultural production in Poland is less mechanised than in the Netherlands and costs of land and labour are low. Poland is supposed to join the EU in 2004 and does not participate in the EU agricultural subsidy system yet.[1]

In Section 2 of this chapter, the method of calculating bioenergy costs and GHG emission reductions per area of land of multi-product crop systems is described. In Section 3, input parameters for the selected multi-product crop systems, are presented and discussed. Primary bioenergy fuel costs and GHG emission reductions of the studied multi-product crop systems are presented in Section 4. The results are shown in relation to specific parameters of material applications, e.g. GHG substitution and market price per tonne of material. A sensitivity analysis, i.e. a Monte-Carlo analysis of the result is covered in Section 4, too. Section 5 finishes with discussion and conclusions concerning the performance of multi-product crops systems.

[1] This was the situation at the time of writing. In May 2004 Poland has joined the EU.

2 Approach

In the first part of this Section, various parameters that can influence GHG emission reduction and primary biomass fuel costs of multi-product crop systems considered in this study are discussed. The second part gives an overview of the calculation method used in this study. A detailed description of the background of input parameters (origin, calculation method and limitation) is given in Section 3. At the end of this section, the method used for a sensitivity analysis is discussed.

2.1 Multi-product crop systems

A multi-product crop production and conversion system basically consists of crop production, energy conversion and material production. Materials and energy out of the multi-product crop substitute reference materials and energy. Figure 2.1 gives an overview of such a system. Consequently, GHG emission reductions and costs of multi-product crop systems depend on the performance of the sub-systems that are influenced by many parameters. These are discussed in detail below.

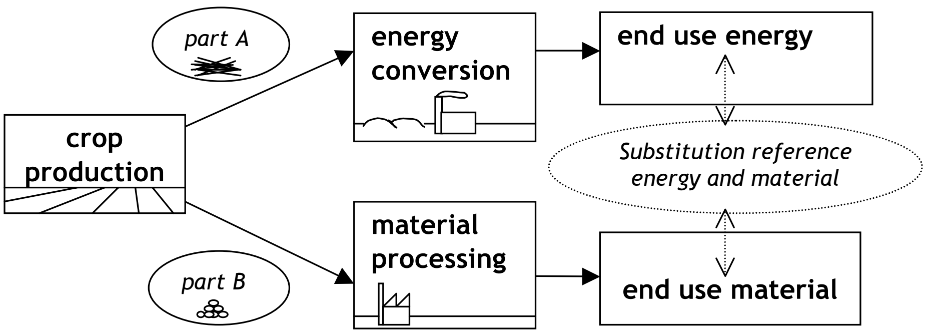


Figure 2.1: Schematic overview of a multi-product crop system

Crop production

The *crop yield per ha and year* and the *share of plant components* (e.g. wheat straw and grain) determine the amounts of biomass available for energy and material purposes. However, crop yields and shares of plant components depend on *agricultural practice* and *local conditions*, e.g. soil and climate. Also, *GHG emissions of crop production* depend strongly on agricultural practice. The *use of machines and materials* (like seeds, fertilisers, pesticides and buildings) leads to GHG emission directly and indirectly. GHG emissions are also caused by the *use of nitrogen fertiliser* in the form of N_2O emissions in the field. Finally, agricultural practice and local conditions also influence *the costs of crop production*. Important costs items in this context are *labour costs, machines costs, land costs* and *material costs*. Apart from that, crop production costs are reduced by *EU agricultural subsidies* in the Netherlands.

Material production and substitution

The *kind of material application* and the *kind of reference materials* substituted determine the possible benefits of using parts of a multi-product crop for material applications. *GHG emission reductions of materials use* are the GHG emissions that would result from the production of the reference material minus the GHG emissions of material production from the multi-product crop. *Market prices of crop components* determine the revenue from material components in the multi-product crop system. These market prices vary with time and can be subject to regional variations.

Production and substitution of energy carriers

Crops can be converted to different energy carriers that substitute the same amount of reference energy carriers thereby reducing GHG emissions. This reduction depends on *the specific GHG emissions of reference energy production*. Energy produced from multi-product crops can be sold for the market price of the respective energy carrier. However, in this chapter we calculate 'gross' bioenergy costs of multi-product crops, i.e. bioenergy costs without revenues from energy sales. Comparing these 'gross' bioenergy costs to common market prices, the competitiveness of bioenergy from multi-product crops can be judged.

2.2 Calculation method

The different multi-product crop systems are compared on the basis of two key indicators:

- Net GHG emission reduction per area of biomass production [Mg CO_{2eq}/(ha*yr)]
- Costs of primary biomass fuel [€/GJ_{LHV}]

These indicators are calculated according to formula 1 and 2. One ha of biomass production and the respective yields of energy and material components are the starting point for the calculations of these indicators. Input data are derived from literature including statistics, LCA analyses and scientific reports.

Crops are usually produced in order to maximise the amount of a certain component, i.e. grain of winter wheat, bark fibres of hemp and total biomass yield of poplar. This is studied in the 'base case' of this chapter. In addition, the multi-product crop systems are studied for some other crop production objectives. These are the production of hemp for seeds and the production of poplar for larger shares of white chips by longer rotation times. For wheat no variation of production method is considered, because wheat varieties with high grain yields have high total biomass yields, too (Lewandowski, 2001).

$$GHG_{red} = (- GHG_{crop} + GHG_{mat} + GHG_{fuel}) / A \quad (1)$$
$$C_{biof} = (C_{crop} - R_{mat}) / LHV \quad (2)$$

GHG_{red}: Net GHG emission reduction per area of biomass production [Mg $CO_{2\ eq}$/(ha*yr)]
GHG_{crop}: indirect and direct GHG emissions due to machine, material and fertiliser use incl. N_2O emissions [Mg CO_{2eq}]
GHG_{mat}: GHG emission reduction due to the substitution of a reference material by a multi-product material [Mg CO_{2eq}]
GHG_{fuel}: GHG emission reduction due to the substitution of solid fossil fuel by the primary biomass fuel [Mg CO_{2eq}]
A: area of biomass production biomass production [ha*yr]
C_{biof}: Costs of primary biomass fuel [€/GJ_{LHV}]
C_{crop}: Crop production costs due to machine, material and fertiliser use and land rent [€]
R_{mat}: revenues from sale of plant components for material use [€]
LHV: lower heating value of plant components for energy use [GJ_{LHV}]

Finally, a sensitivity analysis of the base case results is carried out, because the input data used in the analysis can vary, as shown by e.g. reference energy systems and historical data for market prices. As many different variables with relatively large uncertainties determine the results, a Monte-Carlo analysis is carried out for each crop and country. For a Monte-Carlo analysis a probability distribution for the value of each input variable is defined from which probability distributions of the results are generated. Moreover, the sensitivity of the results (i.e. the contribution to variance) to each input parameter is determined.

Several variables influence the results only at a certain percentage of energy use of the multi-product crop. For example, GHG emission reduction factors of material use don't influence the net annual GHG emission reduction per ha if the crop is used as a pure energy crop. Therefore, three utilisation modes of the multi-product crops are studied in the Monte-Carlo analysis. These modes are (1) 'material crop', i.e. the maximum share of the crop is used for material applications, (2) 'multi-product', i.e. only the main material component is used for material applications, and (3) 'energy crop', i.e. the whole crop is used for energy. To allow calculation of primary biomass fuel costs, it is assumed that in the 'material crop' mode at least 5% of the crop is used for energy purposes.

3 Input data

In this section, input parameters for the analysis of the multi-product crop systems are presented for crop production and material and energy uses. Average values of parameters are used for the calculation of base case results. Also ranges of the value of the parameters are given that are used in the Monte-Carlo analysis.

3.1 Yield, costs, energy demand and GHG emissions of crop production

Yields of crop components in the Netherlands and Poland for wheat, short rotation poplar and hemp are presented in Table 2.1. Values refer to average soil and climatic conditions and average agricultural practice of the respective country. Data on wheat is derived from agricultural statistics, while data on hemp originates from statistics and scientific literature. Poplar is only cultivated on a small scale in the Netherlands and is not cultivated in Poland. Therefore, poplar yields for the Netherlands are taken from feasibility studies. Yields of short rotation poplar in Poland were estimated from yields of short rotation willow in Poland in comparison to willow and poplar yields in the Netherlands.

Table 2.1: Biomass yields and distribution of plant parts of wheat, hemp and poplar

Parameter	**Netherlands**			**Poland**		
	Average	*Range*	*Reference*	*Average*	*Range*	*Reference*
Winter wheat						
Yield whole plant [Mg/(ha*yr)]	12.5	11.1-13.6	(PAV, 2000)	6.5	5.8-7.0	(CSO, 2003) [a]
Wheat straw [% whole plant]	34	32-36	(PAV, 2000)	47	44-48	(CSO, 2003) [b]
Hemp						
Yield stem [c] [Mg/(ha*yr)]	8.1	6-9	(PAV, 2000; Karus et al., 1996)	2.8	2.6-5.6	(FAO, 2004)[d]
Bark fibres [% stem] [e]	25	20-30	(Karus et al., 1996)	25	20-30	(Karus et al., 1996)
Yield seeds [Mg/(ha*yr)] [e]	0	0	(Karus et al., 1996)	0	0	(Karus et al., 1996)
Short rotation poplar						
Yield [Mg/(ha*yr)][f]	7.0	6-10	(Lewandowski, 2001; Biewinga and vd Bijl, 1996; Stolp et al., 1996)	3.6	3.1-5.1	(Kozlowski et al., 2001) [g]
White chips [% whole plant]	75	70-80	(Teeuwisse, 1999)[h]	75	70-80	(Teeuwisse, 1999)[h]

[a] 47% of straw is added to grain yield from statistics.
[b] Straw yields refer to the total of cereals in Poland, which consist to one third of wheat.
[c] Hemp yields consist of seeds and stems (i.e. straw). The stems can be divided into a woody core and the outer bark fibres.
[d] 75% of core is added to fibre yields from statistics. (Until begin of the 1990ies, hemp was cultivated on a much larger scale, i.e. more than 2000 ha compared to 70 ha 2002. At that time yields were higher and reached up to 5.6 Mg/(ha*yr) (FAO, 2004). For current experimental plots Kozlowski (2001) states yields of 10 Mg/(ha*yr).
[e] Distribution of plant components refers to Germany that is comparable to the Netherlands and Poland.
[f] Yields of short rotation poplar in literature differ considerably. While Biewinga and van der Bijl (1996) state actual yields of 7.0 Mg_{dm}/(ha*yr) and attainable yields of 8.5 Mg /(ha*yr) for the Netherlands, Stolp et al. (1996) assume an average yield of 10 Mg_{dm}/(ha*yr). For Germany, that has comparable climatic conditions average yields are 7-9 Mg_{dm} /(ha*yr), while ranges of 6-18 Mg_{dm} /(ha*yr) are possible (Lewandowski, 2001).
[g] Poplar is currently not cultivated in Poland. Therefore, poplar yields are estimated by means of short rotation willow yields, which is a comparable crop. Average yields of willow in Poland are about 4 Mg/(ha*yr)*yr. (Kozlowski et al., 2001). In the Netherlands, poplar yields are about 10% lower than willow yields (Biewinga and van der Bijl, 1996). Poplar yields in Poland are estimated from willow yields by the same percentage.
[h] Value applies to short rotation cottonwood, which is closely related to hybrid poplar.

Costs of crop production are divided into costs of labour, machines, materials and land. In general, crop production costs are calculated from rented means of production, i.e. land, machines, and labour. By this approach market prices of these production factors are taken into account. Hence, profits are implicitly considered. The crop production costs are reduced by EU agricultural subsidies in the Netherlands, while in Poland currently no subsidies are provided.

Data on wheat production costs in the Netherlands and Poland and data on hemp production costs in the Netherlands are from agricultural statistics. Because hemp is currently not cultivated on a significant scale in Poland, data on hemp production costs in Poland originates from experimental plots. For poplar, production data has been taken from scientific literature. For Poland production costs have been estimated based on information on short rotation willow, which has a comparable production method. As poplar is a perennial crop, yearly costs are calculated from present values.[2] For a detailed description of values and assumptions, see Table 2.2.

Ranges of biomass production costs have been estimated. Minimum costs represent a less intensive agricultural production system, i.e. a system without fertiliser and pesticide use. Moreover, costs of machines, labour and materials have been reduced by 10%. For the Netherlands, maximum costs of biomass production are estimated by a variation of 10% and the assumption that no agricultural subsidies are granted. For Poland, maximum costs of biomass production represent a more intensive system like in the Netherlands. Therefore, respective production costs of the Netherlands are used.

Table 2.2: Working hours and costs of biomass production per ha and year

	Wheat – NL[a]		**Wheat – PL**[b]		**Hemp – NL**[a]		**Hemp – PL**[c]		**Poplar – NL**[d]		**Poplar – PL**[e]	
	Hour	*€$_{2000}$*	*Hour*	*€$_{2000}$*	*Hour*	*€$_{2000}$*	*Hour*	*€$_{2000}$*	*Hour*	*€$_{2000}$*	*Hour*	*€$_{2000}$*
Labour	*11.2*	*258*	*32.6*	*94*	*12.8*	*294*	*N/a*	*31*	*4.1*	*100*	N/a	*22*
Establishment	5.4	124	N/a	N/a	5.4	124	N/a	12 [f]	0.8	29	2.7	5
Cultivation	2.6	60	N/a	N/a	0.9	21	N/a	N/a	0.2	4	N/a	N/a
Harvest	3.2	74	N/a	N/a	6.5	149	6.5	19	3.1	67	6.1 [g]	17 [g]
Machines	*9.8*	*793*	*14.0* [h]	*179*	*10.8*	*519*	*N/a*	*357*	*3.5*	*104*	*N/a*	*31*
Establishment	5.4	158	N/a	N/a	5.4	158	N/a	52 [f]	0.2	17	0.5	6
Cultivation	2.6	156	N/a	N/a	0.9	56	N/a	N/a	0.2	12	N/a	N/a
Harvest	1.8	479	N/a	N/a	4.5	305	4.5	305	3.1	75	N/a	25 [g]
Materials	-	*500*	-	*189*	-	*219*	-	*245*	-	*195*	-	*250*
Seeds	-	75	-	26	-	111	-	104	-	60	-	79
Fertiliser	-	132	-	78	-	95	-	132	-	12	-	14
Pest control	-	230	-	43	-	0	-	0	-	2	-	5
Other	-	63 [i]	-	42 [j]	-	13 [i]	-	9	-	121 [k]	-	111
Land [l]	-	*373*	-	*33*	-	*373*	-	*33*	-	*373*	-	*33*
Subsidies	-	*-415*	-	*0*	-	*-646*	-	*0*	-	*-488* [n]	-	*0*
Total	**-**	**1509**	**-**	**495**	**-**	**759**	**-**	**666**	**-**	**284**	**-**	**262**

[a] Cost data are from PAV (2000). However, only total costs for hired machines including labour are given. These have been split into distinct machine and labour costs by average costs of labour of 23.0 € per worked hour (CBS, 2002).
[b] Data are from Skarzynska et al. (2001a) and Skarzynska et al. (2001b). Average farm size for wheat in Poland is quite small, i.e. 3.6 ha, however, the trend in recent years has been that farm size increased and the proportion of labour costs

[2] $Cq_a = [q_a/(1+r)^a]*r/[1-(1+r)^{-a}]$ with Cq_a = yearly costs from cost item q_a, q_a = cost item in year a, r = interest rate

decreased. Machine costs include capital costs and are, thus, comparable to hired machines. Labour costs are calculated by assuming an hourly wage 2.87 €/hour for workers in the agricultural sector (ILO, 2001).
[c] Costs data of hemp production are from experimental plots (Kozlowski, 2001), because no other data were available. While all costs are lower or comparable to costs in the Netherlands, the costs of harvest are extremely high. In the cheapest variant harvest costs are 880 €/(ha*yr) (machines and labour) compared to 429 €/(ha*yr) in the Netherlands. However, it is assumed that these high costs are due to the utilisation of special harvesters on small experimental plots and that machines costs and labour hours comparable to the Netherlands are achievable in practice.
[d] If possible, cost costs are taken from PAV (2000). However, working hours, material costs and harvest costs had to be taken from PAGV (1994). Other costs are derived from Stolp et al. (1996).
[e] Poplar production costs in Poland are taken from short rotation willow production costs (Szczukowski et al., 2000a). (For comparison, Termeer (2002) calculates for the Netherlands willow production costs of 990 €/(ha*yr) and poplar production costs of 890 €/(ha*yr) based on Kozlowski (2001) and Coelman and Pijanowska (1996). Moreover, it is assumed that like in the Netherlands, poplar is fertilised every 4 years, while willow is fertilised every year.) As Szczukowski et al. (2000a) do not take into account stump clearing costs, stump clearing costs from the Netherlands are added to other costs.
[f] Establishment costs include cultivation costs
[g] Harvest costs include cultivation costs
[h] Machine hours do not include average use of horse power, i.e. 0.4 h/(ha*yr).
[i] Drying, taxes, insurance.
[j] Buildings, insurance, capital costs.
[k] Forestry tax and stump clearing at the end of plantation after 20 years.
[l] In 1998 the average rent for arable land was 338 ECU/(ha*yr), while between 1994 and 1998 land prices increased about 3 to 8% (Eurostat, 2000). Assuming a yearly increase of 5% a land rent of 373 €/(ha*yr) in the year 2000 results. Land costs in Poland are from Skarzynska (2001a).
[f] Subsidy for set-aside land, that is applicable for dedicated non-food production.

Energy demands of biomass production result from direct and indirect use of energy for machines and materials. Data on energy demands of crop production in the Netherlands are taken from a study on the environmental impacts of energy crops of Biewinga and van der Bijl (1996). Data on energy demands of crop production in Poland were not available. Therefore, data on production inputs (e.g. machine hours) were combined with specific energy demands of these inputs from scientific literature (e.g. gross energy demand of 1 kg of mineral fertiliser). If possible, these specific energy demands are also taken from the study of Biewinga and van der Bijl (1996) to generate comparable results for the Netherlands and Poland. Information on inputs of crop production in Poland is derived from agricultural statistics. However, information on inputs of hemp production is derived by means of a comparison of production costs in the Netherlands and Poland, assuming that the amount of money spent on a mean of production (e.g. fertiliser costs) is proportional to the energy inputs. Finally, poplar production inputs in Poland have been derived from a study on short rotation willow production.

GHG emissions of biomass production are derived by country-specific emission factors from energy demands; see Table 2.3. These emission factors refer to overall fuel mixes used in each country and are 75 kg CO_{2eq}/GJ for the Netherlands and 90 kg CO_{2eq}/GJ for Poland (IEA, 2002a; Frischknecht et al., 1994). In general, for machine use an emission factor of diesel oil was applied (75 kg CO_{2eq}/GJ). Furthermore, N_2O emissions on the field due to N fertilisation are included in the calculation of GHG emissions due to fertiliser use. Data on these emissions were taken from literature; see Table 2.3.

Also for GHG emissions of crop production ranges have been estimated. The lowest value of GHG emissions refers to agricultural production without fertiliser and pesticide use. For the Netherlands, maximum GHG emissions exceed the average emissions by only 10% because average agricultural production is already quite intensive. However, in the case of poplar maximum GHG emissions exceed the average emissions by 30%, because production of poplar is currently quite extensive compared to the production of wheat and hemp. In Poland, biomass production is relative extensive. Therefore, respective average emissions of the intensive biomass production in the Netherlands are used as higher range of GHG emissions in Poland.

Table 2.3: Energy demand and GHG emissions per ha and year of biomass production

	Wheat – NL [a]		**Wheat – PL** [b]		**Hemp – NL** [a]		**Hemp – PL** [c]		**Poplar – NL** [a]		**Poplar – PL** [d]	
	GJ	*Mg CO_{2eq}*	*GJ*	*Mg CO_{2eq}*	*GJ*	*Mg CO_{2eq}*	*GJ*	*Mg CO_{2eq}*	*GJ*	*Mg CO_{2eq}*	*GJ*	*Mg CO_{2eq}*
Machines	8.7	0.74	3.5	0.26	8.7	0.74	6.0	0.45	6.0	0.51	5.2	0.47
Seeds	0.6	0.05	0.6	0.05	1.6	0.12	1.6	0.12	0.3	0.02	0.4	0.03
Fertiliser [e]	7.9 [f]	2.29	4.6	1.34	3.1 [f]	0.83	4.3	0.99	2.5 [g]	0.59	1.1	0.20
Pest control	0.3	0.02	0.2	0.02	-	-	-	-	0.0	0.00	0.1	0.00
Other	-	-	-	-	-	-	-	-	7.2 [h]	0.54	6.4 [i]	0.57
Total	**17.5**	**3.10**	**8.9**	**1.67**	**13.4**	**1.69**	**11.9**	**1.56**	**16.0**	**1.66**	**13.2**	**1.37**
Range (min)		0.79		0.31		0.77		0.57		1.07		1.17
(max)		3.40		3.10		2.20		1.69		1.82		1.66

[a] Data on energy use from Biewinga and van der Bijl (1996).
[b] For machines use, only general operations hours are given in Skarzynska et al. (2001b). Therefore, the energy use is estimated by means of average nominal motor size of Polish agricultural machines, i.e. 32 kW (CSO, 2003) and an according energy use, i.e. 5.8 kg diesel per hour of normal work (Kaltschmitt and Reinhardt, 1997). Fertiliser utilisation is taken from Skarzynska et al. (2001b) and combined with gross energy requirements from Biewinga and van der Bijl (1996). For seed production, energy demands from the Netherlands are assumed. Moreover, it is assumed that the costs for pesticides are comparable in the Netherlands and Poland and therefore, the energy requirements of pesticides are estimated by the costs.
[c] Energy demands of machine use and fertiliser application could only be estimated by means of cost differences from the respective values in the Netherlands (see Table 2.2). For seed production, the same energy demands as in the Netherlands has been assumed.
[d] Data refers to short rotation willow production (Szczukowski et al., 2000b) with the assumption of fertilisation every 4 years, see footnote Table 2.2.
[e] GHG emissions of fertilisation include N_2O emissions from the field (Biewinga and van der Bijl, 1996). For Poland, it is assumed that N_2O emissions are proportional to the amount of N fertiliser applied in comparison to the amount of N fertiliser applied in the Netherlands.
[f] The energy demand is based on fertiliser application due to average agricultural practice as described in PAV (2000).
[g] The energy demand is based on fertiliser application due to average agricultural practice as described in PAGV (1994).
[h] Chipping and drying.
[i] The energy demand of chipping refers to short rotation willow production in Poland (Termeer, 2002). Because no drying is assumed in the Polish case, energy use from drying in the Netherlands is added here.

3.2 Prices and GHG substitution factors of biomass material

Market prices of plant components for material use and their historical developments in the last ten years are presented in Figure 2.2. Prices of wheat components, i.e. straw and grain, originate from national statistics. Prices of wood chips are import prices of FAO trade statistics (FAO, 2004). However, for hemp no historical development of prices could be found, because cultivation of hemp in the Netherlands is a recent development and in Poland, local markets are small and not very developed. Therefore, price ranges of the German market that imports among others hemp from the Netherlands and Poland are used. In Figure 2.2, these price ranges are depicted as vertical lines. For the calculation of the base case results, current average prices are used, see Table 2.4.

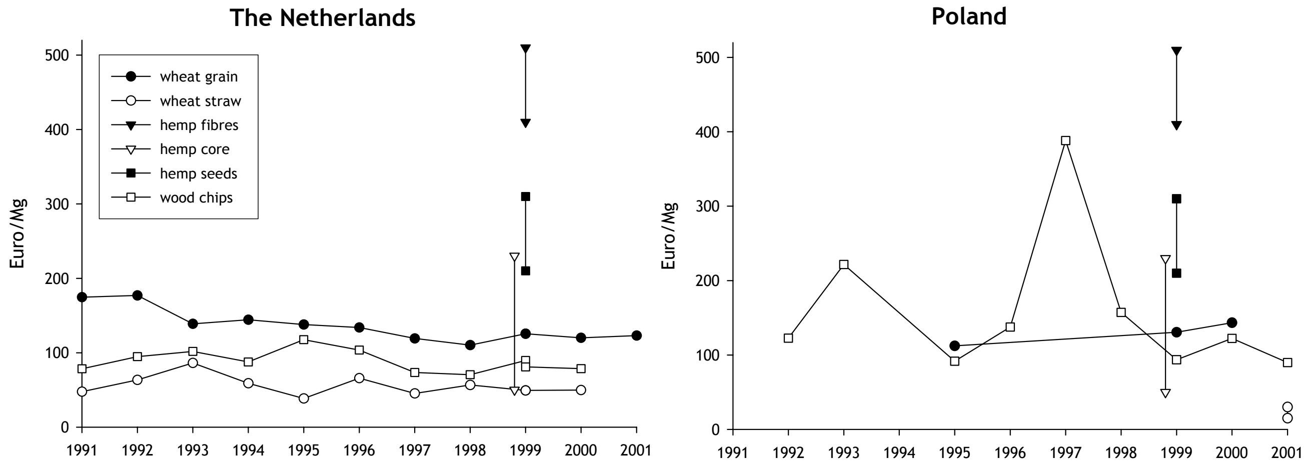


Figure 2.2: Historical development of prices of plant components for material applications in the Netherlands and Poland in Euros of respective years. For hemp components current price ranges are indicated as vertical lines (data from PAV, 2000; CSO, 2003; FAO, 2004; Karus et al., 2000; EC/BREC, 2003).

GHG substitution factors of material application of multi-product crops are presented in Table 2.5. These substitution factors have been determined in several steps. First, it has been defined for which material application the plant component is typically used. Second, a reference material, i.e. a material that is substituted by the biomass material is determined. Finally, the GHG emission reduction per Mg of multi-product crop component is estimated. This GHG emission reduction are the emissions caused by the production of the reference material minus the emission caused by the production of the multi-product crop material. It is taken into account that different amounts of materials are substituted. For example, if used in composites of comparable properties 0.7 kg of hemp fibres replaces 1 kg of glass fibres (Wötzel et al., 1999).

Data on GHG emission reduction by material substitution are taken from LCA studies and other scientific literature. These data refer to average European production processes. Thus, no difference between Poland and the Netherlands is made. Also for GHG emission reduction of material substitution, ranges have been estimated. The minimum reduction is zero, assuming that no material is actually substituted. To account for variations in production processes, we estimate the maximum reduction to be 20% higher than the base value.

Table 2.4: Average material prices of plant components in the Netherlands and Poland in €/Mg material

	Wheat grain	**Wheat straw**	**Hemp bark fibres**[a]	**Hemp core**[b]	**Hemp seeds**[b]	**Poplar chips**[d]
Netherlands	123 [e]	50 [e]	460	60	256	79
Poland	143 [f]	15 [g]	460	60	256	90

[a] Bark fibre prices from Karus et al. (2000) are diminished by average decortication costs of 90 €/Mg. Decortication costs in Germany range from 54 to 117 €/Mg depending on the quality of the fibre gained (Karus et al., 1996).
[b] For hemp core, a price from the lower range is used because the market for high value products is very limited (Karus et al., 2000).
[c] (Karus et al., 2000).
[d] Prices are derived from import prices of wood chips (FAO, 2004). These prices are stated for m^3 and are converted to prices per Mg by an average density of wood of different species of 0.5 Mg/m^3 (Haygreen and Bowyer, 1996).
[e] (PAV, 2000).
[f] (CSO, 2003).
[g] (EC/BREC, 2003).

Table 2.5: GHG emission reduction by material substitution in Mg CO_{2eq} per Mg of multi-product crop component

Biomass	**Used for**	**Substituted material**	**Substitution of GHG**	**Range**
Wheat - grain	Food/fodder	Soya	0.38 [a]	0-0.46
Wheat - straw	Animal litter	Mineral litter[b]	0.17 [c]	0-0.20
Hemp – bark fibre	Reinforced plastic	Glass fibre	0.8 [d]	0-0.96
Hemp – core	Animal litter	Mineral litter[b]	0.17 [c]	0-0.20
Hemp - seeds	Fodder + cosmetics	Soya meal + vegetable oil[e]	0.7 [c]	0-0.84
Poplar - chips	Pulp	Pulpwood	0.04 [f]	0-0.05

[a] This value refers to soya production in U.S. (UBA, 2003). Substitution accords to protein contents, i.e. 13% in soyabean and 12.2 % in wheat.
[b] Bentonit.
[c] (Karus et al., 1996).
[d] Values are taken from Wötzel, et al. (1999). These are reduced by GHG emissions related to the energy demand of fibre processing (decortication), i.e. 2.1 GJ/Mg_{fibre} (Kok, 2001).
[e] Oil of *Oenothere biennis* (evening prime rose).
[f] Pulp and fibre wood mix EU (UBA, 2003).

3.3 GHG substitution factors of biomass energy

GHG substitution factors of energy application of multi-product crops are presented in Table 2.6. The underlying assumption is that crop components are used for heat or power generation as solid fuel. Moreover, crop components replace fossil fuel that otherwise would have been used for heat and power generation. The GHG substitution factor of heat and power generation refers to the respective fossil fuel mixes.[3] These substitution factors are 76 kg CO_{2eq}/ GJ_{LHV} in the Netherlands and 94 kg CO_{2eq}/GJ_{LHV} in Poland (IEA, 2002a; Frischknecht et al., 1994). In the Monte-Carlo analysis a variation of these factors from 0 to 98 kg CO_{2eq}/GJ_{LHV} is taken into account. This range corresponds to the emissions of different energy carriers that may be substituted, i.e. renewable or nuclear energy and coal.

Table 2.6: Lower heating values of multi-product crop components in GJ/Mg_{dm} and GHG substitution factors for energy uses in Mg CO_{2eq}/Mg_{dm} (IEA, 2002a; Frischknecht et al., 1994; ECN, 2003)

	Wheat grain	**Wheat straw**	**Hemp bark fibres**	**Hemp core**	**Hemp seeds**	**Poplar chips[a]**
LHV	17.0	15.5	16.4[b]	16.4[b]	16.8[c]	17.7
Range LHV	15-18	15-18	15-18	15-18	15-18	15-18
GHG substitution factor NL[d]	1.29	1.18	1.25	1.25	1.28	1.35
GHG substitution factor PL[e]	1.60	1.46	1.54	1.54	1.58	1.66

[a] Chips of whole tree.
[b] Heating value of hemp straw comprising core and bark fibres.
[c] Heating value of hemp whole plant.
[d] Assuming a carbon factor of 76 $kgCO_{2eq.}/GJ_{LHV}$, see Section 3.3.
[e] Assuming a carbon factor of 94 $kgCO_{2eq.}/GJ_{LHV}$, see Section 3.3.

4 Performance of multi-product systems

4.1 Net annual GHG emission reductions per ha

Figure 2.3 shows the net annual GHG emission reductions per ha and year in relation to the specific GHG substitution factors of material application. Note that main components of multi-product crops for material application are wheat grain, hemp bark fibre or poplar white chips. 'Base' results calculated from average input values are marked by dots. The ranges of the lines in Figure 2.3 corresponds to the estimated ranges of GHG substitution factors, compare Table 2.5.

The impact of the specific GHG emission factor of material substitution on the net annual GHG emission reduction per ha and year increases with the percentage of biomass used for material production. Therefore, hemp (with only about 25% bark fibre content) is the least influenced by changes in the GHG credits of material substitution. Hemp in the Netherlands has the highest net GHG emission reduction per ha and year in the 'base' results. This can be explained by the high amount of material that is used for energy purposes and the relative low GHG emissions during hemp cultivation. However, hemp in

[3] Therefore, a displacement factor of 1 is implicitly assumed in this study, which applies to co-firing of biomass in fossil power plants as is currently quite common.

Poland achieves a smaller net reduction of GHG emissions per ha and year, because crop yields are lower. Wheat contributes modestly and poplar only slightly to GHG emission reduction if used as a multi-product crop.

Comparing the situation in Poland and the Netherlands, net reduction in GHG emissions are typically lower in Poland. Thus, lower crop yields in Poland are not offset by lower GHG emissions during crop production. Note that for specific GHG emission reduction of material substitution, no differences between the two countries have been considered.

Net annual GHG emission reductions per ha as function of the percentage of multi-product crop used for energy purposes are presented in Figure 2.4. This percentage is varied from zero (material utilisation only) to hundred (complete use for energy). The 'base' figures are marked by dots and assume that the main component of the multi-product crop is used for material, while other components of the plant are used for energy.[4] For poplar, it is assumed that only the main component (i.e. white chips) can be used for material applications. Therefore, no results for the material utilisation of other components (bark, small twigs, etc.) are presented. For wheat and hemp, the utilisation of other plant components as material than the main component is considered, too. For these components other specific GHG emission reduction factors apply, see Table 2.5. As a result, a bend can be observed in the curves of hemp and wheat in Figure 2.4.

With regard to net annual GHG emission reductions per ha, a hundred percent use of the crop for energy purposes is more advantageous than using part of the crops for material applications. However, using the main component of multi-product crops for energy is less advantageous with regard to net GHG emission reduction than using the other components for energy. These results are related to the differences between the specific GHG substitution factors of different material applications and energy purposes, compare Table 2.5 and 2.6.

If the whole crop is used for energy purposes, wheat has the highest net GHG emission reduction per ha and year (12 Mg CO_{2eq}/(ha*yr)). However, poplar and hemp can reduce net GHG emissions per ha and year of biomass production, too. Typically, net GHG emission reductions per ha and year are lower in Poland. This is mainly due to the much lower crop yields.

[4] The 'base' figures have been used also for calculation of net GHG emissions presented in figure 3.

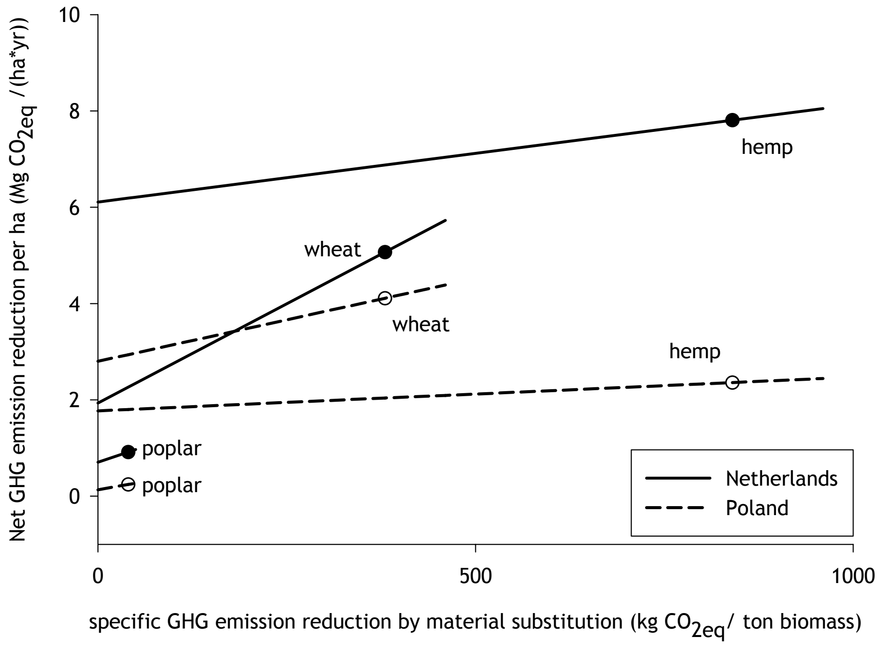


Figure 2.3: Net GHG emission reduction per ha and year (positive values indicate emission reduction) versus specific GHG emission reduction of material use

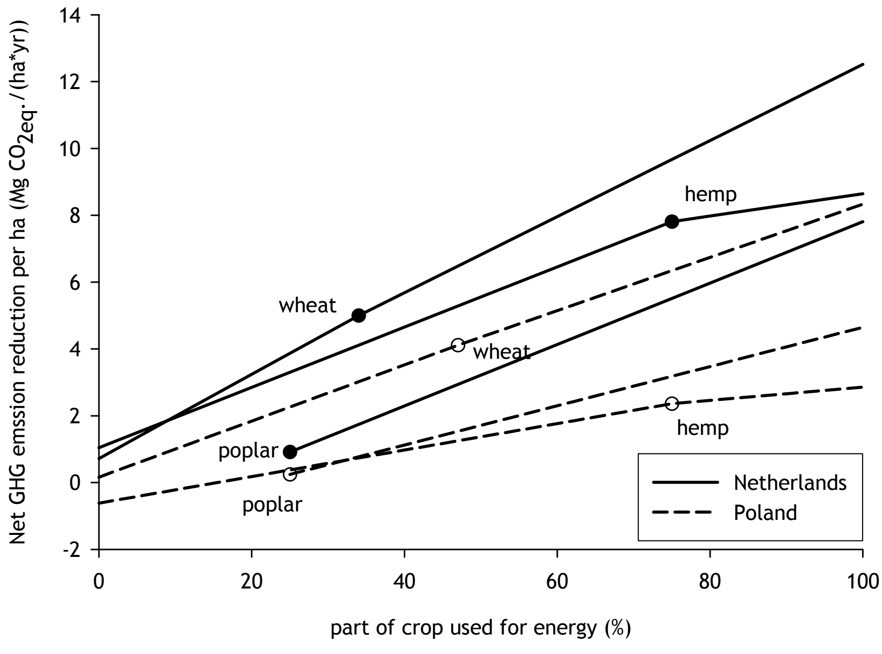


Figure 2.4: Net GHG emission reduction per ha and year (positive values indicate emission reduction) versus the percentage of multi-product crop that is used for energy

4.2 Primary biomass fuel costs

In Figure 2.5, primary biomass fuel costs in relation to the market prices of the main multi-product crop component are presented. 'Base' assumptions, i.e. current average prices are marked by dots. Ranges that are covered correspond to price ranges in the last ten years, see Figure 2.2.

Primary biomass fuel costs appear to be very sensitive to the price of materials, especially in the case of wheat and poplar where large amounts of the crop are used for material applications. Poplar and hemp in the Netherlands have the lowest primary biomass fuel costs, if used as multi-product crop. This can be explained by the high market prices of hemp bark fibre and the low cultivation costs of short rotation poplar.

Including EU agricultural subsidies in the Netherlands in the calculation, most primary biomass fuel costs are higher in Poland than in the Netherlands. Thus, lower crop production costs in Poland cannot compensate for the lower crop yields. However, for wheat this is vice versa and primary biomass fuel costs are lower in Poland than in the Netherlands.

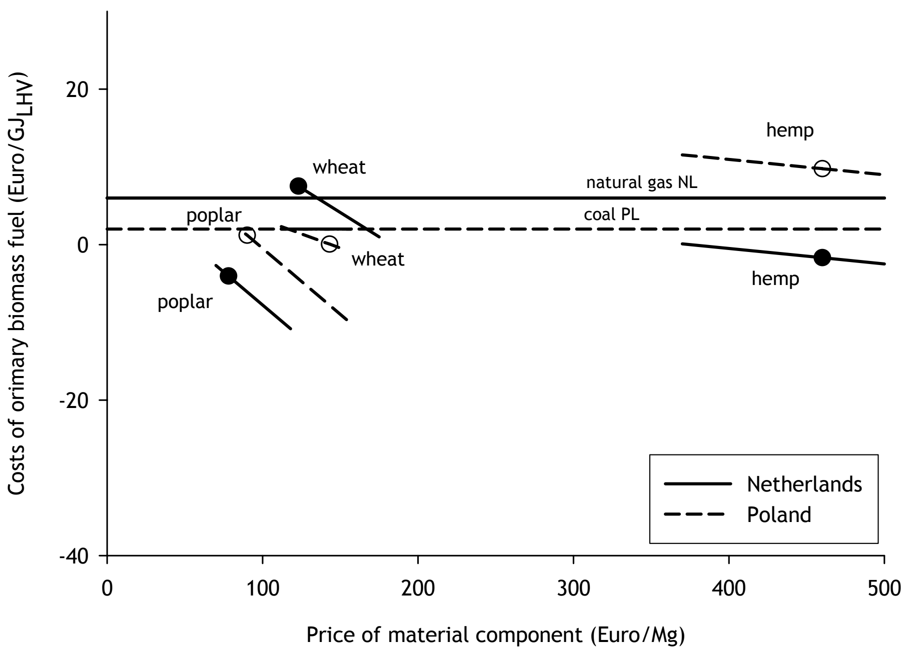


Figure 2.5: Primary Biomass fuel costs versus material prices of main components of multi-product crops

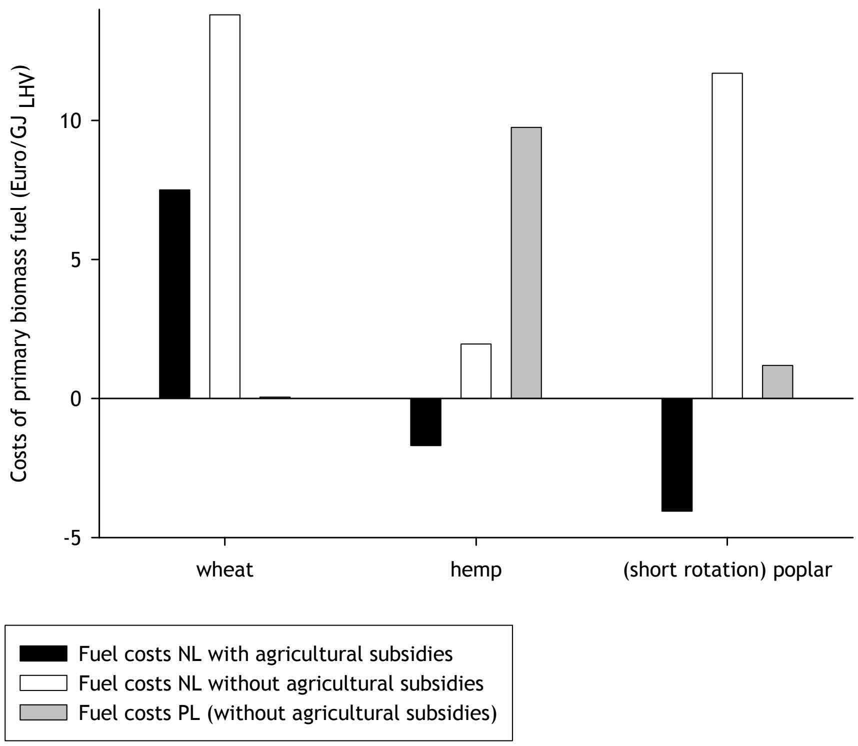


Figure 2.6: Comparison of primary biomass fuel costs in the Netherlands with and without agricultural subsidies to biomass fuel costs in Poland

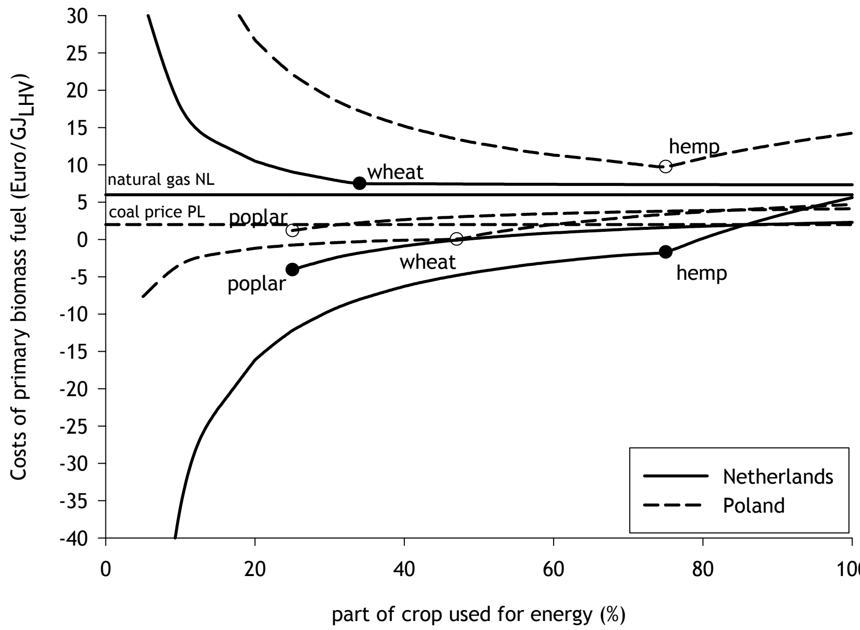


Figure 2.7: Primary biomass fuel costs versus the percentage of multi-product crop that is used for energy

Most primary biomass fuel costs of multi-product crops can compete with the costs of fossil fuels, as shown in Figure 2.5. Only primary biomass fuel costs of wheat straw in the Netherlands and hemp straw in Poland are higher than respective prices of natural gas and coal.

Primary biomass fuel costs in the Netherlands would be much higher without EU agricultural subsidies, exceeding the figure for Poland in the case of wheat and poplar, see Figure 2.6. Especially for poplar, primary biomass fuel costs in the Netherlands are much higher without agricultural subsidies. Only the primary biomass fuel costs of hemp are still lower in the Netherlands than in Poland due to the relatively high production costs and low yields of hemp in Poland.

Figure 2.7 presents primary biomass fuel costs for different percentages of the multi-product crops used for energy purposes. Dots mark the 'base' assumptions on the percentage of a crop used for energy (see also the discussion of Figure 2.4). It is remarkable that the curves of the different crops in the two countries show qualitatively very different behaviours. For wheat in the Netherlands and hemp in Poland, primary biomass fuel costs are lowest if only agricultural residues are used for energy purposes. Main components, i.e. grain and bark fibre, are then used for material applications. However, in the case of wheat the utilisation of grain for energy instead of material (i.e. food) would increase primary biomass fuel costs only slightly. Primary biomass fuel costs of short rotation poplar in the Netherlands and Poland, of hemp in the Netherlands and of wheat in Poland can be decreased if larger parts of the crops are used for material applications.[5] If hemp bark fibres are used for energy instead of material (in the curves at the right hand side of the dots), primary biomass fuel costs increase strongly.

4.3 Variation of crop production objective

In case hemp is grown for seed production and poplar is produced in longer rotation of about 20 years, total yield and shares of plant components change, see Table 2.7. The influence of such a changed crop production scheme on the net GHG emission reductions per ha and year and the primary biomass fuel costs is presented in Table 2.8.

Hemp for seed production has less net annual GHG emissions reduction per ha than hemp for bark fibre production. This is mainly because crop yields are lower. In addition the

[5] Note that for poplar material use is limited to 75% of the plant, see discussion earlier.

specific GHG emission reduction of material substitution is lower for seeds than for fibre. If poplar is cultivated in longer rotations net GHG emission reductions are slightly lower compared to short rotation poplar. This is due to the fact that in this variant less poplar wood is used for energy, while energy use is advantageous compared to material use with regard to GHG emission reduction.

Table 2.7: Biomass yields and distribution of plant parts of hemp for seed production and long rotation poplar

Parameter	Netherlands			Poland		
	Average	*Range*	*Reference*	*Average*	*Range*	*Reference*
Hemp for seed						
Yield stem [Mg/(ha*yr)]	2.7	2.5-3	(Karus et al., 1996)	1.1	0.7-1.4	(CSO, 2003) [a]
Bark fibres [% stem] [b]	10	10	(Karus et al., 1996)	10	10	(CSO, 2003)
Yield seeds [Mg/(ha*yr)]	1.2	0.8-2	(Lewandowski, 2001; Karus, et al., 1996)	0.5	0.3-1.0	(FAO, 2004) [c]
Poplar long rotation (20 year)						
Variation yield [Mg/(ha*yr)]	6.7 [d]	5.3-7.2 [e]	(Jansen, et al., 1996)	3.5	3.5-4.9	(Kozlowski et al., 2001; Jansen, et al., 1996) [f]
Variation white chips[z] [% whole plant]	95	94-96	(Jansen, et al., 1996)	84	81-86	(Jansen, et al., 1996)

[a] Stem yields are estimated from seed yields by assuming the same proportions between seeds and stems as for the Netherlands.
[b] Distribution of plant components refers to Germany that is comparable to the Netherlands and Poland.
[c] Also hempseeds were produced on a larger scale in Poland earlier in the last century. Seed yields then reached up to 1 Mg/(ha*yr) (FAO, 2004).
[d] If poplar is produced in longer rotations, yields diminish a.o. because less trees can be planted per ha and year. Yields refer to average growth classes with 4 m of spacing.
[e] Here, the amount of white chips is the amount of work wood.
[f] No yield tables for Poland were available. Therefore, it is assumed that the yields of longer rotations compared to the yields of shorter rotations decrease with the same proportion than in the Netherlands.

Table 2.8: Net GHG emission reduction per ha and year and primary biomass fuel costs of hemp for seeds and long rotation poplar compared to the base case results, i.e. hemp for fibre and short rotation poplar

Crop production objective	Net GHG emission reduction (Mg CO_2eq /(ha*yr))		Primary biomass fuel costs (€/GJ_{LHV})	
	NL	*PL*	*NL*	*PL*
Hemp for fibre (base case)	7.8	2.4	-1.7	9.7
Hemp for seeds (variation)	2.6	-0.5	10.0	29.2
Short rotation poplar (base case)	0.9	0.2	-4.1	1.2
Long rotation poplar (variation)	-0.9	-0.3	-36.9	-0.3

Assuming a multi-product crop system approach, primary biomass fuel costs of long rotation poplar are much lower than those of short rotation poplar in the Netherlands (see Table 2.8). This is due to the high amount of white chips produced, which can be sold for a high market price. In Poland, the amount of white chips increases less with longer rotation times. Therefore, the reduction of primary biomass fuel costs by longer rotations of poplar is small. In the case of seed hemp, primary biomass fuel costs are higher than in the case of fibre hemp. This is due to lower crop yield and lower market price of hemp seeds compared to hemp fibres.

4.4 Monte-Carlo analysis

Probability distributions of the value of input parameters employed in this Monte-Carlo analysis are based on the data presented in Section 3. In general, triangular distributions are assumed. Average data (used in the preceding base calculations) are used as most likely value, while ranges are used to define the minimum and maximum value of the distribution.[6] The only probability distributions of input variables that are not triangular are those of material market prices of plant components. These probability distributions are fitted to the historical data of the last 10 years.

The sensitivity of the performance of multi-product crop system to different input parameters has been analysed as the contribution to variance of the result of the Monte-Carlo analysis. With regard to net annual GHG emission reductions per ha, the GHG emission reduction factor of energy use has the strongest influence on the results (up to 96% in the case of wheat production in the Netherlands), depending on the percentage of crops that is used for energy purposes, see Table 2.9. The second most important input parameters are the GHG emissions during crop production (up to 70% in the case of wheat production in Poland). The third most important factor is the specific GHG emission reduction of material substitution of the main component (up to 65% in the case of wheat production in the Netherlands). Net annual GHG emission reductions per ha are significantly influenced by crop yields, too (up to 25% in the case of hemp production in Poland). In cases where the whole crop is used for materials, the specific GHG emission reduction factor of material use of secondary crop components becomes less important.

From an analysis of the sensitivity of the primary biomass fuel costs to different input parameters (see Table 2.10), it is concluded that crop production costs have the largest influence (up to 98% contribution to variance). The influence increases with the percentage of the crop that is used for energy purposes. For poplar and wheat, the market price of the main material component, i.e. grain and white chips, is an important factor too (up to 85% in the case of poplar in Poland). However, for hemp the market price of the main material component, i.e. bark fibre, is less important due to the fact that the amount of bark fibre (about 25% of whole plant) is relatively small. Therefore, in this case the market price of

[6] If the triangular distribution is not symmetric, the mean value of the distribution differs from the most likely value. Consequently, the mean results of the Monte-Carlo analysis differ from 'base' results.

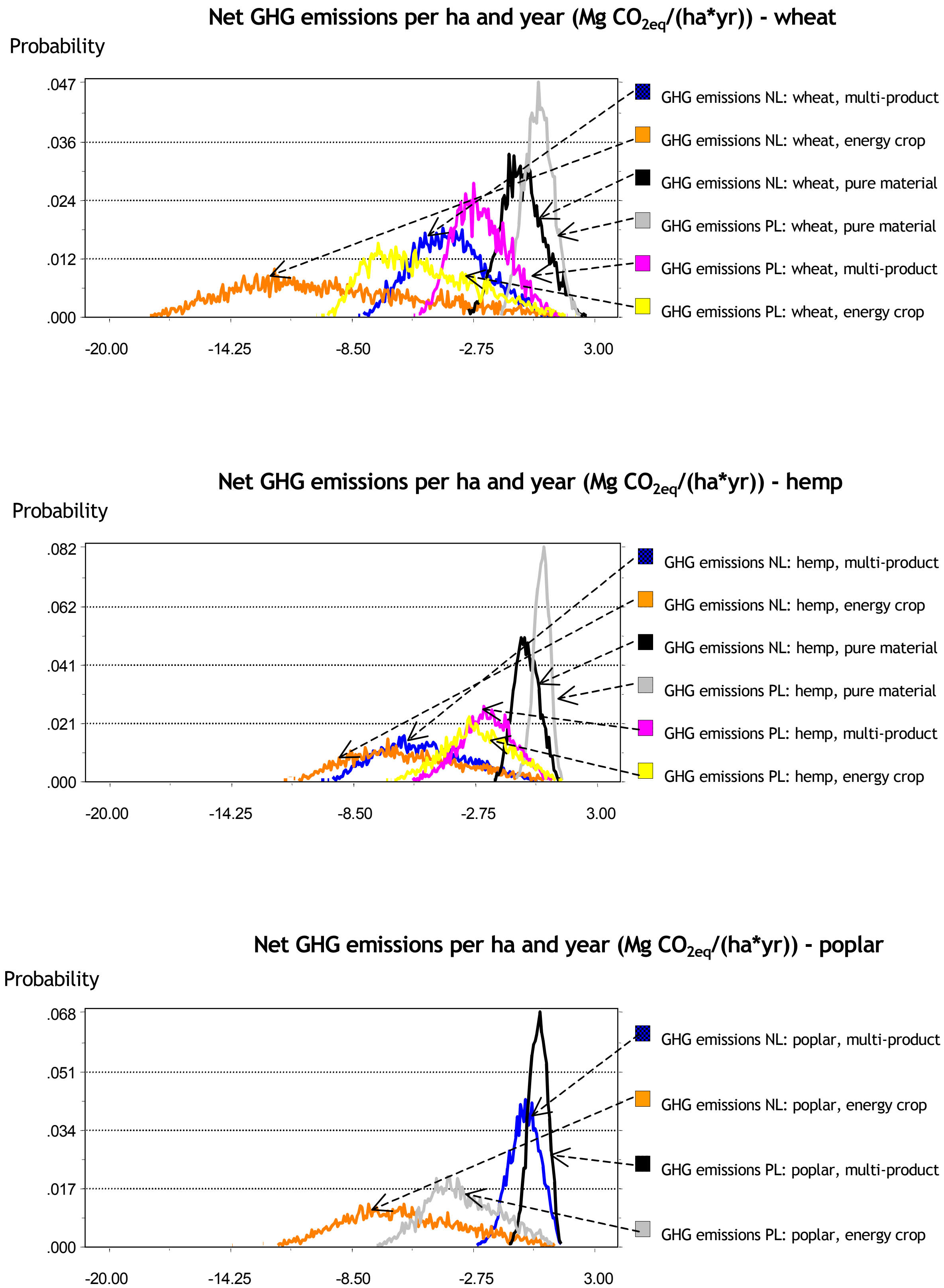
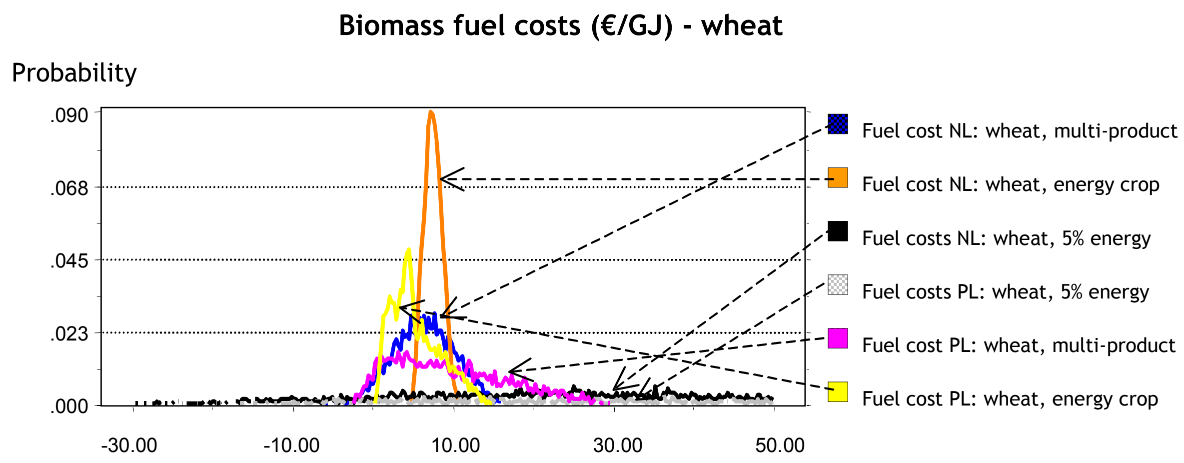
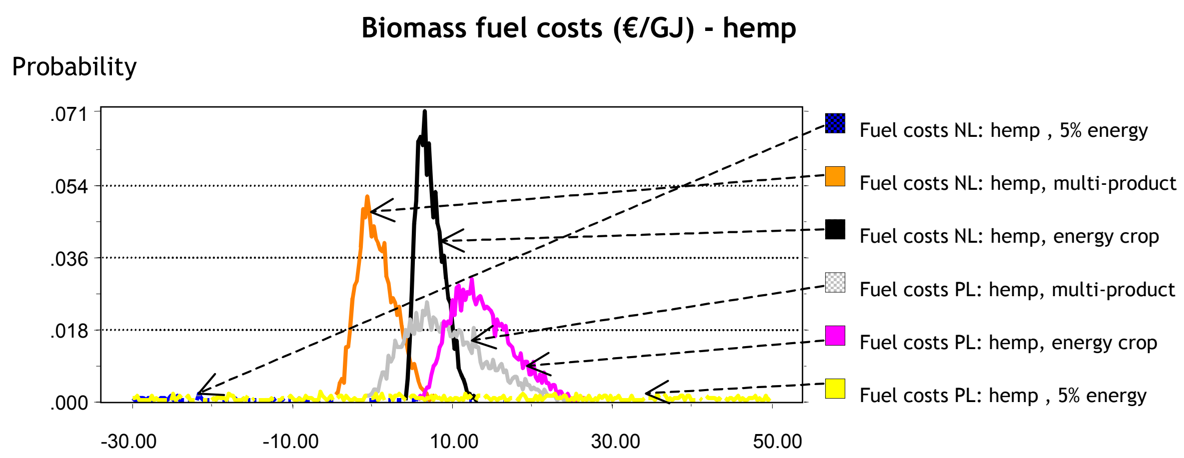


Figure 2.8: Probability distribution (as result of a Monte-Carlo analysis) of net GHG emission reduction per ha and year





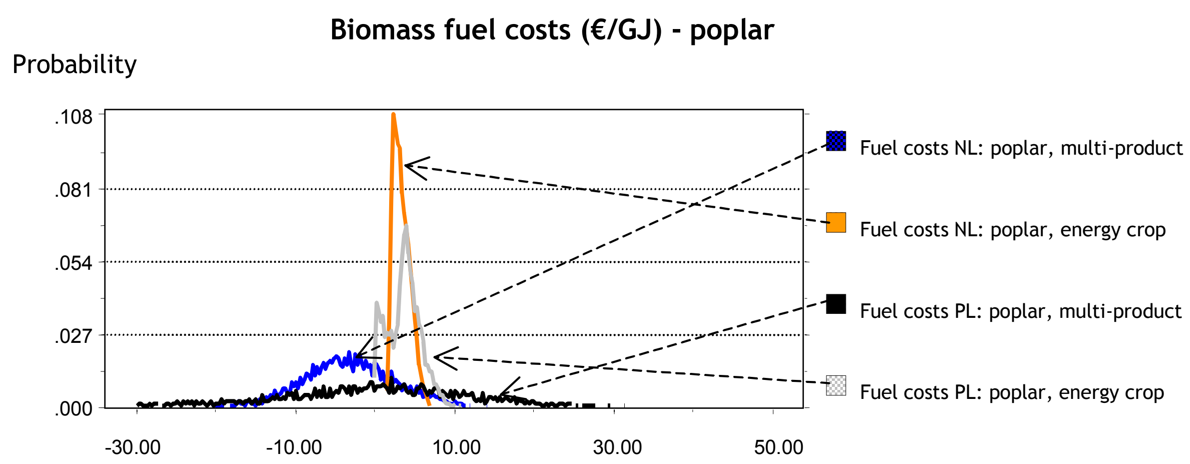


Figure 2.9: Probability distribution (as result of a Monte-Carlo analysis) of primary biomass fuel costs

the secondary material component becomes very important if only 5% of the plant is used for energy. Other factors that influence the primary biomass fuel costs strongly are the crop yield and the share of the material component. Only for wheat, the latter factor is not important because the share of straw varies only little (from 32% to 36%).

Table 2.9: Sensitivity of the net GHG emission reduction per ha and year to the input parameters as percent contribution to variance of the results from the Monte-Carlo analysis

	Wheat NL			**Wheat PL**			**Hemp NL**			**Hemp PL**			**Poplar NL**			**Poplar PL**		
Energy use of crop	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*
Crop yield	1	2	1	1	1	1	7	7	5	19	25	20	N/a	7	9	N/a	10	12
Share of main component in crop	0	0	01	0	0	0	0	2	0	1	0	0	N/a	17	0	N/a	19	0
GHG emissions crop production	30	10	21	70	20	6	2	25	1	44	4	2	N/a	5	0	N/a	5	0
GHG substitution energy use	0	66	96	0	70	92	87	0	93	0	68	77	N/a	69	89	N/a	66	88
GHG substitution material use main comp.	65	21	0	25	8	0	3	49	0	26	2	0	N/a	1	0	N/a	1	0
GHG substitution material use other comp.	3	0	0	4	0	0	0	17	0	10	0	0	N/a	N/a	N/a	N/a	N/a	N/a

m: material use (0% energy use); mp: multi-product use; e: energy use (100% energy use)

Table 2.10: Sensitivity of the primary biomass fuel costs to the input parameters as percent contribution to variance of the results from the Monte-Carlo analysis

	Wheat NL			**Wheat PL**			**Hemp NL**			**Hemp PL**			**Poplar NL**			**Poplar PL**		
Energy use of crop	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*	*m*	*mp*	*e*
Crop yield	6	6	7	1	1	1	7	11	13	33	10	40	N/a	8	2	N/a	5	3
Share of main component in crop	0	0	0	0	0	0	3	10	0	1	0	0	N/a	9	0	N/a	1	1
Crop production costs	62	64	90	96	97	98	38	75	85	48	59	59	N/a	62	87	N/a	30	10
Market price main crop component	29	30	0	2	2	1	2	3	0	0	1	0	N/a	20	0	N/a	63	85
Market price other crop component	2	0	0	0	0	1	50	0	0	17	0	0	N/a	N/a	N/a	N/a	N/a	N/a

m: material use (5% energy use); mp: multi-product use; e: energy use (100% energy use)

The results of the Monte-Carlo analysis show possible ranges and most likely values of net annual GHG emission reductions per ha and primary biomass fuel costs, see Figure 2.8 and 2.9. Possible ranges of net annual GHG emission reductions per ha are the smallest for pure material use and the largest for use as energy crops. In absolute terms, the values may range from large net GHG emission reductions per ha and year (up to 19 kg CO_{2eq}/(ha*yr)) to small net GHG emissions per ha and year (about 2 kg CO_{2eq}/(ha*yr)).

The probability distributions of primary biomass fuel costs have very different shapes. In the case of multi-product and energy crop use, the curves have high and narrow extremes. Thus, primary biomass fuel costs in these cases are hardly influenced by uncertainties. On the other hand, if nearly the whole crop is used for material (i.e. 5% energy use), possible primary biomass fuel costs cover a very broad range. Therefore, in these cases actual biomass fuel costs are difficult to predict. Even though the ranges for primary biomass fuel costs of multi- product crop use are comparatively small, they are quite large in absolute terms, e.g. for hemp they range from –10 to 10 €/GJ.

5 Discussion

In this chapter, the performance of multi-product crop systems with regard to net annual GHG emission reductions per ha and primary biomass fuel costs has been analysed. A case study of multi-product crop systems in Poland and the Netherlands has been carried out to consider different intensities of agricultural systems. While in the Netherlands agriculture is quite intensive with high yields and inputs, Polish agriculture is more extensive. However, within Poland agricultural practices can differ remarkably and so can the related costs and energy uses.

Moreover, the low crop production costs in Poland are mainly due to lower costs of labour and land. As Poland is an EU-member from May 2004 on, it is to be expected that these costs will increase in the future. On the other hand, biomass production costs in the Netherlands are favoured by EU agricultural subsidies that aren't granted in Poland. Plans are to align these subsidy schemes for the new Eastern European member of the European community. However, to forecast the amount and time scheme of these changes due to the Polish EU membership is beyond the scope of this study.

No interaction between the degree of implementation of multi-product crop systems and the market has been considered. However, if multi-product crops would be implemented on a large scale, macro-economic effects emerge. Material markets are subject to change due to price elasticity. Therefore, prices of plant components and also materials that are likely to be substituted by multi-product crop materials (and the related GHG emissions) can change. Moreover, costs of land and the quality of land available for multi-product crop cultivation would change due to increased competition for land for different purposes. To gain insight into the feasibility of large-scale multi-product crop systems, it is desirable to include these aspects into further research.

With regard to material use of multi-product crops, only direct and indirect GHG emissions due to the production of reference materials have been considered. However, some multi-product crop materials replace other plant materials, e.g. wheat grains replace soya and poplar chips substitute wood from conventional forestry. In these cases, no credits for land use have been considered, although it can be assumed that the land occupied for the production of these reference materials could otherwise be used for the production of other biomass materials. These other biomass materials, subsequently could contribute to GHG emission reduction. Considering these subsequent GHG emission reductions by land use, the specific GHG emission reduction of multi-product crop materials could be larger than considered in this study. This effect has been discussed in chapter 3 of this thesis.

Moreover, the contribution of multi-product crop materials to GHG emission reduction is potentially underestimated for another reason, too. In this study, only the production stage of multi-product crop materials is considered, but not the rest of their life cycle. Multi-product crop materials are assumed to be nearly CO_2 neutral. However, if multi-product crop materials are used for a long-term use carbon is stored. If these materials are moreover incinerated with energy recovery after use, carbon is released but the generated energy replaces fossil energy.

6 Summary and conclusions

Crop production costs in Poland are much lower than in the Netherlands. Consequently, if no agricultural subsidies are granted primary biomass fuel costs of wheat and poplar are significantly lower in Poland. For hemp, this is not the case because crop production costs in Poland are based on relatively expensive experimental plots. Including agricultural subsidies, primary biomass fuel costs of the multi-product crop system in the Netherlands are calculated at 7.5 €/GJ for wheat, -1.7 €/GJ for poplar and –4.1 for hemp. Primary biomass fuel costs of multi-product crop systems in Poland are 0.1 €/GJ for wheat, 1.2 €/GJ for poplar and 9.8 €/GJ for hemp. On the other hand, GHG emission reductions per ha and year of multi-product crop systems are calculated to be lower in Poland than in the Netherlands. This due to the low crop yields in Poland. GHG emission reductions of multi-product systems in the Netherlands are 5.0 Mg CO_{2eq}/(ha*yr) for wheat, 0.9 Mg CO_{2eq}/(ha*yr) for poplar and 7.8 Mg CO_{2eq}/(ha*yr) for hemp. For Poland, respective GHG emission reductions are 4.1 Mg CO_{2eq}/(ha*yr) for wheat, 0.2 Mg CO_{2eq}/(ha*yr) for poplar and 2.4 Mg CO_{2eq}/(ha*yr) for hemp. Comparing multi-product crop systems in the Netherlands and Poland, it should be noted that quality of data on Polish crop production systems was low. Many parameters (e.g. costs, yields and GHG emissions) had to be estimated from comparable systems, see Section 3.

In general, the economic attractiveness of multi-product crops depends strongly on the market price of crop components for material use. Apart from these material prices, primary biomass fuel costs are very sensitive to crop production costs, crop yields and the percentage of the crop employed for energy. At current material prices, using components of wheat, hemp and poplar for materials lowers primary biomass fuel costs. For hemp in the Netherlands, material use of all components (i.e. core, bark fibres and eventually seeds) is economically favourable to energy use. For hemp in Poland, only material use of main components (i.e. bark fibres and eventually seeds) is economically favourable to energy use. Also for wheat, only material use of the main component (i.e. grain) lowers the primary biomass fuel costs. On the other hand, poplar has only one possible material component.

Net annual GHG emission reductions per ha of multi-product crop systems depend strongly on the specific GHG emission reduction of multi-product crop material use. Furthermore, net GHG emission reductions per ha are strongly influenced by the specific GHG emission reduction of energy use (i.e. by the reference energy system) and the percentage of the crop employed for energy. Besides, net annual GHG emission reductions per ha depend on GHG emissions during crop production and on crop yields. With the material substitutions assumed in this study, using components of hemp, wheat and poplar for material applications, lowers net GHG emission reductions compared to energy use. Thus, energy use of these crops is favourable to multi-product crop systems. However, this fact may change with other material uses and reference materials and other reference energy systems.

Finally, multi-product use of crops can significantly decrease primary biomass fuel costs compared to energy crops. However, this does not apply in general, but depends on crops and material uses. For the examples analysed in this chapter, GHG emissions per ha and year are not lower for multi-product use of crops than for singly energy use. The case might be different for other crops and material uses. Therefore, multi-product crops are not a priori an option to increase the performance of bioenergy systems. A detailed analysis of specific crops and material applications is necessary to evaluate the attractiveness of an individual multi-product crop system.

Further research on the feasibility of large-scale multi-product crop systems is advisable. Especially related macro-economic effects on land and material markets should be investigated. Furthermore, a more detailed analysis of material substitution systems including land demand of reference materials and waste treatment of multi-product crop materials is desirable. This could estimate the contribution of multi-product crops to net GHG emission reduction estimated more accurately.

CHAPTER 3:

Cost and CO_2-emission reduction of biomass cascading - Methodological aspects and case study of SRF poplar*

Abstract
This study presents and applies a coherent methodological framework to compare biomass cascading chains, i.e. the subsequent use of biomass for materials, recycling and energy recovery, considering land use, CO_2 emission reduction and economic performance. Example cascading chains of short rotation poplar wood are compared to each other on basis of literature data. Results for these chains vary strongly, namely, from CO_2 mitigation benefits of 200 €/Mg CO_2 to CO_2 mitigation costs of 2200 €/Mg CO_2, and from net CO_2 emission reductions per hectare of biomass production of 28 Mg CO_2/(ha*yr) to net CO_2 emissions of 8 Mg CO_2/(ha*yr). Using a present-value approach to determine CO_2 emissions and costs affects the performance of long-term cascading chains significantly, i.e. cost and CO_2 emission reduction are decreased. In general, cascading has the potential to improve both CO_2 emission reductions per ha and CO_2 mitigation costs of biomass usage. However, this strongly depends on the biomass applications combined in the cascading chain. Parameters that significantly influence the results are market prices and gross energy requirements of substituted materials and energy carriers, and the efficiency of biomass production. The method presented in this study is suitable to quantify land use, CO_2 emission reduction and economic performance of biomass cascading systems, and highlights the possible impact of time on the attractiveness of specific cascading chains.

---

* To be published in *Climatic Change*; Co-author: A.P.C. Faaij.

1 Introduction

According to many studies, the use of biomass for energy and material purposes may contribute significantly to the reduction of global carbon dioxide emissions if produced in a sustainable way (IPCC, 2001b; Turkenburg et al., 2000). Biomass can replace fossil fuels not only for the supply of heat, electricity and transportation fuel, but also as a feedstock for material production. Nevertheless, the use of biomass for material and energy applications can have two main disadvantages. Firstly, the costs involved are often higher than the costs of alternatives based on fossil fuel. Secondly, biomass production needs large quantities of often-scarce land resources if it is to contribute substantially to the world's energy consumption. Accordingly, an efficient use of biomass with regard to costs and land use is desirable in order to reduce CO_2 emissions. Such an efficient use of biomass can be realised by biomass cascading. In this context, cascading means that biomass is first used for a material application; next, it may be recycled for several further material applications; and finally, energy is recovered from the biomass. By means of cascading, reference materials and fossil fuels can be saved more than once during the lifetime of the biomass.

While cascading has been proposed for improving the GHG reduction efficiency of biomass systems—see e.g. Fraanje (1997) and Goverse et al. (2001)—few integrated analyses of biomass material and energy systems analysing their efficiency with respect to CO_2 emissions reduction, costs and land use have been carried out. Several studies have investigated carbon balances and land use of biomass materials, in particular from long-rotation forestry wood; see e.g. Schlamadinger et al. (1997) and Börjesson and Gustavsson (2000). In addition, several studies have analysed carbon balances of the utilisation of biomass for energy (Lettens et al., 2003; Ney and Schnoor, 2002). However, none of these studies compared the subsequent use of biomass for materials and energy with the use of biomass for either material or energy. Hence, the effects of biomass cascading on final CO_2 emission reduction and costs with regard to the area of land used for biomass production are not well understood from a bottom-up perspective.

Methodological aspects play an important role in quantifying these effects. For example, the choice of a reference system to determine CO_2 emission reductions of a biomass cascading chain can be very important, as shown by life cycle analyses of bio-material systems (Patel et al., 2003; Finnveden and Ekvall, 1998). Another aspect is the evaluation of CO_2 emission reductions at different moments in time. This becomes particularly important if long-life products are considered and CO_2 emissions can be distributed over a time span up to some hundred years. While for biomass material and energy systems this problem has not been studied extensively, several approaches have been developed to evaluate the sequestration of carbon in forests as function of time (IPCC, 1999; OECD and IEA, 2001; Marland et al., 2001). In conclusion, to enable a quantitative analysis of land use, CO_2 emission reduction and costs of complex biomass cascading chains, a coherent methodological framework is needed but not available.

A wide variety of biomass material applications and possible cascading chains exist, and it is unclear yet, which biomass chain is optimal with regard to costs, CO_2 emission reduction and land demand. This is due to the large number of parameters (e.g. material production process, crop yield, reference energy system) affecting the overall performance of biomass cascading chains. Preliminary analyses have shown that CO_2 emission reduction per hectare of biomass production and the CO_2 mitigation costs differ significantly for different biomass cascading systems (Dornburg and Faaij, 2001a).

Therefore, the methodological framework defined in this study is used to investigate and compare different biomass cascading chains. In order to analyse the effect of combining different material and energy applications in a biomass chain, we demonstrate the methodology with a single biomass crop. For that purpose, we select short rotation (SR) poplar wood. Key criteria for this selection are that the crop should be suitable for a broad variety of material applications, can be cultivated on different land qualities and has a relative high yield in Northern Europe, which is the geographical area considered in this study.

Summarizing, the aim of this study is twofold: *(1) to select and develop a coherent methodological framework for the comparison of different biomass cascading chains in terms of costs, land demand and CO_2 emission reductions and (2) to identify key parameters and issues that influence the efficiency of biomass cascading chains.*

The remainder of this chapter is organised as follows: In Section 2, the methodological framework is defined, while special attention is paid to the inventory and evaluation of different approaches for such a framework. Important aspects of this framework are (1) the boundaries of the biomass cascading system and the respective reference system, (2) inclusion of land use, (3) dealing with the time dimension and (4) definition of reference applications. Section 3 gives an overview of the biomass chains of SR poplar wood that are selected for this study, and presents the reference applications for the material and energy carriers produced from biomass. In the second part of this section, the background of input data for the calculation of costs, land demand and CO_2 emission reductions of the biomass systems is discussed. Results of applying the methodology to the SR poplar chains, together with sensitivity analyses are presented in Section 4. Finally, Section 5 discusses the results and draws general conclusions.

2 Methodology

2.1 System boundaries of biomass and reference system

To determine the CO_2 emission reduction that can be achieved by using biomass, a reference system needs to be defined (IPCC, 2001b). Usually it is assumed that materials and energy carriers from biomass substitute materials and energy carriers that fulfil the same functions; see e.g. van den Broek et al. (2001).

If more than one function is fulfilled in the biomass system—in our case through the production of several materials and energy carriers—two main approaches to define a reference system exist (CML, 2001). First, within a 'mono-functional' approach one 'main function' of the biomass system is compared to a reference function. Impacts related to this 'main function' are determined by allocation. Second, within a 'multi-functional' approach, all functions of the biomass system are compared to functions of the reference system. As an example, combined heat and power production using biomass can be compared to heat and power production from fossil fuels ('multi-functional'). On the other hand, part of the impacts of combined heat and power production using biomass can be allocated to biomass power generation. Subsequently, these allocated impacts of power generation from biomass can be compared to power production using fossil fuels ('mono-functional'). A main objective of this study is to compare different biomass cascading chains that fulfil several functions. Therefore, a *multi-functional approach* is applied.

Some studies that compare different biomass systems use a product-basket approach; see e.g. (van den Broek et al., 2001). This implies that the system boundaries of each biomass cascading system are extended in such a way that all systems fulfil the same functions. For example, to compare a biomass power plant (system A) to particleboard production and electricity recovery by incineration of the board at the end of its life cycle (system B), board production would be added to system A. However, when analysing a number of biomass cascading chains, each one producing a different suite of materials, the resulting product basket would become very complex. Therefore, in this study *every single biomass cascading chain is compared to a single reference system.* This enables a fair comparison between various biomass chains, which is what we want to demonstrate here.

Because a final waste-to-energy step of the biomass is included in the analysis for a fair comparison, the reference materials are assumed to be converted to energy after use too. The amount of energy carriers produced in both systems is evened out by assuming that surplus energy carriers in either the biomass or the reference system would be produced from an average mix of fossil fuels in the respective other system.[1]

Transportation of biomass and materials, as well as collection of waste materials, is not considered in this study, because a detailed analysis of all transportation steps is beyond the scope of this study and in general, the impact of transport on overall energy balances tends to be minor (Biewinga and van der Bijl, 1996; Dornburg and Faaij, 2001b). Therefore, we simply assume that logistics in the biomass and the reference cascading chain are comparable.

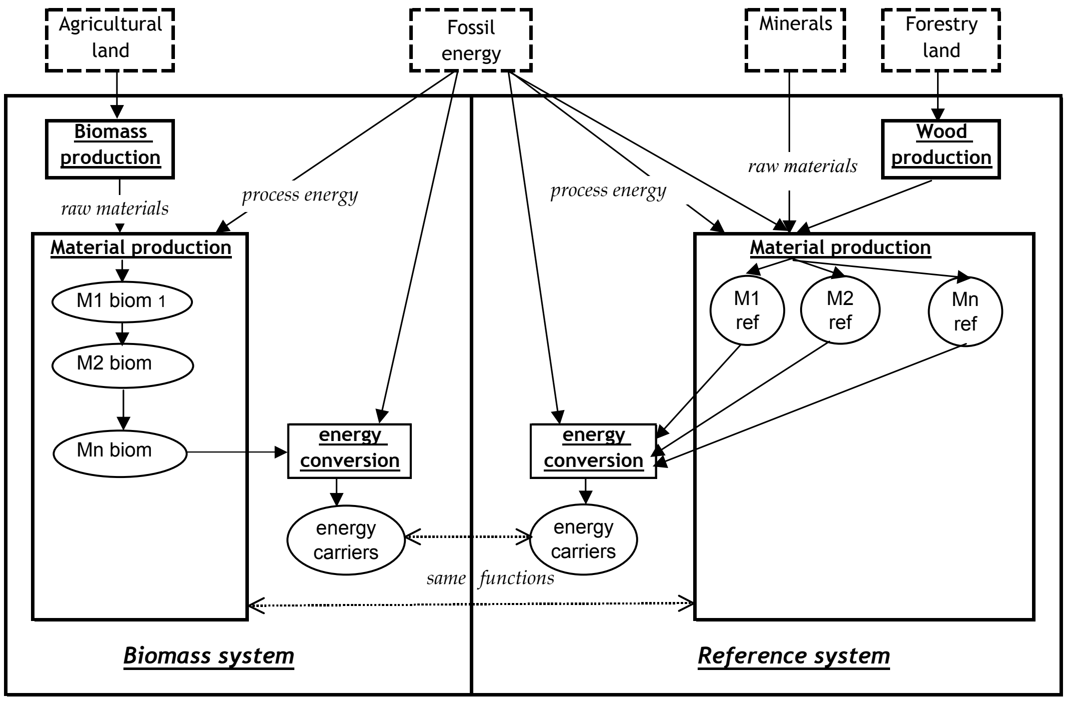


Figure 3.1: Schematic representation of biomass cascading system with according reference system

Figure 3.1 gives an overview of the biomass reference system as investigated here. Within the rest of this study, the biomass system refers to cascading chains of SR poplar. Note that biomass may be used for material production in the reference systems as well, because in

[1] Thus, we assume that biomass energy would most likely replace fossil energy.

some cases products from SR poplar substitute wood products from conventional forestry. The implications of reference products from conventional forestry for the inclusion of land demand into the analysis are discussed in Section 2.2.

2.2 Land use

Land use, i.e. the area of land occupied by biomass production, is a crucial parameter in the analysis and comparison of biomass systems (Schlamadinger et al., 1997). While the inclusion of land use in comparative analyses of biomass energy and material systems is not common practice yet, this issue has been gaining importance in recent discussions; see e.g. Green (2000) and chapter 4 of this thesis. Approaches to include land use in the analysis of biomass systems vary. In a study of van den Broek et al. (2001), the area of land used is a function in the biomass system. Other functions in this biomass system are food and energy carriers produced. Consequently, in the biomass and the reference system, the same area of land is used and the same amounts of food and energy are produced. Other authors limit the use of land to the biomass system. CO_2 emissions and other environmental impacts are then calculated per ha of land used in the biomass system, see e.g. Gärtner et al. (2002).

Roughly, the geographical context considered in this study is Northwest-Europe (i.e. North-France, The Netherlands, Belgium and Germany). These countries have quite comparable conditions for biomass production and, moreover, comparable material markets. It is assumed that short rotation poplar is cultivated on medium quality agricultural land. Thus, in this study 1 ha of *land use in the biomass system* refers to the occupation of 1 ha of *medium quality agricultural land in NW-Europe* for one year.

At any rate, the production of biomass in the biomass system requires land. Moreover, for the production of reference materials, land may be used too. This is clearly the case if SR poplar products in the biomass system substitute wood products from conventional forestry in the reference system. On the other hand, if short rotation poplar products substitute only fossil fuel based or mineral materials, no land is needed in the reference system. In this study, both situations are considered.

If land is used to produce wood products from conventional forestry in the reference system, it has to be accounted for. The area to produce wood products is typically larger than the area to produce short rotation poplar products with equivalent functions. This is due to lower yields in conventional long-rotation forestry (Kaltschmitt and Reinhardt, 1997). Therefore, defining land use in the biomass system as the land use for biomass production minus the land use for conventional wood production may lead to negative net land uses.

As a negative net land use is not a useful unit for comparison, another method is applied to account for the land use in the reference system.

The *land use of conventional forestry* can be *converted to CO_2 emission reductions and cost reductions*, which are the main objectives of biomass systems in this study. In order to do so, the amount of energy crops that could be produced on this forestry land if it would not be used for wood production, is calculated. If these energy crops replace fossil fuels, CO_2 emissions are reduced. (CO_2 emission reductions are the CO_2 emissions due to the use of fossil fuels minus the CO_2 emission due to the production of energy crops.) Similar to the reduction of CO_2 emissions, these energy crops can also reduce costs, i.e. the costs of fossil fuel use minus the costs of energy crop production. By using this approach, specific CO_2 emission reduction and cost reduction per area of reference land demand can be determined.[2]

However, yields of energy crops on forestry land are in most cases lower than yields of energy crops on agricultural land. Therefore, the yields of energy crops on forestry land are corrected by a dimensionless quality factor. For example, with a quality factor of 0.5, yields of energy crops expected on conventional forestry land are half of those expected on medium quality agricultural land.

2.3 Time dimension

(Avoided) CO2 emissions over time

The use of biomass, when derived from well-managed plantations, is considered to be close to having no net impacts on the carbon in the atmosphere, because all carbon sequestered during plant growth is released during energy conversion and vice versa. However, if cascading systems of biomass are considered, the release of sequestered carbon can take place significantly later in time than the moment biomass is harvested. Depending on the applications in the cascading chains this period can vary from several weeks (e.g. paper) to a century or more (e.g. construction wood). Furthermore, CO_2 is emitted at different moments in time in the biomass as well as in the reference system.

---

[2] Another obvious possibility to convert the use of conventional forestry land in the reference system to CO_2 emission reductions exists. Here, one would assume that alternatively the forest would not be harvested and consequently, carbon would be sequestered in the standing trees. This carbon sequestration could be expressed as carbon emission reduction. However, the hypothetical conversion of sequestered carbon to permanent carbon emission reduction is a disputed topic and several approaches exist; compare OECD and IEA (2001). Furthermore, sequestered carbon is never fully equivalent to avoided fossil fuel based carbon emissions. In order to circumvent this issue, we assume that in case the wood is not used for products, energy crop production instead of carbon sequestration takes place on the forestry land.

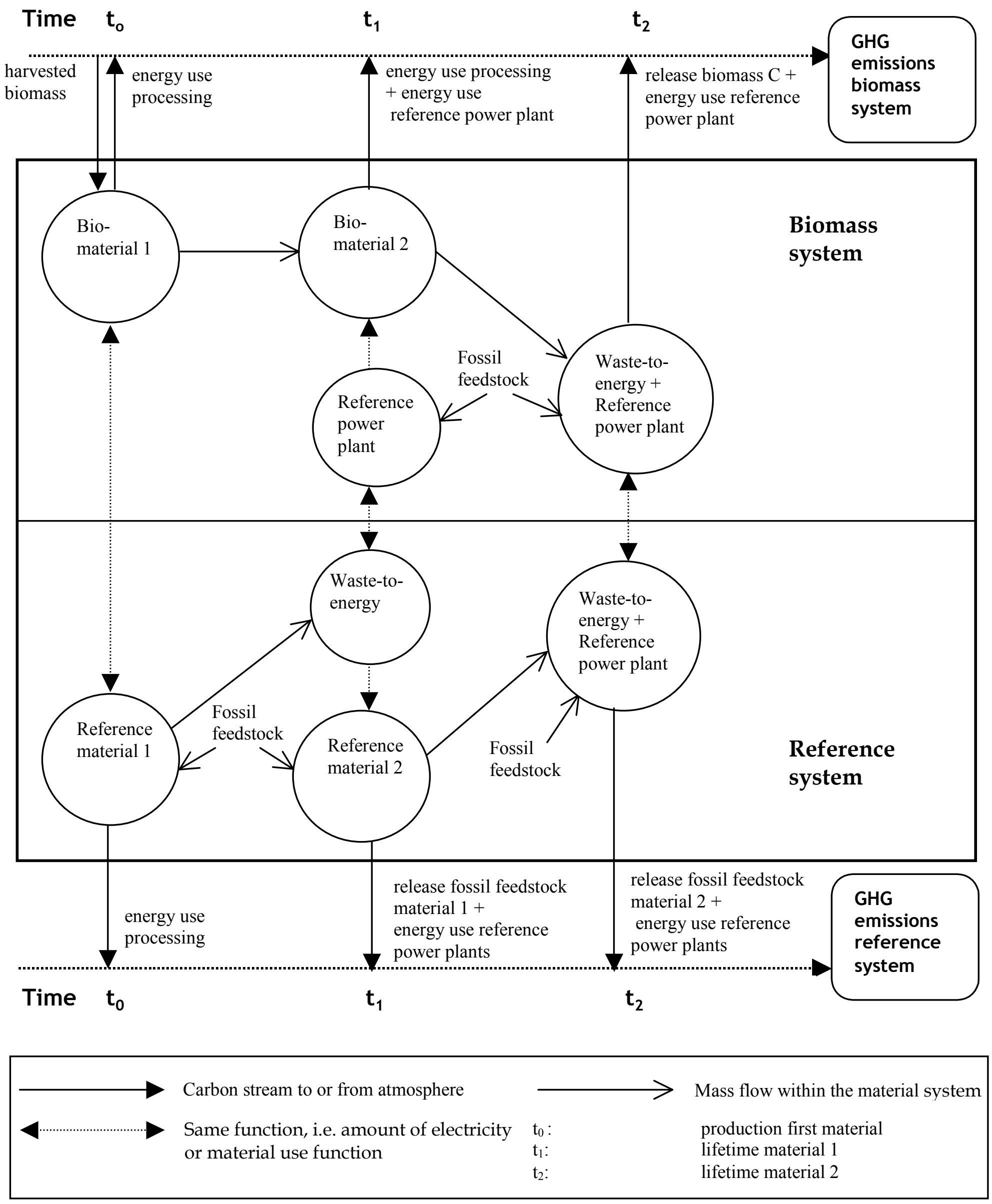


Figure 3.2: Carbon streams in time for a biomass cascading system with two material application steps

Figure 3.2 illustrates schematically CO_2 emissions for a cascading chain with two different material production steps and a waste-to-energy conversion step. The analysis starts with the input of carbon sequestered in the biomass at time t_o, when the first material in the cascading chain is produced, i.e. the biomass is harvested. After the lifetime of this first material, at time t_1, CO_2 emissions occur due to recycling and the production of the second material. At time t_2, after the lifetime of the second material, the biomass is converted to energy and the sequestered carbon is released. At the same moments in time reference materials and energy carriers are produced.

It is obvious that the later CO_2 is emitted to the atmosphere, the later the greenhouse effect due to this emission takes place. To account for CO_2 emissions at different moments in time, an indicator predicting climate change due to a certain emission at a certain moment would be desirable. However, this climate change depends on many external factors such as the concentration of greenhouse gases in the atmosphere and, thus, cannot be determined. (IPCC, 2001a) Alternatively, carbon emissions in time could be evaluated by an economic approach. Marland et al. (1997) evaluated CO_2 emissions by a carbon discount rate representing the potential economic damage of a certain emission. However, this damage also depends on many external and unpredictable factors.

Van den Broek et al. (2001) discussed that physical units that represent monetary values can be converted to present values. They applied this concept to the yields of a eucalyptus plantation. The concept may also be used for CO_2 emissions, since these are tradable and, thus, represent a monetary value, which may vary in time.

In this study, lacking a better instrument, we evaluate CO_2 emissions at different moments in time by assuming that these *CO_2 emissions represent a constant monetary value that can be converted to a present value taking into account an interest rate*. Besides CO_2 emissions, all other *cost items are converted into present values*, too. The interest rate assumed is a crucial parameter. Here, a low discount rate representing annual capital costs in public projects is assumed, because mitigating carbon emissions is considered a public responsibility.[3] To highlight the impact of time on the evaluation of CO_2 emission reductions in biomass cascading chains, the performance of CO_2 emission reduction is also analysed *without* present-value approach.

Technology developments

Because some biomass material applications last for quite a long time, it should be noted that during such a time period technological developments occur. Consequently, the economics and energy balances of processes to produce material and energy carriers out of both biomass and reference raw materials are affected. In such a way technological developments influence the performance of long-term cascading chains. However, data on expected developments of all relevant technologies are not readily available and the analysis of mechanisms behind these developments is beyond the scope of this study. Nevertheless, the potential impact of *possible developments of some technologies* is evaluated *in the sensitivity analysis*.

[3] For a discussion on the choice of interest rate in long-term problems as climate change; see (Portney and Weyant, 1999).

2.4 Definition of functional units

The biomass and the reference system are compared by means of equivalent functions, where functions are services that are embodied in material objects. (CML, 2001) The function of a certain material can be defined in several ways, because materials can have various (desirable) properties, such as the structural and indoor climate functions of a construction element. Typically, wood products have different secondary functions and characteristics than products from fossil fuels or minerals. In general, however, a main function can be identified, e.g. the structural function in construction. Only this *main function of an application* is considered in this study. The *functional unit* is the *function fulfilled by the biomass material produced from one hectare and one year of SR poplar production.* Based on this functional unit, the kind and amount of reference material can be determined. For a more detailed discussion of the reference materials, see Section 3.

2.5 Calculation method

Starting from one hectare and one year of biomass production, costs and CO_2 emissions of the biomass cascading chain in comparison to a reference system are determined. The following three key parameters are calculated for a comparison of biomass cascading chains:

- Annual CO_2 emission reduction per area of biomass production [kg CO_2/(ha*yr)]
- Net annual costs per hectare of biomass production [€/(ha*yr)]
- CO_2 mitigation costs [€/kg CO_2]

CO_2 emission reduction

The CO_2 emission reduction of a biomass cascading chain is the difference between the emissions in the reference system and those in the biomass system. These CO_2 emissions comprise emissions associated with biomass production, reference land use, material production, and the production of energy carriers. CO_2 emissions during biomass production result from the direct energy inputs and the indirect energy inputs of materials (e.g. machines, fertilisers). Moreover, we include N_2O emissions from fertiliser application using its CO_2 equivalent value. Other greenhouse gas emissions (such as CH_4 emissions from decay of roots) play only a small role in overall greenhouse gas emissions of biomass production systems (IPCC, 2001b) and are not taken into account.

Net annual costs

Analogous to the CO_2 emission reduction, the net annual costs (or benefits) of the biomass cascading chain are the difference between the annual costs in the biomass and the reference system. These cost items are biomass production costs and the costs of material and energy production. In general, no taxes or subsidies are taken into account. Net costs of biomass applications (i.e. materials and energy carriers) are defined as the production costs of the biomass application minus the market price of the reference application. However, using market prices introduces uncertainty into the analysis due to possible fluctuations of such prices. This uncertainty is addressed by using ranges of market prices that are derived from relevant statistics.

3 Cascading chains of short rotation poplar

3.1 Applications

For a case study, biomass cascading chains of short rotation poplar have been selected on the basis of a review of possible applications and recycling options, which can be found in Appendix 1. For the selection of applications the following criteria were applied:

- The suitability of SR poplar wood for the investigated application
- A potentially large market volume of the material
- The possibility of cascading
- The likeliness of high CO_2 emission reduction and low CO_2 mitigation costs
- Inclusion of at least one application in key industrial sectors that are suitable for poplar wood (construction, packaging, pulp & paper, chemicals and energy)

Finally, eight application of SR poplar wood are considered in this analysis, i.e. particle lumber (LU), medium density fibre board (MDF), transportation pallets (PA), chemical pulp (PUL), ethylene (ET), methanol (ME), viscose (VI) and electricity (EL), and for each of these applications also a reference application is defined:

- *MDF board* generally substitutes other wood products. Main substitute is structural plywood from softwood that is the reference material in this study. Functional unit for comparison is the volume of board material.
- *Particle lumber* substitutes concrete. Respective amounts needed in equivalent houses with concrete or wooden frames substitute each other in the analysis.
- *Pallets* are not commonly produced from alternative materials, e.g. from HDPE or corrugated fibreboard, and, therefore, wooden pallets from softwood are assumed as a reference material.
- *Chemical pulp* from short rotation poplar replaces chemical pulp made from other fibre sources, i.e. softwood.
- *Ethylene* from SR poplar wood substitutes ethylene from fossil sources, i.e. naphtha, gas oils, liquefied petroleum gas and ethane. Because in Western Europe about 73% of ethylene production was based on naphtha in 1997, ethylene production via steam cracking of naphtha is regarded as the reference process (CEFIC, 2000).

- *Viscose* replaces mainly synthetic staple fibres. The most important staple fibres in terms of production volume are polyester (PES) fibres (CIRFS, 2002). Viscose can be used for most technical applications for which PES can be used. As a consequence, PES fibre is assumed as reference material for viscose. Functional unit for the comparison of PES and viscose fibres is the volume of fibres.
- *Electricity* from biomass is assumed to substitute the average Western European electricity mix in the reference system.
- *Methanol* replaces gasoline, which serves as a reference material. Functional units for the comparison are lower heating values.

Selected applications and reference applications for further analysis are summarised in Table 3.1.

Table 3.1: Applications of SR poplar wood and reference applications included in this study

Abbreviation	SR poplar application	Reference material	Substitution
LU	Particle lumber	Concrete	Fossil, mineral
MDF	MDF board	Plywood softwood	Wood
PA	Pallets	Pallets softwood	Wood
PUL	Chemical pulp	Chemical pulp from softwood	Wood
ET	Ethylene	Ethylene from naphtha	Fossil
ME	Methanol	Gasoline	Fossil
VI	Viscose	PES fibre	Fossil
EL	Electricity, IG/CC	Electricity mix, Western Europe	Fossil

3.2 Cascading chains

In this section, cascading chains of SR poplar are selected for a case study. Here, a 'cascading chain' is the production of one of the selected applications from SR poplar, zero or more recycling steps to subsequently produce other material applications of SR poplar, and finally an energy recovery step. Most chains comprise two or four applications, including energy recovery.

In line with the feasibility of recycling (see Appendix 1), a number of possible cascading chains result. Assuming that a waste-to-energy recovery step of poplar material always takes place, 12 representative chains presented in Table 3.2 are selected for further analysis from all 28 theoretically possible chains. Many of the theoretically possible cascading chains only differ with regard to the last waste-to-energy recovery step, i.e. either methanol or electricity production. Because both direct production of methanol and of electricity are included, it can be concluded whether a final conversion to methanol or electricity is more favourable without studying all these different cascading chains. Therefore, all selected cascading chains comprise electricity production as last cascading step. Moreover, of all possible cascading chains with four successive applications, chains that only differ with regard to one application are represented by a single chain. In only one case, i.e. PA-LU-LU-EL and PA-LU-MDF-EL, it is analysed what difference of results the change of one step causes.

Table 3.2: Cascading chains of short rotation poplar regarded in this study

Abbreviation	Raw material		Primary material		Secondary material		Tertiary material		Energy
EL	SR poplar	⇒							Elec. IG/CC
ME	SR poplar	⇒							Methanol
LU-EL	SR poplar	⇒	Lumber					⇒	Elec. IG/CC
MDF-EL	SR poplar	⇒	MDF					⇒	Elec. IG/CC
PA-EL	SR poplar	⇒	Pallets					⇒	Elec. IG/CC
PUL-EL	SR poplar	⇒	Pulp					⇒	Elec. IG/CC
ET-EL	SR poplar	⇒	Ethylene					⇒	Elec. IG/CC
VI-EL	SR poplar	⇒	Viscose					⇒	Elec. IG/CC
LU-LU-ET-EL	SR poplar	⇒	Lumber	⇒	Lumber	⇒	Ethylene	⇒	Elec. IG/CC
PA-LU-LU- EL	SR poplar	⇒	Pallets	⇒	Lumber	⇒	Lumber	⇒	Elec. IG/CC
PA-LU- MDF-EL	SR poplar	⇒	Pallets	⇒	Lumber	⇒	MDF	⇒	Elec. IG/CC
PA-MDF-ET-EL	SR poplar	⇒	Pallets	⇒	MDF	⇒	Ethylene	⇒	Elec. IG/CC

3.3 Input data

Detailed input data for the analysis of the selected SR poplar cascading chains are presented in Appendix 2. Mainly, data have been derived from comparative studies of material production efficiencies and agricultural and forestry studies on biomass and wood production. If necessary, these data are adapted through our own calculations.

Data on the SR poplar applications and their reference materials are presented in Table A3.1. The substitution between these materials is deduced from the functional units described above and lifetimes of materials are based on lifetimes of the main applications Wood inputs for the production of materials implicitly determine the land use and the impact for agricultural and forestry production of the materials.

On the basis of (environmental) studies on several material production processes, energy inputs for material production are defined. Energy inputs related to the cultivation of wood are not included here, but are accounted for separately; see below. The energy requirements of material production are converted to CO_2 emissions using carbon emission factors that are derived from average European energy mixes.

Energy inputs of material production consist of direct and indirect energy uses. Direct energy uses are electricity, steam and other primary energy carries used in the production process. (The utilisation of wood residues and by-products from various production steps is accounted for either as energy input of the process or by allocation, see Appendix 2.) Indirect energy use is the energy use for raw material production, i.e. feedstock and process energy for the production of these raw materials and their subsequent raw materials.

Production costs of biomass materials are also taken from studies on material production processes, while market prices of reference materials are derived from mainly economic studies. As market prices are subject to changes over time, ranges reflecting those changes in recent years and average prices are given.

Data on costs and efficiency of electricity generation by IG/CC (either from SR poplar wood directly or from used materials) are presented in Table A3.2. Data on wood production within the biomass chain (SR poplar) and the reference system (softwood) are presented in Table A3.3.

Yields, costs and energy inputs of wood production vary depending on production methods and location. Data used in this study refer to NW Europe, i.e. in this case to Germany and the Netherlands. Conventional forestry production of spruce is the reference for softwood production. For the production of SR poplar average values of a four-year rotation cycle are used. Information in literature differs in whether yields and production costs decrease or increase with longer rotation times of 10 to 15 years (REU and FAL, 1996; Teeuwissen, 1999)[4] Therefore, the production data of SR poplar of a four-year rotation cycle is assumed in the case of pallet production too, even though a longer rotation time is necessary for that application.

To convert land use of softwood production into CO_2 emission and cost reductions, it is assumed that otherwise energy crops, i.e. SR poplar would be produced on that land. Potential yields of energy crops are likely to be considerably lower on former forestry land, compared to agricultural land; this is expressed by the quality factor of forestry land. Nevertheless, this factor depends on circumstances like soil composition, climate, location, etc. and can have a broad range; see also Section 5.2. [5]

[4] Teeuwissen (1999) compared data from several sources for different rotation times.
[5] Also during conversion of forestry land to short rotation coppice, a change of soil carbon takes place. However, no studies estimating soil carbon changes in temperate climate of conversion of forestry plantations to short rotation forestry have been carried out. Hansen (1993) starts from the conversion of cropland to SRF and estimates an increase if

4 Results

Using the input data described in Appendix 2, a bottom-up analysis leading to CO_2 emission reduction and cost estimates of single cascading chains has been carried out. Results of this analysis are annual CO_2 emission reductions per hectare, CO_2 mitigation costs, net annual costs per hectare and the respective present values for each cascading chain[6]. The results for all base assumptions are discussed in Section 4.1. In Section 4.2, a sensitivity analyses is presented.

4.1 Base case results

In Figure 3.3, *the annual CO_2 emission reduction per hectare* for the selected chains (see Table 3.2) is presented. (Recall, that the CO_2 emission reduction is based on the comparison of all applications in the SR poplar chain to a reference application. Compare Table 3.1 for reference applications and abbreviations.) Two chains that include the production of particle lumber show net CO_2 emissions, but most cascading chains of short rotation poplar lead to net CO_2 emission reductions. In total, the results range from net emissions of 8 Mg CO_2/ (ha*yr) to net reductions of 28 Mg CO_2/(ha*yr).

Because of relatively low energy requirements of the respective reference materials, the production of particle lumber, ethylene, methanol and electricity have the lowest CO_2 emission reduction potentials. With the given assumptions on the reference energy systems and on the production processes from biomass, electricity production has a higher CO_2 emission reduction per ha and year than methanol production. Therefore, electricity production is the best waste-to-energy conversion step investigated. It is remarkable that the applications that substitute for products from softwood, i.e. MDF board, pallets and pulp, have a higher CO_2 reduction potential than those substituting for non-renewable materials. This is valid under the assumptions made for the utilisation of forestry land for energy cropping.

To add an additional reuse step to a cascading chain can lead to both higher and lower annual CO_2 emission reductions per ha. This depends on the additional application. For example, including particle lumber production in an existing cascading chain (e.g. PA-EL)

10-25 Mg C/ha over a period of 10 to 15 year, while Guo and Gifford (2000) state a change of 18% of soil carbon between cropland and plantations. Assuming an average soil carbon content of temperate forests of 105.5 Mg C/ha (Paustian et. al., 1998) and a duration of 100 years of the land use change to short rotation forestry, the emission of CO_2 from soil can be estimated to be about 0.5 Mg CO_2/(ha*yr).

[6] Results are expressed in relation to Mg CO_2. 1 Mg CO_2 is equivalent to 12/44 Mg C.

lowers the CO_2 emission reduction per ha per year, while including MDF production increases it.

As discussed in Section 2, the annual CO_2 emission reduction per ha and year is presented (1) without discounting CO_2 emission in time and (2) with a present-value approach of these emissions (assuming an interest rate of 5%). Applying a present-value approach to CO_2 emissions lowers the CO_2 emission reductions for a given cascading chain. This applies especially to chains that have a long lifetime, i.e. chains including particle lumber production. The maximal difference between the non-discounted CO_2 emission reduction and the present value of CO_2 emission reduction is about 12 Mg CO_2/(ha*yr) for the chain of PA-LU-MDF-EL. Applying present values also changes the order of best applications with regard to CO_2 emission reductions, i.e. ET-EL becomes better with present value approach than LU-EL.

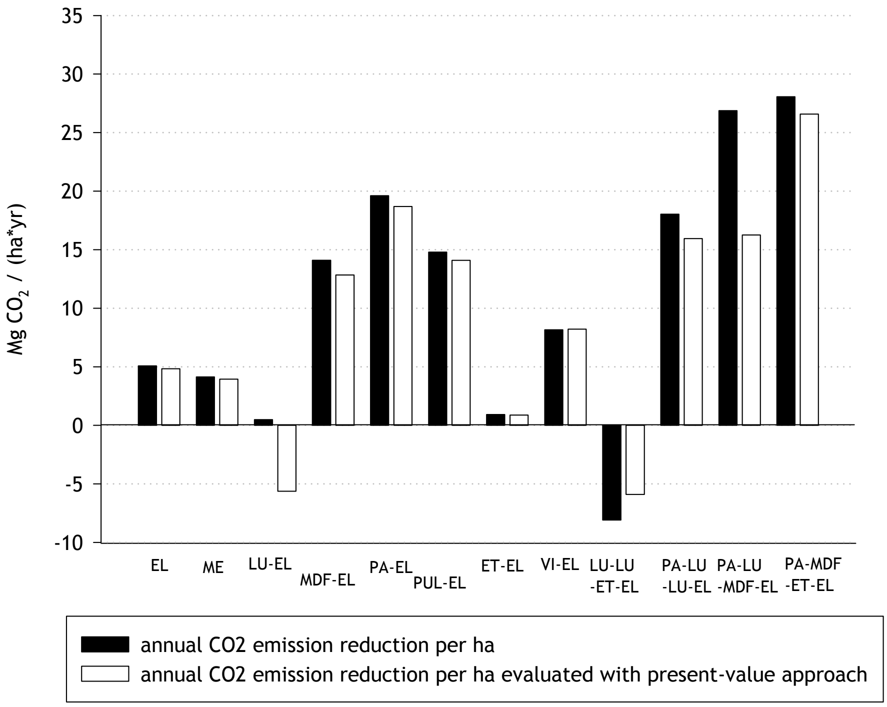


Figure 3.3: Net annual CO_2 emission reduction per ha (+) or net annual CO_2 emissions (-) of the different cascading chains with and without applying a present-value approach

In Figure 3.4, *CO_2 mitigation costs* are shown. (For cascading chains with net CO_2 emissions, no value for mitigation costs is presented.) While some cascading chains lead to net benefits, CO_2 mitigation by means of other chains is found to be very expensive. CO_2 mitigation costs of the different chains vary between net benefits of 200 €/Mg CO_2 and net costs of 2200 €/Mg CO_2.

Market price ranges of reference materials result in large uncertainties in these figures, as indicated by the error bars. These uncertainties can change the ranking of the different chains. For example, electricity production can become more expensive with regard to mitigation costs than pallet production with electricity recovery.

In the base case, the production of MDF board, transportation pallets and pulp, and most chains including a larger number of re-use steps, i.e. PA-LU-LU-EL, PA-LU-MDF-EL and PA-MDF-ET-EL, have the lowest CO_2 mitigation costs. In absolute terms, MDF board production replacing plywood scores best. Hence, also with regard to CO_2 mitigation costs, poplar wood applications substituting softwood products perform well. This depends, of course, on the potential yields of energy crops on the conventional forestry land; see Section 4.2. Electricity production is more favourable than methanol production as waste-to-energy step with respect to CO_2 mitigation costs. Compared to undiscounted mitigation costs, the present value approach does not change the results significantly. This is because both monetary values and CO_2 emission reductions are discounted and the effects partly offset each other.

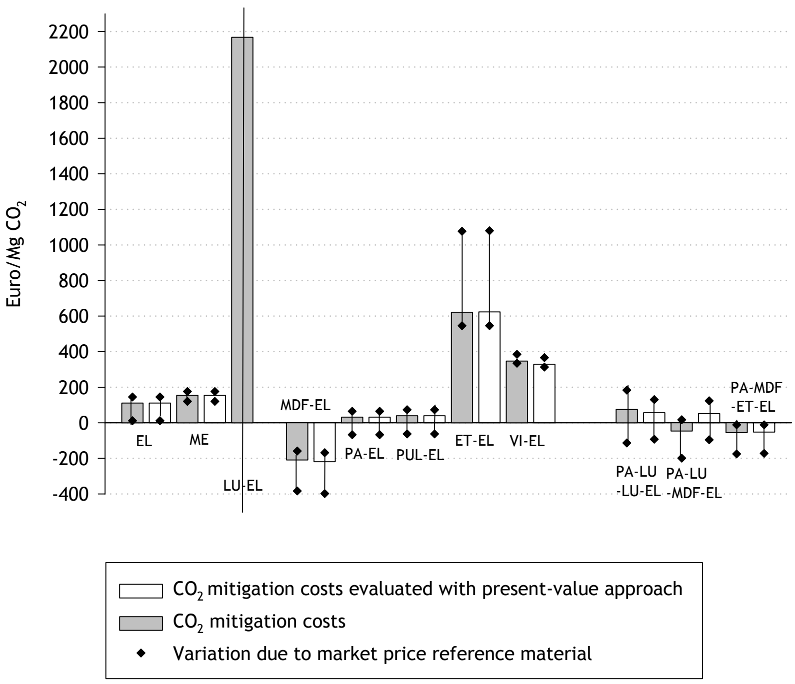


Figure 3.4: CO_2 mitigation costs (+) or benefits (-) of the different cascading chains with and without applying a present-value approach to costs, incomes and CO_2 emission reduction

Finally, the net annual *costs or benefits per ha* of the biomass cascading chains are shown in Figure 3.5. In total, they range from benefits of 3000 €/(ha*yr) to costs of 2900 €/(ha*yr) for the different cascading chains. For comparison, net margins of farmers in Ireland and the Netherlands are about 400 €/(ha*yr) and 1300 €/(ha*yr), respectively (van den Broek et al., 2002).

Also with regard to costs or benefits per ha and year, market prices of reference materials result in large uncertainties. Comparing the present value of costs and benefits to the un-discounted costs and benefits, the preferred order of cascading chains changes, i.e. PA-LU-MDF-EL becomes less attractive than ME, PA-EL, PUL-EL, ET-EL and EL in the present value approach. Highest incomes per ha and year (with and without present value) are achieved by MDF board production, followed by the cascading chain of PA-MDF-ET-EL. Highest costs per ha and year emerge from viscose production.

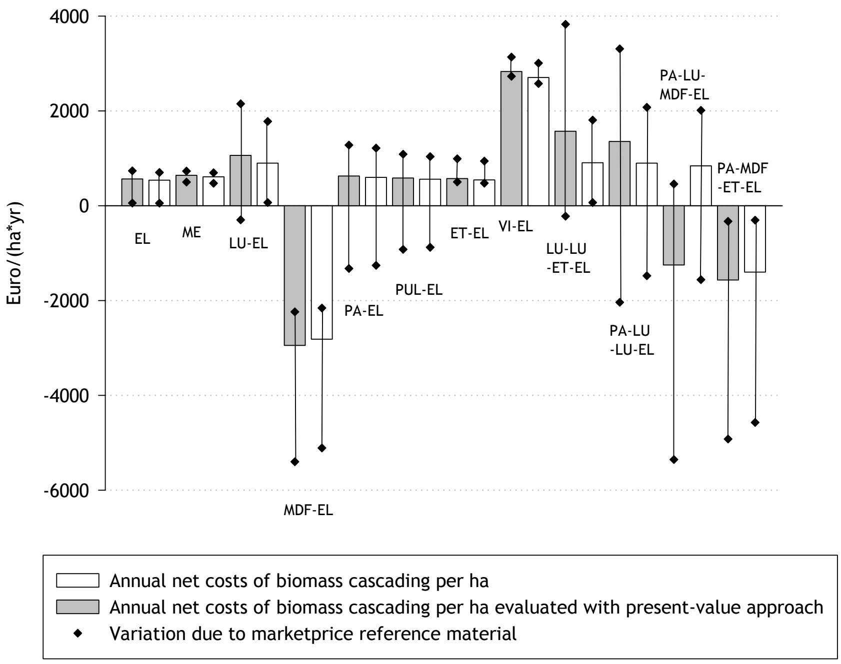


Figure 3.5: Net annual costs (+) or benefits (-) of the different cascading chains per ha with and without applying a present-value approach to costs and benefits

4.2 Sensitivity analysis

In this section, sensitivity to the main parameters is investigated. These main parameters are poplar yield, carbon emission factor of electricity, land price, rent and the quality factor for the conversion of forestry to agricultural land. Apart from that, the influence of technology developments is analysed. The sensitivity analysis focuses on present values of annual CO_2 emission reductions per ha and on present values of CO_2 mitigation costs.

Table 3.3 shows ranges of the main parameters that are varied. Concerning long-term technology improvements, many technologies of material production are well developed and no significant cost or efficiency improvements are foreseen. However, the production of methanol, ethylene and electricity from biomass with IG/CC technology are in a quite early stage of development. Therefore, potential developments of these technologies within a time frame of about 20-30 years are incorporated in the sensitivity analysis and input data are summarised in Table 3.4.

Table 3.3: Ranges of parameters for sensitivity analysis

Parameter	**Base**	**Min**	**Max**
Poplar yield[a] [Mg/(ha*yr)]	8.6	6.4 (74%)	19.3 (224%)
Land prices[b] [€/ha]	220	128 (58%)	338 (154%)
Carbon factor electricity[c] [kg CO_2 /GJ_e]	109	0 (0%)	270.3 (248%)
Interest rate[d] [%]	5	3 (60%)	18 (360%)
Quality factor land [$ha_{agriculture}/ha_{forestry}$][e]	0.66	0.2 (30%)	1 (152%)

[a] Depending on location factors (Lewandowski, 2001).
[b] Low and high average prices in NW-European countries (Eurostat, 2000).
[c] 100% renewable/nuclear power plants or 100% coal power plants (η_e = 45 %).
[d] Inflation rate compared to commercial amortisation rates.
[e] The lower limit is representative for conventional forest yields.

Table 3.4: Long term technological developments considered in the sensitivity analysis

Parameter	**Base**	**Min**	**Max**
Ethylene production [a] [$kg_{bio\text{-}mat}$ /kg_{wood}]	0.076	-	0.089 (117%)
Ethylene production costs[a] [$€_{2002}$ /$Mg_{bio\text{-}mat}$]	1400	1062 (73%)	-
Methanol production [b] [$kg_{bio\text{-}mat}$ /kg_{wood}]	0.447	-	0.525 (117%)
Methanol production costs[b] [$€_{2002}$ /$Mg_{bio\text{-}mat}$]	227	171 (75%)	-
Electric efficiency IG/CC[c] [%]	43.5	-	53 (122%)
Production costs electricity [$€_{2002}$/GJ_e]	10.4	4.5 (42%)	-

[a] Improvements of methanol production process, see below.
[b] Fuel production only 430 $MW_{th\text{-}input}$ plant as modelled in Hamelinck and Faaij (2002).
[c] Advanced IG/CC plant of 215 MW_e (Faaij et al., 1998).

Figure 3.6 presents the *sensitivity of present values of the annual CO_2 emission reduction per ha* to changes of the main parameters and to long-term technological developments. Overall, the variation of the parameters does not change the ranking of the cascading chains with respect to CO_2 emission reductions.

The maximum SR poplar yield considered in the sensitivity analysis leads to the highest possible CO_2 emission reduction per ha and year. SR poplar yields have the largest influence on applications substituting wooden reference materials, i.e. MDF boards, pallets and pulp. This is due to the fact, that poplar yields also influence CO_2 emissions that are calculated on basis of energy crop production on land used in the reference system; see Section 2. However, the sensitivity analysis shows that MDF boards, pallets and pulp are still better than ethylene and particle lumber production, and are comparable to methanol and electricity production if a low potential yield of energy crops on former forestry land is assumed. These low yields are about equivalent to wood yields in conventional forestry.

The influence of the carbon emission factor of electricity on the CO_2 emission reduction varies with the amount of electricity used during material production. Because electricity is used and produced in both the biomass and reference system, lowering the carbon factor of electricity can lead to either an increased or a decreased CO_2 emission reduction depending on the chain. In the chains LU-EL, VI-EL and LU-LU-ET-EL the annual CO_2 emission reduction per ha decreases if the intensity of carbon per unit of electricity increases. Finally, long-term technology developments lead to a slight increase of annual CO_2 emission reductions per ha.

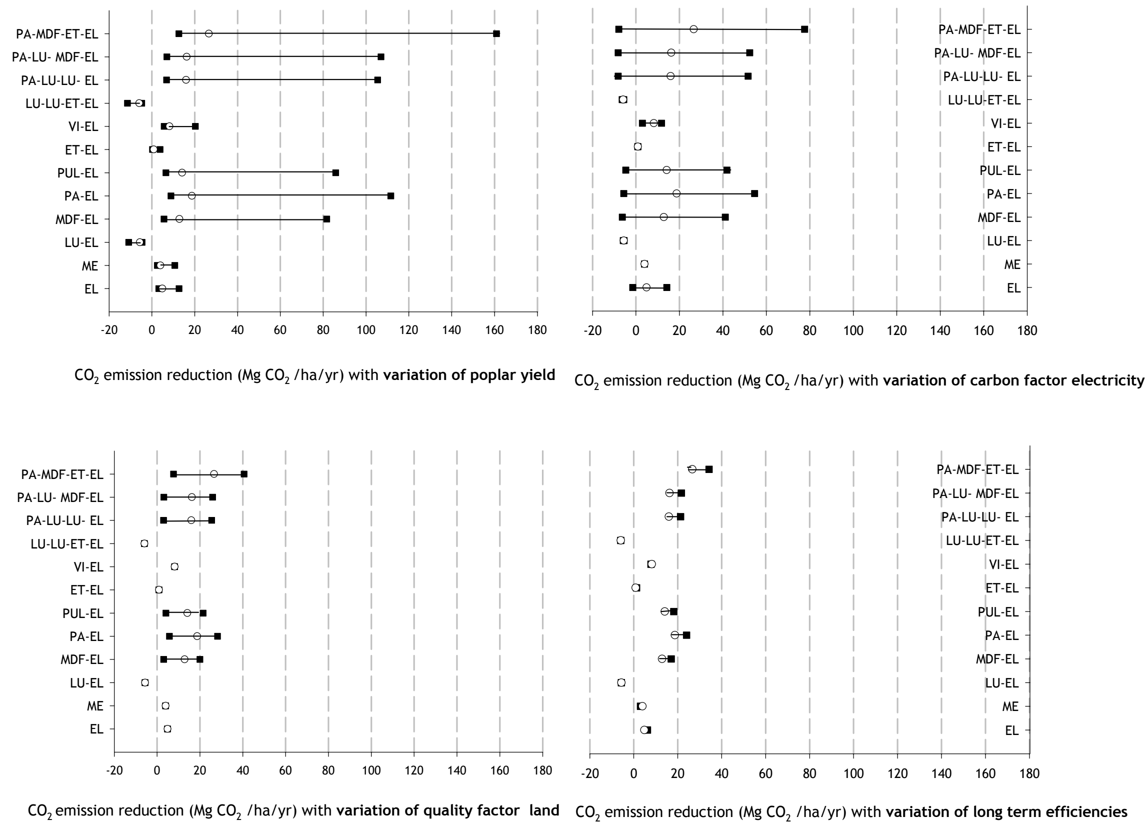


Figure 3.6: Sensitivity analysis with regard to annual CO_2 emission reduction per ha based on a present value approach

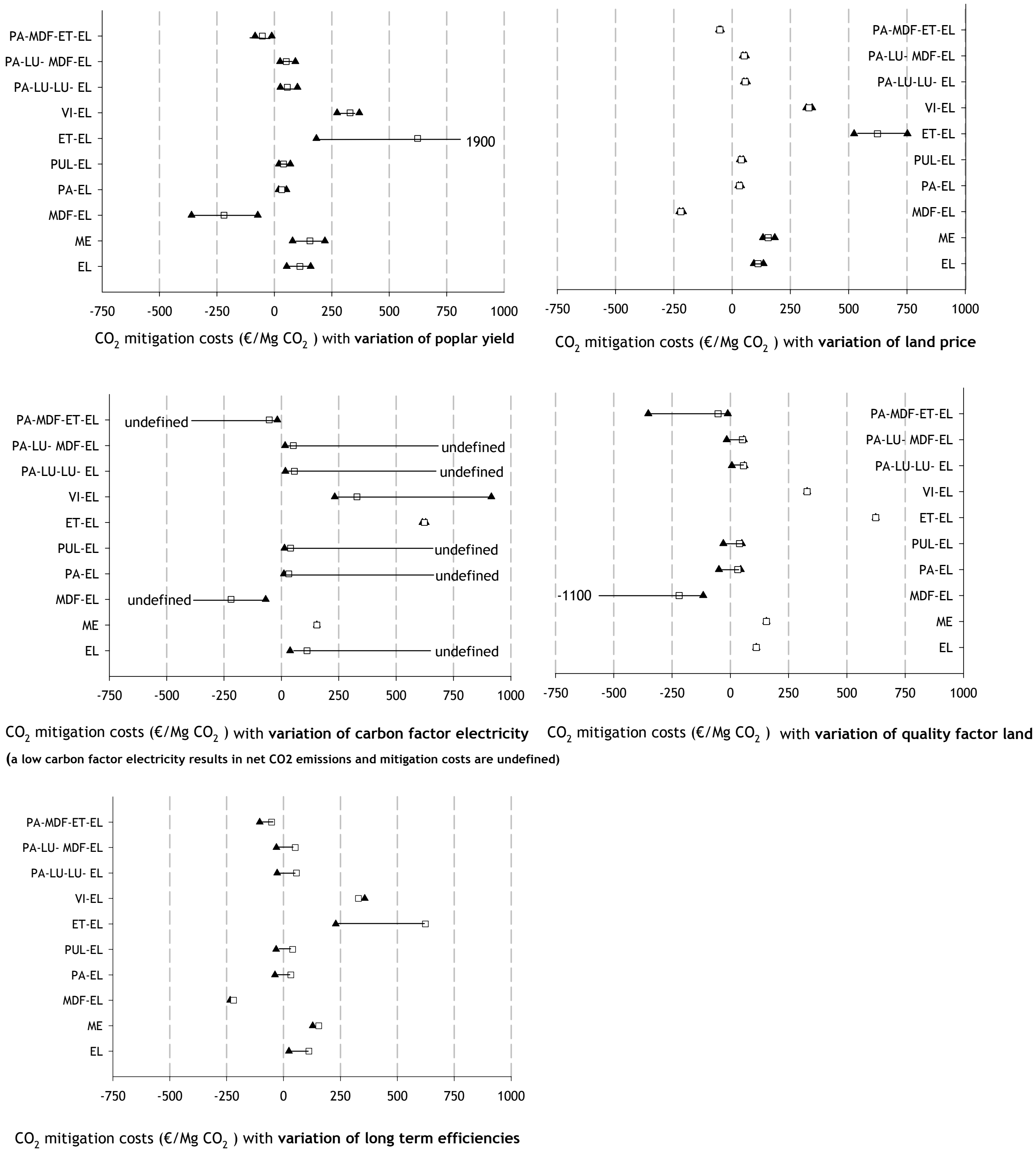


Figure 3.7: Sensitivity analysis of CO_2 mitigation costs to the variation of different parameters assuming a present-value approach

In Figure 3.7, the *sensitivities of present values of CO_2 mitigation costs* are presented. Several parameters show a strong effect on CO_2 mitigation costs. By far the highest CO_2 mitigation costs result from ethylene production assuming a low yield of poplar. This is due to the fact that only a small amount of ethylene is produced per unit of biomass. Consequently, biomass production costs per unit of ethylene produced are very high, if poplar yields are low. On the other hand, a potential low yield of energy crops on forestry land, i.e. a low quality factor of forestry land, increases CO_2 mitigation costs of applications substitution wood products only slightly. In many cases, CO_2 mitigation costs could not be defined for the variations of the carbon emission factor of electricity, because net CO_2 emissions result for the respective cascading chain and the respective value of the carbon emission factor.

If technology improvements are assumed, CO_2 mitigation costs decrease slightly for electricity and methanol production. For the production of ethylene, CO_2 mitigation costs decrease strongly, so that ethylene production becomes more attractive with regard to CO_2 mitigation costs than viscose production on the long term. This can be explained by the fact that the technology performance of ethylene production is assumed to improve significantly in the long term. These fluctuations in CO_2 mitigation costs are directly linked to the data used for the technologies and more thorough analysis is desired on this point. Finally, the variation of the interest rate (from 3 to 18%) had only a minor influence on the results and is, therefore, not presented in Figures 3.6 and 3.7.

5 Discussion and conclusions

In this study, we developed a methodology to evaluate biomass cascading chains with regard to CO_2 emission reduction, land use and costs. In a case study, a wide variety of cascading chains of short rotation poplar have been compared in order to identify key parameters that influence the performance of biomass cascading chains and to quantify costs, land use and CO_2 emissions of the cascading chains.

Key methodological aspects of this study were the definition of system boundaries, the inclusion of (indirect) land use and the inclusion of a time dimension. The method of inclusion of land use allows comparing different utilisation strategies of biomass produced from one hectare of land. Other environmental impacts—apart from CO_2 emissions—that can be important in biomass production and material systems were beyond the scope of this study. Nevertheless, for a complete evaluation of different biomass cascading chains, a complete LCA approach would be appropriate.

5.1 Potential benefits of cascading

In general, cascading of biomass has the potential to improve annual CO_2 emission reductions per ha and to reduce CO_2 mitigation costs. This is the case if every biomass application in the cascading chain is advantageous compared to the reference application, because the advantages add up when several materials are produced in sequence from a biomass resource. Parameters that strongly influence the CO_2 mitigation costs and/or annual CO_2 emission reduction per ha of biomass production are market prices of materials, gross energy requirements of reference materials, performance of material production, characteristics of reference energy systems and yields of biomass cultivation.

Whether adding an additional application to a cascading chain increases or decreases net CO_2 emissions per ha of biomass production depends on the net CO_2 emissions of the respective application. These net CO_2 emissions depend on themselves on the carbon intensity of the reference system and may change in the future. A short cascading chain with high CO_2 emission reduction per hectare of the single application, e.g. pallet and electricity production, can be favourable compared to long cascading chains.

Some poplar applications result in economic benefits of CO_2 mitigation, while others result in costs. Therefore, the effect of prolonging cascading chains by additional applications on the CO_2 mitigation costs can also be both positive and negative.

Fraanje (1997) concludes from a qualitative study of cascading pinewood, that more cascading increases the efficiency of resource use in general. On the contrary, the results of this study show that, depending on the depreciation of CO_2 emission reductions in time and the kind of biomass applications, a shorter cascading chain can be more favourable in terms of CO_2 emission reduction per hectare and CO_2 mitigation costs than a long cascading chain.

5.2 Quantitative results of the case study

The performance of the different cascading chains considered in this study varies strongly. CO_2 mitigation costs range from benefits of 200 €/Mg CO_2 (MDF-EL) to costs of 2200 €/Mg CO_2 (LU-EL). Net CO_2 emissions per unit of land used for biomass production varies from reductions of 28 Mg CO_2/(ha*yr) (PA-MDF-ET-EL) to net emissions of 8 Mg CO_2/(ha*yr) (LU-LU-ET-EL).

Substituting wood products from conventional forestry by short rotation poplar materials, i.e. MDF board, pallets and pulp, is very attractive from a cost and CO_2 emission reduction point of view. MDF board, pallet and pulp production in combination with electricity production and the cascading chains of PA-LU-MDF-EL and PA-LU-LU-EL result in both net annual benefits and net annual CO_2 emission reductions per hectare (assuming that market prices of reference products are offset against production costs of poplar materials). Some cascading chains have net benefits in terms of CO_2 mitigation costs. On the other end of the spectrum are cascading chains (e.g. VI-EL and LU-LU-ET-EL) that result in CO_2 mitigation costs of more than 100 €/Mg CO_2 and that are not likely to be applied in the near future, according to IPCC (2001b).

The utilisation of SR poplar for materials before conversion to energy is mostly favourable compared to pure energy utilisation with regard to annual CO_2 emission reduction per ha of biomass production. Only for ethylene production this is not the case. On the other hand, with respect to CO_2 mitigation costs, pure energy utilisation can be favourable compared to cascading of SR poplar, depending on the applications.

Main uncertainties affecting the results are related to data on market prices of reference materials and production costs of SR poplar materials. First, market prices can vary significantly over time and between regions. Effects of these variations are among others large ranges of possible CO_2 mitigation costs and a change in ranking between the cascading chains; see Section 4.1. Second, production costs of biomass materials are often not publicly available. Consequently, these costs had to be estimated. As a result, CO_2 mitigation costs as calculated in this study are not exact, but represent an order of magnitude. Also, data on biomass production and the reference energy system influence the results strongly, as shown in the sensitivity analysis. The value of these parameters is very site-specific and varies between countries and regarding biomass production even between different locations within a single country.

5.3 Bottom-up analysis

In this analysis, CO_2 emissions and costs are calculated bottom-up. That means that the results are valid for a small-scale application of a cascading chain. If one of these cascading strategies, however, would be implemented on a large scale, then the resulting CO_2 mitigation costs would be different. Several mechanisms play a role in such large-scale implementations, i.e. economies of scale and dynamic markets. On the one hand, production costs decrease with larger scales and on the other hand, market prices of biomass materials decrease, while at the same time market prices of agricultural land increase.

Furthermore, we assumed for the materials that substitute wood products from conventional forestry that the alternative use for conventional forestry land would be short rotation forestry. While this gives a bottom-up estimate of the CO_2 emission reduction potential of a cascading chain, transaction costs of such a land use change would occur during implementation.[7] In summary, this study defines CO_2 mitigation costs on a bottom-up level assuming 'ideal' substitution without transaction costs. A next step would be to calculate the costs of large-scale implementation of cascading chains that have been identified as promising in our study.

5.4 Methodological framework

The method presented in this study is suitable to quantify the CO_2 emission reductions, costs and land use of biomass cascading systems, while such quantification has not been carried out before. As demonstrated, using a present value approach to evaluate costs and CO_2 emission reduction of different strategies over time highlights the possible impact of time on the attractiveness of specific cascading chains. This impact can be significant. For example, present values of CO_2 emission reduction differ up to 12 Mg CO_2/(ha*yr) from the undiscounted results. However, present values are not directly related to the impact of a unit of carbon dioxide emitted at a certain time on global warming. Nor are present values directly related to the rather uncertain developments of future market prices of CO_2 emissions. Thus, a widely accepted methodology to evaluate time aspects in CO_2 reduction by biomass material use still has to emerge from scientific as well as political discussion.

5.5 Further research

To gain further insight in the CO_2 emission reduction potentials of biomass cascading several aspects deserve further research:

- Possible effects of technological learning for long term cascading chains, in particular for material production systems
- Alternative crops and applications other than the applications of short rotation poplar regarded
- More complex cascading chains, including material utilisation of rest products from production processes and including logistics for the different cascading chains

[7] Apart from the transaction costs, there are other aspects (e.g. ecological) that need to be considered in order to evaluate possible land use changes from conventional forestry to short rotation forestry.

- Material and land prices, including elasticity on the market in order to determine CO_2 mitigation costs if biomass materials and non-food crops are introduced on a larger scale
- Costs and effects of land use changes from conventional forestry to short rotation forestry

APPENDIX 1: Application of short rotation poplar

A 1.1 Overview and selection

Short rotation poplar

Short rotation (SR) poplar is typically cultivated with rotation times of 3-4 years (Dix et al., 1993), while rotation times of up to 20 years are possible. Main characteristics of poplar wood from short rotations of 3-4 years are a relatively low wood density, relatively low strength properties, low lignin content, and relatively short fibres (Balatinecz et al., 2001). Typical medium yields in Germany range from 7-9 oven-dry Mg/(ha*yr) (Lewandowski, 2001).

Bulk materials

Bulk materials that can be substituted by short rotation poplar products are mainly round wood, paper & board, cement and plastics. These materials together represent two thirds of total current bulk material production (Phylipsen et al., 2002; UN/ECE and FAO, 2001). The total production of wood products in the EU amounted to 84 thousand Mg in 1999[1]. Particleboard, fibreboard and engineered wood products together have a market share of 44 % of all wood products. These products can be produced from short rotation poplar and are expected to gain a larger market share in the future (UN/ECE and FAO, 2001).

Construction applications

Sawn wood is mainly used as construction material. However, short rotation poplar wood is not suitable for most sawn wood applications due to its relatively low strength characteristics (Balatinecz et al., 2001).

Fibreboard consists of a small amount of bonding resins and pressed wood fibres gained in pulping processes. Main categories of fibreboard (in order of decreasing density) are hardboard, MDF (medium density fibreboard) and insulation board. In principle, these materials can be produced from SR poplar wood (Stolp et al., 1996). MDF is the most important fibreboard on the market, even though it is a relatively new product. Worldwide production capacity in 2001 was about $30*10^6$ m^3/year (UN/ECE and FAO, 2002). Main uses of MDF are furniture panels and non load-bearing construction applications. Because of its market importance and the good suitability of juvenile poplar as a raw material for its production (see Roffael et al., 1992; Dix et al., 1993) MDF is included in this study.

[1] i.e. $161*10^6$ m^3; converted by means of average densities of different wood products

The term *particleboard* comprises a broad variety of materials composed of wood particles and bonding agents that serve different purposes—e.g. material for furniture and interior walls—while *particle lumber* is employed for load-bearing construction applications. SR poplar wood is suitable for all these particle-based applications, even though because of swelling characteristics and reaction wood content[2], other wood species are added (see Balatinecz et al., 2001; Stolp et al., 1996; Samson et al., 1999). Particle lumber is selected for further study, because (1) particleboard including particle lumber is the most important wood product besides sawn wood and (2) particle lumber, as opposed to particleboard, can substitute concrete, which is a relative energy-intensive product (van Heijningen et al., 1992).

Plywood is a panel product made from wood veneers, i.e. thin wood sheets. It has either structural or decorative applications. However, as juvenile poplar tends to discolour (Balatinecz et al., 2001) and has low strength characteristics, it is not very suitable for this purpose. Consequently, plywood is not considered in this study.

Engineered wood products are load-carrying elements that are glued from veneer or smaller timber pieces. Their structural qualities are equal or superior to those of sawn wood. However, due to low strength characteristics, SR poplar is not very suitable for these applications. Thus, engineered wood products are not considered in this study.

Packaging

Wood is mainly used for transport packaging applications, i.e. crates and pallets, while few products are primary packed in wood products. Most wooden transport packages are *pallets* for either single or multiple uses. About $5 * 10^6$ Mg wood, mainly from poplar, spruce and pine, were used for pallet production in Europe in 1994 (Hekkert et al., 2000). Because of this large production volume, pallets are selected as packaging application for this study. However, because wood from a very short rotation is not suitable for pallet production, a rotation time of 10 years would be more appropriate for this application. (Implications of this longer rotation time on poplar production are discussed in Section 3.2)

Pulp and paper

A broad variety of paper and board products can be produced from pulp. Main pulping processes are mechanical, semi-chemical and chemical. Generally, SR poplar wood is suit-

[2] Wood with different properties that develops as a reaction to a tree being tipped from vertical.

able for all pulping processes. Of all chemical pulping processes, sulphate pulping is the most suitable process for poplar wood. Poplar pulps are often mixed with softwood pulps before papermaking, because poplar white chips have both advantages (e.g. lower lignin-content) and disadvantages (e.g. shorter fibres) compared to softwood (Balatinecz et al., 2001). In 1996, European pulp production for paper was 12.8*10^9 m^3 mechanical, 21.5*10^9 m^3 chemical and 1.5*10^9 m^3 semi-chemical (UN, 2002). Because of the high market volumes and the good suitability of juvenile poplar wood, chemical sulphate pulping is regarded in this study.

Chemicals

Most chemical products with a large production volume are so-called intermediates. In terms of production volume *ethylene* is the largest organic petrochemical intermediate. In Europe 17*10^9 m^3 were produced in 1997 (UN, 2002). Ethylene can be produced from SR poplar wood via ethanol fermentation, via methanol production from synthesis gas and a methanol-to-olefin (MTO) conversion, or via flash pyrolysis (Patel, 2000). None of these processes are currently commercially developed and applied. However, considerable efforts are undertaken to develop the technology to produce ethanol from wood via fermentation, while dehydration of ethanol to ethylene has been applied on an industrial scale for a long time. Moreover, the production of methanol from wood is quite well developed and experience with the MTO process on a demonstration scale is available (Patel, 2000). Flash pyrolysis has only been tested on a pilot scale.

Other main organic intermediates in terms of production volume are *propylene* and *benzene*. About 10*10^9 m^3 of propylene and 4*10^9 m^3 of benzene were produced in 1997 in Europe (UN, 2002). These chemicals can be produced by thermo-chemical conversion of wood, e.g. by pyrolysis (Klass, 1998), but this is not commercially developed (DCO, 1999).

Because of the high market volume and the better technological development stage than that of the production of other chemical main intermediates from biomass, *ethylene* production is selected for this case study, even though this process is not applied commercially. The MTO route is analysed, because (1) flash pyrolysis at the current state of the art is not a realistic option and (2) gasification allows using waste wood—which offers cascading possibilities—while fermentation does not. However, if wood fermentation follows the predicted developments, the fermentation route may become very attractive in terms of costs and efficiency (Lynd et al., 1996; Nossin et al., 2002).

Chemicals that, unlike the intermediates discussed above, are derived from existing chemical structures in wood can be produced from SR poplar as well. Most important in terms of production volume are *chemicals derived from cellulose,* of which production is well commercialised since many decades. The largest product category within this group is fibre with 6.0 $*10^5$ Mg production volume in Western Europe in 1999 (CIRFS, 2002), mainly viscose (i.e. cellulose fibre).[3] To include one of these more traditional chemical products from wood, viscose is selected for the analysis, too.

Energy
The main energy carriers that can be produced from wood are heat, electricity and transportation fuels. The attractiveness of producing *heat* is very dependent on local circumstances, i.e. heat demand, existing heat sources, etc. Therefore, this option is not regarded in this study.

Electricity can be generated using a broad range of conversion technologies like combustion, gasification and pyrolysis. Integrated gasification combined cycle systems (IG/CC) are expected to reach high electric conversion efficiencies and to become commercially available at relatively low costs (Faaij et al., 1998). Currently, this technology is in a demonstration stage. Because of the potential attractiveness, IG/CC is considered as energy conversion technology in this study.

Main *transportation fuels* that can be produced from wood are ethanol, methanol, hydrogen and Fischer-Tropsch gasoline. However, the production of ethanol from wood is still in a pilot stage and requires relatively clean source material. Fischer-Tropsch gasoline, methanol and hydrogen can be produced from synthesis gas from wood gasification and allow for the utilisation of waste materials. Methanol production has higher efficiencies and lower costs than the production of Fischer-Tropsch gasoline, while the use of hydrogen requires a special infrastructure (Faaij et al., 2000; Hamelinck and Faaij, 2002). Therefore, *methanol* is regarded in this study, although none of these gasification-based routes is applied commercially at present.

A 1.2 Recycling

Every one of the above selected wood products requires certain raw material properties. Consequently, not every application can be produced from any waste material. In this section, for every selected application of SR poplar wood, raw material requirements and possible cascading steps are described. For particleboard production, particles with a size

[3] Production of viscose is decreasing, while the production of synthetic yarns is growing.

of 5-15 cm are required. Even though some producers are critical about the use of waste wood, waste wood with little contamination, e.g. old board materials, is actually utilised for particleboard production (Boogardt, 2000). It is assumed that these specifications are valid for *particle lumber* production, too. Therefore, waste materials from which slightly contaminated waste wood with required size specifications can be derived, are assumed to be suitable for particle lumber production, i.e. pallets and particle lumber in this case.

The raw material for the production of *MDF boards* is wood fibres produced from particles with a size of 5-15 cm. Non-wood components in the raw material can influence the binding between resin and wood fibres, hence, wood without contamination of sand, preservatives, varnish etc. is required. Possibly, waste wood can be refined for MDF production (Boogardt, 2000). Therefore, it is assumed that waste pallets as well as particle lumber can be utilised for MDF board production.

The production of *pallets* requires large pieces of sawn wood that cannot be derived from waste materials of applications regarded in this study.

Pulp is mainly produced from wood chips or waste paper. Because only non-contaminated fresh wood is a suitable raw material for pulp production, the use of waste wood from other applications is not considered (Boogardt, 2000). The production of pulp from waste paper is also not taken into account, because the material from SR poplar and the reference material are assumed to be identical, namely, chemical pulp. If we would assume that pulp from poplar is processed into paper and then recycled, it would be consistent to assume that the pulp produced from softwood is used in a similar way. Both the poplar and the reference systems would then gain the same benefits with regard to CO_2 reduction and costs from the recycling steps. Consequently, including pulp production from waste paper would not change the result of the cascading chains investigated in this study.

For *viscose* production, the raw material is chemical pulp (Patel, 1994). As chemical pulp cannot be produced from waste wood, it is assumed that viscose cannot be produced from waste materials either.

The production of *ethylene* starts with the production of methanol; see Section 3.1. Thus, ethylene can be produced from the same waste materials as methanol. *Methanol* and electricity from IG/CC are derived from synthesis gas from gasification. Waste products from the applications regarded in this study are suitable raw materials for these processes.

Appendix 2: Input data

Table A 3.1: Input data on SR poplar and reference applications. (In cases where the biomass material and the reference material are identical, differences between these materials can be zero. Respective input data are marked as 'N/a')

	LU: *Particle lumber*	**Reference LU:** *Concrete*	**MDF:** *MDF board*	**Reference MDF:** *Plywood*	**PA:** *Pallets*	**Reference PA:** *Softwood pallets*	**PUL:** *Pulp*
Substitution factors and wood inputs							
Substitution [kg][a]	0.45[b]	1	1.40	1	1	1	1
Lifetime [years]	75[b]	75	10[c]	10	1[f]	1	1[e]
Wood input [kg_{wood}/kg][i]	-	-	-	2.2[j]	-	2.7[k]	-
SR poplar input [kg_{wood}/kg][i]	1.0[j]	-	1.4[m]	-	1.5[h]	-	2.0
Energy inputs and CO_2 emissions of material production							
Electricity input [MJ/kg]	0.34	-	0.63[q]	0.4	N/a	N/a	N/a
Steam input [MJ/kg]	1.95	-	0[q]	3.45[q]	N/a	N/a	N/a
Primary energy input [MJ/kg][t]	-	-	0.84	-	N/a	N/a	N/a
Energy raw materials [MJ/kg][u]	7.2	1.41	5.2	1.6	N/a	N/a	N/a
CO_2 emission [kg CO_2/kg]	0.75[w]	0.51[x]	0.51[y]	0.16[z]	N/a	N/a	N/a
Lower heating value for energy recovery							
Lower heating value [MJ/kg]	16.1	0	15.0[af]	13.5	N/a	N/a	N/a
Production costs and market prices							
Production costs [$€_{2002}$/kg][ah]	0.31[j]	-	0.11[ai]	-	N/a	N/a	N/a
Average market price [$€_{2002}$/kg]	-	0.11	-	0.93	N/a	N/a	N/a
Range of market price [$€_{2002}$/kg]	-	0.06-0.16[am]	-	0.89-1.12[an]	N/a	N/a	N/a

	Reference PUL: *Softwood pulp*	**ET:** *Ethylene*	**Reference ET:** *Ethylene*	**ME:** *Methanol*	**Reference ME:** *Gasoline*	**VI:** *Viscose*	**Reference VI:** *PES fibre*
Substitution factors and wood inputs							
Substitution [kg][a]	1	1	1	2.15	1	1.10	1
Lifetime [years]	1	2[f]	2	0[g]	0	5[h]	5
Wood input [kg_{wood}/kg][i]	2.7[k]	-	-	-	-	-	-
SR poplar input [kg_{wood}/kg][i]	-	14.3[n]	-	2.2[o]	-	2.9[p]	-
Energy inputs and CO_2 emissions of material production							
Electricity input [MJ/kg]	N/a	0[r]	0.12	-	-	8.63	2.3
Steam input [MJ/kg]	N/a	0[r]	- 0.36[s]	-	-	12.0	0
Primary energy input [MJ/kg][t]	N/a	1.16[r]	53.4	-	43.3	3.06	5
Energy raw materials [MJ/kg][u]	N/a	-	-	-	-	-	78.6
CO_2 emission [kg CO_2/kg]	N/a	0.09[m]	3.89[aa]	0[ab]	0.68[ac]	2.31[ad]	5.80[ae]
Lower heating value [MJ/kg]	N/a	N/a	N/a	43.3	19.9	36.5	17.4[ag]
Production costs and market prices							
Production costs [$€_{2002}$/kg][ah]	N/a	1.40[aj]	-	0.23[ak]	-	1.04[al]	-
Average market price [$€_{2002}$/kg]	N/a	-	1.20	-	0.38	-	0.23
Range of market price [$€_{2002}$/kg]	N/a	-	0.56-1.38[ao]	-	0.33-0.46[ap]	-	0.09-0.33[aq]

[a] The amount of biomass in kg that is needed to replace 1 kg of reference material.
[b] Goverse et al. (2001).
[c] Estimation for applications such as furniture, door panels etc.
[d] Estimation. Average use of a returnable pallet is 20 trips. (Hekkert et al., 2000)
[e] Estimation; paper from chemical pulp is often used for packaging.
[f] Estimation; plastic products.
[g] Use as transportation fuel.
[h] Estimation for applications such as clothing.

[i] Wood inputs are stated in kg of wood fresh matter. This is equivalent to 7% moisture content for SR poplar wood and 50% moisture content for softwood.
[j] Efficiency of sawn wood production.
[k] Haygreen and Bowyer (1996).
[l] Balatinecz et al. (2001).
[m] Own calculation based on hardboard production figures of Richter et al. (1995).
[n] Joosten (2001). The amount of ethylene produced per kg of wood is quite low, because during the catalytic conversion of methanol to ethylene large amount of by-products are produced. These by-products are mainly propylene and a C_4 fraction (butenes, butadiene, BTX).
[o] Equivalent to net efficiency of *methanol* of 55% (HHV) (Faaij et al., 2000). Analyses by Katofsky (1993) and Williams et al. (1995) concluded a net HHV efficiency of 54 to 58%. If methanol is produced from waste resources, higher costs and a lower efficiency are to be expected due to additional gas cleaning. However, these losses have not been quantified yet and could, therefore, not be included in the analysis.
[p] Data from NMMO (dissolution in N-methylmorpholine N-methyl oxides) process (Eibl et al., 1996).
[q] Part of the energy demand is supplied by wood residues.
[r] Data include allocation of energy inputs to different outputs on basis of their energy contents, see footnote n.
[s] Steam output due to combustion of by-products.
[t] These are primary energy inputs of fuels, e.g. gasoline, that are not converted to electricity or steam or are embodied in the raw materials.
[u] These are the gross energy requirement (GER) in MJ primary energy per kg of main material that are embodied in the raw materials of the application. Gross energy requirements are feedstock and process energy for the production of these raw materials and their subsequent raw material. (GER of short rotation poplar wood and softwood are not included, but are calculated separately.)
[v] CO_2 emissions are calculated from the energy inputs given above in this table, i.e. electricity, steam, primary energy and gross energy requirements of raw materials. Carbon emission factors used for these conversions are: electricity - 109 kg CO_2 /GJ (average OECD European electricity mix in 1999 (IEA, 2002a)); steam – 95 kg CO_2 /GJ (average energy mix of heat production (IEA, 2002a)); primary – 73.3 kg CO_2 /GJ (factor of crude oil, which is in between the factor of natural gas 56 kg CO_2 /GJ, and coal 95 kg CO_2 /GJ).
[w] CO_2 emission due to process energy use (181 kg CO_2/Mg referring to particleboard (Frühwald et al., 1997) and gross energy requirement of average glue consumption. However, weight percentage of glue per Mg can vary between 6-12%. (equivalent to 353 - 707 kg CO_2/Mg) (Haygreen and Bowyer, 1996) .
[x] Amount of cement (Portland), gravel, sand and iron replaced by the substitution of piles (Goverse et al., 2001) combined with GER values (Heijningen et al., 1992).
[y] Inclusive gross energy requirements of a resin content of 6,5% (Haygreen and Bowyer, 1996) and credits from black liquor utilisation for heat and electricity production in a traditional recovery boiler (Hekkert and Worrell, 1998).
[z] Forintek (1993). This value includes gross energy requirements of 2% resin content (Hekkert and Worrell, 1998). Heat production from wood rests from the production process is accounted for.
[aa] Fuels that result as by-product are accounted for as process energy. The remaining emissions are allocated to the different products of the process, i.e. ethylene, propylene, C_4, BTX, by means of energy content (Joosten, 2001).
[ab] It is assumed that energy needs for the production of methanol are supplied from the biomass raw material.
[ac] IEA (2002c).
[ad] Data from NMMO (dissolution in N-methylmorpholine N-methyl oxides) process (Eibl et al., 1996). It includes the utilisation of the by-product lignin as fuel (Heijningen et al., 1992). In comparison, traditional viscose production emits about 2376 kg CO_2 /Mg (Patel, 1994).
[ae] Gross energy requirement of PET (Heijningen et al., 1992) and the energy use for spinning (Patel, 1994). However, estimates of gross energy requirements of PET vary from 59.4 (Patel, 1999) to 78.6 GJ/Mg (Heijningen et al., 1992) and 88.5 GJ/Mg (APME, 1999).
[af] Refers to hardboard.
[ag] Wiley (1985).
[ah] Production costs are only given for SR poplar materials and do not include the costs of SR poplar production. Production costs are compared to the market prices of reference materials.
[ai] Calculated from wood input prices and MDF board export prices in Europe (FAO, 2004).
[aj] MTO process costs (Joosten, 2001) and methanol production costs (Hamelinck and Faaij, 2002).
[ak] Production costs, excluding biomass costs, are 10-13 $US\$_{2001}$/GJ (Katofsky, 1993).
[al] Own calculation from investment cost of 16.000 DM_{1993}/(Mg*yr) (Patel, 1994), a lifetime of 25 years and an interest rate of 5%.
[am] Average price for pre-cast concrete in the Netherlands between 1994 and 2000, while prices varied about 20% (CBS, 2002). In 1994 the Dutch price was in the middle range of European prices, which is given as range (Eurostat, 1997).

[an] European export prices decreased between 1990 and 1998 and then recovered slightly in 1999 and 2000 (FAO, 2004). Assumed is the market price of 2000, while the full range of the last 10 years is indicated (using an average density of 500 kg/m^3 for conversion).
[ao] Total market price of all cracker products varied between 1100 and 1400 €/$Mg_{ethylene}$ from 1997 until begin 2001 (CEFIC, 2002). Price of 2001 is assumed.
[ap] Equivalent to 0.28 €/l. European gasoline prices excl. taxes were 0.243 to 0.313 €/l in spring 2002 and 0.26 to 0.32 $US\$_{2000}$/l between 1998 and 2000 (IEA, 2002b).
[aq] Average EU price for man-made fibres in 2000 (Eurostat, 2000; CIRFS, 2002). No (historical) price data for PET fibres were available. Ranges are estimated by prices of non-spun PET, i.e. 93-103 €/Mg in 1998 and 1999 (CBS, 2002), and production costs of polyester yarns, i.e. 330 €/Mg (own calculation by investment costs and crude oil prices (Solantausta et al., 1997; Eurostat, 2000)).

Table A 3.2: Input data related to electricity production in an IG/CC

	Unit	Value
Electric efficiency[a]	%	43.5
Production cost[b]	$€_{2002}/GJ_e$	10.4[b]
Market price	$€_{2002}/GJ_e$	8.33, range: 5.55-16.67
CO_2 emission reduction[c]	kg CO_2 /GJ_e	109

[a] Based on large state-of-the-art IG/CC plant (about 150 MW_e) (Faaij et al., 1998).
[b] Own calculation from 1.97 million €/MW_e investment costs (Faaij et al., 1998) - a lifetime of 25 years, an interest rate of 5% rent and a load factor of 80%. According to Solantausta et al. (1997) and DOE (1997), investment cost for even smaller state-of-the-art plants of 62 and 75 MW_e are about the same.
[c] Average OECD European electricity mix in 1999 (IEA, 2002a).

Table A 3.3: Input data related to wood production of SR poplar and softwood

Parameter	Unit	SR poplar	Softwood
Yield	Mg_{wet}/ha*yr	8.6 (MC 7%) [a]	2.6 (MC 50%) [b]
CO_2 emission	kg CO_2/(ha*yr)	1570 [c]	296 [d]
Production costs	$€_{2002}$/(ha*yr)	441 [e]	343 [f]
Quality factor land	$ha_{agriculture}/ha_{forestry}$	N/a	0.66 [g]

[a] Lewandowski (2001).
[b] Moisture content (Hekkert and Worrell, 1998) and yield (Kaltschmitt and Reinhardt, 1997).
[c] Includes equivalents of N_2O emission from fertiliser use (Biewinga and van der Bijl, 1996).
[d] Kaltschmitt and Reinhardt (1997).
[e] Production cost data (Stolp et al., 1996). Costs include arable land rents. In 1998 arable land rents in France, the Netherlands, Belgium and Germany were between 128 and 338 €/ha, a non-weighted average of 220 €/ha is assumed here (Eurostat, 2000).
[f] Estimated on basis of average round wood prices in Europe of 48 $€_{2002}/m^3$ (FAO, 2004) and including a land rent that is derived from average rents and the quality factor comparing forestry land to arable land.
[g] Personal communication, I. Lewandowski (2002). Assumptions are (1) that poplar is cultivated, because short rotation coppice is most suitable for former forestry land and (2) that no extreme (with regard to water availability, slopes, etc.) forest areas are used. If forest areas would be converted to agricultural land, establishment costs and energy uses occur for stump clearing and soil amelioration, which can be significant. However, as these are one-off costs that do not represent the situation of a permanent land use chance, they are not taken into account.

<u>CHAPTER 4:</u>

Comparing the land requirements, energy savings, and greenhouse gas emissions reduction of bio-based polymers and bioenergy - An analysis and system extension of life-cycle assessment studies*

Abstract

This study compares energy savings and greenhouse gas (GHG) emission reductions of bio-based polymers with those of bioenergy on a per-unit of agricultural land-use basis by extending existing life-cycle assessment (LCA) studies. In view of policy goals to increase the energy supply from biomass and current efforts to produce bio-based polymers in bulk, the amount of available land for the production of non-food crops could become a limitation. Hence, given the prominence of energy and greenhouse issues in current environmental policy, it is desirable to include land demand in the comparison of different biomass options. Over the past few years, numerous LCA studies have been prepared for different types of bio-based polymers, but only a few of these studies address the aspect of land use. This comparison shows that referring energy savings and GHG emission reduction of bio-based polymers to a unit of agricultural land, instead of to a unit of polymer produced, leads to a different ranking of options. If land use is chosen as the basis of comparison, natural fibre composites and thermoplastic starch score better than bioenergy production from energy crops, whereas poly(lactides) score comparably well and poly (hydroxyalkaonates) score worse. Additionally, including the use of agricultural residues for energy purposes improves the environmental performance of bio-based polymers significantly. Moreover, it is very likely that higher production efficiencies will be achieved for bio-based polymers in the medium term. Bio-based polymers thus offer interesting opportunities to reduce the utilisation of non-renewable energy and to contribute to GHG emission mitigation in view of potentially scarce land resources.

* Published in: *Journal of Industrial Ecology*, 2003, **7** (3-4), 109-132; Co-authors: I. Lewandowski and M. Patel.

1 Introduction

Polymers, lubricants, surfactants, and solvents account for the largest share of synthetic organic material production in contemporary economies. Today they are almost exclusively produced from fossil feedstock (with the exception of surfactants); however, they could, in principle, also be produced from renewable feedstock. This chapter focuses on polymers, which represent about half of the total production of synthetic organic materials (excluding bitumen) (Patel et al., 1999). A wider use of bio-based polymers could thus become an important way to increase the use of renewable resources. Bio-based polymers are defined here as polymers that are fully or partially produced from renewable raw material. In the 1980s and 1990s, bio-based polymers began playing an increasingly important role in several applications. The environmental (and economic) performance of many bio-based polymers is likely to improve in the future as a result of technological progress and economies of scale; see e.g. Nossin et al. (2002) and Vink et al. (2003). The fact that environmental considerations have been, and will continue to be, an important motivation to develop and introduce bio-based polymers calls for a thorough comparative analysis of their environmental performance.

To this end, numerous life-cycle assessments (LCAs) have been prepared in the last few years for different types of bio-based polymers (Patel et al., 2003). Only a few LCA studies published in the scientific literature on bio-based polymers address the question of land use, however; most studies do not take it into consideration at all. In the first group of studies, the environmental impacts are mostly compared using the amount of bio-based polymer as a functional unit, for example, 1 kg of polymer. Some studies report the amount of land used but do not use it as a basis for comparison of different polymers.[1]

The inclusion of demand for land in such studies is desirable because considerable efforts are currently being made to produce bio-based polymers in bulk. In the longer term, this could result in a substantial demand for agricultural land. In view of land requirements for food production and policy goals to increase energy supply from biomass, the amount of available land for the production of nonfood crops is likely to become scarcer and more expensive. For example, Hoogwijk et al. (2003) estimated that depending on food demands, in 2050 about 0.4 to 3.2 Gha of agricultural and degraded land will be available for biomass production for energy and materials on a global scale, whereas the bio-materials

[1] Van den Broek et al. (2001) compared environmental impacts of three different land-use strategies, that is, organic agriculture, set-aside land, and bioenergy production. In their approach, system extension is used to provide the same amount of food and electricity and to use the same amount of land in all systems. With respect to acidification, energy carrier depletion, and climate change, the strategies producing bioenergy score best.

production in 2050 will demand about 0.4 to 0.7 Gha. Therefore, it will become increasingly important to maximize environmental benefits from the use of land, and this requires indicators of the impacts of biomass utilisation that reflect land use.

Given the prominence of energy and greenhouse issues in current environmental policy and because of the limited data that are available from LCA studies, this chapter only analyses nonrenewable energy use and greenhouse gas (GHG) emissions in relation to land use. We do, however, recognize that other environmental impacts, issues of sustainable agriculture, and economical feasibility are also important criteria for ranking options. For the sake of simplicity, "nonrenewable energy use" is also referred to as "energy use" in this chapter.

The fact that biomass can be used both for the manufacture of materials (here: polymers) and for the production of energy commodities raises the question which of the two options is more advantageous in terms of energy use and GHG emissions. The LCA prepared by Corbière-Nicollier et al. (2001) indicates that the production of polymers based on starch, kenaf, and china reed offer greater opportunities for energy saving and GHG mitigation than the production of bioenergy. In contrast, Kurdikar et al. (2001) argued, in the case of genetically modified corn stover, that the use of stover wastes as an energy source contributes more to GHG emission reduction than the production of PHA.

During agricultural production, large amounts of agricultural residues arise; for example, in the case of corn about 50% of the total dry matter is residue (known as "stover"). The utilisation of agricultural residues is usually not accounted for in LCA studies for bio-based polymers. This approach is often justified because agricultural residues are typically used for low-value applications within agriculture (e.g. animal food or soil improvement), however, more and more agricultural residues are now used for energy purposes (e.g. straw combustion for district heating, as is done on a large scale in Denmark). Given this trend, it is of interest to study to what extent the utilisation of agricultural residues for bioenergy generation could improve the environmental performance of bio-based polymers.

Whenever a process has more than one output, allocation issues may become important in LCA studies. Examples of such by-products in the case of bio-based polymers are proteins, glucose syrup, and vegetable oil. Allocation of impacts to the polymer versus the by-products can have significant effects on the calculated energy savings and GHG emission reduction for the bio-based polymer. Allocation is of particular interest for bio-based polymers because the byproduct streams can be relatively large and different approaches are chosen in various LCA studies.

In this chapter, we analyze LCA studies of various bio-based polymers and calculate for each study the respective energy savings and GHG emission reductions per hectare of land used for biomass production, thereby assuming the substitution of bio-based for petrochemical polymers. These benefits are then compared to the benefits of energy production from dedicated energy crops, hereafter referred to as "bioenergy." In other words, we use the term "bioenergy" only for the exclusive production of heat, electricity, or other mechanical power from agricultural crops. Moreover, this chapter assesses energy savings and GHG emission reductions per hectare resulting from the utilisation of agricultural residues for energy production. This also represents a form of bioenergy use but is to be exclusively referred to in the following as "residue use" or "residue utilisation" in order to avoid confusion with "bioenergy." The effect of the choice of different allocation methods and selected parameters on the results is analyzed in a sensitivity analysis.

2 LCA studies of bio-based polymers

In this assessment, we analyzed 11 LCA studies of bio-based polymers (Dinkel et al., 1996; Würdinger et al., 2002; Estermann et al., 2000; Vink et al., 2003; Gärtner et al., 2002; Gerngross and Slater, 2000; Heyde, 1998; Diener and Siehler, 1999; Wötzel et al., 1999; Pervaiz and Sain, 2003; Corbière-Nicollier et al., 2001). These cover thermoplastic starch (TPS), poly(hydroxyalkaonates) (PHAs), poly(lactides) (PLAs), and natural fibre reinforced composites. The studies compare bio-based polymers to petrochemical polymers either as raw material or as product. The comparison at the raw material level refers to one mass unit of primary plastics (i.e., granules, pellets), whereas the comparison at the product level refers to end products, such as moulded components for automobiles or blown films for packaging. The publications differ considerably in the amount of published background data and the degree of detail regarding explanations about methodology and results. A detailed description of the studies can be found in Patel et al. (2003).

Regarding system boundaries, some studies only analyze the process chain from cradle to factory gate, whereas other studies take a cradle-to-grave approach, thereby covering different types of waste treatment (e.g. incineration with or without energy recovery, recycling, composting, etc.). An important note is that the use phase has been excluded in all of the studies taken into account in this chapter. In this study, we decided to compare all bio-based polymers equally on a cradle-to-grave basis, including incineration without energy recovery in the waste management stage. This choice has been made in view of the fact that direct landfilling of untreated waste containing organic carbon will be prohibited in many industrialized countries, especially in Europe, in the near future (EC, 1999). Recycling, reuse, and other waste management options such as digestion and composting (the latter two are only relevant for biodegradable polymers) are still rarely used, and this has not appreciably changed in the last few years. As a consequence, in Europe waste incineration is likely to become the standard waste treatment technology that is applicable to all the bio-based and petrochemical polymers considered. The reasons for neglecting energy recovery are that not all facilities recoup and export energy and that energy recovery yields are in general still poor. Average energy conversion efficiencies of incineration plants in Europe have been estimated at about 12% heat and 12% electricity on a lower heating value (LHV) basis of the waste input (Phylipsen et al., 2002).

2.1 Non-renewable energy savings and GHG emission reduction

Non-renewable energy savings and GHG emission reduction per kilogram of bio-based polymers as presented in the LCA studies, and if necessary recalculated to a cradle-to-grave basis, including incineration without energy recovery, are shown in Table 4.1. This table allows us to compare the ranking of the different bio-based polymers with regard to energy savings and GHG reduction per kilogram of polymer to the ranking with regard to benefits per hectare of cultivated biomass. The type and amount of the substituted petrochemical polymers have generally been taken as given in the original studies. To ensure a consistent comparison of the LCA, data for petrochemical polymers, that is, low-density polyethylene (LDPE), high-density polyethylene (HDPE), and expanded polystyrene (EPS), were all taken from a single source, that is, the Association of Plastics Manufacturers in Europe (APME, 1999). These data sets, commissioned by the association and elaborated by the Boustead consultancy, are to our knowledge the most extensive and authoritative sources for LCA data on petrochemical polymers. Data for the conventional counterparts of natural fibres used in composites, that is, fibreglass and acrylonitrile butadiene styrene (ABS), were taken as given in the original LCA studies because no other authoritative data sets were available.

2.2 Inclusion of land demand and agricultural residues

As Table 4.2 shows, most studies did not take into account land demand in their assessment of environmental impacts. Furthermore, four studies did not indicate a reference crop yield per hectare on which the assessment is based. In the studies, different approaches are used to account for land demand. Gärtner et al. (2002) included land demand

Table 4.1: Summary of LCA study results: Energy savings and CO_2 emission reduction on a cradle-to-grave basis including polymer incineration without energy recovery per kg of bio-based polymer (negative values represent energy savings and GHG emission reduction by bio-based polymers relative to their petrochemical counterpart)

Bio-based versus petrochemical polymer				**Bio-based polymer**					**Petrochemical polymer**	
Type of polymers		*Non-renewable energy use [MJ/f.u.]*	*GHG emissions [kg CO_2eq/f.u.]*	*Functional unit (f.u.)*	*Product*	*Reference*	*Non-renewable energy use [MJ/f.u.]*	*GHG emissions [kg CO_2eq/f.u.]*	*Substituted amount per f.u.*	*Reference*
TPS	Vs. LDPE	-55.2	-3.90	1 kg	Pellets	Dinkel et al., 1996	25.4	1.14	1 kg	APME, 1999
TPS	Vs. LDPE	-24.5	-1.99	1 kg	Film	Dinkel et al., 1996	25.4	1.14	(150μm) 0.62 kg	APME, 1999
TPS[a]	Vs. EPS	-8.8	0.28	1 kg	Loose fills	Würdinger et al., 2002	18.9	1.10	(0.08 $m^{3)}$ 0.33 kg	APME, 1999
TPS[b]	Vs. EPS	-1.3	-0.76	1 kg	Loose fills	Estermann et al., 2000	36.5	0.37	(0.1 m^3) 0.45 kg	APME, 1999
PLA	Vs. LDPE	-23.6	-1.20	1 kg	Pellets	Vink et al., 2003[c]	57	3.84	1 kg	APME, 1999
PLA	Vs. PE	N/a	N/a	N/a	Pellets	Gärtner et al., 2002[d]	N/a	N/a	N/a	N/a
PHA	Vs. HDPE	1.1	N/a	1 kg	Pellets	Gerngross and Slater, 2000[e]	81.0	N/a	1 kg	APME, 1999
PHA	Vs. HDPE	-13.8	N/a	1 kg	Pellets	Heyde, 1998	66.1	N/a	1 kg	APME, 1999
Natural fibre/PP[f]	Vs. Fibreglass	-45.1	N/a	1 kg	Fibres[g]	Diener and Siehler, 1999	9.6	N/a	1 kg	Diener and Siehler, 1999
Natural. Fibre/EPS	Vs. ABS	-72	-1.0	1 kg	Automotive parts	Wötzel et al., 1999	89	5.1	1.37 kg	Wötzel et al., 1999
Natural. fibre/PP	Vs. Fibreglass /PP	-48.5	-2.8	1 kg	Composite	Pervaiz and Sain, 2003	47.9	2.8	1 kg	Corbière-Nicollier, et al. 2001
Natural fibre/ PP	Vs. Fibreglass /PP	-53.7	N/a	1 kg	Transport pallet	Corbière-Nicollier, et al. 2001	61.3	N/a	1.27 kg	Corbière-Nicollier, et al. 2001

[a] Including 13% polyvinyl alcohol.
[b] Including 15% polyvinyl alcohol.
[c] No reference polymer is specified in the original source (Vink et al., 2003). For the preparation of this table the petrochemical reference has therefore been assumed to be PE as in Gärtner et al. (2002). (For TPS applications, LDPE use has been assumed.)
[d] In this study, only aggregated values per ha of biomass cultivation are presented.
[e] In this study substitution of PET and PS is considered possible, too.
[f] PP: polypropylene.
[g] Fibres are utilized to reinforce PP. However, the functional unit of the study is the amount of natural fibre.

Table 4.2: Inclusion of land demand and key data on agricultural production in the LCA studies considered

Study	Type of polymer	Inclusion land demand	Crop	Country[a]	Crop yield *[Mg/(ha*yr)]*[b]	Crop input[a] *[kg crop /kg polymer]*
Dinkel et al., 1996	TPS	Energy and GHG savings calculated per ha	Potato, corn	CH	Potato: 37.5 (fm)[c], corn: 12.5 (dm)[d]	Potato: 2.23 + corn: 0.385
Würdinger et al., 2002	TPS	Impact category: natural area use	Corn	DE	Corn: 6.45 (dm)	0.786
Estermann et al., 2000	TPS	No	Corn	FR	Corn: 8.2 (dm)	0.971
Vink et al., 2003	PLA	No	Corn	US	Corn: 9.06 (dm)	1.74
Gärtner et al., 2002	PLA	Impacts calculated per ha	Corn	DE	N/a	N/a
Gerngross and Slater, 2000	PHA	No	Corn	US	Corn: 7.7 (dm)	5.06
Heyde, 1998	PHA	No	Sugar beet	DE	N/a	N/a
Diener and Siehler, 1999	Flax/PP	No	Flax	DE	N/a	N/a
Wötzel et al., 1999	Hemp/EPS	No	Hemp	DE	N/a	0.49
Pervaiz and Sain, 2003	Hemp/PP	No	Hemp	CA	Hemp: 2 (dm)	0.65
Corbière-Nicollier et al., 2001	Miscanthus/PP	Energy savings calculated per ha	Miscanthus	CH	Miscanthus: 17-20 (dm)	0.75

[a] International internet country codes.
[b] Yields and amounts of plants needed refer to the typically used crop part, that is, potato - tubers, corn - grain, sugar beet – beet, hemp – fibres and miscanthus – above ground biomass.
[c] Fresh matter (fm).
[d] Dry matter (dm), refers to plant as harvested with a moisture content of about 12%.

Table 4.3: Summary of assumptions about utilisation of agricultural residues and by-products in the LCA studies considered

Study	Crop	Residue	Accounted for residue?	By-products at	Accounted for by-products?
Dinkel et al., 1996	Potato, corn	Foliage, stover	No	Starch production	Impacts allocated to energy content
Würdinger et al., 2002	Corn	Stover	Reduced fertiliser need	Starch, semolina production	Impacts allocated to economic value
Estermann et al., 2000	Corn	Stover	No	Starch production	Impacts allocated to energy content
Vink et al., 2003	Corn	Stover[a]	No, but inputs allocated to stover and grain	No details given	Unknown
Gärtner et al., 2002	Corn	Stover	No	Starch production	Substitution of sunflower products
Gerngross and Slater, 2000	Corn	Stover	No	Glucose production	Impacts allocated, no details given
Heyde, 1998	Sugar beet	Leaves	Unknown	Unknown	Unknown
Diener and Siehler, 1999	Flax	Stalks	Unknown	Unknown	Unknown
Wötzel et al., 1999	Hemp	None	No	Fibre preparation	Unknown
Pervaiz and Sain, 2003	Hemp	None	No	Fibre preparation	No
Corbière-Nicollier et al., 2001	Miscanthus	None	No	Fibre preparation (grinding)	Penalty for energy use of disposal

[a] Stem, husks, leaves, etc.

in their analysis by calculating environmental impacts per hectare of biomass cultivation. Corbière-Nicollier et al. (2001) and Dinkel et al. (1996) reported the environmental impacts per kilogram of bio-based polymers and the energy savings per hectare. Moreover, Dinkel et al. (1996) also calculated the GHG emission reductions per hectare. Würdinger et al. (2002) applied the concept of "natural area demand," where land is categorized into different classes of natural quality.

Table 4.3 summarizes whether and how agricultural residues and by-products from the material production process were accounted for in the analysis. Although many studies accounted for by-products from crop processing and polymer production, only Würdinger et al. (2002) and Vink et al. (2003) also considered the use of agricultural residues.

Only Gärtner et al. (2002) assumed that the by-products substitute for equivalent products originating from other production processes (i.e., products from sunflowers); the other authors distributed the environmental impacts among the products ("allocation" in the strict sense). Concerning agricultural residues, only Würdinger et al. (2002) and Vink et al. (2003) accounted for their potential value. Würdinger et al. (2002) assumed that corn stover, which is usually left on the field, substitutes for artificial fertiliser. Vink et al. (2003) allocated a small part of the biomass production impacts to the residues but did not specify the basis of this allocation.

3 Methodology

Because the different studies deal in very different ways with land use and agricultural residues, we related energy savings and GHG emissions to land demand in a consistent way and, moreover, extended the system boundary to include the use of agricultural residues for energy production. To address land use, the area of medium-quality agricultural land occupied for biomass production is used as a functional unit. Different biopolymers can then be compared with regard to their environmental performance per unit of (possibly scarce) agricultural land. Other important functions of land, for example, erosion pre vention and habitat, are outside the scope of this study. Schemes for the systems studied are presented in Figure 4.1 and are explained below in more detail.

3.1 Energy savings and GHG emission reduction without utilisation of residues

To determine energy savings of bio-based polymers without residue use, the non-renewable energy use for the production of a bio-based polymer (left box in Figure 4.1, top) is compared to the non-renewable energy use for the production of a (functionally equivalent) petrochemical polymer (right box in Figure 4.1, top). Energy use within the system includes direct energy inputs for crop production, crop processing and polymer production (process energy and feedstock energy), and indirect energy inputs that are energy inputs for the supply of materials needed for production, for example, machines and fertilisers. These energy requirements lead to GHG emissions. Moreover, non-CO_2 GHG process emissions (N_2O and CH_4) that mainly result from agricultural crop production are also taken into account. Data on these energy uses and GHG emissions were taken from the LCA studies reviewed (Table 4.1).

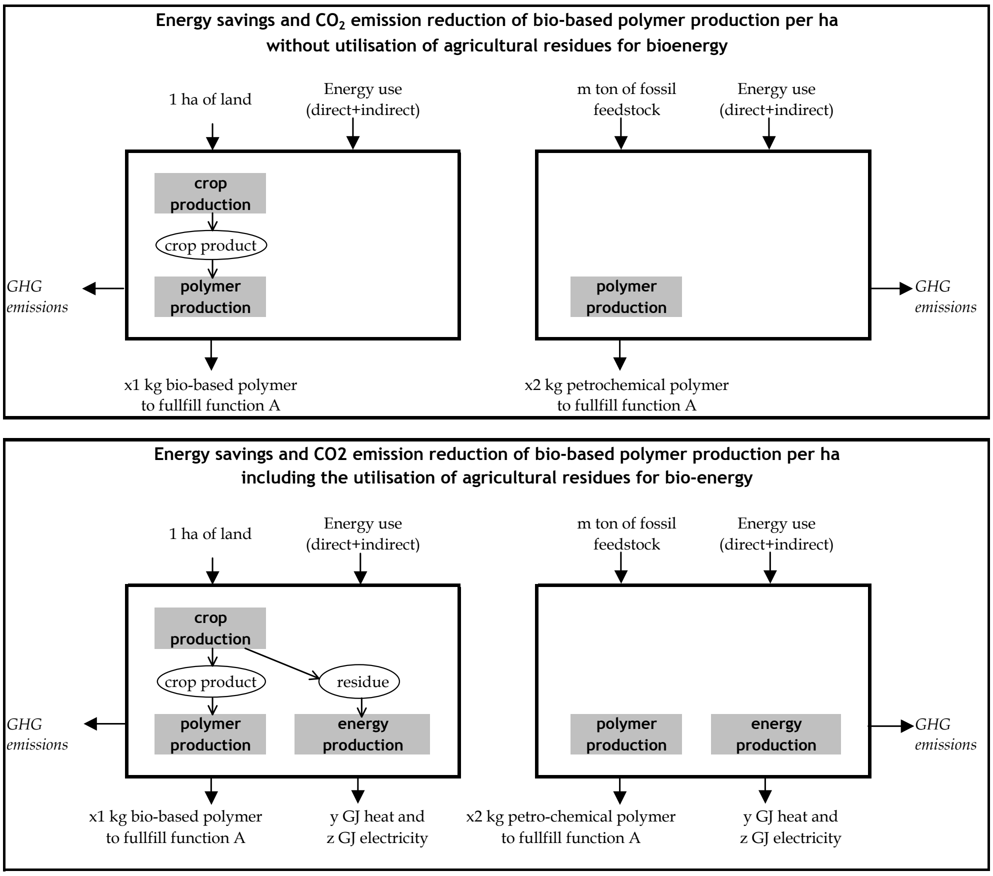


Figure 4.1: Systems studied to determine energy savings and CO_2 emission reduction of bio-based polymer production per hectare of agricultural land.

The amount of bio-based polymer (*x*1) that can be produced from 1 ha of biomass cultivation substitutes for an amount of petrochemical polymer (*x*2) that can fulfill the same function (e.g. a certain amount of pellets, volume of loose fill, or an automotive component). The quantities *x*1 and *x*2 can be the same, but this is not necessarily the case. For example, less favourable material properties for a given thermoplastic bio-based polymer may result in the need for more material (mass) than its petrochemical counterpart in order to fulfill the same function. This may occur for starch polymers in certain applications. On the other hand, natural fibre composites are typically lighter than the substituted material, that is, fibreglass composites.

The results of environmental impacts caused by the production of bio-based polymers as found in the 11 LCA studies can be expressed per unit of land demand (in hectares). This approach has been applied earlier by Gärtner et al. (2002), Corbière-Nicollier et al. (2001) and Dinkel et al. (1996). To correct for regional differences in agricultural productivity, we recalculated the key results of the LCA studies using a uniform crop yield drawn from the agricultural literature. This value is computed as the average of the "medium yield" shown in Table 4.4 (hereafter referred to as "medium yield"). We thereby assume that the LCA studies refer to good-practice agricultural production methods on land of average fertility, as no exceptional cases have been mentioned. In practice, however, yields per hectare can vary significantly depending on local conditions, for example, climate and soil quality. The influence of this yield variation on the energy savings and GHG emission reduction per hectare of land is analyzed in a sensitivity analysis.

Even if total land requirements were compared, the land use in the petroleum-based reference systems would be negligible. Therefore, land requirements are taken into account only for the bio-based polymers (Figure 4.1). If a bio-based polymer is compared to another bio-based polymer or bioenergy, however, the land demand for each of these bio-based products is 1 ha. Alternative use of the land in question as set aside or for food production is not considered here.

3.2 Energy savings and GHG emission reduction by utilisation of residues

If the agricultural residues are used for energy production, then this energy is assumed to substitute heat and power from an average energy mix in Europe (see Figure 4.1, bottom). The GHG emission reduction is determined on the same basis.

In practice, agricultural residues often remain on the field. When studying the effect of residue removal on energy use and GHG emissions, three aspects need to be taken into account: (1) the withdrawal of nutrients from the field, (2) the energy requirements and the emissions related to the collection of the residues on the field and their transportation to the energy conversion installation, and (3) the conversion of residues to secondary energy that substitutes for fossil energy.

The amount of agricultural residue can be calculated on the basis of the yield of the crop utilized for bio-based polymer production and the harvest index, that is, the proportion of different crop components, as found in the agricultural literature (see Table 4.4). The amount of nutrients removed when residues are recovered is determined by assuming typical residue nutrient content in terms of nitrogen (N), phosphorous (P), potassium (K), and calcium (Ca). Next, we assume that synthetic fertilisers must be used to replace the withdrawn nutrients and that the environmental impacts of fertiliser production must be accounted for in the analysis. Energy use and related GHG emissions during harvest and transportation of the agricultural residues are based on the necessary machinery use and average transport distances. Furthermore, for every agricultural residue a representative commercial conversion technology has been defined: Relatively dry agricultural residues (10% to 50% moisture content) are assumed to be combusted in small sized combined heat and power (CHP) plants. Wet agricultural residues (50% to 90% moisture content) are digested on a small scale, and the biogas is converted to heat and power by a gas engine. The heat and electricity produced from agricultural residues is assumed to replace heat and electricity from an average European energy mix (see Table 4.8) because most LCA studies considered were prepared for Europe.

3.3 Comparison with bioenergy

Finally, the replacement of fossil fuels by either bio-based polymers or bioenergy is compared on the basis of land area units. This is done by comparing energy savings and GHG emission reduction of bioenergy production from dedicated energy crops to the results from bio-based polymers. For bioenergy, a selection of ethanol, CHP, and power production processes has been considered. Energy savings and GHG emission reduction of bioenergy relative to respective reference energy commodities (i.e., gasoline, heat, and power) were taken from the published literature.

In detail, for ethanol production from corn and short-rotation woody crops (SRWCs), production efficiencies were taken from an energy balance for the United States and Europe, respectively (Shapouri et al., 2002; Faaij et al., 2000). The amount of ethanol produced per hectare is converted to energy savings and GHG emission reduction by the substitution of gasoline on an LHV basis. For ethanol from sugar beet and CHP from perennial grasses, energy savings and GHG emission reduction per hectare are derived from a German LCA, where the energy carriers were compared to an average energy mix (Kaltschmitt and Reinhardt, 1997). For power production in a biomass-integrated gasification system with combined cycle fuelled by SRWCs, power generation efficiencies are derived from (Faaij et al. 1998). The power produced is converted to energy savings and GHG emission reductions by substitution of the average European electricity mix.

4 Agricultural production and energy use of agricultural residues

This section presents all complementary data that we used for our calculations of the environmental impacts per area of land used and the use of agricultural residue for energy production (as described in Figure 4.1).

The different yield classes (low, medium, high) as shown in Table 4.4 refer to Germany, being representative of the temperate climate in central Europe.[2] The medium-level yield has been used in our calculations.

Table 4.4: Yield classes of crops and agricultural residues (Hydro Agri Dülmen, 1993; Scheer-Triebel et al., 2000; Lewandowski et al., 2000; Karus et al., 1996; Corbière-Nicollier et al., 2001)

Crop[a]	**Medium yield** *[Mg/(ha*yr)]*	**Low yield** *[Mg/(ha*yr)]*	**High yield** *[Mg/(ha*yr)]*	**Residue**	**Proportion** [*Crop : Residue*]
Potato (tubers, fm)	35 – 50	25 - 35	50 – 60	Foliage (fm)	5:4
Corn (grain, dm)	6.5 - 8	5 – 6.5	8 - 9.5	Stover (dm)	1:1.3
Sugar beet (beet, fm)	50 - 70	40 – 50	70 - 80	Leaves (fm)	4:3
Flax (fibres, dm)	1.4 - 1.6	1 – 1.4	1.6 – 2	Straw (dm)	1:4
Hemp (fibres dm)[b]	1.5 - 2.25	1.25 - 1.5	2.25 - 2.5	Straw (dm)	1:4
Miscanthus (dm)	12 - 20	6 – 12	20 - 30	Grinding res. (dm)	1:0.3

[a] fm = fresh matter, dm = dry matter

[b] For hemp, yield classes were determined from average yields (Karus et al., 1996) assuming same proportions as for flax.

In general, we assumed that 100% of the agricultural residues can be removed; however, the final effect of residue removal on soil fertility is a complicated issue and is the subject of intense debate at the moment (see discussion). Table 4.5 presents the nutrient and energy content of residues. Because fibre plants, that is, flax, hemp, and *Miscanthus*[3], are usually harvested as a whole crop and no residues are left on the field, no credits for avoided fertiliser use have been introduced for these plants. The energy contents given in Table 4.5 are stated as an LHV if the residues are combusted directly and an LHV of biogas production if the residues are digested.

---

[2] These yields are also possible in parts of the United States and Canada with a very broad range of climatic conditions and consequently widely varying crop yields. The studies concerning bio-based polymer production in the United States and Canada (Vink et al., 2001; Gerngross and Slater, 2000; Pervaiz and Sain, 2003) do not specify yields for agricultural production, however. For comparability reasons, the same yield data have been used to calculate environmental impacts per hectare of land demand in this chapter.

[3] *Miscanthus* is a tall perennial grass that is widely being investigated as a bioenergy crop.

Table 4.5: Characteristics of agricultural residues: Average nutrient and energy content (Hydro Agri Dülmen, 1993; LWK 2002; Kuhn, 1995; Kaltschmitt and Reinhardt, 1997; Kaltschmitt and Hartmann, 2001)

Residue	**Nutrient content** [kg/Mg residue (fm)]				**Water content** [%]	**Energy content**
	N[a]	*P*	*K*	*Ca*		
Potato foliage	2.4	0.44–0.87	4.1-5.8	19.9	75	15.8 $GJ_{LHV\text{-}biogas}/Mg_{dm}$
Corn stover	4.2	2.18–3.06	12.4–20.7	3.6–5.0	14	15.7 GJ_{LHV} /Mg_{dm}
Sugar beet leaves	1.7	0.35–0.48	3.3-5.8	5.0–10.0	84	12.9 $GJ_{LHV\text{-}biogas}/Mg_{dm}$
Flax straw	N/a	N/a	N/a	N/a	N/a	16.9 GJ_{LHV}/Mg_{dm}
Hemp straw	N/a	N/a	N/a	N/a	N/a	15.6 GJ_{LHV}/Mg_{dm}
Miscanthus straw	N/a	N/a	N/a	N/a	N/a	16.9 GJ_{LHV}/Mg_{dm}

[a] Amount of nitrogen that is available for plant growth and can replace fertiliser if agricultural residues are left on the field, this is about 60% of the total content.

The energy use and GHG emissions due to the removal of agricultural residues are summarized in Table 4.6. Energy use and GHG emissions due to synthetic fertiliser were derived from research by Kaltschmitt and Reinhardt (1997) and were combined with the nutrient contents in Table 4.5. Residues of hemp, *Miscanthus,* and flax are usually removed from the field. Therefore, no extra energy use or emissions were included for their removal.

To calculate impacts related to the collection of residues, machine hours and fuel use for a large field of approximately 40 ha were taken from the research of Kaltschmitt and Reinhardt (1997). We have assumed that the collection of corn straw is comparable to the collection of wheat straw. Machine hours for clearing and collecting sugar beet leaves were derived from research by PAV (2000). No data were available for the harvesting of potato foliage because it is a very uncommon operation. As potato foliage is relatively wet and has to be cleared and collected from the field like sugar beet leaves, the same process has been assumed.[4]

Energy use and GHG emissions related to transportation have been derived from the fuel use of trucks per megagram and kilometre with an empty return. Data were taken from research by Kaltschmitt and Reinhardt (1997). Sugar beet leaves and potato foliage were assumed to be utilized in a small-scale digestion facility converting the biogas (mainly CH_4) to heat and power in a small-scale gas engine. Hence, truck capacity and transportation distances are assumed to be small, that is, 7.5 Mg and 15 km. Corn straw is assumed to be utilized for CHP generation in a medium scale combustion facility (ca. 20 MW thermal

[4] Potato foliage can be cleared mechanically, chemically, or thermally. Thermally, foliage is burned on the field and is consequently not available for utilisation. Chemically treated foliage is quite dry, but contaminated with herbicides. Therefore, a mechanical clearance is assumed here.

input on an LHV basis). Given the necessity of transport from several farms, the results for corn stover in Table 4.6 are based on a medium truck capacity of 23 tonnes and an average transportation distance of 50 km. Finally, residues of flax, hemp, and *Miscanthus* occur in large quantities at fibre processing plants. Thus, the truck capacity for transport of these residues is the largest possible, that is, 40 tonnes. We assumed that the CHP plant for utilisation is located relatively nearby (15 km).

Table 4.6: Non-renewable energy use and GHG emissions due to the removal of 1 Mg of agricultural residue (Kaltschmitt and Reinhardt, 1997; own assumptions)

	Non-ren. energy use [MJ/Mg residue]				**GHG emissions [kg CO_2eq /Mg residue]**			
	Fertiliser[a]	*Harvest*[b]	*Transport*[c]	*Total*	*Fertiliser*[a]	*Harvest*[b]	*Transport*[c]	*Total*
Potato foliage	168.0	58.5	65.4	292	25.8	4.3	4.9	35
Corn stover	345.3	61.1	112.2	519	39.7	4.5	8.3	53
Sugar beet leaves	122.8	58.5	65.4	247	16.6	4.3	4.9	26
Flax straw	N/a	N/a	16.1	16	N/a	N/a	1.2	1
Hemp straw	N/a	N/a	16.1	16	N/a	N/a	1.2	1
Miscanthus straw	N/a	N/a	16.1	16	N/a	N/a	1.2	1

[a] Penalty to account for the removal of nutrients together with the residue.
[b] Collection of the residue on the field.
[c] Transportation from the field to a decentralized energy conversion facility.

Table 4.7 shows the average efficiencies of the energy conversion plants that were assumed for the utilisation of agricultural residues. Table 4.8 presents average European Union data (for 1998) for conventional power production. For the production of heat, the replacement of small central heating systems (50% oil and 50% gas) was assumed. For comparison, the respective values for the United States are 2.62 GJ/GJ_e and 205 kg CO_2 equivalent/GJ_e. At the other extreme, in Switzerland only about 1.31 GJ/GJ_e and 11 kg CO_2 equivalent/GJ_e are substituted due the country's high share of hydroelectric and nuclear power (UBA, 2003). Characteristics of bioenergy production are presented in Table 4.9. Note that some of these processes use only part of the crops and thus do not include residue utilisation, whereas other processes use the whole crop.

Table 4.7: Energy conversion efficiencies for the utilisation of agricultural residues (Rösch and Wintzer, 1997; Kaltschmitt and Reinhardt, 1997)

Conversion technology	**Residues**	**Net efficiency, power** *[%]*	**Net efficiency, heat** *[%]*
Digestion	Potato, sugar beet	25 (biogas LHV)	55[a] (biogas LHV)
Combustion	Corn, hemp, flax, miscanthus	18 (biomass LHV)	64 (biomass LHV)

[a] Own estimate: total efficiency of gas engine 80%.

Table 4.8: Conventional heat and power production assumed to be substituted by energy from agricultural residues (based on UBA, 2003)

Non-renewable energy use, power *[GJ/GJ_e]*	**Non-renewable energy use, heat** *[GJ/GJ_{th}]*	**GHG emission factor, power** *[$kgCO_2eq./GJ_e$]*	**GHG emission factor, heat** *[$kgCO_2eq./GJ_{th}$]*
2.48	1.36	126	87

Table 4.9: Characteristics of processes of bioenergy production compared to bio-based polymer production

Bioenergy	Crop	Crop yield *[Mg/(ha*yr)]*	Technology	Conversion efficiency
Ethanol[a]	Corn (grain)	7.3 (dm)	Fermentation	86% of max. ethanol[h]
Ethanol[b]	Sugar beet (beet)	56.2 (fm)	Fermentation	86% of max. ethanol
Ethanol[c]	SRWC (whole plant)[e]	10.0 (dm)	Pretreatment + ferment.	46% ethanol 4%power
CHP[b]	Grass (whole plant)[f]	16.7 (dm)	Combustion + steam cycle	18% heat, 64% power
Power[d]	SRWC (whole plant)[g]	10.0 (dm)	BIG/CC	43% power

[a] (Shapouri et al., 2002). This is a cradle-to-gate energy balance for U.S. ethanol production plants. Values per ha are estimated with the average yields used in this study.
[b] (Kaltschmitt and Reinhardt, 1997). Data are taken from this complete bioenergy LCA.
[c] (Faaij et al., 2000). Net energy yields of crops and energy conversion rates are converted by our own calculation with the energy factors from Table 8 and a substitution factor of 74.1 kg CO_2/GJ ethanol derived from gasoline.
[d] (Faaij et al., 1998), calculation see comment c.
[e] SRWC = short rotation woody crop, fast growing trees, for example, poplar, willow, with a rotation time of usually 3-4 years.
[f] Fast growing perennial grasses, for example, miscanthus, switchgrass.
[g] BIG/CC = Biomass integrated gasification with combined cycle.
[h] Maximum ethanol yield is the complete conversion to ethanol; value corresponds to an assumed starch content of 64%.

5 Results

Figure 4.2 shows the annual energy savings and GHG emission reductions per hectare for bio-based materials (polymers and composites) relative to their petrochemical and mineral counterparts. The results both with and without the utilisation of agricultural residues are presented. The figure also shows the net benefits of using bioenergy instead of conventional (mainly fossil) energy commodities.

5.1 Energy savings and GHG emission reduction per hectare

Non-renewable energy savings of the different bio-based polymers on a per-hectare basis have a broad range from about 1,100 to -2 GJ/(ha yr), with the highest value achieved by *Miscanthus* transportation pallets. On the other hand, PHA and TPS loose fills offer comparatively low non-renewable energy savings of less than 30 GJ/(ha yr).

The outstanding performance of the *Miscanthus* composite can be explained by the very high yield of *Miscanthus* and the large part of the harvested biomass (ca. 70%) that is usable for composite production. By contrast, in the case of hemp and flax, only about 25% of the harvested biomass (i.e. the fibres) is utilised in composite production.

For PLA, two studies (a and b) are considered. Energy savings per hectare that result from these studies differ considerably. In the case of the first study (Vink et al., 2003), based on the current production technology of Cargill Dow, the energy savings per hectare have

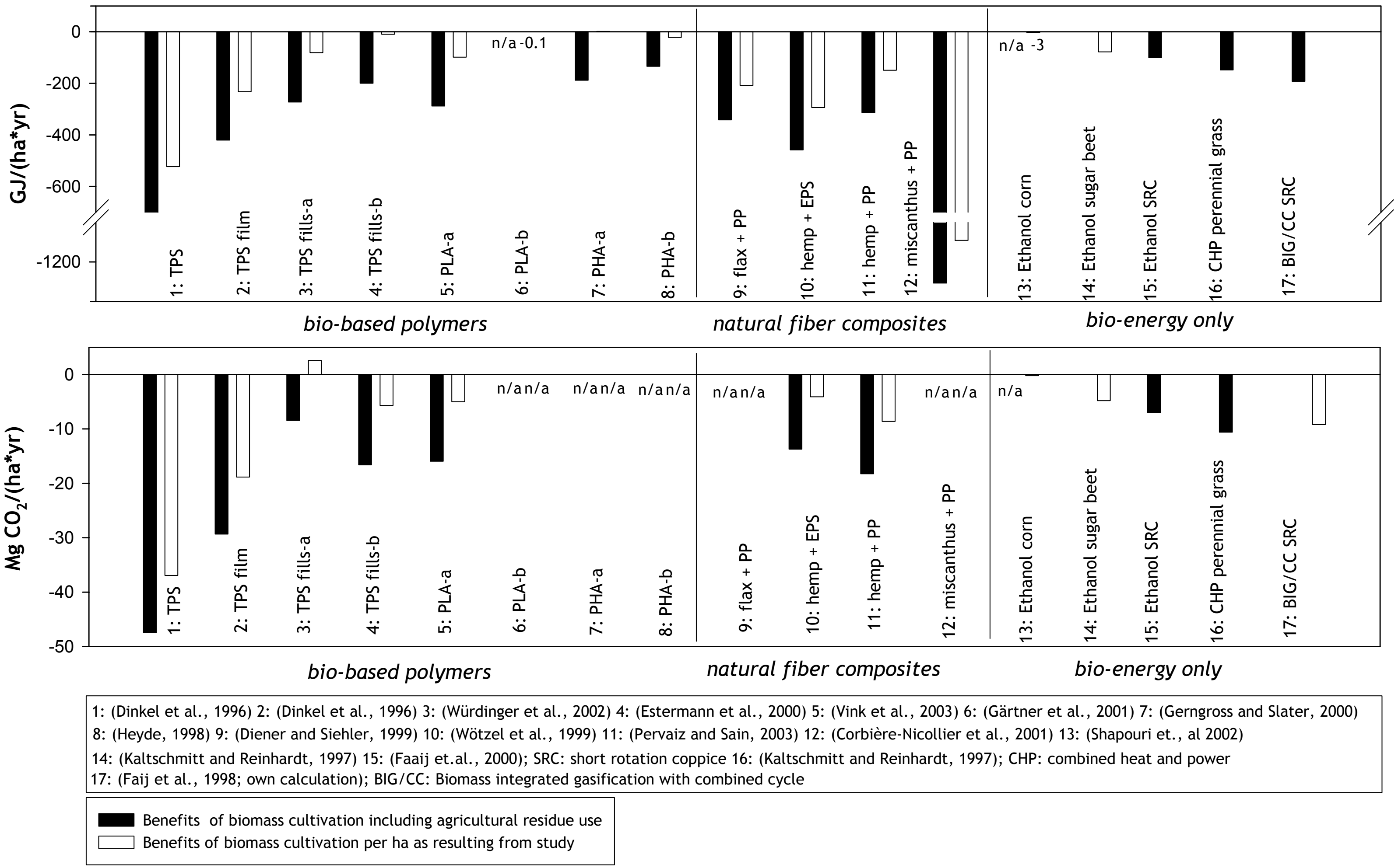


Figure 4.2: Non-renewable energy use and GHG emissions of bio-based polymers relative to their fossil counterparts, cradle to grave including incineration without energy recovery. The results are presented per hectare of biomass cultivation with medium crop yields as achieved in central Europe. Negative values represent energy savings or GHG emission reductions compared to use of fossil-fuel-based alternatives.

been calculated as explained in the methodology. In the case of the second study (Gärtner et al., 2002), only aggregated energy savings per hectare were published, with very little additional information. Therefore, it is not possible to explain whether the differences relative to Vink et al. (2003) are as a result of methodological or empirical discrepancies, for example, lower crop yields.

As the differences between the grey and black bars in Figure 4.2 show, the utilisation of agricultural residues increases the non-renewable energy savings per hectare considerably. Additional benefits are in the range of up to 190 GJ/(ha yr). For the production of PLA and PHA, these benefits are even larger than the benefits of polymer production.

In Table 4.10, the bio-based polymers are ranked on the basis of energy savings per hectare and on the basis of energy savings per kilogram of bio-based polymer, both with and without residue use for energy production. The table shows that the rankings computed on these bases differ significantly. This is due to two factors: (1) the polymer yield (kilogram polymer per kilogram of biomass input) differs for the different kinds of bio-based polymers (Table 4.1), and (2) the polymers are produced from crops with very different yields (Table 4.4).

Miscanthus transportation pallets, which already have quite high savings per kilogram, score even better than hemp/EPS composites and TPS with regard to energy savings per hectare both with and without energy use of residues. Also, all TPS materials improve their relative position when ranked by savings per hectare. PHA from sugar beet (Heyde, 1998) scores less well on a per hectare basis compared to a per-kilogram basis.

Table 4.10: Comparison of ranking of energy savings of bio-based polymers on basis of area of biomass production and kilogram of bio-based polymer[a]

Bio-based polymer[b]	**Ranking with residue use**		**Ranking without residue use**	
	*Based on GJ/(ha*yr)*	*Based on GJ/kg*	*Based on GJ/(ha*yr)*	*Based on GJ/kg*
1: TPS pellets	2	5	2	2
2: TPS film	4	9	4	6
3: TPS fills-a	8	10	8	9
4: TPS fills-b	9	11	10	10
5: PLA-a	7	7	7	7
7: PHA-a	10	1	11	11
8: PHA-b	11	4	9	8
9: flax + PP	5	6	5	5
10: hemp + EPS	3	2	3	1
11: hemp + PP-a	6	3	6	4
12: miscanthus + PP	1	8	1	3

[a] See Table 4.1 for energy savings per kg of bio-based polymers without residue use.
[b] Legend of references, see Figure 4.2.

The difference in results between the area-based versus the mass-based approach is particularly large if agricultural residues are used for the production of heat/electricity (Table 4.10). This drastically improves the reported environmental performance if expressed relative to one mass unit of bio-based polymer. This can be observed, for example, in the PHA production from corn case (see line 7 of Table 4.10). The amount of corn needed to produce 1 kg of PHA is relatively high. Consequently, the amount of agricultural residue and the amount of energy that can be produced from it are quite high as well. At the other extreme, for *Miscanthus*-reinforced polypropylene (PP), only small amounts of *Miscanthus* are needed, and hence only small amounts of residues are generated. That is, relative to other biopolymers, less fuel is created per kilogram of bio-based polymer. Therefore, *Miscanthus*-reinforced PP scores relatively poorly relative to other biopolymers when compared on a per-kilogram basis with the utilisation of residues for energy included. Thus, a mass-based metric may lead to misinterpretation in the case of systems with large energy use of residues.

The levels of GHG emission reduction from the bio-based polymers relative to their petrochemical counterparts also change significantly when computed on a per-hectare basis. Values range from about -37 to 3 Mg CO_2 equivalent/(ha yr) without including the utilisation of residues for energy. Fewer results are available for GHG emission than for renewable energy use. The highest GHG emission reductions per hectare are achieved by TPS pellets and TPS films. The utilisation of residues reduces carbon emissions further to around 47 Mg CO_2 equivalent/(ha yr). The small number of materials for which GHG emission data are available makes a comparison of rankings between the surface-based and the mass-based approach less significant. As a consequence, the results are not discussed here.

5.2 Comparison of bio-based polymers and bioenergy only

The benefits, that is, energy savings and GHG emission reductions, per hectare from exclusive bioenergy production based on dedicated energy crops and the subsequent substitution for fossil fuels are shown on the right-hand side of Figure 4.2. Non-renewable energy savings per hectare of biomass cultivation can be up to a factor of 6 higher for the production of bio-based polymers (*Miscanthus* composites) than for the production of bioenergy. This is because most bio-based polymers replace petrochemical polymers that are more energy intensive than fossil energy generation. Not all bio-based polymers achieve higher benefits per hectare than bioenergy, however. Including residue utilisation, both records for PHA and PLA as calculated by (Gärtner et al. 2002) still have lower energy savings per hectare than bioenergy. Contrary to this, according to (Vink et al., 2003), PLA without residue utilisation is comparable to bioenergy production. Similarly, whereas TPS loose fills score better than bioenergy if the residues are utilized, they score worse without residue use.

Comparing bioenergy and bio-based polymers on the basis of GHG emission reduction per hectare produces different results than the comparison based on energy savings. This is probably because of the use of different carbon emission factors for energy consumption, that is, different emission factors for primary energy, electricity, and heat (units of kg $CO_{2\,eq}$/GJ) in the various studies of bio-based polymers. Including residue utilisation, all bio-based polymers, except for TPS loose fills (Würdinger et al., 2002), result in GHG emissions reductions comparable to bioenergy applications. Without agricultural residue utilisation, PLA, hemp-EPS, and all TPS loose fills have lower GHG emission reductions than bioenergy production. No GHG emission data are available for PHA.[5]

For ethanol, Figure 4.2 shows a wide range of values both for (non-renewable) energy and (fossil) GHG emissions (see numbers 13 to 15). The benefits of corn-based ethanol in the United States (based on Shapouri et al. (2002)) are very low compared to all the other bioenergy production routes. This finding corresponds to a net energy value of 21,000 Btu/gal. Furthermore, Shapouri et al. (2002) compared nine other studies of ethanol production in which net energy values ranged from-34, 000 to 30,000 Btu/gal. Based on reported data, ethanol from sugar beet (data refer to western Europe) is clearly better than corn-based ethanol. In the longer term, even larger benefits might become available by making use not only of the starch yield from corn (Figure 4.2, number 13) but also of the ligno-cellulosic crop parts, that is, the corn stover. In the United States, current major research and development (R&D) projects are focusing on this option; for example Dale (2002). Energy efficiencies of ethanol production from stover are about 49% with an additional co-production of 5% electricity (Atherton et al. 2002). If methods of using ligno-cellulosic plant components are successfully developed, fast-growing SRWCs could also be used as a feedstock for ethanol production. Yields of SRWCs and corn stover are comparable, amounting to about 10 Mg/(ha yr). For the advanced production of ethanol from SRWCs, Faaij et al. (2000) estimated energy yields of about 53% ethanol and 8% electricity. This would allow savings on the order of about 130 GJ/(ha yr) and 9 kg CO_2 equivalent/(ha yr). Comparing these values to the energy savings and GHG emission reductions

[5] Kurdikar et al., (2001) investigated the production of PHA from genetically modified corn. The agricultural residue, that is, stover, is used for PHA extraction, and the process waste is converted to energy. This results in an emission reduction of about 5.4 Mg CO_2 equivalent/(ha yr), which is comparable to the lower results for TPS loose fills but lower than bioenergy production.

of bio-based polymers as presented in Figure 4.2, it can be concluded that the production of most polymers from biomass is more advantageous than advanced bio-ethanol production with pretreatment, that is, conversion of lignocelluloses to fermentable sugars from SRWCs or corn stover.

In general, energy savings of bioenergy production are limited by crop yields. For a high-yield crop such as *Miscanthus*, average yields in central Europe are about 270 GJ/(ha yr). In an ideal situation, biomass can substitute for fossil fuel on a 1:1 basis[6], which leads to energy savings of about 270 GJ/(ha yr). The energy savings related to bio-based polymers can exceed this value as the results for TPS, hemp-EPS, and *Miscanthus*-PP composites in Figure 4.2 show. This is due to the fact that the energy requirements (i.e., feedstock and process energy) for petrochemical polymers can be much higher than for the corresponding bio-based polymers.

6 Sensitivity analysis

Measures of biopolymer performance such as energy savings and GHG emission reduction per hectare of biomass cultivation are highly sensitive to crop yield. Crop yields vary depending on local conditions and agricultural practices. Whereas medium crop yields were used for the calculations discussed above, sensitivity analyses using high and low crop yields are now presented.[7] Accounting for the possible range of yields due to different agricultural conditions in Western Europe changes the overall result by a factor of 2, as shown in Table 4.11. This range of results exceeds by far the maximal benefits from agricultural residue utilisation. Even with low crop yields, however, *Miscanthus* pallets and TPS pellets are still much better in terms of energy savings than bioenergy production, whereas the performance of other TPS products, PLA, and hemp/flax composites is comparable to that of bioenergy production when low crop yields are assumed.

Another uncertain factor is the amount of agricultural residues that can be removed from the field. The mechanisms determining the final effect of residue removal on soil fertility are very complex and are currently the subject of debate (Sheehan et al., 2002). For the results in Figure 4.2, we assumed that 100% of the agricultural residues can be removed without any long-term adverse effects on soil fertility, provided that the lost nutrients are

[6] Even slightly higher substitution rates are possible if biomass is used as solid fuel in a more efficient energy conversion process than the reference.
[7] In this analysis, only the crop yields are varied to account for site difference, but no variations due to site-specific crop production methods are taken into account.

replaced by artificial fertiliser. Kurdikar et al. (2001), however, argued that only 60% of corn stover can be removed or soil quality decreases. Reduction in the fraction of residue recovered results in a proportionate reduction in the added benefits of residue recovery relative to the non-recovery cases as shown in Figure 4.2.

Table 4.11: Sensitivity analysis of benefits of biopolymers per hectare including utilisation of agricultural residues with different crop yields per hectare[a]

	Non-renewable energy use (GJ/ha*yr)			**GHG emissions (Mg $CO_{2\,eq}$ /(ha*yr)**		
	Base case [b]	*With low crop yield*[c]	*With high crop yield*[c]	*Base case* [b]	*With low crop yield*[c]	*With high crop yield*[c]
1: TPS pellets	-711	-452	-966	-51	-32	-69
2: TPS film	-420	-267	-571	-33	-21	-45
3: TPS fills-a	-273	-188	-357	-12	-9	-16
4: TPS fills-b	-200	-138	-262	-21	-14	-27
5: PLA-a	-288	-199	-378	-20	-14	-26
7: PHA-a	-188	-130	-247	N/a	N/a	N/a
8: PHA-b	-134	-89	-179	N/a	N/a	N/a
9: Flax + PP	-342	-228	-455	N/a	N/a	N/a
10: Hemp + EPS	-458	-286	-573	-17	-11	-21
11: Hemp + PP-a	-313	-196	-392	-22	-13	-27
13: Miscanthus + PP	-1253	-470	-2350	N/a	N/a	N/a

[a] References for the information in this table can found in Figure 4.2.
[b] Including the use of agricultural residues, results as presented in Figure 4.2.
[c] Low and high crop yields are the extremes of yield classes given in Table 4.4.

Apart from data uncertainties, methodological choices made when preparing an LCA study also need to be considered. In the case of multi-product processes, the allocation of impacts to a certain product and its by-products can have significant effects on the environmental performance calculated. As mentioned earlier, the allocation can be based on mass flows, energy contents, monetary values, or other indicators. Although not all of the LCA studies specify exactly the procedure applied, it is still obvious that different allocation methods have been used (Table 4.4).

Lack of information on the exact allocation procedure did not allow us to recalculate the results using a common approach (this proved not to be a limiting factor for the conclusions, as is discussed below). Therefore, the original results as calculated in the respective studies were used. Moreover, for the sake of simplicity and comparability, we allocated all land use to the bio-based polymers. It is interesting, however, to consider the consequence of applying a more detailed allocation procedure that allocates parts of the land used for biomass cultivation to the various by-products. In this case, the benefits per hectare would be even greater than those reported in Figure 4.2. Hence, our simpler approach can be regarded as conservative, tending to underestimate the true benefits.

This is shown in Table 4.12, which presents the results if the total area of land is allocated to the different by-products according to the allocation methods that were used in the respective study. These methods are either (1) no allocation (i.e., all impacts are assigned to the bio-based polymer being the main product), (2) allocation on the basis of energy contents, or (3) allocation based on economic values of products and by-products (Table 4.3). Table 4.12 contains only those bio-based polymers for which the allocation method used in the original study is known. The allocation key used for TPS pellets and films by Dinkel et al. (1996) assigns the least impacts of starch production to the bio-based polymer. If the same allocation procedure is applied to the area of land used, the energy savings and GHG emission reduction per hectare increase by about 70% for TPS pellets and TPS films (rows 1 and 2 of Table 4.12). For TPS loose fill, the sensitivity to the allocation of hectare of biomass cultivation to by-products is lower (0% to 25%).

Moreover, the kind of allocation method used to assign life-cycle flows to by-products also plays a role. Table 4.13 shows the energy savings and GHG emission reduction for TPS pellets with different allocation methods. The results can differ by about a factor of 2. The by-products associated with other bio-based materials were not sufficiently well known to perform a similar sensitivity analysis for those materials.

Table 4.12: Sensitivity analysis of benefits of biopolymers per hectare with and without allocation of area of biomass cultivation to by-products within the production process

	Energy savings (GJ/ha*yr)		**GHG emission reduction (Mg $CO_{2\,eq}$ /(ha*yr)**	
	Base case without agricultural residues use, no allocation of ha	*Results including allocation of ha to by-products*	*Base case without agricultural residues use, no allocation of ha*	*Results including allocation of ha to by-products*
1: TPS pellets	-523	-888	-37	-63
2: TPS film	-232	-392	-19	-32
3: TPS fills-a	-81	-100	3	3
4: TPS fills-b	-10	-10	-6	-6

[a] References for the information in this table can found in Figure 4.2.

Table 4.13: Energy savings and GHG emission reduction of TPS (Dinkel et al. 1996) relative to its petrochemical counterpart with different allocation methods (applied to energy use, GHG emission and area of biomass cultivation)

	Energy savings (GJ/ha*yr)				**GHG emission reduction (Mg $CO_{2\,eq}$ /(ha*yr)**			
	No allocation	*Energy content*	*Economic value*	*Mass*	*No allocation*	*Energy content*	*Economic value*	*Mass*
1: TPS pellets	-460	-888	-580	-944	-34	-63	-42	-67

7 Discussion

7.1 Methodological differences between the LCA studies

This chapter discusses the results of various LCA studies of bio-based polymers with one surface unit of land (1 ha) as the basis of comparison. The LCA studies used for the analysis presented here vary considerably both in scope and detail. Although the LCAs of Würdinger et al. (2002) and Dinkel et al. (1996) are very detailed, other studies are less explicit about methodology and data used. Moreover, the studies differ with regard to the methodologies used and also contain several data uncertainties.

In this chapter, we standardized several assumptions. First, to ensure a uniform basis for comparison, we used one single data source for LCA data on petrochemical polymers (APME, 1999), although the data used in the studies differ considerably.[8]

Second, we assumed common crop yields that represent medium values for western and central Europe. Yields can vary considerably, even within the same geographic region, and sensitivity analyses show that the influence of variation in yield on the savings per hectare is very significant. In the case of the United States, the average crop yields can differ considerably from those in central Europe, but average corn yields in the United States, that is, 7.6 Mg/(ha yr) (Shapouri et al., 2002), are more or less comparable to the medium yields in central Europe that we assumed, that is, 7.3 Mg/(ha yr). Many European studies assume a crop yield slightly above or below the medium yields we assumed here, however. The advantage of assuming uniform crop yields as done in this chapter is that the different bio-based polymers can be compared among one another with regard to efficient land use.[9]

Third, we chose to compare only one method of waste management within the cradle-to-grave system boundaries, that is, waste incineration without energy recovery. Some of the LCA studies also include a variety of waste management technologies. Although it can be very useful to evaluate bio-based polymers in a specific waste management regime appropriate for a specific polymer or a specific region, it is more appropriate to assume uniform waste treatment for this initial comparison. A further investigation of the influence of waste management technologies and especially of those predetermined for biodegradable polymers, that is, composting and digestion, is desirable, however.

---

[8] For a comparative overview of life-cycle energy and CO_2 data for chemical products across various sources, see research by Patel et al. (2003), and for a discussion of the influence of inventory data sets on LCA results, see the work by Peereboom et al. (1998).

[9] This is justified because we suppose that all of the studies assume comparable agricultural production methods.

On the other hand, it was not possible to make a correction for all methodological differences found in the LCA studies. Because the fuel mix for power generation is country specific, the studies assume different carbon emission factors for electricity, which has not been corrected. Another important aspect in this context is that different allocation methods have been applied in the LCA studies. This can have a substantial impact on the results (Table 4.13). Moreover, it was not possible to allocate land demand to the by-products, and therefore the benefits per hectare of land tend to be underestimated (Table 4.12). The lack of data made it impossible to apply one uniform allocation method. The results presented in this study provide a first conservative estimate of the performance of bio-based polymers on a per-hectare basis. Further research in this context is necessary.

Patel et al. (2003) carried out an analysis of 20 studies on bio-based polymers containing most of the studies considered in this chapter. In spite of different approaches, end products, allocation methods, and system boundaries of the LCA studies reviewed, the results of this meta-analysis show a uniform picture for different bio-based polymers. We are therefore confident that the energy savings and GHG emission reduction per hectare calculated from the original study results per kilogram of bio-based polymers provide a reliable overall picture.

7.2 Removal of agricultural residues

A few more aspects have to be considered when interpreting the results. First, the data used to calculate the penalties for residue removal are uncertain. This is because several assumptions had to be made, that is, machine use for harvest, amount of nitrogen in the residues that is available to plants, and the transportation distances to the energy conversion facility. Total penalties do not play an important role on the overall energy balance, however, as they are less than 3% of the energy savings per hectare due to agricultural residue utilisation.

Much more important, and controversial, are the amounts of residues that can be removed without significant losses in soil fertility. As the base case, we assumed 100% removal. This reflects agricultural practice for certain crops; for example, fibre crops are typically harvested as whole plants (excluding roots) and sugar beet leaves are sometimes used as fodder. Kurdikar et al. (2002) assumed in their study on bio-based polymers from genetically modified corn in the United States, however, that only 60% of the residues can be harvested. On the other hand, Sheehan et al. (2002) argued that soil carbon would only slightly decrease (less than 3 metric tonnes of carbon equivalent per hectare [MTCe/ha] in 100 years) if corn stover were completely removed. Without stover removal, soil carbon would increase (about 30 MTCe/ha in 100 years) and the soil would be rebuilt. According to (Linden et al., 2000), it depends on local climate and soil conditions whether corn stover removal decreases or increases corn yields; however, if residues are well incorporated into the soil, and fertiliser application is adjusted, leaving residues on the field would generally have positive effects on soil fertility. We consider the removal of all residues to be a good base-case assumption to indicate maximal benefits from agricultural residue utilisation. Nevertheless, it should be noted that this removal is not possible in all cases, not only for soil fertility reasons but probably also because of the extra costs incurred for collection and logistics. As shown in the sensitivity analysis, this can considerably decrease the savings per hectare of agricultural land use. Therefore, a more detailed study of the local agricultural circumstances and the implications of residue removal today and in the long term is necessary in order to determine the possible benefits of residue utilisation in specific settings.

7.3 Technological development

A successful development of pretreatment technology, where pretreatment refers to conversion of lignocelluloses to fermentable sugars, would not only offer new opportunities for bio-ethanol production but also for bio-based polymers. For example, Cargill Dow intends to use pretreated lingo-cellulose for PLA production in the future. This is expected to clearly improve the environmental performance (Vink et al., 2003); however, the overall environmental performance of pretreatment may depend on whether and how genetically modified organisms are used.

Moreover, it should be emphasized that the material properties, and hence the possible applications of the bio-based polymers covered in this chapter, are not comparable. Because of their sensitivity to moisture and other disadvantageous material properties, starch polymers have a more limited range of applications than PHA and PLA. This generally limits starch polymers to niche applications. Regarding natural fibre composites, the studies considered cover products that are in different stages of development. Although the use of hemp and flax composites has been successfully adopted by the automotive sector, the technical feasibility of *Miscanthus* transportation pallets has not yet been proven.

7.4 Secondary savings

Finally, it needs to be pointed out that the use phase has been excluded from this analysis. Depending on the application area, however, so-called "secondary savings," that is, savings during the use phase, can be very important. These are especially important for natural fibre composites used for transportation applications, as they are typically lighter than their glass-fibre-reinforced counterparts. For example, secondary savings associated with the *Miscanthus* pallets can be about 90% of the energy required of their production if a transport distance of 5,000 km is assumed (Corbière-Nicollier et al., 2001). Thus, the inclusion of secondary savings could increase the energy savings and GHG emission reduction per hectare of natural fibre composites considerably.

8 Conclusions

In this chapter, non-renewable energy savings and GHG emissions reductions per kilogram of bio-based polymers (relative to the petrochemical polymers that they replace) are compared to the same savings computed on the basis of the area of land used for biomass production. These benefits are based on state-of-the-art production data. Comparing biopolymers on the basis of energy savings and GHG emission reduction per hectare of biomass cultivation changes their ranking relative to a ranking referring to 1 kg of bio-based polymer. *Miscanthus* composites and TPS pellets rank higher, whereas PHA, PLA, hemp composites, and flax composites are comparatively less good. For the polymers studied, most changes in the ranking are moderate, although some changes are quite large.

The utilisation of agricultural residues can increase the benefits per hectare of biomass cultivation significantly (by about 190 GJ/(ha yr) and 15 Mg CO_2 equivalent/(ha yr)) but does not change the ranking of biomass polymers. Depending on local circumstances (soil, climate, etc.) that influence soil carbon contents and economic considerations, however, it might not be possible to remove 100% of residues, and the benefits of residue utilisation would thus be lowered by the percentage of residues that remain in the field. Further research is required regarding the amount of residue that can be removed under a sustainable regime.

Referring energy savings and GHG emission reduction of bio-based polymers to a unit of agricultural land used, instead of to a unit of polymer produced, can lead to a different ranking of options (Table 4.10). Therefore, referring these benefits to a unit of land area provides additional insights into the performance of bio-based polymers. Moreover, it can help to optimize the use of agricultural land if it is scarce and expensive. A consistent allocation method is needed to more accurately compare bio-based polymers and should be used in future research.

If compared on a hectare basis and without residue utilisation, most bio-based polymers score better in terms of energy savings and GHG emission reduction than bioenergy production from energy crops. This is clearly the case for natural fibre composites and TPS pellets and films. Energy savings and CO_2 emission reduction for PLA on a per-hectare basis are in the range of the benefits for bioenergy production and are worse than bioenergy applications only in the case of PHA. If compared on a per-hectare basis *with* residue utilisation, even the benefits of PHA production are in the range of the benefits of bioenergy production. Bio-based polymers such as PLA are in an early stage of commercial development, however, and PHA is just about to reach the stage for bulk polymer applications. On the other hand, many bioenergy technologies have already reached commercial status.

Therefore, in the medium to long term, technological progress will therefore most likely lead to higher efficiency gains for bio-based polymers than for bioenergy production, and as a consequence, this would also result in higher energy savings and GHG emission reductions. This has mainly to do with technological progress, the long process chain for bio-based polymers (compared to bioenergy), and developments in waste management. We therefore conclude that the production and use of bio-based polymers offer very interesting opportunities to reduce the utilisation of non-renewable energy and to contribute to GHG mitigation. Furthermore, the amount of land that can be used for the production of non-food crops is limited and might not be sufficient to supply the total energy demand (Turkenburg et al., 2000). Energy in the form of fuels, electricity, and heat can also be supplied by renewable sources other than biomass, although chemical feedstock cannot. For this reason, the production of bio-based polymers seems a good strategy to reduce the overall use of non-renewable energy, provided that it can compete on economic terms.

CHAPTER 5:

Economics and GHG emission reduction of a PLA bio-refinery system – Combining bottom-up analysis with price elasticity effects*

Abstract

This chapter analyses energy savings, GHG emission reductions and costs of bio-refinery systems for poly lactic acid (PLA) production. The systems include multi-functional use of biomass resources, i.e. use of agricultural residues for energy consumption, use of by-products, and application of recycling and waste-to-energy recovery. We evaluate the performance of these systems per kg of bio-based polymer produced and per hectare of biomass production. The evaluation is done using data of Poland, assuming that biomass and PLA production is embedded in a European energy and material market. First, the performance of different bio-refinery systems is investigated by means of a bottom-up chain analysis. Second, an analysis is applied that derives market prices of products and land depending on the own-price elasticity of demand. Thus, costs of the bio-refinery systems depending on the demand of land and material are determined. It is found that all PLA bio-refinery systems considered lead to net savings of non-renewable energy consumption of 70 to 220 GJ/(ha*yr) and net GHG emission reductions of 3 to 17 $MgCO_{2eq}$/(ha*yr). Most PLA bio-refinery systems considered in this study lead to net costs of the overall system of up to 4600 €/(ha*yr). PLA production from short rotation wood leads to net benefits of about 1100 €/(ha*yr) if a high amount of a high value product, i.e. fibres, is produced. Multi-functionality is necessary to ensure the viability of PLA bio-refinery systems from biomass with regard to energy savings and GHG emission reduction. However, with regard to costs, the multi-functional use of biomass does not contribute much to overall incomes. Own-price elasticity of the demand for materials influences the overall costs of the bio-refinery system strongly. The own-price elasticity of land demand markets could become important if bio-refineries or biomass production in general are introduced on a large scale.

---

* Co-authors: A.P.C. Faaij, M. Patel and W.C. Turkenburg.

1 Introduction

Biomass is suitable for many material and energy applications, e.g. construction materials, chemicals, liquid fuels or electricity. One of the reasons to investigate the potential use of biomass for such applications is the reduction of greenhouse gas (GHG) emissions—especially CO_2—that goes along with the replacement of fossil fuels by biomass, provided that the biomass is produced in a sustainable way. In this context, the potential savings of non-renewable fuels, the resulting GHG emission reductions and the associated costs are key indicators to evaluate the effectiveness of biomass use. This proves to be a complex issue in which many aspects play an important role, e.g. the choice between energy and materials application, the reference systems that are replaced by biomass systems and the time-frame of the analysis; see chapter 3 of this thesis. In addition, if biomass is to be used in large volumes to contribute significantly to GHG emission reduction, it needs to be cultivated requiring relatively large areas of agricultural land. Consequently, the amount of agricultural land used for biomass production is a decisive parameter as well.

Given the importance of low costs and high efficiency of land use and GHG emission reduction, optimising the performance of biomass systems is vital for successful deployment at large scale. Many options to improve the performance of biomass systems deal with parts of the system, for example, the development of more efficient energy crop production systems or increasing the energetic efficiency of conversion technologies. However, multi-functional use of biomass resources can be applied to optimise biomass systems too, e.g. by multi-product use and cascading. Multi-product use is defined using different parts of biomass resources for different applications. These applications can cover a broad range of materials and a broad range of energy carriers. Cascading is defined as using biomass resources sequentially for several material applications ending typically with an energy conversion step to fuels, heat and/or electricity.

A particular type of multi-functional biomass system, namely, the so-called 'bio-refinery', has recently received much attention in literature. This term is derived from petroleum refineries that maximise the economic benefits by obtaining a diverse mix of products from the feedstock. The concept of 'bio-refinery' is not well defined though and it can comprise various processes such as production of fodder from a crop (Annetts and Audsley, 2003) or pyrolysis of biomass (Scott et al., 1997). In literature, bio-refinery is e.g. defined as the production of various products with innovative technology (Benjamin and Weenen, 2000) or the production of fuels and chemicals (Elliot, 2004) from biomass resources. In general, bio-refineries are considered to produce a range of products, see e.g. DTO (1997), Elliot (2004), NRC (1999), Wyman (2003) and Benjamin and Weenen (2000). Many definitions explicitly include the production of chemicals or fuels in analogy to the petrochemical refinery (e.g. Scott et al., 1997; Elliot, 2004; Wyman, 2003).

Bio-refineries as referred to in this study combine two aspects of multi-functional biomass systems. The first aspect is the use of parts of a whole crop to produce different products and the second aspect is cascading, if products of the bio-refinery are recycled (usually by so-called down-cycling) and their wastes are finally converted to energy carriers.

It has been indicated in earlier studies that multi-product use and cascading may improve the efficiency of biomass utilisation with regard to GHG emission reductions, energy savings and costs (chapter 2 and 3 of this thesis). Wyman (2003) has carried out an economic analysis of a bio-refinery producing fuels and chemicals from sugar, identifying possible benefits of multi-product use. These studies are focused on parts of bio-refinery systems, i.e. multi-product use or cascading. If the parts involved would be incorporated in a more complex bio-refinery system, as outlined above, the overall performance may change compared to the simple sum of performances. For example, material recycling might on the one hand reduce process energy demand of material production. On the other hand, this reduced process energy demand might lead to excess heat produced from biomass residues that cannot be used if there is no additional heat demand in the vicinity of the facility. If biomass materials are recycled, one might assume that respective reference materials will also be recycled, leading to different benefits of the biomass system compared to a “cradle-to-factory-gate” analysis of multi-product use. So far, however, little attention has been paid in scientific literature to quantifying the performance benefits of more complex bio-refinery systems.

The economics of a bio-refinery system depend on the above-mentioned factors, but also on its scale. This is partly due to the well-studied mechanisms of economies of scale of conversion processes and equipment use (see e.g. Faaij et al., 1998), and the logistics of biomass supply (see e.g. Aden et al., 2002; Hamelinck et al., 2003). Another mechanism influencing the economic performance is that the size of the market and the amount of products and inputs (in particular land) in the bio-refinery system determine the market prices of in- and outputs. These changes of market prices can influence the economic efficiency of biomass systems significantly (Wyman, 2003; Green, 2000; chapter 2 of this thesis). There has been considerable interest in the interactions between biomass utilisation and market mechanisms. Equilibrium models have been used to investigate the demand and supply of various products as a function of market prices. For example, Walsh (2000) modelled demand and supply of biomass fibres in the USA as function of biomass material prices. Azar and Berndes (1999) estimated market prices of food as a function of carbon emissions penalties and subsequent demands for biomass and agricultural land. Gielen et al. (2003) estimated the global demand of biomass for energy, food and materials depending on GHG emission mitigation policies. However, this type of studies does not analyse market prices in relation to the supply, i.e. scale of a biomass material system and thus, do not quantify the effect material and land markets can have on the costs of certain biomass system as determined by technological bottom-up analysis. Studies addressing this subject could not be identified in scientific literature.

In conclusion, multi-functional bio-refinery systems may help to increase the performance of biomass systems, but potential benefits have been poorly studied, quantitatively. Consequently, the following research question is essential:

What are potential quantitative benefits of multi-functional bio-refinery systems with regard to GHG emission reductions, savings of non-renewable energy consumption and costs in relation to production scale and market volumes?

Two factors play a key role in this question:
1. The set-up and composition of the bio-refinery system, e.g. by-product use, recycling rates, etc.
2. The structure of the markets in which the bio-refinery system is embedded, i.e. the market for final products, for by-products and for agricultural land.

To answer the research question taking into account these key factors, we carried out a case study. Subject of the case study is a bio-refinery producing a bio-based polymer, i.e. poly lactic acid (PLA). A previous study has shown that bio-based polymer production has an interesting potential for energy savings and GHG emission reductions with regard to agricultural land use (Chapter 3 of this thesis). Out of the group of bio-based polymers that are suitable for bulk applications, PLA is selected, because it is currently the only commercially produced polymer apart from thermoplastic starch, which is less suitable for a number of bulk applications. Currently, PLA is mainly used for packaging applications. The global production capacity is about of 0.2 million tonnes per year (Crank et al., 2004) compared to 19 million tonnes per year of polyethylene (PE) (UN, 2002). As PLA could partly replace PE and other large volume petrochemical polymers like polyethylene teraphthalate (PET), it has the potential to contribute significantly to savings of non-renewable energy consumption and to GHG emission reduction.

In our case study, a whole crop is defined as input for the production system. Consequently, all parts of the crop are used for the production of PLA, by-products, i.e. fodder products, or energy in the form of electricity and heat. This is how the feature of multi-product crops is accounted for in the system. In addition, after their use, parts of the PLA products are recycled and parts are digested or incinerated with energy recovery. Thus, cascading is part of our PLA bio-refinery system.

In the first place, the performance of the bio-refinery system in terms of GHG emission reduction, non-renewable energy saving and costs are calculated by means of a bottom-up analysis. These parameters are presented both per kg of bio-based polymer produced and per ha and year of agricultural land needed for biomass production. Several configurations of the bio-refinery system (e.g. kind of crop production system, recycling rates, kind of products) are investigated, such that the influence of the structure of the bio-refinery system on the performance can be analysed. In the second place, market prices of products and land depending on the scale of the bio-refinery are determined by own-price elasticity. By this, the overall economic performance of the bio-refinery system in relation to elastic markets of products and land can be determined.

Geographically the case study is situated in Poland. Currently, PLA production takes place mainly in the U.S. while several initiatives for PLA production exist in Asia. This raises the question whether such a system may also have potential in Europe. One argument is that the Eastern European states[1] that accessed the European Union in May 2004 are particularly interesting for new biomass systems, because of relative low biomass production costs, relatively highly available land surfaces and an agricultural system with low productivity being in transition (FAO, 2003; Skarzynska and Augustynska-Grzymek, 2001b).

This chapter is structured as follows: Section 2 describes the methodology for the chain analysis of the bio-refinery system and the methodology to analyse the elasticity of markets in which the bio-refinery system is embedded. The following section describes the case of a PLA production bio-refinery system including input data for the analysis. Section 4 presents the results of the analysis: the performance of the PLA production system, depending on the system configurations, and the GHG mitigation costs in relation to the scale of the PLA production system with and without market influences. Section 5 and 6 contain discussion and conclusions.

[1] The Czech Republic, Estonia, Hungary, Latvia, Lithuania, Poland, the Slovak Republic, and Slovenia joined the European Union on 1st May 2004.

2 Methodology

The analysis of the bio-refinery system is carried out in two distinct steps as presented in Figure 5.1. In the first place, GHG emission reductions, savings of non-renewable energy consumption and overall costs of the bio-refinery system are calculated. These calculations are performed for different configurations of the PLA bio-refinery system as described in Sections 3 and 4. Subsequently the performance of these configurations is compared and moreover, GHG mitigation costs are analysed in relation to scale by chain analyses including economies of scale. In the second place, GHG mitigation costs in relation to scale are determined, taking into account variable market prices for land and products. These market prices are derived from market analysis using demand elasticity. Thus, the influence of the market on the results of the chain analysis can be singled out. Following, the methodologies of the chain and market analyses applied in this study are described in detail.

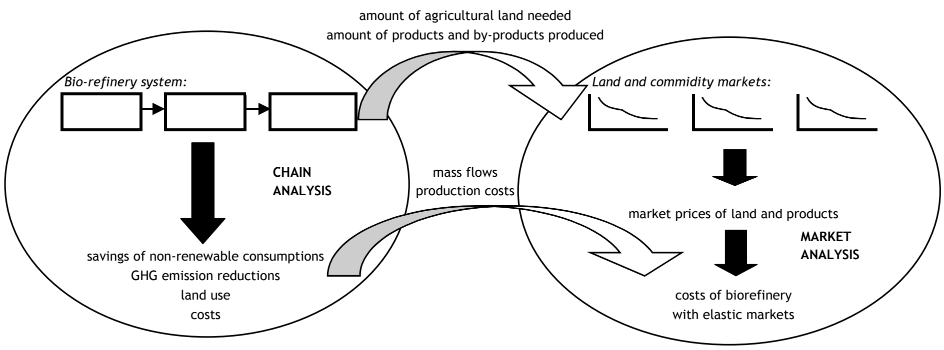


Figure 5.1: Approach to investigate bio-refinery systems – a combination of a bottom-up analysis and an analysis of elastic markets

2.1 Chain analysis

In the chain analysis, the performance of a bio-refinery system is investigated. Or rather, savings of non-renewable energy consumption, GHG emission reductions and costs are calculated. These parameters are related to either the amount of main product that is produced, i.e. in our case study the amount of bio-based polymer, or to the area of land used for biomass production.

In order to determine savings of non-renewable energy consumption and GHG emission reductions of the bio-refinery system, a reference system for comparison needs to be defined. Figure 5.2 shows schematically the bio-refinery system and the reference system belonging to it. Basis of the comparison is that all outputs of the bio-refinery system substitute equivalent products from the reference system. These outputs of the bio-refinery system are the main product and by-products (both are usually materials), electricity from biomass residues[2], and electricity that is recovered by waste treatment of the main product. As for the main products in the bio-refinery system, the main product in the reference system is incinerated with electricity recovery after its end of life. However, by-products in our case study are fodder products, which cannot substitute any fossil reference products, but substitute other conventional fodder products. Consequently, biomass and land are inputs for these fodder products in the reference system. A more detailed description of products and substitution reference products in the case is given in the following chapter on the PLA bio-refinery system regarded.

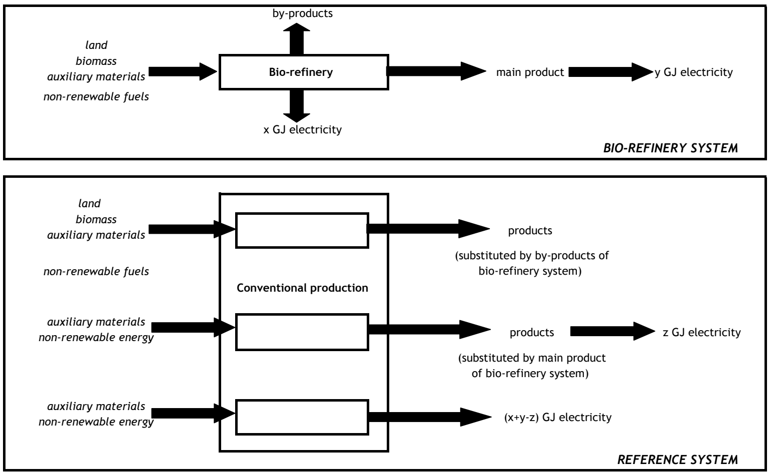


Figure 5.2: Schematic overview of a bio-refinery system and its reference system

To calculate savings of non-renewable energy consumption of the bio-refinery, the energy uses of all parts of the bio-refinery system, e.g. biomass production, main product production, waste treatment etc., are added up and subtracted from the sum of energy uses in the reference system. Overall GHG emission reductions obtained by the bio-refinery system are calculated like the savings of non-renewable energy consumption. In general, GHG emissions of auxiliary material, reference material and energy carrier production are calculated from life cycle analysis data. These data take into account direct and indirect emissions of greenhouse gases summarised as CO_2 equivalents in terms of global warming potential. For the production of biomass, direct and indirect GHG emissions of inputs, e.g. machine use, fertilisers, seeds, etc., are considered. Also, N_2O emissions from the application of N-fertiliser are taken into, while CH_4 emission from agricultural production are small and therefore, neglected.

[2] Heat that is produced from residues is assumed to be used within the bio-refinery, but is not considered as an output that used outside the facility.

In contrast to savings of non-renewable energy consumption and GHG emissions reductions, costs are not calculated by means of substitution products. Instead, revenues of products, by-products and bioenergy produced are based on their respective market values. Finally, land demand of the bio-refinery system is the area of agricultural land used for the production of its biomass inputs. In the reference system, this agricultural land is not used for crop production. From this land use for biomass production, the agricultural land use in the reference system (for the production of products that are substituted by by-products from the bio-refinery) is subtracted.

To calculate the performance characteristics of bio-refinery systems in comparison to the reference system, input data from statistics and scientific literature are used. For every process in the system, e.g. PLA production from grain or waste incineration, inputs and outputs, e.g. land, fuel, steam, electricity, auxiliary materials and products are defined. These in- and outputs are then converted to primary energy consumptions and GHG emissions by conversion factors from life cycle analysis studies, e.g. primary energy consumptions per kg of artificial fertiliser or GHG emissions per GJ electricity used. Cost data refer to the whole process, e.g. euro per kg PLA produced from grain or euro per kg waste treated. A detailed description of input data is presented in Section 3.

The geographical scope of our case study covers Poland and, thus, biomass and PLA production takes place there. Consequently, data on biomass production and agricultural land refers specifically to Poland. However, in our study other input data (such as energy mixes, production of reference products or material markets) refer to the European average situation. This is due to the fact, that non-biomass input of PLA production, e.g. electricity will rather originate from a European market than from Poland only. Moreover, products of the bio-refinery system will substitute products from the European market, too.

2.2 Market analysis

In order to analyse the market price changes of land and products of the bio-refinery system in relation to its scale of production, demand curves are applied. Demand curves describe the relation between demand and market price. In other words, they define which amount of a product could be sold for a certain price or vice versa. This leads us to the problem of how demand curves can be employed to determine market prices of land and products of a bio-refinery in relation to the size of the market demand and supply. Here, two cases are distinguished. First, the case of products that have a variable supply is investigated, i.e. different amounts of these products could be supplied by adapting the production capacity. Second, the case of land that has a fixed supply is discussed, i.e. the amount of agricultural land is limited to the total amount of agricultural land available.

The first case is presented in Figure 5.3. Typically, in an ideal market the price and amount of a commodity are at equilibrium at $P_{(eq)}$ and $Q_{(eq)}$. In other words, the market price of a commodity in an ideal market equals its production costs. If the production of a commodity is increased to $Q_{(scale)}$, for example by installing additional facilities, the market price decreases to $P_{(scale)}$ in the short term. This new market price is the price that corresponds to the new total amount of production on the demand curve. However, on the long term markets will return to a new equilibrium.

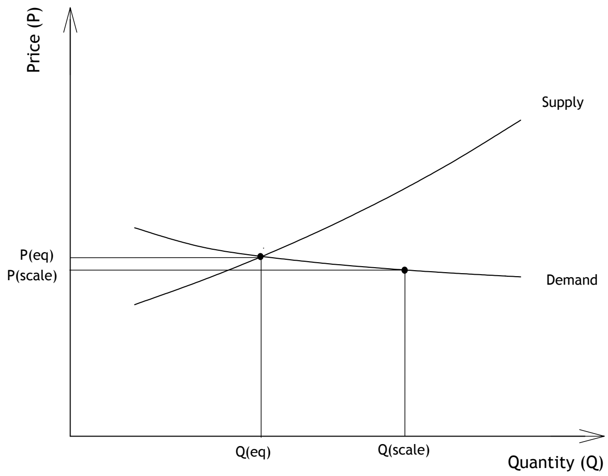


Figure 5. 3: Equilibrium price $P_{(eq)}$ and quantity $Q_{(eq)}$ of a commodity with variable supply and its new price $P_{(scale)}$ if an increased quantity $Q_{(scale)}$ is produced

The second case, of a commodity with fixed supply, is depicted in Figure 5.4. In our case the commodity is agricultural land. All available agricultural land Q_e is already used at its equilibrium price P_e. The demand for land increases to $Q_{(biomass)}$ if agricultural land is needed for biomass production. However, as no more agricultural land is available, the demand curve shifts with this additional demand. The new market price of land can be determined by the intersection of this shifted demand curve with the supply curve. As a consequence, the market price of land increases if additional agricultural land is needed for the production of biomass.

Thus, prices of agricultural land and bio-refinery products are determined by using demand curves in this analysis. Generally, different shapes of demand curves relating demand to market prices are conceivable. A standard form of a demand curve is defined by equation (1) (Franssen, 1999).

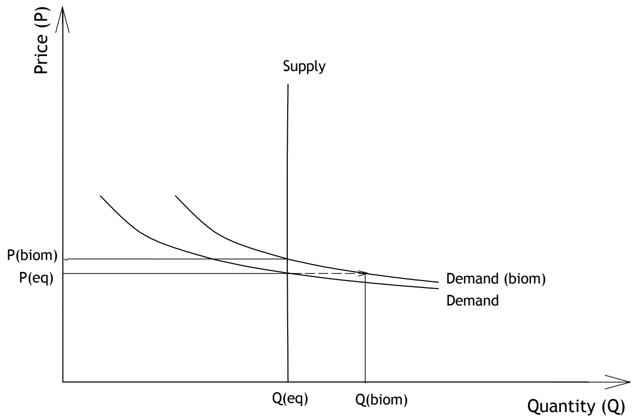


Figure 5.4: Equilibrium price $P_{(eq)}$ and quantity $Q_{(eq)}$ of a commodity with fixed supply—e.g. agricultural land—and its new price $P_{(biom)}$ if an increased quantity $Q_{(biom)}$ is demanded

$$P(Q) = C^*Q^{1/\varepsilon} \tag{1}$$

P: price of commodity; Q: quantity of commodity; C: constant; ε: own-price elasticity of demand

In this formula the own-price elasticity ε is a measure of percentage changes of price in relation to percentage changes of demand at the current demand and the current market price. The own-price elasticity is typically a negative constant for a commodity. The constant C is a factor that translates these percentage changes to actual quantities and prices. If the own-price elasticity of a commodity is known, this constant C can be derived from the current demand (i.e. amount of commodity sold) and the current market price. In this chapter we use this type of demand curve to determine what change of price may be induced by a change of quantity of the commodity.

With regard to own-price elasticity, the demand curve of a commodity depends on many factors, e.g. availability of substitution possibilities, share of budget devoted to the product, kind of product, the time horizon and the geographical scale (Franssen, 1999). Demand curves can be derived econometrically from empirical data (e.g. time series analysis or cross-sectional analysis) or economic models. These are top-down models that simulate the production in several sectors taking into account factors, like investment, labour, cross-price elasticity, etc.

However, it is beyond the scope of our study to determine demand curves for a bio-refinery system by econometrics or macro-economic models. Therefore, a review of own-price elasticity for the demand of land and for products of the bio-refinery as discussed in scientific literature is the basis of own-price elasticity used in our study; see Section 4.6.

3 PLA bio-refinery system

The PLA bio-refinery system investigated in this study is presented in Figure 5.5. It consists of several main components, i.e. production of biomass, PLA and bioenergy, end product use, recycling and waste treatment. In this chapter, a specific design of the components, e.g. kind of crop, recycling technology, waste treatment, is selected in order to investigate the performance of different system designs. The selected components of the PLA bio-refinery system and related input data are described in the following sections. In addition, data for the market analysis of the system is discussed in the last section of this chapter.

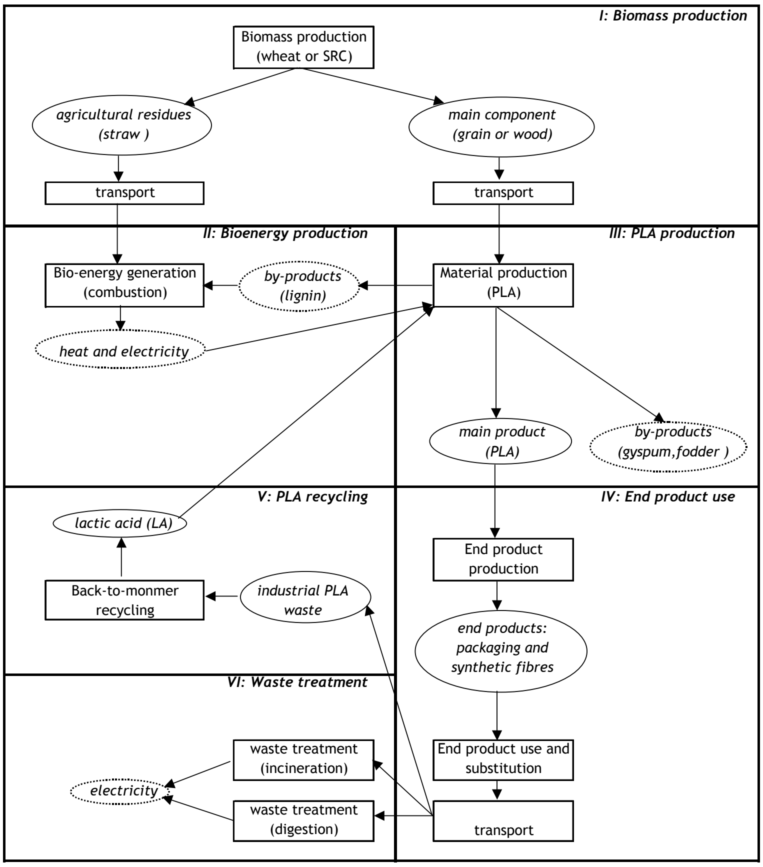


Figure 5.5: PLA bio-refinery system with either wheat or wood as feedstock

3.1 Biomass production

PLA can be produced either from starch or lingo-cellulose plants. In this study, wheat and short rotation (SR) wood were selected. We focus on these crops, because of existing agricultural experience and the possibility of obtaining high yields in Poland.

Data on agricultural land and crop production are presented in Table 5.1 and 5.2. The figures refer to the situation in Poland. Crop yields are given for different qualities of agricultural land. For that purpose, agricultural land is divided into three categories of low, medium and high quality lands by means of 'land valorisation values' on a provincial level (FAO, 2003). In addition, crop yields, costs, non-renewable energy consumption and GHG emissions of crop production are differentiated between intensities of agricultural production, i.e. intensive, average and non-intensive.

Wheat yields for average production intensity are representative for those parts of Poland where agricultural practice is comparable to Western European standards. Wheat yields of non-intensive production are yields in not intensively managed areas of Poland, while wheat yields of intensive production are the potentially obtainable yields (FAO, 2003). Energy use, costs and GHG emissions of wheat production are based on Polish agricultural statistics as discussed in Chapter 2 of this thesis.

As short rotation wood is not produced on a large scale in Poland, wood yields are based on experimental plots and adapted to different production intensities by factors obtained from Polish wheat production (Szczukowski et al., 1998). Also inputs of wood production are based on experimental plot data (Szczukowski et al., 2000b).

Table 5.1: Input data on agricultural land for different land quality categories

Amount [million ha]			**Rent** [€/(ha*yr)][b]		
Low quality[a]	*Medium quality* [a]	*High quality* [a]	*Low quality* [b]	*Medium quality* [b]	*High quality* [b]
4.1	8.5	2.7	27	33	39

[a] Quality factors are divided by land valuation values, which made up of soil quality, agro climate, relief and soil amelioration. While the average land valorisation value for Western Europe is defined as 100, the average for Poland is 67 (FAO, 2003). We defined land with a valorisation value of 55-63 as low quality, of 64-72 as medium quality and of 73-81 as high quality.
[b] Land rents for medium quality lands are average rents paid for land for wheat production, which is the crops with the highest land rents in Poland. On the other hand, lower land prices refer to land used for rye production, which is a crop with low land rents. The difference between medium and low quality land, i.e. 6 €/(ha*yr) has been added to medium quality land rents estimate high quality land rents (FAO, 2003).

Table 5.2: Input data on crop production

Parameter	**Unit**	**Winter wheat**			**Short rotation wood**		
Variation crop production system		*Non-intensive*	*Average*	*Intensive*	*Non-intensive*	*Average*	*Intensive*
Energy use	GJp/(ha*yr)	6.9[a]	8.9[a]	10.9[a]	10.7[b]	13.2[b]	15.7[b]
GHG emissions	$kgCO_{2eq}$/(ha*yr)	1350[a]	1670[a]	1990[a]	1050[b]	1370[b]	1690[b]
Production costs	€/(ha*yr)	412[a]	462[a]	512[a]	237[b]	262[b]	287[b]
Factor crop yield	-	0.75[d]	1[d]	1.25[d]	0.75[d]	1[d]	1.25[d]
Factor straw yield	-	0.75[d]	1[d]	1.25[d]	-	-	-
Variation agricultural land quality		*Low*	*Medium*	*High*	*Low*	*Medium*	*High*
Crop yields	Mg/(ha*yr)	3.9[c]	4.4[c]	4.9[c]	4.8[f]	5.8[f]	7.0[f]
Straw yields	Mg/(ha*yr)	3.5[e]	3.9[e]	4.4[e]	-		

GJ_p: Giga Joules primary

[a] Average intensity inputs are taken from chapter 2 of this thesis. The difference in fertiliser consumption between non-intensive and normal agricultural practice is about 25 kg NPK/(ha*yr) this is equivalent to 7.8 €/(ha*yr) (FAO, 2003), 0.57 GJ/(ha*yr) and 207 kg CO_{2eq}/(ha*yr) (Biewinga and Bijl, 1996). The respective difference in fertiliser use the difference between normal and intensive agricultural practice is about 30 kg NPK/(ha*yr) (FAO, 2003) or 7.8 €/(ha*yr). (FAO, 2003) Including the variation of machine use, we assumed differences between normal and more extreme agricultural practice to be about 50 €/(ha*yr), 2 GJ/(ha*yr) and 320 kg CO_{2eq}/(ha*yr).
[b] Inputs and costs of average production are taken from Chapter 2 of this thesis. Because costs of SR wood production with average intensity are lower than cost of wheat production, lower cost ranges of about 25 €/(ha*yr) are assumed. On the other hand, a larger range is assumed for energy use, i.e. 2.5 GJ/(ha*yr), which is quite high for short rotation willow production due to harvesting, while the range for GHG emissions is the same as for wheat production.
[c] These values are equivalent to 80% of the potential yield (FAO, 2003).
[d] For non-intensive management this is equivalent to 60% of potential yields and for intensive management to 100% of potential yield (FAO, 2003).
[e] This refers to a proportion of grain: straw of 1:0.89, which is derived from average cereal straw yields in Poland (CSO, 2003).
[f] Data on wood yields are data from intensive short rotation willow production (Szczukowski et al., 1998). These yields are converted to average agricultural production by the crop yield factor defined in this table.

With the data given in Table 5.1 and 5.2, production costs of about 51 €/Mg wheat and 65 €/Mg short rotation wood are found on medium quality land and normal intensity of agricultural production. For a more detailed discussion of crop production input data, see Chapter 2 of this thesis.

3.2 *Bioenergy*

Depending on the crop, either straw (from wheat) or lignin (from short rotation wood) is converted to steam and electricity in the bio-refinery system by combustion. It is assumed that steam and electricity are used within the PLA production process and that excess electricity is sold to the grid. Also it is assumed that the latter substitutes electricity based on the European generation capacity.

Table 5.3: Input data on steam and electricity production from wheat straw and lignin

		Straw (wheat) - grate firing	**Lignin (SR wood) - CFB-combustion**
Scale (base facility)	TJ_{input}/year	2880[a]	6209[f]
Net electric efficiency	GJ_e/GJ_{input}	0.31[b]	0.16[g]
Net steam efficiency	GJ_{th}/GJ_{input}	0.59	0.54[g]
GHG emissions	Mg $CO_2/Mg_{biomass}$	0.001[c]	0.001[c]
Investment costs	$€_{2002}/kW_{input}$	864[d]	641[h]
Annual operational costs	Part of investment	0.04	0.04
Lower heating value	GJ/Mg	15.6[e]	20.1

[a] Based on large straw grate firing of 100 MW_{th} as currently used in Denmark (Lans et al., 2000). Load factor is assumed to be 7000 h/yr. Investment costs in the analyses are calculated from this basic scale, by a scale factor of 0.8 and the actual size of the bioenergy facility.
[b] Calculated from trend line generated from efficiencies of different biomass combustion plants (Dornburg and Faaij, 2001b). For comparison Kumar et al. (2003) estimate the electric efficiency of a 450 $MW_{e\ plant}$ to be 34%, while the approach above results in an efficiency of about 36%. Net steam efficiency is derived from an average overall thermal efficiency of 90% (Broek et al., 1995).
[c] Biomass is seen as CO_2 neutral, because the carbon that is emitted during combustion has been sequestered during plant growth. Emission for production and transport of biomass is accounted for separately (see Section 3.1 and 3.6). Indirect emissions of bioenergy plants in general are in the magnitude of 3 kg/GJ_e (Mann and Spath, 1997).

[d] Investment costs for a 100 MW_{input} plant, based on a scaling factor of 0.8 and investment costs of a 450 MW_e straw combustion plant of 1300 US$2000/$kW_e$ (Kumar et al., 2003). These data are based on Broek et al. (1995), DOE (1997) and Wiltsee (2000).
[e] LHV as received, i.e. 10% moisture (ECN, 2003).
[f] Based on lignin combustion unit from ethanol production process of 203 MW_{th} (Aden et al., 2002). Load factor is assumed to be 8000 h/yr. Investment costs in the analyses are calculated from this basic scale, by a scale factor of 0.8 and the actual size of the bioenergy facility.
[g] These efficiencies correspond to the NREL design for ethanol production from corn stover with a scale of about 2000 $Mg_{biomass}$ /day and efficiencies (HHV) of about 15% electric and 51% steam (Aden et al., 2002). For comparison, efficiencies (HHV) of black liquor gasification are about 22 % electric and 41% steam and efficiencies (HHV) of Tomlinson boiler (as used in pulp and paper mills) are about 10% electric and 52% steam.
[h] Investment costs correspond to the NREL design for ethanol production for a boiler of about 203 $MW_{input.}$ Equipment costs for boiler and steam turbine are about 38.3 million $US\$_{2000}$. To these cost we added project investment costs of the total facility by the share of combustion equipment costs to total equipment costs. To adopt costs to the scale of lignin combustion that corresponds to a PLA production facility of a certain scale, a scale factor of 0.7 is applied.

Data on costs and electric efficiency of grate firing for straw combustion and circulating fluidised beds for lignin combustion are taken from literature on state-of-the-art technology (see Table 5.3). The annual costs of those conversion systems are calculated using investment costs, average O&M costs as a percentage of investments, an annuity assuming an interest rate of 5% and a lifetime of 20 years minus the market revenues of electricity produced and the market value of steam utilised in the PLA production facility. Thus, the costs of bioenergy are the additional costs or profits that result from using bioenergy in the PLA production process instead of using fossil energy.

3.3 PLA production

In principle, PLA production consists of biomass conversion to fermentable sugars, fermentation of these sugars to lactic acid, purification of lactic acid and its polymerisation. Currently, PLA is produced commercially from starch crops, while it is not produced from ligno-cellulose. However, Cargill Dow envisages producing PLA from ligno-cellulose in the next 5—8 years, because this process has higher efficiencies (Vink et al., 2003). Therefore, in this study, both processes, i.e. from starch and from ligno-cellulose, are investigated.

Only few process data on PLA production are publicly available. Therefore, input data often had to be estimated on basis of available aggregated data as discussed in detail below. Input data are shown in Table 5.4, while the most important mass and energy flows are summarised in Figure 5.6.

For PLA production from wheat, data on starch production is taken from an LCA of bio-based polymer production in Germany (Würdinger et al., 2002) and data for sugar production from a summary of LCA data on starch products (aAc, 2001). For the process of PLA production from sugar detailed process data have not been published. For this reason aggregated data from an LCA of Cargill Dows production facility in the U.S., being by far the largest installation with a capacity of 140000 Mg_{PLA}/yr, are used (Vink et al., 2003). Moreover, no information on investment costs of PLA production is available, but only estimates of production costs. These costs are adjusted to the scale of our facility by a general scaling factor (0.7).

Also for the production of PLA from ligno-cellulose, e.g. from short rotation wood, only limited aggregated process data are available. Based on a confidential NREL study, Cargill Dow expects that the production of PLA from ligno-cellulose will increase the production efficiency significantly (Vink et al., 2003). Therefore, we estimate the performance of this process on basis of publicly available NREL data on ligno-cellulose pre-treatment (Wooley et al., 1999; Mc Aloon et al., 2000) and data on PLA production from ligno-cellulose (Vink et al., 2003). Concerning production costs, projections of Cargill Dow for a ligno-cellulose production facility are used (Crank et al., 2004) and scaled by a general scaling factor, too.[3]

Table 5.4: Input data on PLA production

PLA production from		**Winter wheat**	**SR wood**
Base scale facility	$Mg_{biomass}$	285714[a]	323316[b]
Products			
PLA	$Mg_{PLA}/Mg_{biomass}$	0.49[c]	0.68[d]
By-product: gypsum	$Mg_{by\text{-}prod}/Mg_{biomass}$	0.49[e]	0.68[e]
By-products: milling	$Mg_{by\text{-}prod}/Mg_{biomass}$	0.21[f]	-
By-product: gluten meal	$Mg_{by\text{-}prod}/Mg_{biomass}$	0.09[f]	-
By-product: pulp	$Mg_{by\text{-}prod}/Mg_{biomass}$	0.50[f]	-
By-product: lignin	$Mg_{by\text{-}prod}/Mg_{biomass}$	-	0.26g[g]
Energy use			
Heat use	$GJ_{heat}/Mg_{biomass}$	12.9[h]	1.4[i]
Fuel use	$GJ_{fuel}/Mg_{biomass}$	10.8[h]	0.00[i]
Electricity use	$GJ_{e}/Mg_{biomass}$	1.8[h]	6.8[i]
Production costs			
Production costs at base scale	€/M_{gPLA}	2310[a]	1038[j]
Scale factor production costs	-	0.7[k]	0.7[k]

[a] Currently, production cost of PLA are still quite high due to the start-up of the first large-scale facility. However, for the near future lower costs are projected. Leaversuch (2002) states 2002 production costs at 1.30 US$/lb, while Cargill Dow aims to produce PLA below 1 US$/lb in the near future. We use the latter estimate as basis production costs for a facility of about 140000 Mg per year such as the Cargill Dow facility in Nebraska. However, parts of these production costs are feedstock costs for corn that are subtracted from the production costs. Costs of corn (for ethanol production) are about 1.94 US$ per bushel or 76.4 US$/Mg (Mc Aloon et al., 2000).
[b] Cargill Dow envisages to reach a total capacity of 500 million kg PLA per year with three production facilities, two producing PLA from sugar/starch crops and one from ligno-cellulose feedstock (Crank et al., 2004). As the first starch facility has a capacity of 140 million kg per million kg, we assume that the ligno-cellulose facility is scaled at 220 million kg.
[c] From data given in Vink et al. (2003) it can be concluded that Cargill Dow uses about 1.74 kg US corn per kg PLA. The amount of wheat has been calculated by average starch contents, i.e. 72% (Mc Aloon et al., 2000) and 60% for wheat (Würdinger et al., 2002).

[3] No factors for technological learning are applied, as the production costs estimates for PLA production from wheat as well as from SR wood are for the next facility to be built. Therefore, technological learning is implicitly included in these estimates.

[d] Wood from short rotation willow contains about 60.5% cellulose and 29.8% hemi-cellulose (ECN, 2003) of which about 80% or 95% respectively can be converted to fermentable sugars by pre-treatment (Wooley et al., 1999). We assume that all of the produced C_5 and C_6 sugars can be fermented with the same efficiency as glucose from starch due to technological development in fermentation technology as Cargill Dow envisages an optimised lactic acid production step by then (Vink et al., 2003).
[e] Few data on gypsum production are available. (Garlotta, 2001) estimates that up to 1 Mg/Mg lactic acid can be produced. As no lower range is given in literature, we use this upper range value.
[f] (Würdinger et al., 2002).
[g] Average lignin content of willow wood is about 26% (HHV: 21.4 GJ/Mg_{dm}) (ECN, 2003).
[h] The only information of energy use of PLA production from dextrose is an aggregated total of 39.5 MJ/kg_{PLA} (Vink et al., 2003). About 14.9 MJ/kg_{PLA} of this total energy amount is used for fuel and electricity in the facility, while the rest is related to operating supplies and wastewater treatment. We assume that this rest is fuel use. The largest part of energy use in the facility is steam, because lactic acid is purified with vacuum distillation and steam is used for the heating of the fermentation reactor (Gruber and O'Brien, 2002). We assume that about 95% of energy use in the facility is steam and the rest is electricity. Moreover, we assume that the data given by Vink et al. (2003) are related to primary energy. We recalculate these data to energy uses by the energy factors stated in this table. Energy use for starch production is 1.65 MJ electricity, 0.005 MJ fuel oil and 7.05 MJ steam per kg wheat (Würdinger et al., 2002). At the same time biogas is produced in wastewater treatment that is used for steam production. We assume that about 90% of biogas is converted into steam i.e. 0.55 MJ/kg. Finally, also the energy use for production of liquid sugars from starch is added. Gross energy requirement is about 3 MJ/kg of sugars (aAc, 2001) or about 1.8 MJ/kg wheat. This energy demand is assumed to be heat for liquefaction.
[i] Only highly aggregated data on energy use could be found. Using future lactic acid production technology reduces energy use for PLA production about 5.3 MJ/kg_{PLA} (Vink et al., 2003) i.e. the energy use for PLA production is 24.2 MJ/kg_{PLA} if original energy use is 29.5 MJ/kg_{PLA}. From the total energy use about 21.8 MJ/kg_{PLA} is electricity. In this case we assume that the remaining energy use is equivalent to steam use for the heating of the fermentation reactor and the wastewater treatment. Moreover, we assume that the data given by Vink et al. (2003) are related to primary energy and recalculate this to energy uses by the energy factors stated in this table.
[j] For 2010 Cargill Dow projects production costs of about 1.30 €/kg_{PLA}, i.e. costs competing with PET production costs (Crank et al., 2004). These production cost contain feedstock costs of corn stover. From the composition of corn stover, we can estimate that about 2.15 kg of stover yield 1 kg of PLA, while costs of corn stover are about 35 US$ per Mg (Mc Aloon et al., 2000). Production costs are recalculated to current costs of 2003 by an average deflator of 2%.
[k] (Faaij et al., 1998).

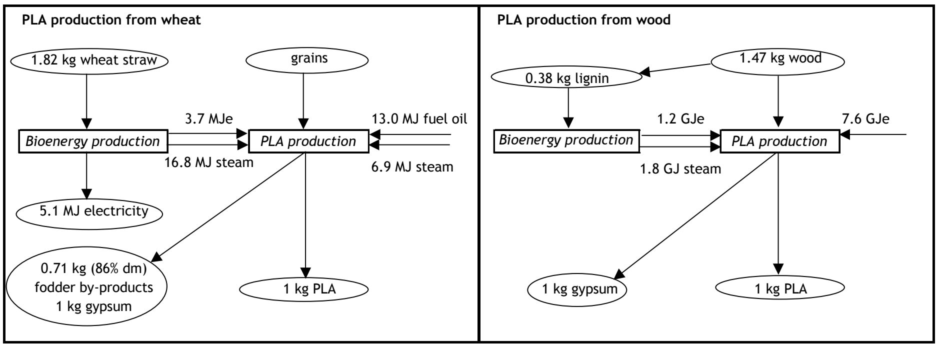


Figure 5.6: Overview of main energy and mass flows for PLA production from wheat and short rotation wood

3.4 Substitution of reference products

As described in the methodology section, it is accounted for the fact that main and by-products of the bio-refinery system replace reference products. In this section, reference products of by-products are discussed first, and reference products of PLA next. Data on the substitution of by-products, end products and energy are taken from life cycle assessments or similar studies and from market price surveys, see Table 5.5. For the use of energy in the process (heat, electricity and fuel oil) and for the substitution of non-renewable fuel, data on European energy balances are used.

Table 5.5: Input data on substitution of reference products and energy

Products and energy uses of bio-refinery	Functional unit	Kind of substitution product	Primary energy use [GJ_p]	GHG emissions [kg CO_{2eq}]	Agricultural land demand [ha]	Market prices [€]
Heat	Per GJ_{th}	Heat mix chem. ind.	1.35[a]	123 /GJ[a]	0	5.6[b]
Fuel oil	Per GJ_{fuel}	EU mix oil products	1.11[c]	83[c]	0	N/a
Electricity	Per GJ_e	EU mix electricity	2.48[d]	126[e]	0	8.9
Milling by-prod.	Per $Mg_{by\text{-}p}$	Wheat grains	1.2[g]	224[g]	0.33[h]	83.6[i]
Gluten	Per $Mg_{by\text{-}p}$	Fish meal	0.05[j]	2374[j]	0	971.3[f]
Pulp	Per $Mg_{by\text{-}p}$	Industrial by-products	0	0	0	4.3[i]
Packaging	Per Mg_{PLA}	HDPE pellets	76.6[k]	5070[l]	0	902[m]
Fibres	Per Mg_{PLA}	PET pellets	77.2[k]	5702[l]	0	1726[n]

[a] (UBA, 2003). No data on the European steam production mix are available. Therefore, we use data for the German chemical industry mix. For comparison, Polish heavy oil boilers have an efficiency of about 1.57 MJ/MJ.
[b] Average costs of heat production in fossil fuel boilers of 0.02 €/kWh in Europe are assumed (EC, 2000).
[c] European Union oil product mix for production (UBA, 2003). For CO_2 emissions we added 74.1 kg/GJ CO_2 emissions due to carbon content of fuel.
[d] European Union electricity mix (UBA, 2003).
[e] (IEA, 2002c). Data on CHP and electricity production, allocation of CO_2 emissions to heat and electricity on an energy basis.
[f] Average European production costs of 0.032 €/kWh (EC, 2000).
[g] Average nutrition value of milling by-products is about 5.54 MJ/kg, while wheat grains that are substituted have a nutrition value of 8.03 MJ/kg (Hydro Agri Dülmen GmbH, 1993). Energy use and GHG emissions of wheat production are taken from input data on crop production from this chapter.
[h] Calculated from average grain yield on medium quality land in Poland, see Section 3.1.
[i] (Würdinger et al., 2002)
[j] Substitution is made on basis of protein content, i.e. 3% for gluten and 70% for fish meal (Hydro Agri Dülmen GmbH, 1993). Energy use of fish meal production is derived from Carlsson-Kanyama and Faist (2000), and GHG emission factors of the European energy mix are applied.
[k] (APME, 1999). It is assumed that PLA substitutes HDPE and PET on a 1:1 weight basis. Thus, the values given are values per Mg HDPE and PET, respectively. The energy use of HDPE production is between energy use of LLDPE and LDPE production.
[l] Emissions consists of emissions during production and emissions during incineration. For HDPE these are 1928 and 3142 kg CO_{2eq}, respectively, and for PET 3410 and 2291 kg CO_{2eq}. Data on emissions during production are derived from UBA (2003) and data on incineration calculated from carbon content of polymers, i.e. 62.5% for PET and 85.7% for HDPE; LHV of HDPE is 43.3 GJ/Mg and for PET 22.1 GJ/Mg (Patel et al., 2002).
[m] PE price (Leaversuch, 2002).
[n] PET price (Clarnival, 2002).

The extraction of starch from wheat results in by-products from milling, gluten and pulp, whereas during conversion of ligno-cellulose to fermentable sugars lignin arises. Unlike lignin that is combusted for energy production, milling by-products, gluten and pulp are used as fodder. These by-products are assumed to be sold at current market prices and to substitute reference fodder products. Milling by-products are mostly used for cattle fodder. It is assumed that they replace wheat grains, which are closest in nutritional composition. Gluten has a relatively high protein content and can therefore replace other protein-rich fodder products such as fishmeal. Finally, pulp from starch production has a relatively low nutritional value. It mainly replaces other industrial by-products as pig fodder. Consequently, no energy or GHG emission credits for pulp are taken into account.

Besides milling by-products or lignin, also gypsum is produced as by-product within the PLA production process during the separation of lactic acid by calcium hydroxide or calcium carbonate (Garlotta, 2001). However, for gypsum the same argumentation as for pulp applies, because most gypsum used is a by-product from industrial processes, e.g. from flue gas cleaning, and gypsum has a low market value. As a consequence, no credits for substitution or market revenues are taken into account for gypsum.

PLA as the main product of the process is suitable for the production of a broad range of end products, e.g. agricultural plastics, diapers and electric appliances. The largest categories of current use are packaging (70%) and fibres (3-28%) (Crank et al., 2004). Cargill Dow expects that in the future these proportions will be more or less reversed. Therefore, both packaging and fibres are regarded as end products with different shares.

While theoretically many petrochemical polymers can be substituted depending on the specific application, for this study the most likely substitution products (due to market shares and substitution possibilities) are selected. In the case of packaging this is HDPE and in the case of fibres this is PET. Because it is assumed that processing pellets into packaging or fibres is comparable for PLA and petrochemical polymers to the processing of petrochemical polymers, PLA is compared to the petrochemical polymers on a pellet basis.

3.5 Recycling and waste treatment

In this study, part of the PLA waste is recycled, while the rest is treated by either incineration or digestion. As recycling technology, back-to-monomer (BTM) recycling of PLA is selected, because it is discussed as most promising recycling strategy for PLA (Gruber and O'Brien, 2002). Energy recovery from PLA waste is covered in the waste treatment part by waste incineration with electricity recovery. However, for back-to-monomer recycling relatively pure PLA waste is necessary that can be obtained by separate collection and upgrading processes. While such collection and upgrading structures for municipal solid waste are installed for plastic types with a very large market share, such systems are too costly for plastics with a low market share. Consequently, it is assumed that such a waste collection structure does not come into place in the medium term for PLA and that only industrial waste, i.e. waste that results from processing of PLA and waste that arise at the commerce level, is recycled.

However, BTM recycling of PLA is still in a laboratory stage and only few experiments have been undertaken, e.g. Fan et al. (2003) and Tsuji et al. (2003). PLA is converted into

L-LA by a short hydrolysis at low temperatures, i.e. about 90% LA can be recovered at 250° C in 10 to 20 min (Tsuji et al., 2003). The L-LA is then polymerised to PLA again. However, no process data on energy uses, GHG emissions and costs are available on BTM recycling and we had to estimate these data on basis of chemical properties and knowledge of other recycling processes; see Table 5.6.

In this study, two different waste treatment technologies for PLA are alternatively investigated. These are digestion and incineration with energy recovery. Digestion makes use of the biodegradable properties of PLA and recovers part of the energy contained in the polymer, while incineration on its part has the potential to reclaim a large amount of energy. Even though digestion can be selected as preferred waste treatment technology for PLA, its maximum share depends on the collection and separation rates of organic wastes and is therefore limited. The remaining waste is incinerated with energy recovery.

For waste treatment performance, data from literature of state-of-the-art technologies in Europe are used, because PLA and petrochemical reference polymers will be treated there. Moreover, Polish government envisages in its environmental policy the compliance of waste management with EU standards in the future (DHV, 2001). Concerning residues of digestion, no credits are taken into account, because PLA ideally degrades completely to water and gases. All input data on recycling and waste treatment technologies are shown in Table 5.6.

Table 5.6: Input data on recycling and waste treatment, i.e. digestion and incineration with energy recovery

		BTM recycling	**Digestion**[a]	**Incineration**
Share of separate collection	Mg_{PLA}/Mg_{PLA}	0.07/0.2[b]	0.3[c]	-
GHG emissions	kg $CO_{2eq.}$/ Mg_{PLA}	74[d]	37.4[e]	0
Energy use	GJ_p/ Mg_{PLA}	0.6[f]	0.87	-
Electricity production	GJ_e/ Mg_{PLA}	-	0.525	5.37[g]
Treatment costs	€/Mg_{PLA}	190[h]	33.5[i]	80[j]
PLA production efficiency	Mg/ Mg_{PLA}	0.9[k]	-	-

[a] State-of-art digestion plant (AOO, 2002). All values refer to medium composition of municipal organic waste.
[b] First value defines average collection rate and second value the maximum collection rates. It is assumed that only industrial waste can be recycled. Possible recycling rates are estimated by averages of European plastic recycling. About 21% of total plastic wastes in Europe arise in the industry and distribution sector, from which currently about 34% are recycled without energy recovery (APME, 2003). Thus, we assume a maximum recycling rate of 20% of all PLA products and a standard recycling rate of 7%.
[c] In the Netherlands, the objective for separate organic waste collection is about 55%, while 53% were realised. Because Poland is less urban, and less experience has been gained we assume a maximum collection rates of 30%. The share of incineration is the share of PLA waste that is not composted or digested.
[d] We assume that CO_2 emissions are related only to energy, because no special materials are used for the recycling process. Emissions are calculated with the carbon factor for steam, see section 3.3 on PLA production.
[e] GHG emission for a digestion installation with air cleaning by bio-filter are 1.1 $kgCH_4/Mg_{waste}$ and 0.046 kgN_2O/Mg_{waste} (AOO, 2002). Carbon that is sequestered during biomass production is assumed to be released during biodegradation and during the use of the residue for soil amelioration.

[f] We assume that for hydrolysis of PLA steam is used for heating, melting and depolymerisation and that efficiency of primary energy use for these steps is about 80%. Specific heat capacity is of most substances is around 1 kJ/(kg*K); we assume this to be valid for PLA, too. Moreover, polymerisation energy of PLA is 27.0 kJ/mol (Brandrup et al., 1999), while general purpose PLA of Cargill Dow has 168 kg/mol (Garlotta, 2001). The melt enthalpy of PLA is 93 kJ/kg (Södergård and Stolt, 2002).
[g] electric efficiency of a state-of the art waste incineration plant complying with EU standards of 30% (Dornburg and Faaij, 2001c). Electricity production calculated with a lower heating value of PLA, i.e. 17.9 GJ/Mg (Patel et al., 2002).
[h] No cost estimates are available on this recycling process are available. We therefore assumed current average costs of mechanical and feedstock recycling of plastics, which are about 190 €/Mg (Eggels et al., 2001).
[i] (AOO, 2002). These are average costs for waste treatment facilities in the Netherlands. Costs depend among others on scale and technology.
[j] Investment costs for a 235 MW_{th} installation are about 2.26 million $€_{2000}$ (Zeevalkink and van Ree, 2000). If 50% of the Polish municipal solid waste would be incinerated in two facilities, scale of the facilities would be 700 $MW_{th.}$ Assuming a scale factor of 0.7, operation and maintenance costs of 4% of investment, a life time of 20 years, 5% interest rate and an average lower heating value of waste of 10 GJ/Mg results in treatment costs of 80 $€_{2002}/Mg_{waste}$ without including sales of electricity. For comparison, in the Netherlands treatment costs for waste incineration are average about 100 Mg_{waste} (AOO, 2002).
[k] Short hydrolysis of PLA at low temperatures leads to highest yields of L-lactic acid, while at different temperatures racemisation occurs which negatively influences the quality of PLA (Fan et al., 2003). About 90% L-lactic acid can be recovered by hydrolysis at 250° C and a processing time of 10 to 20 min (Tsuji et al., 2003).

3.6 Transport

Transport operations in the PLA bio-refinery system that are different from transports needed to produce and discard reference products are:

- The transport of biomass to the PLA production facility
- The transport of PLA waste to the recycling facility
- The possible difference between the transportation of reference plastics to a waste incineration plant and the transportation of PLA to a digestion unit

Table 5.7: Input data on transportation of biomass and PLA

		Truck transport *per Mg*km*	**Truck transfer** *per Mg*	**Train transport** *per Mg*km*	**Train transfer** *per Mg*	**Waste collection** *per Mg*km*
Energy use	GJ_p	0.00125[a]	0.005[b]	0.00058[e]	0.01[g]	0.0075[h]
GHG emissions	kg CO_{2eq}	0.096[a]	0.48[c]	0.039[e]	0.96[c]	0.71[c]
Costs	Euro	0.05	2.4[d]	0.11[f]	1[g]	1.27[h]

[a] (UBA, 2003). Includes indirect emissions and energy uses, data refer to Germany, and an empty return.
[b] Average truck load transfer data (Suurs, 2002).
[c] Calculated from energy use with a direct emission factor of 74 kg CO_{2eq}/GJ and 21 kg CO_{2eq}/GJ of indirect emissions.
[d] Costs refer to the situation in the Netherlands and have been derived from (Suurs, 2002) assuming a load of 120 $m^{3.}$
[e] (UBA, 2003). Includes indirect emissions and energy uses, data refer to electric trains in Germany, and an empty return.
[f] (Hamelinck, 2003). Transportation costs by train differ considerably with different loads and distances. Costs refer to an average distance of 200 km, while in general costs per Mg*km are (16.9 €/Mg+0.0275 €/(Mg*km)*distance in km)/ distance in km.
[g] Average train load transfer data (Suurs, 2002).
[h] Data refers to waste collection in the Netherlands with a distance of about 20 km (VVAV, 1994).

Transportation distances are estimated using the radius of a circle in which the material is distributed. The land used for biomass production is the agricultural land around the facility (about 8-30% depending on the land qualities used). For recycling the per capita consumption of plastics, for example about 4.4 Mg/km^2 for packaging (APME, 2003), defines this area resulting in distances of about 180 km for recycling at the base scale of a PLA facility 285 million kg biomass input. Finally, for waste treatment we assume that Poland will in short term install more incineration and digestion facilities to comply with EU legislation and resulting average transportation distances of 230 km for digestion and 140 km for waste incineration. The means of transportation are assumed to be waste collection vehicles, trucks for distances up to 200 km and trains for exceeding distances. Data on energy use, GHG emissions and costs of transport are state-of-the-art data from a European context. They are presented in Table 5.7.

3.7 Demand curves

In this section, demand curves for the market analysis are presented. These demand curves are determined by own-price elasticity and estimates of current market prices and quantities, see Section 2.2. Reviews of own-price elasticity in different sectors and for different regions show a range of values. Ciaian et al. (2002) found values in economic literature that range from –0.045 to –1.49 for the agricultural sector of different countries. Franssen (1999) reports own-price elasticity of about 0 to –0.3 for energy commodities and transportation fuels in economic literature. Gielen et al. (2000) argue that most own-price elasticity of energy and materials from biomass are between –0.1 to –0.5.

The value of own-price elasticity can have a large impact on calculated market prices if the production capacity is increased significantly. For example, if the total production capacity is increased by 1% an elasticity of –0.1 leads to a 10% decrease of the market price, while an elasticity of –0.5 will result in a 2% decrease of the market price. Consequently, the selection of appropriate own-price elasticity is crucial for the quantitative results of the market analysis. In this study, demand curves and subsequently own-price elasticity are applied to fodder by-products of PLA from wheat, to agricultural land and to PLA.

Agricultural land and fodder products are commodities in the agricultural sector. Moreover, both commodities can be assumed to be limited to the Polish market. Ciaian et al. (2002) derived an own-price elasticity of –0.24 for the Polish agricultural sector from a partial equilibrium model. This value is used in our analysis for fodder by-products and agricultural land.

Unfortunately, in the case of PLA, no specific estimates of the own-price elasticity are available. PLA is bio-based polymer that replaces petrochemical polymers in a European or even global market. In a study on European (biomass) material and energy use, a generic marginal elasticity of – 0.5 for all kind of materials and energy carriers is assumed

(Gielen et al., 2000). An analysis of historical prices and volumes of polypropylene and polyethylene from 1970-2000 (Crank et al., 2004) results in own-price elasticity of –0.5 and –0.9 respectively. However, for a generic estimate of elasticity without econometric data, an own-price elasticity of –0.2 to –0.3 might be more realistic (Vöhringer, 2004).
On the other hand, PLA producers and potential producers estimate market prices and production capacities in 2010 (Crank et al., 2004). If all changes in these market price estimations would be due to own-price elasticity, an average value of –2.5 would result.[4] Note, that this value seems very optimistic, as the producers have assumed only a very small decrease of market price with growing production capacity due to an expected development of large new markets for PLA. In this study, the rather broad range of –0.2 to –2.5 is used for the market analysis of the bio-refinery system to show the broad range of uncertainty with an average value of –0.5.

Finally, elasticity factors are converted to market prices by using the current market prices and production capacity or agricultural land area as calibration point; see Section 2. These data are summarised in Table 5.8.

Table 5.8: Basic input data used for composing demand curves

Commodity	Market price	Production capacity
Low quality agricultural land	27€$_{2002}$/(ha*yr) (rent)	4.1 million ha/yr[a]
Medium quality agricultural land	33 €$_{2002}$/(ha*yr) (rent)	8.5 million ha/yr[a]
High quality agricultural land	39 €$_{2002}$/(ha*yr) (rent)	2.7 million ha/yr[a]
By-product: milling	84 €$_{2002}$/Mg	405.2 million kg /yr[b]
By-product: pulp	971 €$_{2002}$/Mg	173.7 million kg /yr[b]
By-product: gluten	4 €$_{2002}$/Mg	964.77 million kg /yr[b]
PLA	3000 €$_{2002}$/Mg[c]	143.5 million kg/yr[d]

[a] This is the total amount of available agricultural land, see Table 5.1.
[b] The amount of by-products currently produced is estimated by the amount of wheat flour produced in Poland, i.e. 1.5 million Mg/year in 2000 (UN, 2002) which is equivalent to 1.9 million Mg/yr wheat processed (Würdinger et al., 2002).
[c] (Crank et al., 2004).
[d] Installed global capacity 2003 (Crank et al., 2004). However, mid 2004 probably about 193,500 Mg/yr capacity of PLA production will be installed and the production capacity investigated in this study has to been seen as additional to that. For comparison, the global production of PET fibres is about 910 million kg/yr (CIRFS, 2002). Moreover, the average price of 3.0 €/kg for sales of Cargill Dow is assumed as current market price, while market prices of PLA range from 2.2 to 3.4 kg (Crank et al., 2004).

4 Results

4.1 Performance of the PLA bio-refinery system

The options of PLA bio-refinery systems that are investigated vary with regard to the crop used for PLA production, the intensity of crop production, the quality of land used, the

[4] This value has been calculated by fitting equation (1) to the current market volumes and prices and the projected future market prices and volumes.

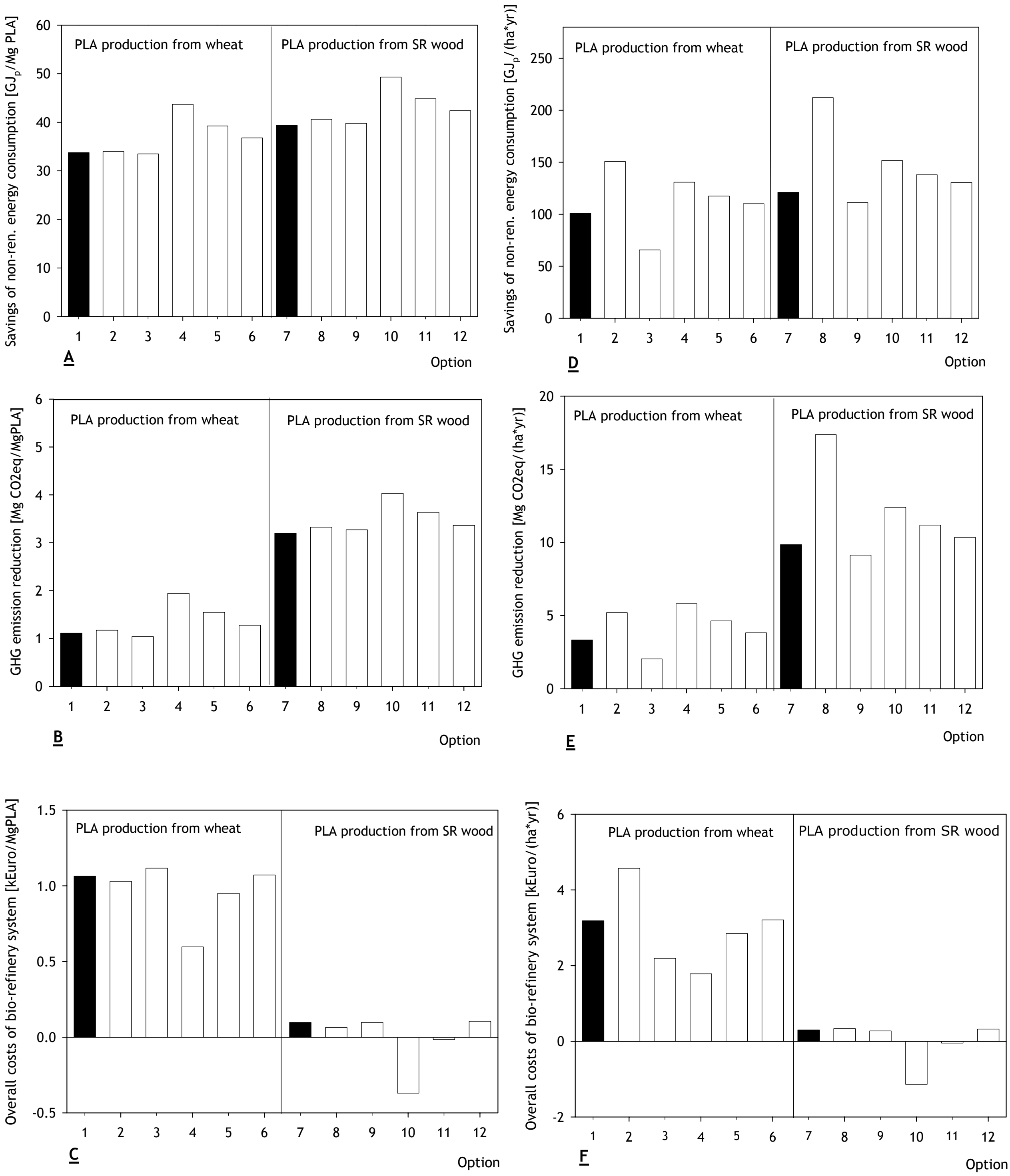
share of PLA fibre production, the recycling rate and the kind of waste treatment. The various combinations are summarised in Table 5.9. Scale of the PLA production facility is set at 285,000 Mg biomass input per year, which is equivalent to the scale of the current Cargill Dow production facility for corn, i.e. 140,000 Mg PLA per year (Vink et al., 2003).

Table 5.9: Options for PLA bio-refinery systems studied in this analysis

Name	Crop	Crop production	Land use[b]	End product [c]	Recycling[d]	Waste treatment
1 Base Case wheat	Wheat	Normal	L,M,H	0.7 packaging	Medium	Digestion
2 Intensive crop prod wheat	Wheat	Intensive	M,H	0.7 packaging	Medium	Digestion
3 Non-intensive crop prod wheat	Wheat	Non-intens.	L,M	0.7 packaging	Medium	Digestion
4 End prod fibres wheat	Wheat	Normal	L,M,H	0.2 packaging	Medium	Digestion
5 Recycling high wheat	Wheat	Normal	L,M,H	0.7 packaging	Max	Digestion
6 Waste incineration wheat	Wheat	Normal	L,M,H	0.7 packaging	Medium	Incinerate
7 Base Case SRC	SRC[a]	Normal	L,M,H	0.7 packaging	Medium	Digestion
8 Intensive crop prod SRC	SRC	Intensive	M,H	0.7 packaging	Medium	Digestion
9 Non-intensive crop prod SRC	SRC	Non-intens.	L,M	0.7 packaging	Medium	Digestion
10 End prod fibres SRC	SRC	Normal	L,M,H	0.2 packaging	Medium	Digestion
11 Recycling high SRC	SRC	Normal	L,M,H	0.7 packaging	Max	Digestion
12 Waste incineration SRC	SRC	Normal	L,M,H	0.7 packaging	Medium	Incinerate

[a] SRC: short rotation coppice, i.e. PLA production from short rotation wood
[b] L: low-quality land, M: medium-quality land, H: high-quality land; In the base case all types of land are used for biomass production, while in the intensive crop production scenario only high-quality and medium-quality land is used, and in the extensive crop production scenario only low-quality and medium-quality land is used.
[c] These shares refers to current (70%) and potential future market shares (20%) for packaging as estimated by Cargill Dow (Crank et al., 2004). It is assumed that all the rest is used for fibre production
[d] Medium recycling refers to an overall recycling rate of 7%, while max refers to a recycling rate of 20% (see Section 4).

In Figure 5.7, the performance of these options of a PLA bio-refinery system is shown. Savings of non-renewable energy consumption, GHG emission reductions and costs are given per kg of PLA produced (A-C) and per ha of land used to produce the biomass feedstock (D-F). In general, the results per kg of PLA and per ha of biomass do not differ substantially except for the options regarding different crop production systems. The intensive systems result in about 40 to 50% higher savings of non-renewable energy consumption and GHG emission reductions per ha than the respective base cases. Ranges of savings of non-renewable energy consumption are between 70 to 220 GJ/(ha*yr) and ranges of GHG emission reductions are between 3 and 17 Mg CO_2eq/(ha*yr). Hence, all options of PLA bio-refinery systems considered result in net energy savings and net GHG emission reduction, but most options resulted in net costs of the overall bio-refinery system of up to 4600 €/(ha*yr). Only two options of short rotation wood studied lead to net profits of about 50 €/(ha*yr) and 1100 €/(ha*yr), respectively. These are the options with a high share of recycling (no.11) and with high share of PLA fibre production (no. 10). This is due to the fact that production costs of PLA from wood are low compared to wheat and that market prices of PET for which the fibres are sold are relatively high.



1-6: PLA production from wheat; 7-12:PLA production from SR wood
1,7: Base case; 2,8: Intensive crop production; 3,9: Non-intensive crop production;
4,10: Fibre production 5,11: High recycling; 6,12: Waste incineration

Figure 5.7: Savings of non-renewable energy consumption (A, D), GHG emission reductions (B, E) and costs (C, F) of 'multi-functional' PLA production systems relative to the reference system. For a detailed explanation of options, see Table 5.9.

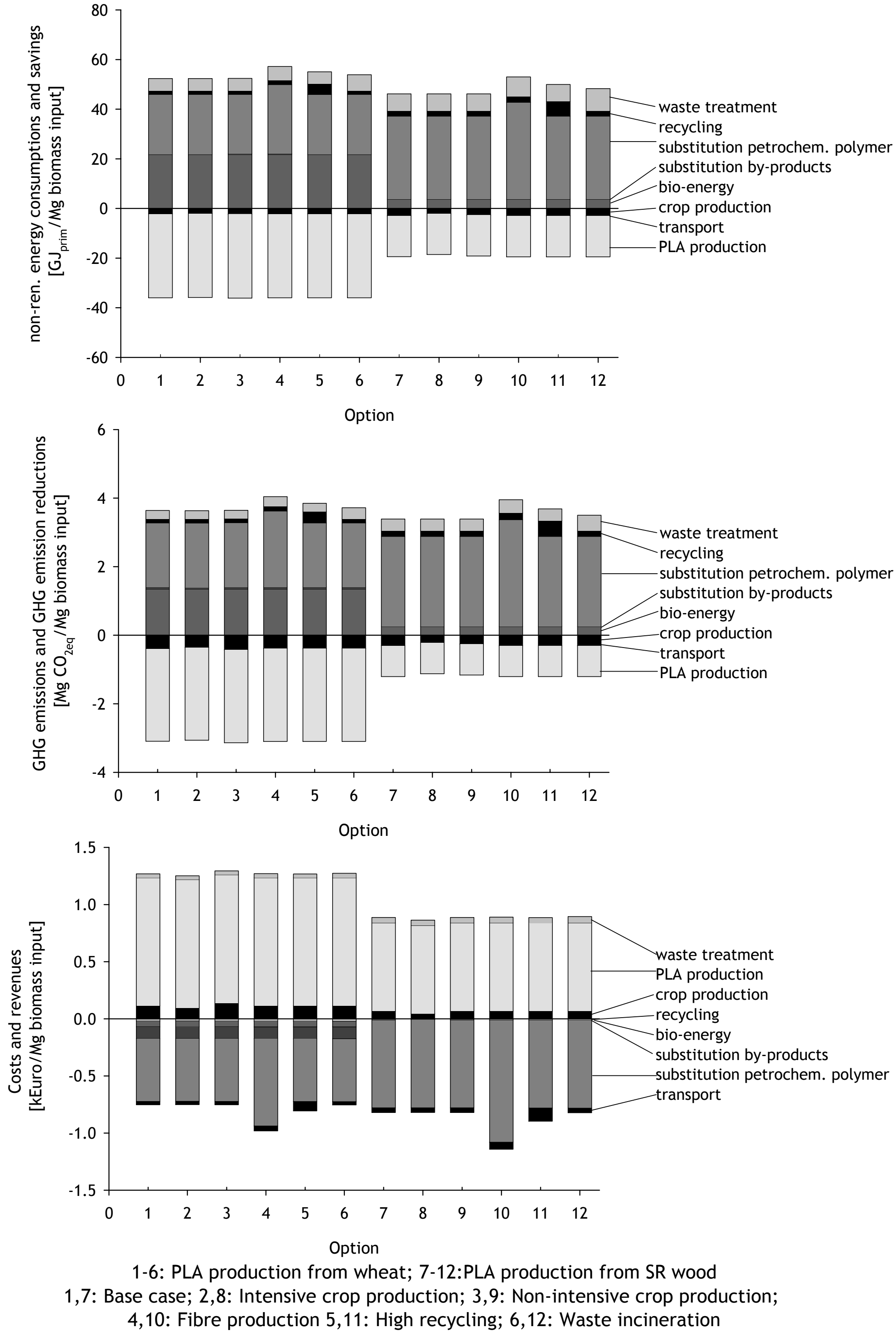


Figure 5.8: Breakdown of energy and greenhouse gas balances and costs structures of multi-functional PLA production systems per kg of biomass input. For a detailed explanation of options, see Table 5.9.

From the comparison of the different options among each other, conclusions can be drawn which system elements have the larges (positive) impact on the overall performance. If we compare PLA production from wheat (no. 1-6) with PLA production from short rotation wood, it can be concluded that with regard to savings of non-renewable energy consumption and GHG emission reductions, the latter is favourable. The PLA bio-refinery system with wood has about 25% higher savings of non-renewable energy consumption than the bio-refinery system with wheat in the base case and about twice the GHG emission reductions. From a cost point of view, PLA production from short rotation wood is much more favourable than PLA production from wheat.

Comparing biomass production systems, intensive crop production (no. 2+8) on high quality lands is most favourable with regard to non-renewable energy consumption and GHG emission reductions. In contrast, non-intensive crop production (no. 3+9) on low quality lands is favourable with respect to costs on a per ha basis but not on a per kg of PLA basis. The production of more fibres instead of packaging materials (no. 4+10) and the application of a higher recycling rate (no. 5+11) have a positive effect on savings of non-renewable energy consumption, GHG emissions and costs and exceed the base case by about 10% to 50%. Finally, waste incineration (no. 6+12) instead of digestion of PLA waste (no. 1+7) leads to slightly higher savings of non-renewable energy consumption and GHG emission reductions, but also to marginally higher costs of the bio-refinery system.

Figure 5.8 presents the break-down of the results presented in Figure 5.7. In other words, it shows non-renewable energy consumption and savings, GHG emissions and reductions, and costs and profits per kg of biomass input for every main step of the process chain considered. The energy balance and the CO_2 balance are similar in the distribution of savings over the main components and are discussed together. The substitution of petrochemicals, the production of PLA and bioenergy production are the most important components in the energy and GHG emission balance. However, the energy production potential and resulting GHG emission reductions of biomass residues are quite large and nearly completely compensate for the energy use during PLA production from wheat (no. 1-6). For PLA production from short rotation wood, energy use and GHG emissions are much lower due to a more efficient production process, but also the benefits from bioenergy production are much lower as less residues become available for direct energy production. Crop production accounts only for a small part of energy use and for a slightly larger part to GHG emissions, due to N_2O emissions resulting from fertiliser use. Recycling and energy recovery from waste treatment of PLA contribute for a smaller part to the energy savings and GHG emission reductions of PLA bio-refinery systems. Finally, the substitution effect of by-products only plays a very small role in the total energy and GHG emission balances, while transport is not significant at all with a share of less than 1%.

The break-down of costs of the PLA bio-refinery systems reveals that PLA production represents the largest share of costs. On the other hand, the profits from the substitution of petrochemical polymers, i.e. the revenues from PLA sales, are the largest part of the revenues, too. PLA production costs from short rotation wood are lower than from wheat and moreover, revenues from PLA sales are higher due to a more efficient production process. If from the biomass input a higher share of fibres is produced, higher revenues of PLA are achieved. Revenues of by-products, which arise in the wheat production options, are more significant in the overall cost balance than their contribution to energy savings or GHG emission reduction, while revenues of recycling and waste treatment are less important. Bioenergy production does not contribute significantly to revenues in contrast to the energy yields and GHG emission reductions. Finally, also in the cost balance, crop production accounts only for a small part and transportation is not significant.

4.2 Costs in relation to scale

In this section, costs per Mg of GHG emission reduction as achieved by PLA bio-refinery systems are investigated in relation to PLA production scale. For this analysis, the best PLA production systems from wheat and from short rotation wood with respect to costs per Mg of avoided CO_2 equivalent emissions from the previous analysis are considered. These PLA bio-refinery systems use intensive crop production, produce about 70% of fibres, have a maximum share of recycling and incinerate the PLA waste.

In Figure 5.9, the dotted line show the relation between costs and the production capacity of the PLA plant due to logistics and economies of scale. The solid line depicts this relation also taking into account the own-price elasticity of the demand of material and land. If scale effects are regarded without considering market price changes due to the larger production capacity (dotted line), mitigation costs decrease with scale and become profitable at around 200 Mg biomass input per year for SR wood and 400 Mg biomass input per year for wheat. However, if market elasticity is taken into account (solid line), costs decrease less with increasing scale. At the larger scales regarded in this analysis, the decrease of costs becomes marginal and costs are rather constant with scale. This effect is explained mainly by the gradually decreasing market price of PLA with larger scales.

It should be noted that costs at the ‘base’ scale, i.e. 140,000 Mg_{PLA}/yr the scale of analysis in Section 4.1, are higher in the analysis with own-price elasticity than in the analysis without own-price elasticity. In the analysis without own-price elasticity, the market price of PLA is equivalent to the market price of petrochemical reference polymers, while with the demand curve used here the market price of PLA is lower.

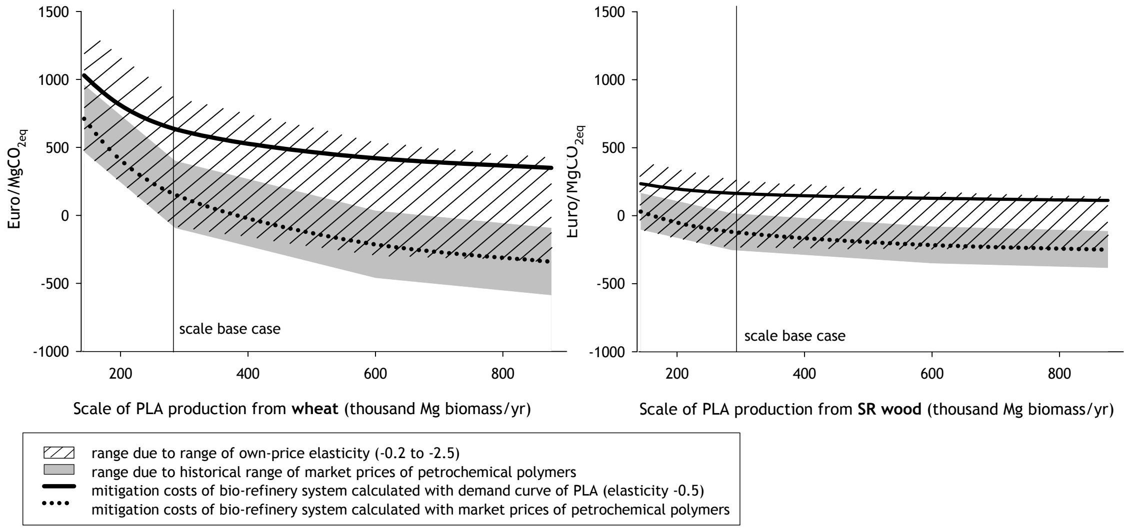


Figure 5.9: Costs (positive) or income (negative) per Mg of avoided CO_2 equivalent emissions in relation to the market supply of a PLA production facility.

Figure 5.9 also shows ranges of the mitigation cost estimates. For mitigation costs that are calculated with elasticity taken into account, these ranges correspond to the ranges of own-price elasticity for PLA (see Section 3.7). For mitigation costs that are calculated without elasticity (dark grey), ranges correspond to historical price changes of petrochemical polymers. Historical price changes were due to many factors, like the price of competing materials, income developments, but also the own-price elasticity of demand. Ranges used in this analysis are based on prices of ethylene, which is an important intermediate for both HDPE and PET. These prices varied from 1995 to 2002 about +/- 35% (CEFIC, 2003). It is obvious that both elasticity factors of PLA and market prices of petrochemical reference materials influence the (subsequently) calculated GHG mitigation costs significantly.

With respect to the market analysis with own-price elasticity of the PLA bio-refinery system, market price changes of PLA play an important role, as discussed above. On the other hand, market price changes of land and by-products are much less important, because crop production costs and by-product revenues are quite small compared to PLA revenues. Nevertheless, land rents increase up to about 6% and by-product revenues even decrease up to 80 % for a PLA bio-refinery system with 900,000 Mg biomass input per year.

5 Discussion

In this study, quantified GHG emission reduction, primary energy savings and costs of PLA bio-refinery systems in different configurations have been presented. Even though our analyses provide a good insight into the mechanisms relevant for bio-refinery systems and into the possible ranges of benefits, some uncertainties due to the quality of input data remain. In principle, data on crop production, bioenergy conversion, petrochemical polymer production and waste treatment are relatively well-known. Less well-known are the specifications of PLA production and recycling.

The current technology to produce PLA from starch has only been used at large scale in a single plant since 2001, while the technology starting from ligno-cellulose is not employed on a large scale, yet. Therefore, not many process data exist. Variations in the data on PLA production can have a significant impact on the overall results, because PLA production is one of the largest components in the balances of energy, GHG emissions and also costs. In general, the total energy savings are quite sensitive to the energy use of PLA production: an increase of 10% in PLA production leads to a relative decrease of savings of non-renewable energy consumption of 5% for short rotation wood and 20% for wheat.

In addition, also for PLA production costs only estimations of total costs are available without any specifications of e.g. investments costs. In order to evaluate economies of scale, production costs are calculated with a scaling factor, although some cost factors are to a smaller degree subject to economies of scale, e.g. process inputs or reactors above a certain size. As a consequence, economies of scale of the PLA plant tend to be overestimated in this analysis. Another important aspect in the context of PLA production costs is that manufacturers in such a small market may not reveal their real costs to their competitors, creating uncertainties in cost assessments. Total costs of the bio-refinery system would increase about 25% to 30% if production costs were 10% higher than estimated in this study.

The reason for the poor data availability on the recycling step is that the technology is still in the laboratory stage and no process data exist. However, as recycling does not play a large role in the calculation of overall energy savings, GHG emission reductions and costs, variations in these data do not lead to large variations in the overall results. A main reason for that is that only small parts of PLA are considered to be recoverable. However, if on the long term the recycling structure for plastics for PLA would be expanded, the performance of recycling will become more important. For illustration, if 100% of the PLA is recycled and after its second use incinerated with energy recovery—with the assumptions used here— energy savings of the total bio-refinery chain would double and costs would decrease more than 70% resulting in net profits in the case of PLA production from short rotation wood.

Unlike recycling, the substitution of petrochemical polymers plays an important role in the overall energy savings and GHG emission reductions. Non-renewable energy consumption for the production and energy recovery from the incineration of petrochemical polymers can be quite different depending on the kind of polymer produced and the waste treatment system applied. It should be noted that the precise type of material to be substituted by a new product could not be defined exactly, because a bio-based polymer will replace several materials depending on technological substitution potentials for specific application and the market prices of competing materials. Thus, in practice a mix of materials will be substituted. For example, PLA is suitable to replace nylon 6 (Crank et al., 2004), which has comparably high gross energy requirements for its production but also high lower heating values (APME, 1999). If the PLA produced in the bio-refinery system would replace nylon 6 instead of HDPE and PET, the total energy savings would increase about 5 to 10% compared to the base case.

Another aspect in the context of substitution is the use of factors to convert energy consumptions to non-renewable fuel consumptions and GHG emissions. While in this study European aggregated mixes of energy sources are used, for some aggregated data sets used in this study, e.g. for substituted reference fodder products, different energy mixes may have been used. However, this difference does not influence the overall results significantly.

Vink et al. (2003) give fossil energy consumptions for PLA production. Producing PLA from corn (without recovering energy from biomass residues) uses about 54 MJ/kg PLA. PLA production from lignocellulose, i.e. corn stalk, with steam use from lignin combustion leads to fossil energy consumptions of 29 MJ/kg PLA. The results of our study are for the PLA production process based on the study of Vink et al. (2003). However, in our study we used other data for the mix of primary energy sources applied and for the biomass production. Furthermore, the use of steam and electricity from biomass residues in the production of PLA was taken into account for both wheat and short rotation wood. Because of this bioenergy use, respective non-renewable energy consumptions of PLA production from cradle-to-factory-gate are lower, i.e. 28 MJ_{prim}/kg PLA for the production from wheat and 23 MJ_{prim}/kg PLA for the production from short rotation wood. Including the savings from recycling and waste incineration, these values are even lower and are about 20 MJ_{prim}/kg PLA for wheat and 14 MJ_{prim}/kg PLA for short rotation wood.

A disadvantage of the approach to use own-price elasticity from literature is that the elasticity figure used is not specifically for a product from a bio-refinery or a specific type of agricultural land. Therefore, own-price elasticity figures used in this study are uncertain. However, they influence the result significantly, as shown in the sensitivity analysis in Section 4.2. The approach presented in this study is therefore first of all a methodological demonstration of incorporating market effects in the analysis.

6 Conclusions

This chapter investigates non-renewable energy consumptions, GHG emissions and costs of bio-refinery systems for PLA production in comparison to a conventional mainly fossil fuel-based production system. The analyses provide a good insight into the mechanisms that influence the performance of bio-refinery systems. As a result, the contribution of 'multi-functional' components to the performance of the overall could be identified.

It is concluded that bioenergy production from biomass residues can contribute significantly to savings of non-renewable energy consumption and GHG emission reduction. The benefits from by-product use, however, are very small, with the largest benefit in a reduction of overall system costs. Therefore, the expected benefits of bio-refineries that would be obtained by producing a range of products are not found in the case of PLA production, because by–products are relative low-value fodder products. Recycling and waste treatment with energy recovery play an important role in the reduction of energy uses and GHG emissions of PLA bio-refinery systems. However, recycling and energy recovery from waste are less important than bioenergy production from biomass residues. If much larger recycling rates of back-to-monomer recycling would become possible by setting up a thorough (and probably costly) structure to collect PLA waste, recycling could result in very large benefits with regard to energy savings and GHG emission reductions.

Concerning the PLA bio-refinery system, the most favourable options emerged from the analysis. In detail, intensive crop production systems on high-quality land are favourable with regard to energy savings and GHG emission reductions, while non-intensive crop production systems are more favourable with regard to cost. While this conclusion is true for the Polish agricultural system, for other countries with a different relation between crop production costs and yields it might diverge. Moreover, fibre production replacing PET as petrochemical reference polymer is better than packaging replacing HDPE. Also, waste incineration is better than digestion with regard to energy savings, GHG emission reduction and costs. A possible production of PLA from short rotation wood instead of wheat is better in view of GHG emissions and much better with respect to economic performances.

All PLA bio-refinery systems lead to net savings of non-renewable energy consumption of up to 220 GJ/(ha*yr), from which about 30—60% relate to multi-functional elements, i.e. energy production from biomass residues, by-product use, recycling of PLA, and energy recovery from PLA waste. Besides, all options regarded lead to net GHG emission reduction with reductions of up to 17 kg CO_2eq/(ha*yr) and a contribution of 20—60% of multi-functional elements to these reductions. Contrary to GHG emission reductions and energy savings, most PLA bio-refinery systems considered in this study lead to net costs of up to 4600 €/(ha*yr). Only 5—20% of the revenues in the overall bio-refinery system result from multi-functional elements. However, PLA production from lignocellulose leads to net benefits if a high amount of the high-value product fibre is produced.

Under Polish conditions, the profitability of PLA production depends strongly on the market price of PLA products. This market price can change due to the kind of application, fluctuations on the market, price elasticity with increasing production capacity, etc. It has been shown that if it is taken into account that PLA prices are likely to decrease due to increased production capacities, the decrease of costs of PLA bio-refinery systems due to economies of scale is marginal at large scales. For example, not taking into account economies of scale, overall costs increase about 3% if the market volume is doubled with the base assumptions of elasticity used here. The shape of costs versus scale is very sensitive to the own-price elasticity of PLA demand curve.

Land costs are not an important element in the total costs of a PLA bio-refinery system. However, considering own-price elasticity of land, land rents increase significantly, even at moderate scales of biomass production. Consequently, these changes of land prices can become very significant if the use of biomass for energy or materials, e.g. for bio-refineries, is introduced on a large scale.

Summarising, a multi-functional use of biomass feedstock is necessary to achieve a viable PLA bio-refinery systems with regard to saving of non-renewable energy consumption and GHG emission reduction. However, with regard to costs, the multi-functional use shows only a small contribution to overall profits. Own-price elasticity of material markets is crucial for the economic performance of PLA bio-refinery systems. A more detailed analysis of material markets is desirable in order to quantify these effects more precisely.

CHAPTER 6:

Estimating GHG emission mitigation supply curves of large-scale biomass use on a country level*

Abstract

This study evaluates the possible influences of a large-scale introduction of biomass material and energy systems and their market volumes on land, material and energy market prices and their feedback to GHG emission mitigation costs. GHG emission mitigation supply curves for large-scale biomass use are compiled using a methodology that combines a bottom-up analysis of biomass applications, biomass cost supply curves and market prices of land, bio-materials and bio-energy carriers. These market prices depend on the scale of biomass use and the market volume of materials and energy carriers and are estimated using own-price elasticities of demand. The methodology is demonstrated for a case study of Poland in the year 2015. The case study applies different scenarios on economic development and trade in Europe that impact biomass supply and markets of land, materials and energy carriers. For the key technologies considered, i.e. medium density fibreboard, poly lactic acid, electricity and methanol production, and scenarios investigated in this study, GHG emission mitigation costs increase strongly with the scale of biomass production. It is found that the influence of a large-scale introduction on the development of biomass supply costs and market prices of land, materials and energy carriers, reduces the GHG emission reduction potential at costs below 50 €/Mg CO_{2eq} with about 13—70% depending on the different scenarios. Bio-material production accounts for only a small part of the total GHG emission mitigation potential at low costs. This is due to relatively small material markets and the subsequent strong decrease of market prices of bio-materials at large scale of production. GHG emission mitigation costs depend strongly on biomass supply curves, own-price elasticity of land and market volumes of bioenergy carriers. This analysis shows that these influences should be taken into account for developing biomass implementations strategies. However, literature estimates of own-price elasticities are highly uncertain and market volumes of biomass applications depend on their competitiveness. To counteract these uncertainties, a combination of a bottom-up analysis with an analysis of market effects is recommended.

---

* Co-authors: J. van Dam and A.P.C. Faaij.

1 Introduction

The use of biomass for energy may contribute significantly to the reduction of greenhouse gas (GHG) emissions (IPCC, 2001b; Johansson, 2000). It has been shown in earlier bottom-up analyses, that the use of biomass for materials combined with the energy utilisation of residues and material wastes may increase the efficiency of GHG emission reduction—i.e. increasing the amount of GHG emissions per area of land used for biomass production and/or decrease the GHG emission mitigation costs—if suitable biomass applications are selected (Chapter 2 and 3 of this thesis). [1]

However, increasing the amount of biomass produced and subsequently the amount of bio-materials and biomass-based energy may lead to an increase of agricultural land prices and a decrease of material and energy prices, depending on the market volumes involved. Especially, markets for bio-materials are often quite small (ECCP, 2001) and therefore, expected decreases of material prices are relatively large; see e.g. Chapter 5 of this thesis. Market volumes and subsequent market price changes may decide which options are economically most attractive to reduce GHG emissions by large-scale introduction of bio-materials and bio-energy carriers. Furthermore, the use of agricultural land for biomass production may influence land prices significantly as the production of food and fodder on agricultural land will compete with the production of biomass for energy and materials; see e.g. Keith (2001). As a consequence, these effects may increase GHG emission mitigation costs of biomass systems with the scale of biomass utilisation.

Various biomass supply curves have been calculated for different geographical scales; see e.g. Hoogwijk et al. (2004a) and Walsh (2000). These calculations are based on the availability of land and its quality and, typically, do not consider changing market prices with increased biomass production. Beside, also supply curves for GHG emission mitigation costs by carbon sequestration in forests have been determined; see e.g. Sedjo et al. (2001) and Stavins (1999). Some of the GHG emission mitigation cost supply curves take increase of land prices due to increased sequestration activities into account by various approaches; see the review by Richards and Stokes (2004). One of these approaches is the use of a demand curve of land that specifies market prices in relation to demand by means of own-price elasticity[2] (Richards and Stokes, 2004). Also the relation

[1] This increase of efficiency is in the range of avoiding about several tens of Mg CO_{2eq} per ha and yr additionally and lowering GHG emission mitigation costs about several hundreds of € per Mg CO_{2eq} compared to single bioenergy utilisation (see chapter 2 and 3).

[2] Own-price elasticity is the percentage change of demand divided by the according percentage change of price on the demand curve of a commodity.

between the amount of biomass use and its market price has been investigated. Otto and Gallagher (2001) estimate the market price of fodder by-products from ethanol production using own-price elasticity, if ethanol production is increased. De La Torre Ugarte et al. (2003) calculate the demand for agricultural crops depending on their prices using a.o. own-price elasticity.

For many different options, e.g. increases of energy efficiency, carbon capture and storage, and renewable energy supplies, GHG emission mitigation costs have been calculated. These calculations have been done using either bottom-up or top-down approaches including various market effects; see e.g. the review of studies in IPCC (2001b). Studies considering a relation between the demand and the price of goods that use a top-down approach typically start with prices. From these prices—e.g. material prices, energy prices or carbon taxes—the demands for goods are calculated from demand curves. The GHG emissions and costs of a system producing these demands of goods are than compared to those of other possible systems with different demands. Thus, various GHG emission mitigation scenarios have been developed; see e.g. RIVM (2001), Bollen et al. (2004) and Gielen et al. (2003).

Studies that calculate GHG emission mitigation costs of biomass systems starting from an exogenous demand, e.g. of biomass products, taking market effects into account could not be identified. Such an approach, however, may create new insights, because it could produce GHG emission mitigation cost curves for bio-material and bioenergy application by varying the amount of biomass utilisation exogenously. Also, biomass supply curves could be integrated into the analysis, leading to overall GHG emission mitigation cost curves of bio-material and bio-energy uses with growing biomass use considering effects on land, energy and material markets. Another advantage of this approach is that market effects for different biomass applications can be analysed explicitly, e.g. a GHG emission mitigation cost supply curve for bio-fuels may be different than that for bio-materials.

The objective of this study is, therefore, *to evaluate the possible influences of a large-scale introduction of biomass material and energy systems and their market volumes on market prices of land, materials and energy carriers and subsequently on GHG emission reduction costs.*

For this purpose, a methodology to estimate GHG emission mitigation supply curve for large-scale biomass use is proposed. The methodology incorporates (1) a bottom-up analysis of bio-material and bioenergy applications, (2) scenario-dependent biomass cost supply curves and (3) estimations of market prices of land, bio-materials and bio-energy carriers depending on the scale of biomass use and the market volume of materials and energy carriers using own-price elasticities of demand. Because biomass supply curves as well as markets of land, materials and energy carriers depend strongly on economic development and trade, these parameters are varied for different scenarios that follow the SRES scenario families of IPCC (2000).

The methodology is demonstrated for bio-material and bio-energy use on a country-level. Subject of this case study is Poland in the year 2015, because in a short term new Eastern European member states of the European Union may play an important role in European biomass production, as many of these countries have relatively large areas of available agricultural land and low biomass production costs. Poland is a representative example of a Central Eastern European country with a rather high biomass production potential.

We analyse GHG emission mitigation cost curves for four selected bio-material and bio-energy applications. Key criteria for the selection are that the application (1) has a potentially large market volume in the year 2015, (2) potentially reduces a large amount of GHG emissions per unit of biomass used and (3) has rather low initial GHG emission mitigation costs. Moreover, for the simplification of biomass supply curves and the calculation of GHG emission mitigation cost curves, only applications are selected that can use the same type of biomass, i.e. short rotation wood. From earlier reviews of GHG emission reduction of bio-materials and bioenergy carriers (see Chapter 2, 3 and 5 of this thesis), the following four bio-material and bio-energy applications are investigated:
- Poly lactic acid (PLA) with waste-to-energy recovery
- Medium density fibreboard (MDF) with waste-to-energy recovery
- Methanol
- Electricity

In Section 2, an overview of the approach is given, while Section 3 presents input data on the biomass supply and the selected biomass applications based mainly on scientific literature, market statistics, and agricultural production data. Section 4 presents GHG emission mitigation costs of the different biomass systems and finally, Section 5 and 6 finish with discussion and conclusions.

2 Method

To calculate GHG emission mitigation cost supply curves, various calculation steps are necessary. In Figure 6.1, the various steps to calculate GHG emission mitigation supply curves are presented.

1. Biomass supply curves describing the possible amount and costs of biomass production in Poland are determined.
2. The effects of increased biomass production on the market prices of agricultural land are investigated.
3. The GHG emission mitigation costs of selected bio-materials and bio-energy applications are calculated.
4. The changes of market prices of materials and energy carriers due to an increased production of bio-materials and bio-energy are estimated.
5. The results of these four steps are combined in a GHG mitigation supply curve.

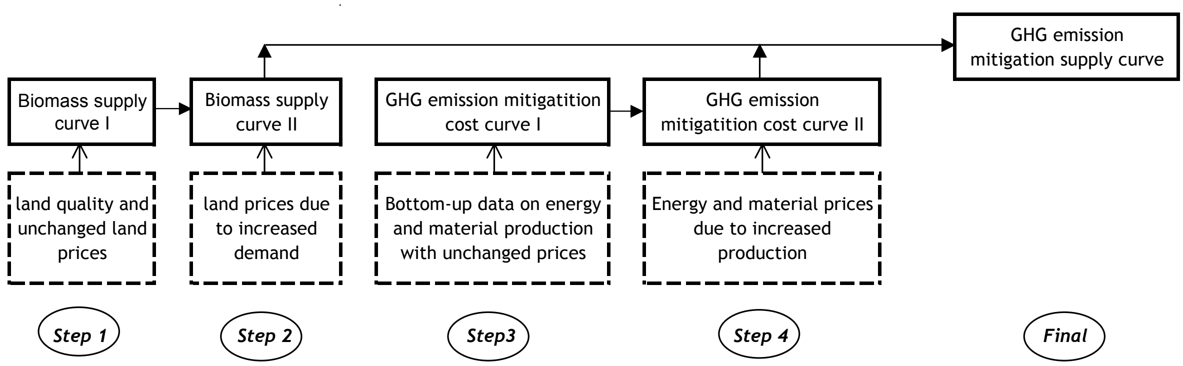


Figure 6.1: Overview of the main steps to calculate GHG emission mitigation supply curves of bio-material and bioenergy applications in which price elasticity effects are incorporated

The first step leads to the *'biomass supply curve I'* describing biomass production costs (€/Mg $_{biomass}$). It is based on the production costs per ha, the available agricultural land of different qualities with different crop yields and land rents. However, with a growing production of biomass for energy and material applications, agricultural land rents rise. In step 2, these increased land rents lead in combination with the production costs to the *'biomass supply curves II'*. Step 3 results in the *'GHG emission mitigation cost curve I'* describing the basic GHG emission mitigation costs (€/Mg CO_{2eq}) of the bio-material and bio-energy applications. These costs are determined from the difference between production costs and market prices of the bio-material applications and from the difference between the GHG emissions of biomass application and a reference application. As for the land rents, an increased production of bio-materials or bio-energy may increase their market prices.[3] With these new market prices of bio-materials and bio-energy and the data from

[3] An increased production of a good typically leads to a decrease in market prices, because selling the additional production necessitates an additional demand and the relation between an increase in demand and market prices is

the GHG emission mitigation cost curve I, GHG emission mitigation costs of the *'GHG emission mitigation cost curve II'* are calculated in step 4. Step 5 then determines the final *'GHG emission mitigation supply curve'* by summing the costs of the biomass supply curve II and the GHG emission mitigation costs II. GHG emission mitigation supply costs are calculated as *marginal costs* for avoiding an additional unit of GHG emissions, see also equation 1. By comparing the marginal GHG emission mitigation costs of the different biomass application and selecting the respective lowest costs at each additional amount of biomass used, an overall GHG emission mitigation cost supply curve can be composed.

$$C_{GHG}\ (S) = (C_{bios}\ (S) + C_{land}\ (S) + C_{bioa} - R_{bioa}(S))/(-GHG_{bios} - GHG_{bioa} + GHG_{sub}) \qquad (1)$$

C_{GHG}: Marginal costs of GHG emission mitigation (€/kg CO_{2eq})
S: Scale of biomass system (kg $_{biomass}$/yr)
C_{bios} (S): Marginal costs of biomass production in relation to scale due to the quality of available land (€)
C_{land} (S): Marginal costs of agricultural land in relation to scale due to land demand (€)
C_{bioa} : Marginal costs of the production of bio-materials and bioenergy (€)
R_{bioa}(S): Revenues of bio-material and bioenergy sales in relation to the market size and their subsequent market prices
GHG_{bios}: GHG emissions during biomass production (kg CO_{2eq})
GHG_{bioa}: GHG emissions during production of bio-materials and bioenergy (kg CO_{2eq})
GHG_{sub}: GHG emissions during production of reference applications that are substituted by bio-materials and -energy (kg CO_{2eq})

Possible biomass supply curves as well as the bio-energy and material markets depend strongly on the trade of food, materials and energy and technological developments in the agricultural sector, which are difficult to predict. To accommodate this variability in a bottom-up calculation, biomass supply curves and market developments are differentiated for four scenarios that reflect possible political developments in Eastern Europe. Following, the methodology used in the various steps of our approach is discussed.

2.1 Biomass supply curve I

The biomass supply curve I is estimated for the production of short rotation coppice (SRC) in Poland in the year 2015.[4] The methodology for the calculation of this biomass supply curve I is summarised in figure 6.2. Food demand and international trade determine the demand for food production in Poland. The available agricultural land in Poland is divided into four different quality categories with subsequent different crop yields and land

usually negative. However, if an additional demand can be created without lowering market prices—e.g. by substituting fossil reference energy carriers—market prices may also stay constant, see Section 2.4.

[4] The methodology and the data used to estimate the biomass supply curves I are based on research on possible future biomass supplies in Central Eastern Europe (CEE) carried out at the Copernicus Institute, Utrecht University. This research is carried out in the context of the European Commission supported research project: *VIEWLS - Clear Views on Clean Fuels, Data, Potentials, Scenarios, Markets and Trade of* Biofuels (NNE5-2001-00619).

costs. These categories are: very suitable (VS), suitable (S), medium suitable (MS) and marginally suitable (mS). This agricultural land is allocated to food production in order to achieve a most efficient land use in terms of total hectares; see Smeets et al. (in press).[5] Following, agricultural land that is available for energy crop production in the four different quality categories, i.e. land that is not used for food production, is determined. From the available land, biomass yields, land costs, and assumptions on the agricultural production system, amounts and costs of biomass production are calculated.[6] The amount and costs of biomass are finally summarised in the biomass supply curve I.[7]

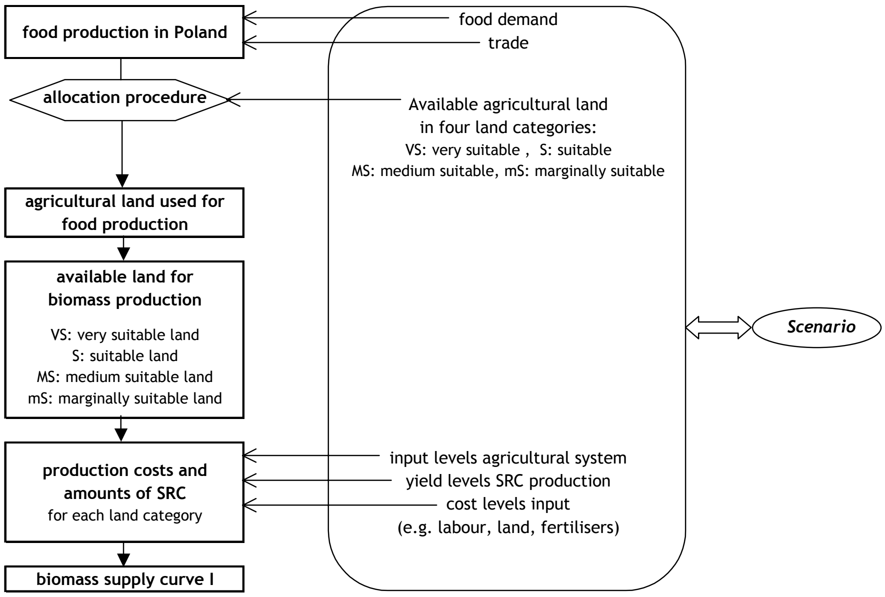


Figure 6.2: Schematic overview of the calculation of biomass supply curves I

Because agricultural production and the demand for food are strongly related to economic and demographic trends, we determine the biomass supply curve I for four different scenarios. These scenarios are related to the SRES emission scenarios (IPCC, 2000) and are translated for Europe. The scenarios differ with regard to an economic (V1, V2) versus an environmental orientation (V3, V4) and with regard to a global (V1, V3) versus a regional orientation (V2, V4). In table 6.1 the main characteristics of these scenarios and the most important assumptions for the calculation of biomass supply curves are presented. (In brackets the most closely related SRES scenario family is indicated.)

[5] Note, that the suitability of land is crop dependent.
[6] In the V1 and V4 scenario, also a small part of agricultural land is reserved for the growth of forestry and urban areas. In the V3 scenario, part of agricultural land (0,5% per land suitability type) is reserved for energy crops before the allocation procedure.
[7] Input data on available land, biomass yields and production costs are summarised in table 6.3 in Section 3.1.

Table 6.1: Characteristic of scenarios for the analysis of biomass supply

Characteristic	V1 (A1)	V2 (A2)	V3 (B1)	V4 (B2)
Main economic characteristic	Fast growing economy, international trade	Slow economy, CEE lacks behind Western Europe	EU economy, scenario based on CAP reforms	EU economy, high level of self-sufficiency in protected market
Trade of agricul. products	Liberal trade on world market	Market oriented CAP reform	Market oriented CAP reform	Self-sufficiency, import reduced substantially
Agricul. production system	High – tech advanced	Intermediate	High input	High –tech advanced
Machinery and labour input	Advanced machinery, low labour input	Current situation CEE	SOTA machinery, low labour input	SOTA machinery, low labour input
Yield levels SRC	+30% of high input system yields	-30% high input system yields	high input system (Data from IIASA)	+30% of high input system yields
Cost level production inputs	Decrease of EU prices (increased competition)	Current cost levels CEE	Current cost levels EU	Increase of EU prices (protected market)
Land costs	Current land rents USA (open market)	Current land rents CEE	Current land rents EU	Increase of EU prices (protected market)
Labour costs	Increased costs (strong economy)	Current cost levels CEE	Current cost levels EU	Increased costs

SRC: short rotation coppice; CEE: Central Eastern Europe; CAP: Common Agricultural Policy of EU; SOTA: State of the Art; IIASA: International Institute of Applied System Analysis

2.2 Biomass supply curve II

The costs of biomass in the initial biomass supply curve I am calculated using fixed land prices differentiated by the quality of the agricultural land and varying per scenario. In the biomass supply curve II, we take into account that the price of agricultural land increases if the demand for land increases, e.g. due the production of biomass.

In an ideal market, the price of a good, e.g. agricultural land, and the demand for it are related negatively. This relation can be described by a demand curve and the ratio between the percentage change of demand and the percentage change of price is the so-called own-price elasticity. This own-price elasticity can vary for different demand levels of a good, but often demand curves are simplified by assuming constant own-price elasticity. This assumption is used in this chapter, too. Equation (2) shows such a demand curve for agricultural land.

Biomass production can be regarded to lead to an additional demand for agricultural land apart from food production. We assume that at the given land rents—used for the calculation of the biomass supply curve I—all agricultural land is used for non-biomass production. Furthermore, it is assumed that in the short term, the amount of agricultural land is fixed and that every increased demand for agricultural land leads to an increased price. The new price of agricultural land can be calculated depending on the own price elasticity of land; see equation (3).[8] This formula describes, in fact, a movement of the original demand curve of agricultural land to a new demand curve. In this new demand curve, the demand of land for a certain price is higher than in the original one, and the difference is about the additional demand of land due to biomass production. For the fixed amount of available agricultural land, the price on this new demand curve then is higher than in the original demand curve; see chapter 5 of this thesis.

$$P_{\mathrm{L}\text{-}curr} = C_L * Q_{\mathrm{L}\text{-}T}{}^{1/\varepsilon L} \qquad (2)$$

$P_{\mathrm{L\text{-}curr}}$: current price of agricultural land rents [€/(ha*yr)]
C_L: Constant [€*(ha*yr)$^{(-1/\varepsilon L-1)}$]
$Q_{\mathrm{L\text{-}T}}$: total amount of agricultural land available per year [ha*yr]
εL: own-price elasticity of agricultural land

$$P_{\mathrm{L}\text{-}new} = P_{\mathrm{L}\text{-}curr} * (Q_{\mathrm{L}\text{-}T}/(Q_{\mathrm{L}\text{-}T} + Q_{\mathrm{L}\text{-}add\text{-}bio}))^{1/\varepsilon L} \qquad (3)$$

$P_{\mathrm{L\text{-}new}}$: new price of agricultural land rents [€/(ha*yr)]
$Q_{\mathrm{L\text{-}add\text{-}bio}}$: additional demand for land due to biomass production [ha*yr]

2.3 GHG emission mitigation cost curve I

For each of the biomass applications, i.e. MDF, PLA, methanol and electricity, the GHG emission mitigation cost curve I is calculated. The GHG emission reduction is determined by comparison of the biomass application system with a non-biomass reference system. In the reference system, the same functions are fulfilled as in the biomass system. Costs are calculated from the difference between the production costs of the bio-materials and bio-energy carriers and their market prices. This approach to calculate GHG emission mitigation costs and the type of input data necessary has been demonstrated in Chapter 5 of this thesis. For the GHG emission mitigation cost curves I, the market prices are fixed. Because we are mainly interested in the market effects of bio-material and bio-energy introduction instead of in the development of biomass applications, no scenario-dependent technology developments or subsidies are taken into account. Also other dynamics such as technological learning and the developments of new markets during large-scale implementation of biomass technologies are not considered; the timeframe until the year 2015 is too short for these effects to be pronounced and, moreover, world markets in general are hard to predict.

---

[8] In this chapter, different land quality classes are used for biomass production. As it is assumed that a demand for agricultural land in any of the classes will lead to increased prices on the whole land market, the increase of the average land price is calculated. From the new average land price and the ratio between the current average land price and the current land price of the land class, the new land price of the land class is calculated.

Figure 6.3 gives an overview of the biomass and the references systems considered in this study. For agricultural land, the biomass supply curves are based on the assumption that necessary food/fodder is produced and that on the remaining land biomass can be produced for other purposes. Therefore, it is assumed that the reference land use of agricultural land is set aside. In the reference system, forestry land is used for the production of plywood, which is utilised as construction material. However, if in the biomass system agricultural land is used to produce an alternative construction material, i.e. MDF board, the conventional forestry land is assumed to produce wood for electricity production.[9]

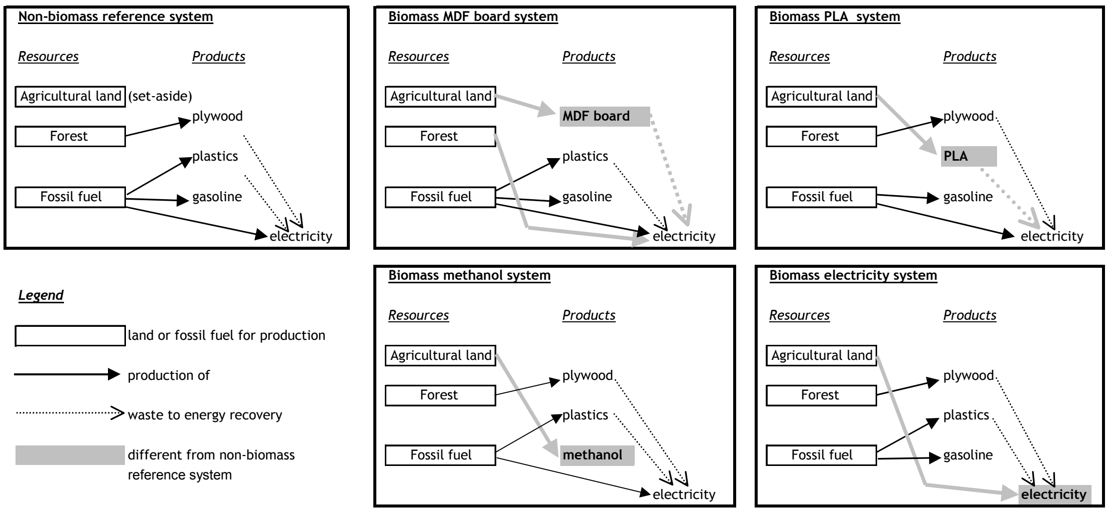


Figure 6.3: Selected biomass systems and the non-biomass reference system to determine GHG emission reductions

Bio-materials in the biomass system are produced and used in a 'multi-functional' way. This means that besides the bio-materials, bio-energy is produced by the utilisation of residues and waste materials. Both bio-materials and bio-energy carriers are then compared to reference materials and energy carriers. In the *MDF biomass system*, MDF board is produced from ligno-cellulosic biomass. This MDF board replaces plywood from softwood produced in conventional forestry on a board volume basis. After utilisation both plywood and MDF board are incinerated with electricity recovery; see Chapter 3 of this thesis for a description of MDF and plywood production. In the *PLA biomass system* bio-based PLA replaces polyethylene (PE) on a weight basis; see also Chapter 4 of this thesis for a description of PLA and PE production. Like in the MDF production system it is assumed that

[9] In fact, the forestry land could be used for the production of other forest products, for the production of energy, converted to agricultural land or not harvested at all. With our assumption of electricity production, the material and energy carriers in the different systems can be easily compared.

both PLA and PE are incinerated for electricity recovery after use. Finally, in the *electricity and methanol biomass systems*, electricity from the grid and gasoline are replaced on the basis of energy content of the energy carriers.

2.4 GHG emission mitigation costs II

In the GHG emission mitigation cost curve II, market price changes of materials and energy carriers are incorporated. The lower the market price of a good, the larger the demand for this good. In an ideal market, the market price is equivalent to the costs of supply and the quantities sold are equivalent to the demand of the respective market price. However, if a larger amount of a good is produced, e.g. through government interventions to reduce GHG emissions, the market price of the good will decrease until the demand equals the amount of production.

This decrease of market prices applies for the bio-materials investigated in this study, i.e. MDF board and PLA. However, for bio-energy carriers, i.e. methanol and electricity, the situation is different. Currently, a large amount of transportation fuels and electricity are produced from other (mainly fossil) resources. If bio-energy is produced, it substitutes the production from these sources.[10] Thus, the production of bio-energy carriers does not lead to an increased supply of energy carriers. As a consequence, market prices stay constant. However, if no more energy carriers from other sources can be replaced, because the total energy consumption is supplied from biomass, increased production of bio-energy carriers leads to a decrease of market prices. This decrease of market prices is described in a similar way as the decrease of market prices for bio-materials. The consequences of this approach are evaluated in the sensitivity analysis.

The decrease of market prices due to an increased supply of a good can be described by a simplified demand curve with constant own-price elasticity, see Section 2.2. If the amount of bio-material or bio-energy production is increased, the new market price of the good $P_{G\text{-new}}$ can be calculated from the total current market volume of the commodity $Q_{G\text{-T}}$, the current price $P_{G\text{-curr}}$, the own-price elasticity εG and the additional production of the commodity $Q_{G\text{-add-bio}}$; see equation (4). Also this approach has been described in Chapter 5 of this thesis.

[10] This substitution depends of course on whether bioenergy carriers can compete with alternative energy carriers. In a real market, the amount of bioenergy carriers produced will be limited to the amount that can compete with other energy sources. However, in this study we want to illustrate the effect of increased bioenergy production on GHG mitigation cost and, therefore, higher amounts of bioenergy production and substitution of alternative energy carriers are assumed.

$$P_{G\text{-}new} = P_{G\text{-}curr} * ((Q_{G\text{-}T} + Q_{G\text{-}add\text{-}bio})/Q_{G\text{-}T})^{1/\varepsilon G} \quad (4)$$

Thus, to estimate the new market price of a bio-material or bio-energy carrier at an increased production level, we need to know three parameters, i.e. the elasticity, the total current market volume of the commodity and the current market price of the commodity.

The own-price elasticity of a good refers to a specific market, i.e. the type of good (e.g. agricultural land, forestry land) and the size of the market (e.g. only Poland or European Union). Own-price elasticity is often derived by econometric analysis from historical data on quantities sold on the market and their prices; see e.g. van Driel et al. (1997). Two main uncertainties are inherent to this methodology. First, factors that influence the own-price elasticity—like the availability of goods for substitution or income—may be different in the future. Second, often the historical data used do not refer to the specific market investigated in our analysis referring to other geographical scopes or products. For example, food demand may have been investigated instead of agricultural land demand, or global demands for gasoline may have been analysed instead of gasoline demand restricted to the European Union.

The total current market volume and the current market price depend on the assumptions about market size and trade. These assumptions are made for the different scenarios in accordance to the assumptions made for the production of biomass; see Section 2.1. Moreover, for each biomass application considered, assumptions are adapted to the specific market of that application; see Table 6.2. MDF board is currently traded globally. However, regional markets for forest products differ, as can be seen from the differences in market prices; see FAO (2004). Therefore, assumptions on MDF markets in the scenarios follow the assumptions for food markets in Section 2.2, i.e. a world market in scenario V1 and V3 and a limited market to Central Eastern Europe and the EU-25 in scenario V2 and V4. Plastics and transportation fuels are typically traded on a global market with global market prices. Because these commodities are usually produced from crude oil, a limitation of markets to Europe seems unrealistic. As a consequence, a world market is assumed for PLA and methanol in all scenarios. Electricity, finally, is traded on regional markets, for example within Europe, due to transportation constraints. As largest market, therefore, a limited market to the EU-25 is assumed in the V1, V3 and V4 scenario. In the V2 scenario, the market is limited to Central Eastern Europe. Input data for market volumes and prices are discussed in Section 3.

Finally, the market volumes of Polish bio-material and bio-energy production also depend on assumptions about the development of markets in other countries. While the additional production of biomass application in Poland is analysed, other countries may also increase their production of biomass application. For bio-materials, this increased production of all countries in the respective market is the additional amount of production leading to a changed market price. For bio-energy carriers, the production of all these countries replaces alternative energy carriers and, finally leads to a decrease of market prices. It seems unrealistic to assume that the growth of bio-material or bio-energy production is exclusively limited to Poland. Therefore, we assume that all countries increase their production of biomass applications. As an approximation for the share of Poland of this increased production, we assume that the current market share of Poland for a certain good stays constant.

Table 6.2: Scenarios and their influence on the biomass energy and material markets

	V1 (A1)	V2 (A2)	V3 (B1)	V4 (B2)
Basic characteristic	Fast growing economy, international trade	Slow economy, CEE lacks behind Western Europe	EU economy, scenario based on CAP reforms	EU economy, high level of self-sufficiency in protected market
MDF market	World	CEE	World	EU-25
PLA market	World	World	World	World
Methanol market	World	World	World	World
Electricity market	EU-25	CEE	EU-25	EU-25

3 Input data

3.1 Biomass supply and land markets

For the calculation of food production in the different scenarios, background data on food demand, GDP growth and trade are data from SRES scenario projections (IPCC, 2000; RIVM, 2001) combined with projections from the FAO on food demands and GDP growth in Eastern Europe (FAO, 2003). Yield data of agricultural crops on a grid cell level (50 km x 50 km) are based on data from IIASA combined with agricultural production data from FAO and EUROSTAT statistics.

Key parameters for the production of biomass, i.e. short rotation willow, are crop yields, amounts of suitable agricultural land in Poland, land rents and biomass production costs; see Table 6.3.[11] Crop yields depend on the intensity of the production system in the respective scenarios. Base data on crop yields are taken from studies of Nonhebel (2002), Londo et al. (2004) and REU and FAL (1996). The suitable areas for energy crop

[11] The input data discussed here are also used in research carried out at the Copernicus Institute, Utrecht University in the project *VIEWLS - Clear Views on Clean Fuels, Data, Potentials, Scenarios, Markets and Trade of Biofuels.* A more detailed report on biomass supply curves is forthcoming.

production are also from IIASA and have been adapted to water stresses for willow production. Land rents for Europe and the U.S. are taken from Eurostat (2003) and USDA (2003), while Polish land rents are obtained from the Institute of Agricultural and Food Economics (IAFE) in Warsaw. Finally, production costs are calculated from a reference case of willow production on a current input level in Poland (EC-BREC, 2004). These production costs are adapted to different intensities of agricultural production and different qualities of agricultural land by assumptions on the amount of agricultural production inputs used, e.g. fertilisers based on Kaltschmitt and Reinhardt (1997) and labour based on ILO (2001).

Table 6.3: Key input data on biomass production in the different scenarios

	Short rotation wood yield	**Suitable agricultural land for biomass production**[a]	**Land rents**	**Production costs**[b]
	Mg/ha*yr	Million ha	€/(ha*yr)	€/(ha*yr)
Scenario V1				
Very suitable land (VS)	15.0	7.20	116	314
Suitable land (S)	11.2	4.58	54	267
Medium suitable land (MS)	7.6	3.01	43	233
Marginally suitable land (mS)	2.4	2.20	25	166
Scenario V2				
Very suitable land (VS)	10.5	8.43	113	112
Suitable land (S)	7.9	3.15	35	89
Medium suitable land (MS)	5.3	3.44	29	68
Marginally suitable land (mS)	1.7	3.58	10	39
Scenario V3				
Very suitable land (VS)	12.8	7.74	165	496
Suitable land (S)	9.5	4.16	111	433
Medium suitable land (MS)	6.5	3.03	100	374
Marginally suitable land (mS)	2.1	2.13	84	303
Scenario V4				
Very suitable land (VS)	15.0	7.45	235	680
Suitable land (S)	11.2	4.41	177	601
Medium suitable land (MS)	7.6	2.93	164	528
Marginally suitable land (mS)	2.4	2.20	145	443

VS very suitable land, S: suitable land, MS: medium suitable land, mS: marginally suitable land.
[a] Total amount of agricultural land without subtracting land for food demands.
[b] Production costs stated here exclude land rents. The production costs are based on data from EC-BREC (2004) in which production costs are about 281 €/(ha*yr) and have been adapted for the different scenarios, characterised by different land qualities and production systems. Main assumption is that the intensive production systems require a high input of machinery and relatively less labour input. The V2 scenario is based on the current situation in Poland. Input data for the total production costs as presented in this table are: (1) interest rates, ranging from 6% for V1 to 4% for V2 (Eurostat, 2004); (2) rotation periods (ranging from 21 years for VS land for V1 and V4 to 25 years for S land for the V2 scenario) and harvest cycle (G. Kunikowski EC-BREC, personal communication 2004; Szczukowski et al., 2002a, Larsson and Lindegaard, 2003); (3) fertiliser use, which is related to yield levels based on the formulas from Kaltschmitt and Reinhardt (1997); (4) cuttings per ha, ranging from 18.000 cutting / ha for S land for the V1 and V4 scenario to 12.000 cuttings / ha for S land for the V2 scenario (Ledin, 1996; DEFR, 2003; Szczukowski et al., 2002a); (5) pesticide use is based on Szczukowski et al. (2002a), Kaltschmitt and Reinhardt (1997) and Stańczyk and Ludwik (2003), and only differentiated for V2 scenario assuming a decrease of inputs; (6) fertiliser costs are € 0.44 / kg for V2 (current cost level in Poland), € 0.52 for V1 (assumption is that cost levels go down with an open market compared to average EU price level

because of increased competition), € 0.60 / kg for V3 (average EU price level) and € 0.75 / kg for V4 (assumption is that cost levels increase because of decrease of competition for European manufacturers), ranges are based on data from EC-BREC (2004) and Eurostat (2003); (7) pesticide costs range from € 3.37 to € 13.28 / litre for *Roundup*, based on the same assumptions as mentioned for fertiliser costs, data are from Stańczyk and Ludwik (2003), PAV (2000) and G.V. Roman (University of Agronomic Sciences and veterinary Medicine in Bucharest, personal communication 2004); (8) labour costs range from € 2.52 / hour for V2 scenario (current wages in Poland) to € 12.22 / hour for V3 scenario (average EU level) and € 14.63 for V1 and V4 scenario (increase compared to average EU level because of strong economy and more efficient production system), data are from ILO (2001) and EC-BREC (2004); (9) Machinery and labour input for harvesting are based on data from EC-BREC (2004), WSRC (2004) and Ledin (1996) and differentiated per scenario based on yield levels and costs for wages and machinery. Input data range for machinery from 3.75 €/t_{dm} per rotation for V2 to 11.06 €/t_{dm} per rotation for the V1 and V4 scenario); (10) Insurance and miscellaneous costs are based for the V3 scenario on data from EC-BREC (2004), assuming a 10% increase for the V1 and V4 scenario and a 5% decrease for the V2 scenario.

Finally, the available agricultural land for energy production depends on the suitability of land for energy crop production and the amount of land that is already used for food production. The available agricultural land for willow production in the different scenarios is summarised in Figure 6.4 (as resulting from the bottom-up approach described in Smeets et al., in press).

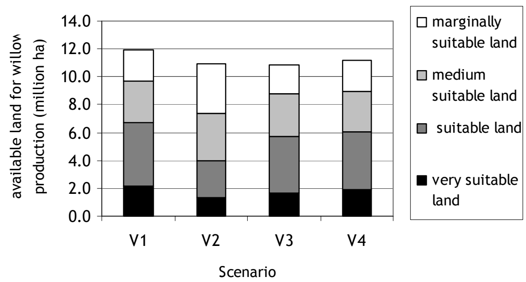


Figure 6.4: Projected available agricultural land in 2015 for the production of biomass (i.e. short rotation wood) after subtracting land for food demands

In the analysis of land markets, new rents of agricultural land are calculated. Input data for these calculations are the base rents, the total amount of agricultural land (see Table 6.3) and the elasticity of agricultural land. In scientific literature, no estimates for own-price elasticity of land could be identified. Rather, own-price elasticity of food is used for the agricultural sector; see Ciaian et al. (2002). This is due to the fact that the demand for agricultural land is closely related to the demand for food. Estimates for the own-price elasticity of food vary considerable.[12] Based on the results of Finke et al. (1984), Ciaian et al. (2002) estimates elasticities of –0.18, -0,24 and –0.3 for the Czech Republic, Poland and Slovakia, respectively. We use the medium value of –0.24 as estimate for Central Eastern Europe in the V2 scenario. In the V4 scenario, we assume a slightly higher value of about –0.2 as presented for the Netherlands by van Driel et al. (1997) to be applicable for the EU-25. Finally, no global estimates for food demand could be identified. As in the global V1

[12] Finke et al. (1984) estimate elasticity factor between -0.03 and 0.64. For the U.S., the authors estimate an elasticity of –0.03, while Driel et al. (1997) determine this elasticity to be –0.45.

scenario current U.S. land rents are assumed, we assume an elasticity factor for food demand in the U.S. of –0.45 (van Driel et al., 1997) in the global land markets of the V1 and V3 scenario.

3.2 Biomass applications and their markets

In this section, input data on the four bio-material and bioenergy systems i.e. medium density fibreboard (MDF), poly lactic acid (PLA), methanol and electricity production are described. Key parameters for the characterisation per unit of biomass application are the input of short rotation wood, the production costs and the GHG emission reduction in relation to the reference system; see Table 6.4. Furthermore, MDF and PLA and their reference materials, i.e. plywood and PE, are converted to electricity after their use. The resulting electricity is assumed to replace electricity from the grid and, thus, to contribute to GHG emission reduction and revenues from sales.

Data on material and energy markets are summarised in Table 6.5. These are market volumes, market prices and Polish market shares of the biomass applications based on statistics and own-price elasticities. While the growth of GDP may lead to developments of the consumption of materials and energy carriers in 2015, in this chapter no changes of material and energy demands are assumed, but current market volumes are used. On the one hand, this is because the demand for materials and energy carriers does not develop one by one with the growth of GDP, since energy efficiencies increase. Also the growth of GDP may cause sectoral changes influencing material and energy intensities, see Groenenberg (2002). On the other hand, keeping market volumes of materials and energy carriers constant, enables us to investigate market effects on GHG emission mitigation costs without disturbing influences. However, in the sensitivity analysis the influence of this assumption is evaluated, see Section 4.5.

In general, GHG emissions are calculated with carbon emission factors representing indirect and direct greenhouse gas emissions of average European energy use in 2000. [13] For fuel, the average EU oil product mix for production of 83 kg CO_{2eq}/GJ and for electricity, the average EU mix of electricity production of 126 kg CO_{2eq} /GJ are used (UBA, 2003). The GHG emissions of biomass production are derived from production inputs of short

[13] GHG emission factors of energy use vary within geographical regions and are likely to change in the future. Moreover future specific GHG emissions are depending on economic developments and governmental policies and are, therefore, scenario specific; see e.g. Bollen et al. (2004). In our analysis, GHG emission factors of energy use are kept constant in order to investigate market effects on GHG emission mitigation costs without disturbing influences of varying GHG emission factors.

rotation willow production in Poland (Szczukowski et al., 2000a) and generic GHG emissions for machine uses, fertilisers, etc. mainly from a study from Biewinga and van der Bijl (1996). The resulting GHG emissions are about 0.23 kg GHG per ha; see Chapter 2 of this thesis for a more detailed description of GHG emissions from biomass production.[14]

MDF

The input of wood for MDF is based on a study of hardboard (Richter et al., 1995). Concerning GHG emission reduction, it is assumed that MDF replaces plywood with the volume of board as functional unit.[15] The GHG emission reduction is then calculated from the GHG emissions during plywood and MDF production[16] (Kaltschmitt and Reinhardt, 1997; Forintek, 1993; Hekkert and Worrell, 1998; Haygreen and Bowyer, 1996). Also, the GHG emissions that could be saved if forestry wood for plywood production would be used for electricity production are added to the GHG emission reductions. Production costs of MDF are estimated to be the difference between the prices of the raw material, i.e. wood chips, and the export prices of MDF board in Europe derived from statistics on wood products trade (FAO, 2004). Data on the production and GHG emission reduction of MDF as summarised in Table 6.4 have already been discussed in a study of short rotation wood cascading; see Chapter 3 of this thesis.

The market volume, i.e. the consumption of MDF boards is currently (in 2002) about 23.3 million m^3 in the world, 8.0 million m^3 in Europe and 1.5 million m^3 in Central Eastern Europe (FAO, 2004). Market volumes in Table 6.3 are converted to Mg with an average density of MDF board of 0.65 Mg/m^3 (Haygreen and Bowyer, 1996). Polish market shares of MDF board production in comparison to MDF board consumption have been 5.3% globally, 15.6% in whole Europe and 84.6% in Central Eastern Europe (FAO, 2004). Market prices are based on the import prices for fibreboard in the world, Europe and Central Eastern Europe, respectively, in 2002 (FAO, 2004).[17] The own-price elasticity of MDF board demand has been estimated from historical MDF board import prices and consumption by regression analysis; see Figure 6.5. These data are available for 1995—2002 (FAO, 2004).

---

[14] Different biomass production systems for Poland, i.e. 'current input', 'high input' and 'high advanced input' are assumed in the scenarios. These production systems use different levels of inputs per hectare, e.g. machinery and fertilisers, but also lead to different levels of short rotation willow yields per hectare. Due to a lack of data on GHG emissions of the various production systems, it is assumed that GHG emission per unit of biomass produced is constant. This assumption can be justified by findings for miscanthus for which the share of energy input (including drying) at the end use energy varies only about 8 – 14% for different production systems (Lewandowski and Heinz, 2003).
[15] This is equivalent to 1 kg of MDF board replacing about 0.71 kg of plywood (Haygreen and Bowyer, 1996).
[16] Data on MDF board and plywood production take into account that processing residues are used for process heat generation.
[17] Exchange rate used for conversion to € is the average rate in 2002 of 1.06 US$/€.

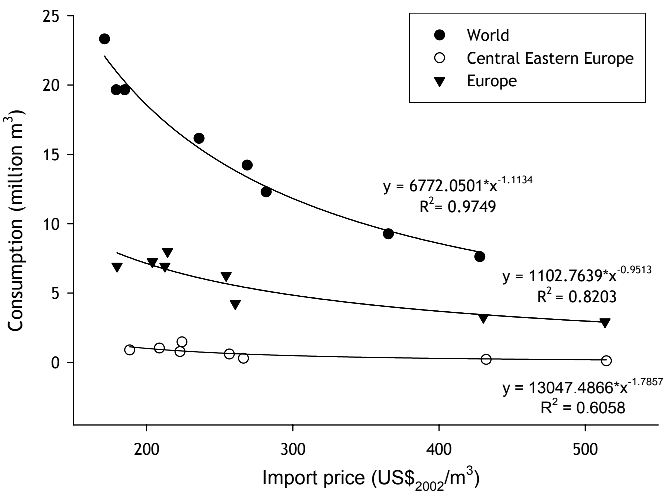


Figure 6.5: Historical data on demand and prices of MDF board used for the estimation of own-price elasticity; data from FAO (2004)

PLA

The efficiency of PLA production is based on current ligno-cellulose pre-treatment (Wooley et al. 1999; Mc Aloon et al., 2000) and current estimations about PLA production from ligno-cellulose (Vink et al., 2003). It is assumed that 1 kg of PLA replaces 1 kg of HDPE. GHG emissions are the emissions of PLA production caused by energy uses (Vink et al., 2003) but accounting for the fact that lignin from short rotation wood is used for process heat and electricity production. Subsequently, net emissions are determined by subtracting GHG emissions of HDPE production APME (2003). Production costs of PLA are taken from projections of Cargill Dow planning to produce PLA from corn stover (Crank et al., 2004). Because HDPE has a much higher heating value than PLA, the net electricity recovery from waste is negative for PLA production. All analyses on energy and GHG balances of PLA production are reported in a study of a PLA bio-refinery; see Chapter 5 of this thesis.

The global market volume of PLA is the current global production of PLA in 2003 of about 0.14 million Mg (Crank et al., 2004). Current global market prices are taken from a study on bio-based polymers (Crank et al., 2004). At the moment no PLA is produced in Poland. Therefore, we use Polish market shares of polyethylene production as fictive market shares of PLA production (UN, 2002). Because the production of PLA (and other bio-based polymers) is a rather new development, no historical data on market prices and production volumes are available. However, as PLA has the potential to substitute PE on a large scale, the own-price elasticity of PE is used in this analysis for PLA. This own-price elasticity is estimated from historical figures of global PE production and market prices by regression; see Figure 6.6 (UN, 2002; Crank, et al., 2004).

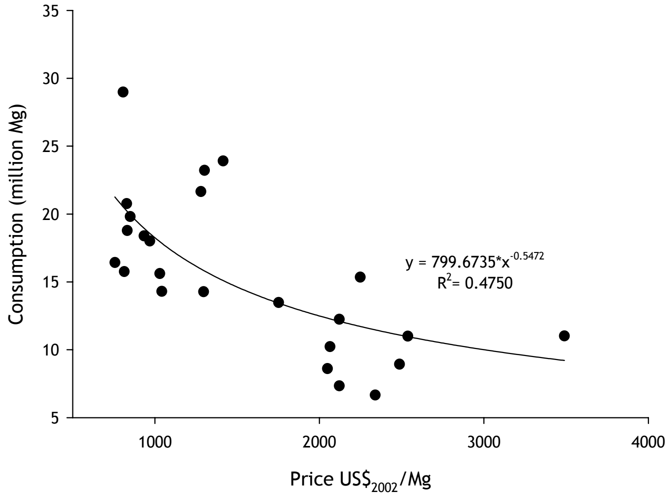


Figure 6.6: Historical data on global demand and prices of Polyethylene used for the estimation of own-price elasticity; data from UN (2002) and Crank, et al. (2004).

Methanol

Data on methanol production refers to an advanced methanol production concept without electricity co-production and a conversion efficiency of 57% (HHV) as investigated by Hamelinck and Faaij (2002). Production costs include distribution costs in order to make the production costs comparable to the market prices for end-users used in this analysis. [18] To determine GHG emission reduction, it is assumed that 1 GJ_{LHV} methanol replaces 1 GJ_{LHV} gasoline.

The methanol is assumed to be sold for market prices of gasoline in advanced economies without taxes.[19] The market volume for methanol is the global consumption of petroleum products for road transport, i.e. 58.8 EJ in 2000 (IEA, 2003a). In 1999, the Polish gasoline production was about 0.3 EJ (UN, 2002).

[18] The methanol production technology considered is currently under development and is likely to be commercial in 2015-2020. The selected concept uses an atmospheric indirectly fired gasifier, wet gas cleaning, steam reforming and liquid phase methanol production (Hamelinck and Faaij, 2004). Production costs are based on an interest rate of 5%, a scale of 400 MWth, a base load of 8000 h/yr, an economical lifetime of 15 years and a technical lifetime of 25 years. Assuming biomass costs of 2 US$/$GJ_{HHV}$, methanol production costs of 7.2 US$/$GJ_{HHV}$ result (Hamelinck and Faaij, 2004). Distribution costs of 2.1 €/GJ_{HHV} (Hamelinck, 2004) are added to the production costs.

[19] Market prices are averaged from prices in USA, Canada, Japan, France, Germany, Spain, Italy and the UK in 2003-2004 (IEA, 2004b).

Espey (1998) compares more than 300 estimates of short-run own-price elasticity of gasoline demand, i.e. describing the relation between prices and demand in the short to medium term. These estimates are derived from economic models as well as time series analysis. The median of all estimates is –0.23 and is used in our analysis.[20]

Electricity

The data for electricity production are based on state-of-the-art IG/CC plant (about 150 MW_e) with a net electric efficiency (LHV) of 43.5% (Faaij et al., 1998). Also for electricity, production costs include distribution costs.[21] Electricity from short rotation wood replaces electricity from the European grid on a kWh basis.

The market volumes of electricity are the consumption of electricity in the year 2000, in the EU-25 and Central Eastern Europe, respectively (IEA, 2002a; IEA, 2002b). Market prices for electricity vary between consumers and countries.[22] In our analysis we use about average market prices without taxes that apply to large-scale household and medium industrial-scale users. In the EU-25 these are about 0.07 €/kWh and in Central Eastern Europe about 0.05 €/kWh (Goerten and Beranek, 2004a and 2004b). Polish electricity consumption in the year 2000 was about 0.35 EJ_e (IEA, 2002a).

Own-price elasticity estimates in economic literature are usually given for limited markets but not for whole regions as the EU-25 or global. However, estimates for elasticity of electricity in scientific literature are in a comparable range. Kamerschen and Porter (2004) estimate the elasticity of total electricity demand in the U.S. between –0.13 and –0.15 and find that this value is within the same range as other US based studies. Wolfram (1999) analyses the British spot market, concluding that the data suggest a price elasticity of approximately –0.1. SEO (1998) investigates industrial electricity use in the Netherlands resulting in an own-price elasticity of –0.2. Given this information, own-price elasticity of electricity is not differentiated between the scenarios, but a value of –0.15 is assumed.

---

20 The estimates of the short-term elasticity range from 0 to –1.36 (Espey, 1998).

21 The lower heating value of short rotation wood is assumed to be 17 GJ/Mg. Production costs of electricity are calculated from the investment costs of the IG/CC plant, i.e. 1.97 million €/MW_e investment costs (Faaij et al., 1998), a lifetime of 25 years, an interest rate of 5% rent and a load factor of 80%. Also for electricity, distribution costs are added to the production costs as market prices used are for end-users. In Western Europe, i.e. in DK, D, F, the Netherlands, UK distribution costs for large-scale consumers are about 0.01-0.02 €/kWh (ECN, 2001). In this study, distribution costs of 1.5 cent/kWh (4.2 €/GJ) are assumed.

22 For example, in the EU-25 market prices without taxes vary from 0.035 €/kWh for large-scale industrial users in Latvia and 0.350 €/kWh for small-scale household users in Norway (Goerten and Beranek, 2004a and 2004b).

Table 6.4: Input data of biomass options for the use of SR wood to reduce carbon emissions

	MDF		PLA		Methanol		Electricity	
SR wood input (dm)	1.3	kg_{wood} /kg	0.68	kg_{wood} /kg	0.10	kg_{wood}/GJ	0.13	kg_{wood}/GJ
GHG em. reduction	0.82	kg $CO_{2\,eq}$/kg	3.97	kg $CO_{2\,eq}$/kg	42.6	$kgCO_{2eq}$/GJ	94.2	$kgCO_{2eq}$/GJ
Production costs[a]	250	€/kg	1210	€/kg	6.2	€/GJ	14.6	€/GJ
Electricity recovery[b]	1.62	GJ_e/kg	-7.65	GJ_e/kg	-		-	

[a] Production costs are without costs of biomass inputs, because biomass costs vary within the different scenarios.
[b] In the A1 and B2 scenarios, electric efficiency of waste incineration is 30% (LHV), which is State-of-the-Art in Europe. Lower heating values are 15 GJ/Mg for MDF, 13.5 GJ/Mg for plywood, 43.4 GJ/Mg for HDPE and 17.9 GJ/Mg for PLA.

Table 6.5: Input data of biomass options for the use of SR wood to reduce carbon emissions

	MDF	PLA	Methanol	Electricity
Market volume				
V1	15.2 million Mg (World)	All scenarios:	All scenarios:	8.9 EJ (World)
V2	1.0 million Mg (CEE)	0.14 million Mg (World)	58.8 EJ (World)	1.3 EJ (CEE)
V3	15.2 million Mg (World)			8.9 EJ (World)
V4	5.2 million Mg (EU-25)			8.9 EJ (World)
Market price				
V1	279 €/Mg	All scenarios:	All scenarios:	19.4 €/GJ
V2	453 €/Mg	3000 €/Mg	8.9[b] €/GJ	13.8 €/GJ
V3	279 €/Mg			19.4 €/GJ
V4	366 €/Mg			19.4 €/GJ
Market share Poland				
V1	5%	All scenarios:	All scenarios:	4%
V2	85%	0.3%	0.5%	27%
V3	6%			4%
V4	16%			4%
Own-price elasticity				
V1	-1.11	All scenarios:	All scenarios:	All scenarios:
V2	-1.79	-0.55	-0.23	-0.15
V3	-1.11			
V4	-0.95			

4 Results

For each scenario, all steps of the calculation, i.e. the biomass supply curves I+II, the GHG emission mitigation costs curves I+II and the resulting GHG emission mitigation supply curves for PLA, MDF board, methanol and electricity are shown in figure 6.7 to 6.10.

4.1 Biomass supply curves

The biomass supply curves I show biomass production costs in Poland. In the V1 scenario, a large amount of biomass of about 88 million Mg_{dry} is available at relatively low prices below 36 €/Mg_{dry}.[23] Production costs, like labour and land, are even cheaper in the V2 scenario in which economic development of Eastern Europe stagnates. However, as yields are

[23] 1 Mg_{dry} of biomass, i.e. short rotation willow has a higher heating value of about 18.4 GJ/Mg. Biomass costs of about 2 €/GJ_{HHV} are, therefore, equivalent to about 36.8 €/Mg_{dry}.

also lower and food production is in principle self-sufficient, a total amount of biomass of 59 million Mg_{dry} are available at costs below 28 €/Mg_{dry}. In the V3 scenario biomass production is quite expensive as levels of input, i.e. machinery, labour, fertilisers, etc. and their costs per unit are high. Thus, only 21 million Mg_{dry} of biomass can be produced at lowest possible costs of about 51 €/Mg_{dry}. The V4 scenario is characterised by even higher production costs and high average land rents. As a consequence, the lowest biomass production costs are 60 €/Mg_{dry} for which about 28 million Mg_{dry} of biomass are available.

The biomass supply curves II, also take into account that land rents will increase if biomass for material and energy is produced on a larger scale. Adding the increase of land costs to the biomass supply curves increases the biomass production costs considerable, especially at large scales of biomass production. The higher the basic land rents in a scenario, the higher is the increase of land rents by increasing biomass production. For example in the V2 scenario with low land rents, the increase of biomass production costs is relatively small, not exceeding 70 €/Mg_{dry} even at large scale.

4.2 GHG emission mitigation cost curves

The GHG emission mitigation cost curves I show these costs not considering variable market prices of products and land. Also, biomass supply curves are not taken into account for the calculation of the GHG emission mitigation cost curve I. Thus, the results are indifferent to the volumes of biomass application produced and represent the GHG emission mitigation costs without the possible influences of a large scale introduction of biomass material and energy system.

The GHG emission mitigation cost curves I are depicted for the fixed amount of available biomass given in the biomass supply curves. PLA production has the technical potential to avoid by far the largest amount of GHG emissions using the available biomass. In decreasing order, MDF board production, electricity production and methanol production have lower technical potentials to avoid GHG emissions.

PLA production at current market prices results in all scenarios in low GHG emission mitigation costs of about –500 €/Mg CO_{2eq}. In the V2 scenario, high market prices of MDF board and low market prices of electricity in a Central Eastern European market are assumed. These assumptions result in GHG emission mitigation costs of about

–200 €/Mg CO_{2eq} for MDF production. [24] In the V4 scenario, biomass production costs are assumed to be relatively high. As a result GHG emission mitigation costs of electricity production are with 100 €/Mg CO_{2eq} relatively high, too. The remaining options in the different scenarios have GHG emission mitigation costs of around zero.

The GHG emission mitigation cost curves II show the influence of material and energy markets on these costs by changing bio-material and bio-energy prices depending on the volume produced.

In all scenarios, the GHG emission mitigation costs of PLA production first increase strongly and then stay nearly constant, with increasing amount of GHG emissions avoided. This is due to the relative small market volumes of PLA. With increasing PLA production, market prices of PLA decrease rapidly until the demand curve becomes nearly constant. [25] For MDF board, methanol and electricity, a similar but less pronounced effect can be observed in figures 6.7 to 6.10. The smaller the market volumes of the bio-material or bio-energy carrier are, the larger is the increase of GHG emission mitigation costs with the amount of GHG emissions avoided. For methanol and electricity, however, in the first part of the curves II, the GHG emission mitigation costs are constant, because it was assumed that alternative energy carriers are replaced at constant market prices; see Section 2.4. In all scenarios, the amount of GHG emissions that can be avoided without an increase of GHG emission mitigation is larger for electricity than for methanol.

4.3 GHG emission mitigation supply curves

In the GHG emission mitigation supply curves, costs of biomass production from the biomass supply curve II are combined with costs of the GHG emission mitigation cost curves II. Only GHG emission mitigation costs curves up to 300 €/Mg CO_{2eq} are depicted, as measures with higher costs are very unrealistic to be implemented.[26] Because MDF, PLA, methanol and electricity production technologies use different amounts of biomass per unit of GHG emission reduction, the influence of the biomass production costs on these options varies.

[24] Negative GHG emission mitigation costs result from the fact, that revenues from material and energy sales are higher than production costs of these materials and energy carriers.
[25] This increase is not visible at the scales in Figure 6.7 to 6.10, but the GHG emission mitigation costs appear as constant.
[26] According to the IPCC, costs of promising GHG emissions reduction options are at present in a range of up to 60 US$/Mg CO_{2eq} (IPCC, 2001b). However, at costs of 60 US$/Mg CO_{2eq}, only a small part of the total GHG emission mitigation supply curve would be visible. Therefore, costs of up to 300 €/Mg CO_{2eq} are shown.

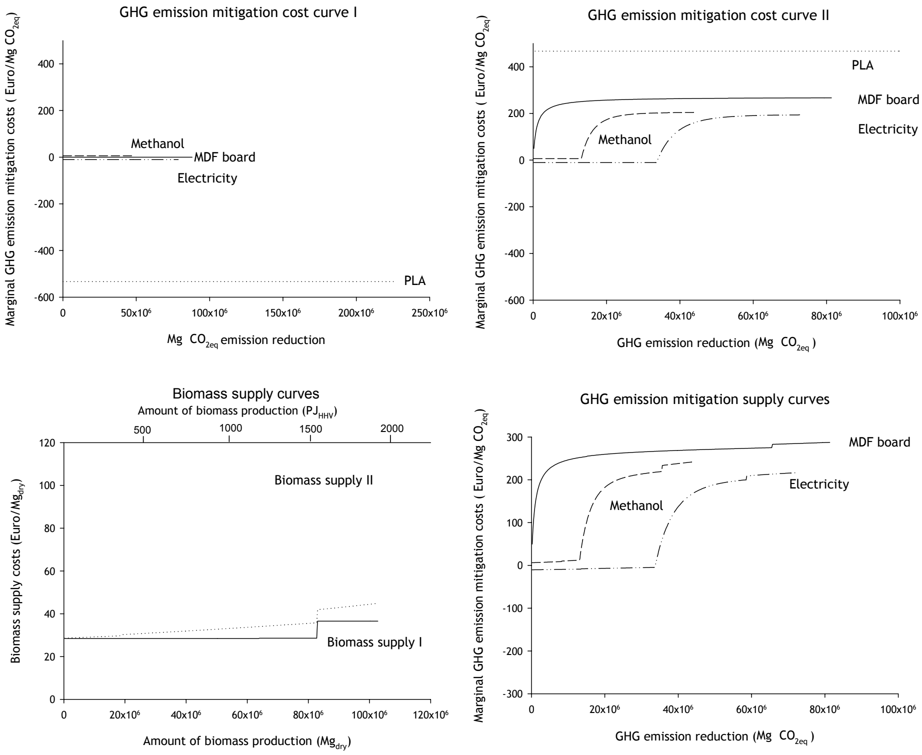


Figure 6.7: GHG emission mitigation cost curves, biomass supply curves and overall supply curves for GHG emission mitigation assuming scenario V1

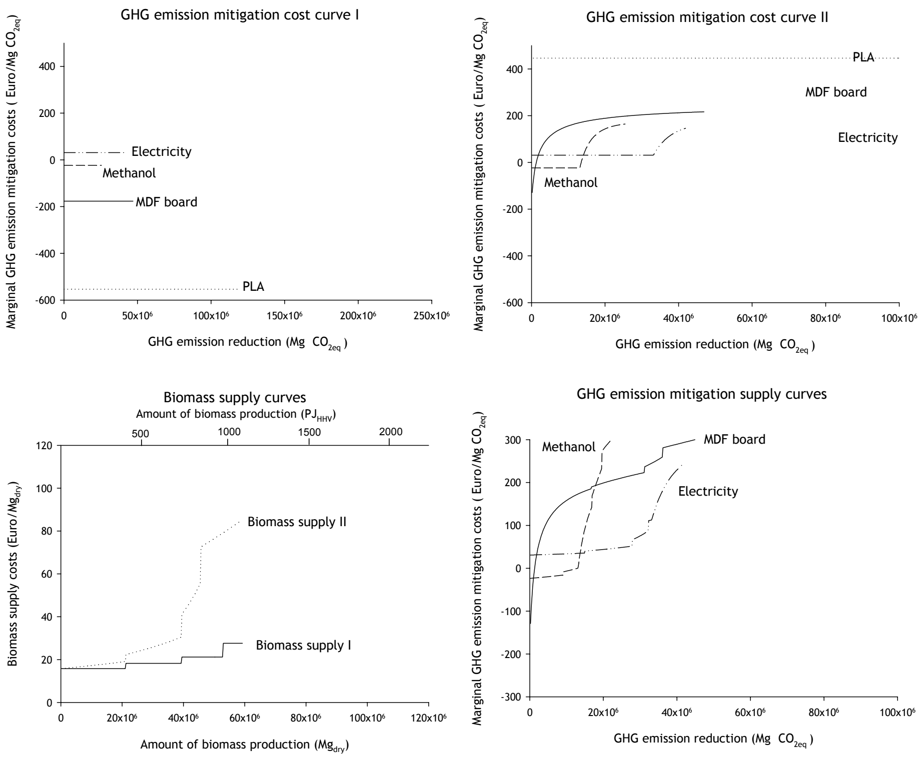


Figure 6.8: GHG emission mitigation cost curves, biomass supply curves and overall supply curves for GHG emission mitigation assuming scenario V2

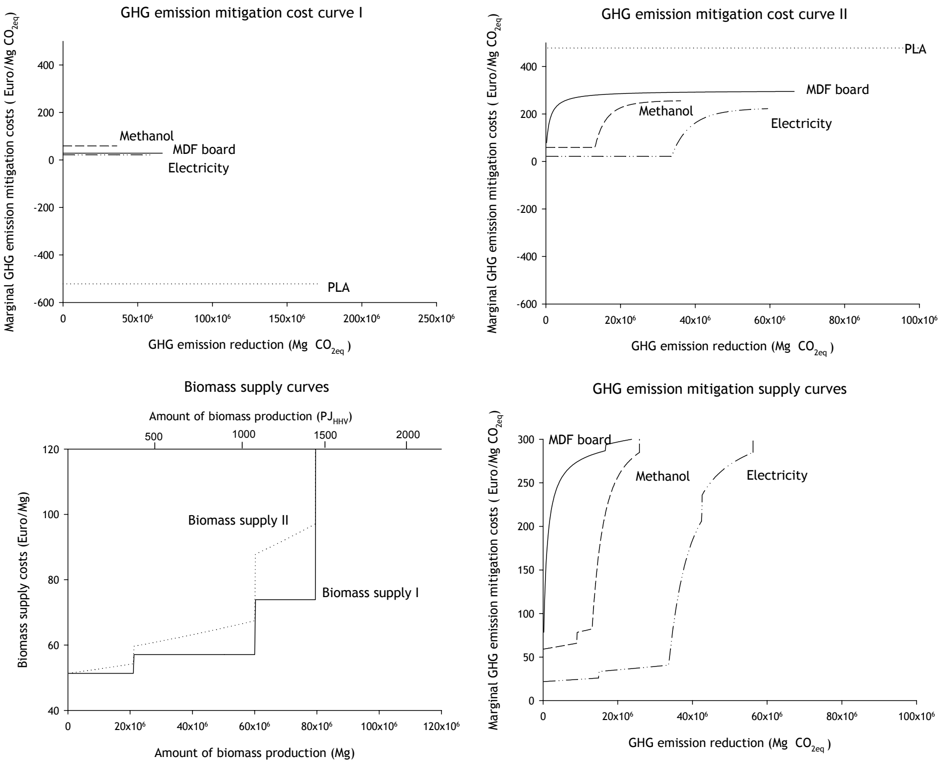


Figure 6.9 GHG emission mitigation cost curves, biomass supply curves and overall supply curves for GHG emission mitigation assuming scenario V3

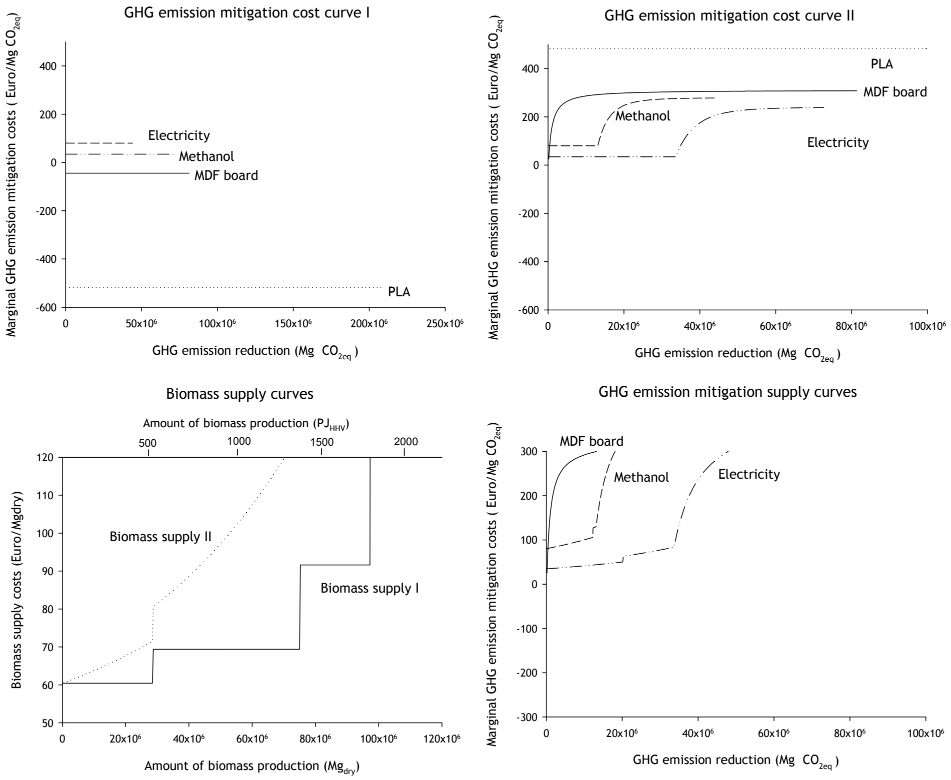


Figure 6.10: GHG emission mitigation cost curves, biomass supply curves and overall supply curves for GHG emission mitigation assuming scenario V4

4.4 Comparison of scenarios

Figure 6.11 depicts the 'integral' GHG emission mitigation cost supply curves for the different scenarios, i.e. for each amount of biomass used, the cheapest biomass material or energy option of the considered technologies is applied.

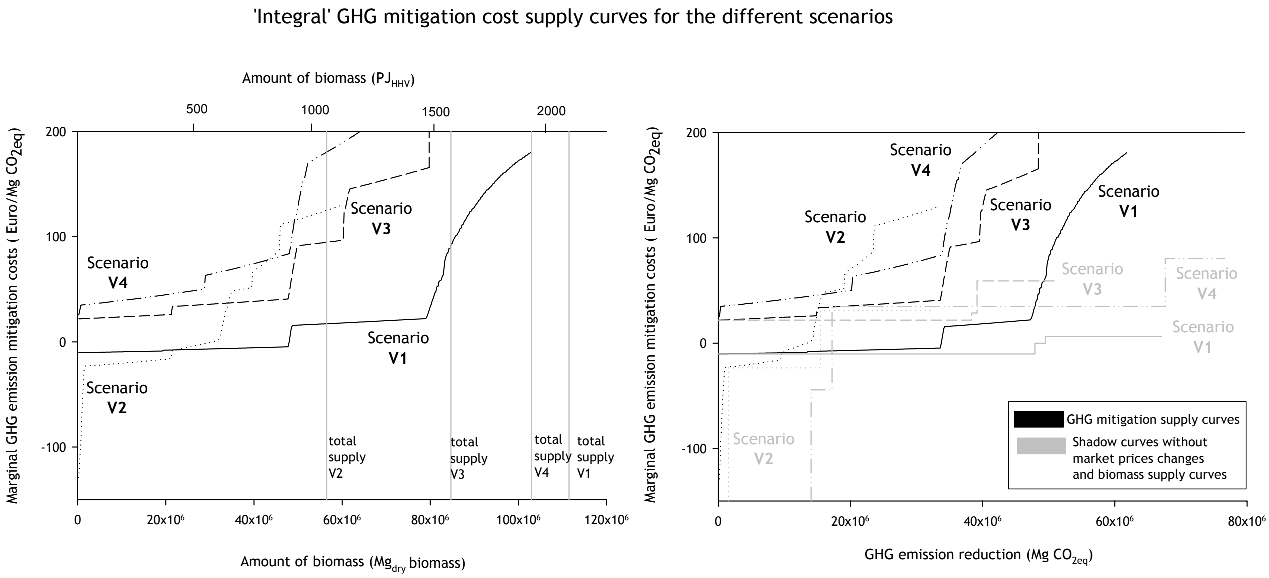


Figure 6.11: Comparison of overall GHG emission mitigation supply curves for the different scenarios

The specific GHG emission reduction, i.e. Mg CO_{2eq} avoided per unit of biomass use, differs only slightly between the technologies used for GHG emission mitigation in the scenarios. Therefore, the shape of the 'integral' GHG emission mitigation supply curves per unit of biomass used (left part of Figure 6.11) and per unit of GHG emissions avoided (right part of Figure 6.11) differs hardly.

In the V1, V3 and V4 scenario, in the first instance electricity production from biomass is preferred for GHG emission mitigation as long as electricity prices stay constant. Second, methanol production from biomass is used for the part of the market volume in which methanol prices stay constant. Finally, MDF, electricity and methanol production are applied in the last part of the integral GHG emission mitigation supply curve. The use of the different technologies alternates. This is due to the fact that with increasing use of one technology its GHG emission mitigation costs increase and, finally, exceed the GHG emission mitigation costs of another technology. In the V2 scenario, first MDF production is applied for a small part of GHG emission mitigation. Second and third, methanol and electricity production are used.

GHG emission mitigation costs in the V1 and V2 scenario are relatively low. In the V1 scenario about 49 and 34 million Mg CO_{2eq} are avoided for costs below 50 and 0 €/Mg CO_{2eq}, respectively. In the V2 scenario, these amounts are about 18 and 13 million Mg CO_{2eq} avoided. In the V3 and V4 scenario, however, biomass production costs are relatively high, especially at large scales of biomass production. As a consequence, GHG emission mitigation costs in these scenarios are higher and no GHG emissions are avoided at costs below 0 €/Mg CO_{2eq}. Moreover, in the V3 and V4 scenario about 34 and 20 million Mg CO_{2eq}, respectively, are avoided at costs below 50 €/Mg CO_{2eq}.

To show the impact of the large-scale introduction of biomass, shadow curves of GHG emission mitigation costs are depicted in the right part of figure 6.11. These curves show the GHG emission mitigation costs in the different scenarios without considering market price changes of materials, energy carriers and increasing costs of biomass supply. The same amount of each technology, i.e. MDF, PLA, methanol or electricity, is used for the shadow curves as in the respective scenarios. The shadow curves show, that market mechanisms and increasing biomass supply costs lowers the GHG emission reduction potential at low costs considerable. The potential GHG emission reduction at costs below 50 €/Mg CO_{2eq} decreases from 67 to 49 million Mg CO_{2eq} in the V1 scenario, from 33 to 18 million Mg CO_{2eq} in the V2 scenario, from 39 to 34 million Mg CO_{2eq} in the V3 scenario and from 68 to 20 million Mg CO_{2eq} in the V4 scenario.

Poland is a country with large biomass production potential. While the total primary energy consumption of Poland in 2000 is about 3.8 EJ (IEA, 2002a), the total biomass supply (short rotation wood) in the different scenarios varies between 1.1 and 2.0 EJ. [27]However, at relatively low GHG emission mitigation costs of below 50 €/Mg CO_{2eq} only about a half to two third of this biomass can be used with the options considered in this study. As biomaterials have only a small potential of GHG emission mitigation at low cost, the production of other bio-energy carriers, e.g. heat, would be necessary to use a larger part of this biomass potential.

4.5 Sensitivity analysis

In a sensitivity analyses, the influences of assumptions about the material and energy market on the integral GHG emission mitigation costs supply curves are analysed. Also, many other factors influence the final GHG emission mitigation supply costs, e.g. biomass

[27] In these potentials agricultural and forestry residues are not included, which amount to about 0.2 to 0.6 EJ.

production costs, efficiency of material production, the reference system and market prices. In this study, however, our main interest is the possible change of GHG emission mitigation costs through the variability of market prices if biomass material and/or energy uses are introduced on a large scale. Therefore, only the main factors influencing these markets, i.e. elasticity, the total market volume and the Polish market shares are investigated here. An overview of the variation of parameters in the sensitivity analysis is given in Table 6.6. For illustration, this sensitivity analysis is carried out for the V1 scenario that has the largest GHG emission mitigation potential at low costs.

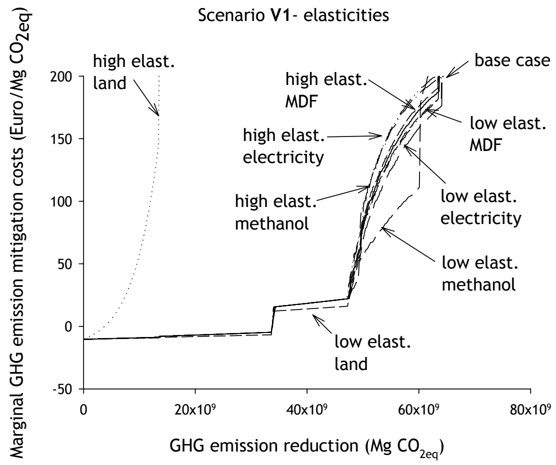
Table 6.6: Variation of elasticity factors and market volumes in the sensitivity analysis of scenario V1

	Land	**MDF board**	**PLA**	**Methanol**	**Electricity**
Base elasticity	-0.45	-1.11	-0.55	-0.23	-0.15
Range elasticity	-0.03 to –0.64	-0.5 to –1.8	-0.5to –2.5	-0.05 to –1.36	-0.1 to –0.2
Base market volume	N/a	15.2 million Mg	0.14 million Mg	58.7 EJ	45.6 EJ
Range market volume	N/a	±10% of volume	±10% of volume	±10% of volume	±10% of volume
Range market share	N/a	+100%/-50% of share	+100%/-50% of share	+100%/-50% of share	+100%/-50% of share

As discussed in Section 3, values of elasticity for the different markets are quite uncertain and various estimates exist. These often-broad ranges are used for the sensitivity analysis. For land markets the range of elasticities presented in Finke et al. (1984) is assumed. For all kind of bio-materials, Gielen et al. (2000) estimate own-price elasticity to be –0.5. This value is used as higher range for MDF board and PLA. The lower range of elasticity for MDF is the lowest value derived from historical data. For PLA, the lower value has been derived from price and volume forecast from producers in Crank et al. (2004); see also Chapter 5 of this thesis. For methanol, the range of elasticity found in the meta-analysis of Espey (1998) is used.[28] Finally, for electricity the values estimated by Wolfram (1999) and SEO (1998) are used as ranges.

Market volumes of materials and energy carriers in our analysis may vary because the total demand changes without related market price changes, e.g. by income variations. Also the market share of Poland on the respective market may increase or decrease depending strongly on the competitiveness of Polish production. To account for these possible changes, a variation of the market volume of 10% and a variation of market shares from half to twice the value is considered for the sensitivity analysis.

[28] The lowest value of gasoline elasticity is zero (Espey, 1998). However, we use a value of -0.05 close to zero that allows for calculation with the elasticity function.



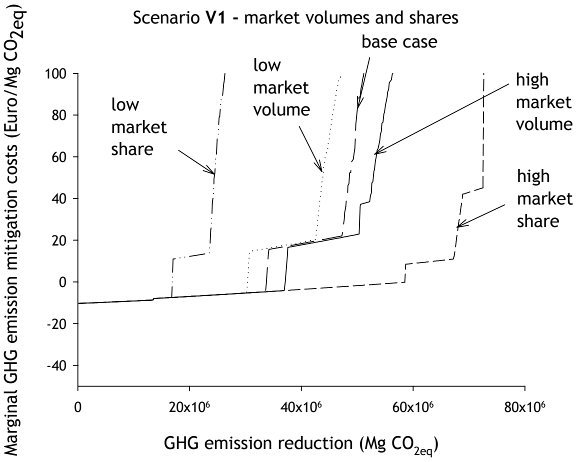


Figure 6.12: Variation of integral GHG emission mitigation cost supply curves (of scenario V1) due to the variation of elasticity factors and market volumes

In Figure 6.12, the influence of the variation of elasticities (left part of Figure 6.12), market volumes and market shares (right part of Figure 6.12) on the integral GHG emission mitigation costs in scenario V1 is presented. The sensitivity analysis shows that the own-price elasticity of the land demand has a strong influence on the GHG emission mitigation costs. If this elasticity is in the upper range, GHG emission mitigation costs increase rapidly reducing the potential of GHG emissions that can be avoided at costs below 50 €/Mg CO_{2eq} by 80%. The influence of elasticity factors on the integral GHG emission mitigation costs is limited to the part of the curve in which market prices are not constant. With low own-price elasticities, the increase of GHG emission mitigation costs with increasing scale of biomass use is slightly less than in the base case, while the opposite applies to high own-price elasticities. At GHG emission mitigation costs of about 100 €/Mg CO_{2eq} the GHG emission potentials varies about 0.01 to 10% through the variation of own-price elasticities of materials and energy carriers. The variation of market volumes and market shares has a large influence on the integral GHG emission mitigation costs. A higher market volume or share increases the amount of GHG emissions that can be reduced without a decrease of market prices of bioenergy carrier significantly, while a lower market volume decreases this amount of GHG emissions. If the Polish market share is doubled, the potential GHG emission reduction at costs below 50 €/Mg CO_{2eq} also doubles.

5 Discussion

The method used in this chapter is suitable to highlight the possible influences of large-scale introduction of biomass material and energy systems on market prices of materials and energy carriers and their feedback to GHG emission reduction costs. To determine the amount of GHG emission reduction costs for various scenarios, input data on technology performance and biomass production costs play a crucial role. The data used in this study give an estimation of possible costs. Even though technology development until 2015 will be limited and the possible differences in the developments of biomass production systems have been included in the scenario analysis, uncertainties in these input data may remain. For example, CO_2 intensities of electricity generation may change in the medium-term future due to e.g. the increased use of renewable energies and the replacement of less efficient fossil energy uses. These uncertainties, however, have not been explored in this analysis.

Instead, our analysis concentrates on the influence of market volumes, market shares and own-price elasticity of demand on the GHG emission mitigation costs. It has been shown in the sensitivity analysis that own-price elasticities of demand of agricultural land and market volumes of bioenergy carriers influence the GHG emission mitigation costs of bio-material and bioenergy utilisation strongly.

However, not all uncertainties in the developments of markets could be addressed. Elasticity factors in literature for land, materials and energy carriers show a very broad range. Moreover, own-price elasticity for MDF and PLA had to be estimated by a simple regression analysis. The influence of other factors than own-price elasticity on demand and market prices—e.g. developments in markets of substituting goods—could not be identified in this regression analysis. For this purpose, an econometric analysis would be necessary, which is beyond the scope of the study. For PLA, the results for PLA production may be pessimistic as the market is still developing and may grow.

For market volumes and Polish market shares of bio-energy carriers it has been assumed that the current volume of alternative energy carriers can be substituted. However, this substitution depends on the competitiveness of bio-energy production with other energy carriers and on the competitiveness of Polish bio-energy production with the production in other countries. This competitiveness may be evaluated by establishing supply curves for energy production. However, such an analysis is beyond the scope of this study.

While in principle the effects of increased bio-material and bio-energy production on their market prices have been included in our approach, other interactions between the large-scale introduction of biomass systems and GHG emission mitigation costs have not been included. For example, with increasing agricultural land prices, food prices increase as well, which may lead to a loss of welfare. Also, it has been assumed that bio-materials and bioenergy carriers substitute reference materials and energy carriers without accounting for any net effects of consumer or producer surpluses.

It may also be possible that the production of reference materials and energy carriers do not equally decrease with the production of biomass applications. Moreover, reference systems may change at large scales of biomass utilisation. For instance PLA may substitute poly(ethylene) first, and if this substitution potential is used, PLA may substitute paper packaging. As a result, the amount of GHG emissions reduced by biomass utilisation changes. Finally, the substitution of different amounts and kinds of reference materials and energy carrier may influence the shape of demand curves for biomass applications. To include these types of interaction a more detailed top-down model would be necessary, which is beyond the scope and objective of this study.

Also the time dimension of the large-scale introduction of biomass use has not been included in our study. The results depict the GHG emission mitigation costs in relation to the scale of biomass production at a certain fictive moment in time, i.e. the year 2015. To implement biomass systems on a large scale, however, will take a certain time span. During this implementation period, technological learning is likely to take place, lowering the resulting GHG emission mitigation costs. These effects have not been considered, because the main objective of this study was to analyse GHG emission mitigation cost changes due to a large-scale utilisation of biomass in a country rather than technological development during implementation strategies. Moreover, the time until the year 2015 is a rather short period for substantial learning to take place.

6 Conclusions

This study evaluates the possible influences of a large-scale introduction of biomass material and energy systems and their market volumes on market prices of land, materials and energy carriers and subsequently on GHG emission reduction costs. In first instance, it can be concluded that GHG emission mitigation costs from biomass on a country level may increase considerable with the scale of biomass production and utilisation. The potential for GHG emission mitigation below costs of 50 €/Mg CO_{2eq} for the four biomass applications considered is 49 million Mg CO_{2eq} in the V1 scenario (related to SRES scenario family A1), 18 million Mg CO_{2eq} in the V2 scenario (related to SRES scenario family A2), 34 million Mg CO_{2eq} in the V3 scenario (related to SRES scenario family B1) and 20 million Mg CO_{2eq} in the V4 scenario (related to SRES scenario family B2). Without the influence of a large-scale introduction on the development of biomass supply costs and market prices of land, materials and energy carriers, the GHG emission reduction potential at costs below 50 €/Mg CO_{2eq} would be about 13-70% higher, depending on the scenarios.

The increase of GHG mitigation costs depends mainly on biomass supply curves, the increase of agricultural land costs and the decrease of market prices of material and energy carriers. Biomass supply costs increase between 20 and 100 €/Mg_{dry} in the different scenarios, if all land that is not necessary for food production is used for biomass production. Additionally, the increase of agricultural land rents due to increased biomass production adds up to 50 to 100 €/Mg_{dry} to biomass production costs at large scales. At large scales of biomass use that exceed the volumes of current markets for energy carriers, GHG emission mitigation costs increase rapidly due to changes of market prices of material and energy carriers. At these scales, GHG emission mitigation cost levels rise to very high values.

Bio-material production covers only a small part of GHG emission mitigation at low costs. This is due to relatively small material markets and the subsequent strong decrease of market prices of bio-materials at large scale of production. Instead, mainly bio-energy production is applied for GHG emission mitigation as energy markets are comparably large and alternative energy carriers can be substituted at a large scale without decreasing market prices. Therefore, both supply and demand of materials and especially energy carriers should be analysed jointly to quantify the amounts that realistically can be used in a country/region.

GHG emission mitigation costs depend strongly on own-price elasticity of land and market volumes of bio-energy carriers. To a lesser degree, GHG emission mitigation costs also depend on the own-price elasticity of materials and energy carriers and market volumes of bio-materials. However, literature estimates of own-price elasticities are highly uncertain and market volumes of biomass applications depend on their competitiveness.

This analysis shows the importance to consider both biomass supplies and potential demands and markets simultaneously to get a more realistic picture of optimal biomass utilisation strategies for GHG emission mitigation. Hence, it would be ideal to combine the bottom-up approach demonstrated here with an analysis of market effects using top-down modelling.

Summary and conclusions

Biomass can play a role in mitigating greenhouse gas emissions by substituting conventional materials and supplying biomass based fuels. However, currently only a modest amount of biomass is used for these applications in industrialised countries. Main reason for the low share of biomass applications in Europe (about 6 EJ in the EU-15) is their often-high production costs, among others due to the relatively low availability of agricultural land. Therefore, in the short to medium term more efficient and cost effective routes for the introduction of biomass are needed. One of these routes may be the further development of multi-functional biomass systems and the shift of biomass production to more favourable areas, for example, Central Eastern Europe. Multi-functional biomass systems use the concepts of 'multi-product use' and 'cascading'. Multi-product use is defined as using biomass for different applications, while cascading is the subsequent use of biomass for a number of applications, i.e. materials, recycling of materials and energy recovery. Important parameters for the efficiency of multi-functional biomass systems are savings of non-renewable energy consumption, GHG (greenhouse gas) emission reduction, (agricultural) land use and total costs of the system. However, only very few studies have explicitly addressed multi-functional biomass systems analysing these parameters quantitatively. Therefore, the central research question of this thesis is: *What is the potential of multi-functional biomass systems to improve the costs and the land use efficiency of saving non-renewable energy consumption and reducing GHG emissions in quantitative terms?* Two main aspects play an important role in answering this central question. First, methodologies to account for costs, land use, GHG emissions and non-renewable energy consumptions need to be adapted for the evaluation of multi-functional biomass systems. Particularly, issues of allocation of environmental impacts and costs or system expansion, accounting for the time dimension and the integration of market price changes due to the large-scale introduction of biomass systems deserve special attention. Second, the potential benefits depend on the kind of biomass system regarded and the mechanism of this dependence have to be studied in order to identify promising multi-functional biomass systems.

In this thesis, the performance of multi-functional biomass systems with regard to GHG emissions, non-renewable energy consumption, agricultural land use and costs is quantified. This analysis is carried out by several case studies of multi-functional biomass systems that appeared to be promising from a first review. The case studies are situated in Europe and concentrate on Poland. In such a way the potential of biomass production in Central Eastern Europe can be investigated, because these states have (at present) large areas of agricultural land, potentially high to medium crop yields and comparably low costs of land and labour.

Chapter 2 investigates the concept of multi-product crops. Multi-product crops, i.e. using a crop partially for energy and partially for material purposes, can possibly create additional incomes as well as additional GHG emission reductions. In this chapter, the benefits of multi-product crop systems in comparison to energy crop systems are investigated systems, focussed on the primary biomass fuel costs and GHG emission reductions per area of biomass production. For this analysis a case study approach is followed and the sensitivity of the results is investigated by means of a Monte-Carlo analysis. The multi-product crops studied are wheat, hemp and poplar in the Netherlands and Poland. GHG emission reductions of these multi-product systems are found to be between 0.2 and 2.4 Mg CO_{2eq} per hectare per year in Poland and between 0.9 and 7.8 Mg CO_{2eq} per hectare per year in the Netherlands, while primary biomass fuel costs range from –4.1 to –1.7 €/GJ in the Netherlands and from 0.1 to 9.8 €/GJ in Poland. Results show that the economic attractiveness of multi-product crops depends strongly on material market prices, crop production costs and crop yields. Net annual GHG emission reductions per hectare are influenced strongly by the specific GHG emission reduction of material use, reference energy systems and GHG emissions of crop production. Multi-product use of crops can significantly decrease primary biomass fuel costs. However, this does not apply in general, but depends on kind of crops and material uses. For the examples analysed here, net annual GHG emission reductions per hectare are not lowered by multi-product use. Consequently, multi-product use of crops is not a priori an option to increase the performance of bioenergy systems.

Chapter 3 has a twofold aim: (1) to select and develop a coherent methodological framework for the comparison of different biomass cascading chains in terms of costs, land demand and CO_2 emission reductions and (2) to identify key parameters and issues that influence the efficiency of biomass cascading chains using the methodology developed. For the second purpose, the approach is applied to a case study. A number of cascading chains of short rotation poplar wood are compared to each other on basis of literature data. Results for these chains vary strongly, namely, from net benefits of CO_2 mitigation of 200

€/Mg CO_2 to net costs of CO_2 mitigation of 2200 €/Mg CO_2, and from net CO_2 emission reductions per hectare of biomass production of 28 Mg CO_2 per hectare per year to net CO_2 emissions of 8 Mg CO_2 per hectare per year. Using a present-value approach to determine CO_2 emissions and costs affects the performance of long-term cascading chains significantly, i.e. cost and CO_2 emission reduction are decreased. Throughout, cascading has the potential to improve both CO_2 emission reduction per hectare and CO_2 mitigation costs of biomass usage. However, this strongly depends on the biomass applications combined in the cascading chain. Parameters that significantly influence the results are market prices and gross energy requirements of substituted materials and energy carriers, and the efficiency of biomass production. The method presented in this study is suitable to quantify land use, CO_2 emission reduction and economic performance of biomass cascading systems, and highlights the possible impact of time on the attractiveness of specific cascading chains.

Chapter 4 compares savings of non- renewable energy consumption and greenhouse gas (GHG) emission reductions of bio-based polymers with those of bioenergy on a per unit of agricultural land-use basis. In view of policy goals to increase the energy supply from biomass and current efforts to produce bio-based polymers in bulk, the amount of available land for the production of non-food crops could become a limitation. Hence, given the prominence of energy and greenhouse issues in current environmental policy, it is desirable to include land demand in the comparison of different biomass options. Over the past few years, numerous life-cycle assessment (LCA) studies have been prepared for different types of bio-based polymers, but only a few of these studies address the aspect of land use. Therefore, this chapter extends existing LCA studies by taking into account land requirements in the analysis of energy savings and GHG emission reductions. The results show that referring energy savings and GHG emission reduction of bio-based polymers to a unit of agricultural land, instead of to a unit of polymer produced, leads to a different ranking of options. If land use is chosen as the basis of comparison, natural fibre composites and thermoplastic starch score better than bioenergy production from energy crops, whereas poly(lactides) score comparably well and poly (hydroxyalkaonates) score worse. If the use of agricultural residues for energy purposes is included, the environmental performance of bio-based polymers improves significantly. Moreover, it is very likely that higher production efficiencies will be achieved for bio-based polymers in the medium term. Bio-based polymers thus offer interesting opportunities to reduce the utilisation of non-renewable energy and to contribute to GHG emission mitigation in view of potentially scarce land resources.

Chapter 5 analyses the potential quantitative benefits of multi-functional bio-refinery systems with regard to GHG emission reductions, savings of non-renewable energy consumption and costs in relation to production scale and market volumes. For this analysis a case study of bio-refinery systems for poly lactic acid (PLA) production is carried out. The systems include multi-functional use of biomass resources, i.e. use of agricultural residues for energy production, use of by-products, recycling, and waste-to-energy recovery. We evaluate the performance of these systems per kilogram of bio-based polymer produced and per hectare of biomass production. The evaluation is done using data of Poland assuming that biomass and PLA production is embedded in a European energy and material market. First, the performance of different bio-refinery systems is investigated by means of a bottom-up chain analysis. Second, an analysis is applied that derives market prices of products and land by means of the own-price elasticities. Thus, costs of the bio-refinery systems are determined depending on the demand of land and materials. It is found that all PLA bio-refinery systems considered lead to net savings of non-renewable energy consumption of 70 to 220 GJ per hectare per year and to net GHG emission reductions of 3 to 17 Mg CO_{2eq} per hectare per year. Most PLA bio-refinery systems considered in this study lead to net costs of the overall system of up to 4600 € per hectare per year. On the contrary, PLA production from short rotation wood leads to net benefits of about 1100 € per hectare per year if a high amount of a high-value product, i.e. synthetic fibres, is produced. Multi-functionality is necessary to ensure the viability of PLA bio-refinery systems from biomass with regard to energy savings and GHG emission reduction. However, with regard to costs, the multi-functional use of biomass does not contribute much to overall incomes. Own-price elasticity of the demand for materials influences the overall costs of the bio-refinery system strongly. The own-price elasticity of land demand could become important if biomass systems are introduced on a large scale.

Chapter 6 evaluates the possible influences of the large-scale introduction of biomass material and energy systems and their market volumes on land, material and energy market prices and their feedback to GHG emission mitigation costs. GHG emission mitigation supply curves for large-scale biomass use are compiled using a methodology that combines a bottom-up analysis of biomass applications, biomass cost supply curves and market prices of land, bio-materials and bioenergy carriers. These market prices depend on the scale of biomass use and the market volume of materials and energy carriers and are estimated using own-price elasticities of demand. The methodology is demonstrated for a case study of Poland in the year 2015. The case study applies different scenarios on economic development and trade in Europe that impact biomass supply and markets of land, materials and energy carriers. For the key technologies considered, i.e. medium density fibreboard, poly lactic acid, electricity and methanol production, and for the scenarios investigated in this chapter, GHG emission mitigation costs increase strongly with the scale of biomass production. It is found that the influence of a large-scale introduction on the development of biomass supply costs and market prices of land, materials and energy carriers, reduces the GHG emission reduction potential at costs below 50 €/Mg CO_{2eq} with about 13—70% depending on the different scenarios. Bio-material production accounts for only a small part of the total GHG emission mitigation potential at low costs. This is due to relatively small material markets, that lead to a strong decrease of market prices of bio-materials at large scale of production. GHG emission mitigation costs depend strongly on biomass supply curves, own-price elasticity of land and market volumes of bioenergy carriers. Our analysis shows that these effects should be taken into account for the development of strategies to implement the use of biomass. However, literature estimates of own-price elasticities are highly uncertain and market volumes of biomass applications depend on the competitiveness of the applications. To counteract these uncertainties, a combination of a bottom-up analysis with an analysis of market effects is recommended.

Summarising, we have analysed various multi-functional biomass systems and their potential benefits with regard to GHG emission reduction, energy savings, land use and costs. Results for the different systems vary considerable from plain benefits to no distinct advantage of multi-functional biomass use. The benefits of multi-functional biomass systems are influenced by many factors. These factors depend on the one hand on the type and structure of the biomass systems such as type of material or energy carrier produced or efficiency and costs of production. On the other hand, these factors depend on external circumstances, like market volumes and prices of materials or the CO_2 intensity of reference systems.

Most multi-functional biomass systems regarded in this thesis, however, increase the potential benefits of biomass use in terms of costs, GHG emission reductions and agricultural land use. In comparison to single bioenergy systems, the multi-product systems investigated in chapter 2, decrease primary biomass fuel costs by about 5 to more than 50 €/GJ_{LHV} in cases of very high-value material applications. (For comparison, primary biomass fuel costs of bioenergy systems are about 2—15 €/GJ_{LHV}, while coal prices are about 2 €/GJ_{LHV}.) With regard to GHG emission reductions, these multi-product systems lower the reductions with about 3—10 Mg CO_{2eq} per hectare per year of biomass production. The systems of cascading of short rotation wood that are analysed in chapter 3 show a very broad range of results. Due to cascading, the GHG emissions avoided alter by about –13 to 23 Mg CO_2 per hectare per year and modify GHG mitigation costs by about –300 to 2000 €/Mg CO_2 compared to single energy use. For comparison, the use of short rotation wood for bio-electricity production results in avoided emissions of about 5 Mg CO_2 per hectare per year at costs of about 100 €/Mg CO_2. The use of agricultural residues for energy production that is investigated in Chapter 4 increases the benefits of bio-based polymers production, i.e. GHG emission reductions increase by up to 15 Mg CO_{2eq} per hectare per year. Finally, multi-functional biomass use in the PLA bio-refinery systems analysed in Chapter 5 leads to additional benefits of about 4-12 Mg CO_{2eq} per hectare per year and 0-200 €/Mg biomass input.

In addition, it can be concluded that within the structure of biomass systems, the main material application has the largest influence on the overall performance of multi-functional biomass systems. Also, the utilisation of agricultural residues for energy production can significantly improve the performance of biomass systems. Of course, the evaluation of multi-functional biomass systems depends strongly on the alternative reference systems. The type of materials and energy carriers that are substituted and the waste management system have proven to be crucial for costs as well as for GHG emission reductions obtained for biomass systems.

In the case of bio-materials that have a relatively long lifetime, time dimensions of carbon storage in these materials can play an important role. For example, if time is considered by using a present value approach, net GHG emission reductions of cascading systems decrease. Also, market prices of land, materials and energy carriers influence the economic performance of multi-functional biomass systems strongly. Research in Chapter 6 indicates that with a growing use of biomass for materials and energy, GHG emission mitigation costs of these options may increase. While many multi-functional biomass systems already have quite high GHG emission mitigation costs, the economic potential of these systems to mitigate GHG emissions might even be further limited by these market effects. This is especially the case for bio-materials that have comparably small markets. Moreover, it should be noted that market interactions can influence the type of reference materials and energy carriers, e.g. the type of fossil fuel, that are substituted.

Also, methodological lessons can be learned from the analyses in this thesis. It can be concluded that the inclusion of agricultural land use in the comparison of biomass systems may provide valuable insights in their ranking. Because often bio-based materials, for which land is used for their production, are substituted in the reference system, it is necessary to account for this land use in the reference system as well. The approach used in this thesis, i.e. the assumption of alternative production of bioenergy on this land, is suitable for the comparison of biomass systems. However, an agreement on a standard methodology how to deal with this issue in scientific research would be desirable. Also, a standard methodology needs to be developed for the inclusion of the time dimension in GHG emission balances, because bio-material systems may store carbon for a relatively long time. Our analysis using a present-value approach shows the potentially large influence of the time dimension on the results. Finally, the scale of biomass production and use influences the costs of the system as well as GHG emission balances due to changes of prices and of substituted reference applications. The bottom-up analysis combined with a simple analysis of land, material and energy demands presented in this thesis shows important trends for the dependency of biomass systems on the scale of biomass use.

In conclusion, to use biomass efficiently in terms of GHG emission reduction, (agricultural) land use and total costs of the system, multi-functional biomass systems can be an attractive option if carefully designed, depending on reference systems and land, material and energy markets. The best multi-functional biomass systems analysed in this thesis increase the GHG emission reduction per unit of agricultural land used by a factor 5 compared to single biomass uses and decrease the total systems costs by about the same factor. However, for the performance of biomass systems at a large scale of biomass use, the interactions of biomass use with land, material and energy markets need to be better understood. Therefore, further research on optimal biomass systems for GHG emission mitigation should combine bottom-up information of biomass system with knowledge on market mechanisms from top-down analyses.

Samenvatting en conclusies

Biomassa kan een rol in de vermindering van broeikasgas-emissies spelen, zowel door conventionele materialen te substitueren als door bio-brandstoffen te leveren. In geïndustrialiseerde landen wordt momenteel echter slechts weinig biomassa gebruikt voor deze toepassingen. Een belangrijke reden hiervoor is dat de productiekosten van biomassatoepassingen vaak hoog zijn, onder andere door de relatief lage beschikbaarheid van landbouwgrond. Voor de introductie van biomassa op de korte tot middellange termijn zijn dus efficiëntere en meer kosteneffectieve routes nodig. Deze routes kunnen de ontwikkeling van aantrekkelijke multifunctionele biomassasystemen en de verplaatsing van biomassaproductie naar gunstigere gebieden, b.v. Centraal-Oost-Europa, zijn. Multifunctionele biomassasystemen behelzen de concepten 'multiproductgebruik' en 'cascadering'. Multiproductgebruik wordt gedefinieerd als het gebruik van de geproduceerde biomassa voor verschillende toepassingen, terwijl cascadering het opeenvolgend gebruik van biomassa voor meerdere toepassingen is, dat wil zeggen, materialen, materiaalrecycling en het terugwinnen van energie. Belangrijke parameters voor de efficiency van multifunctionele biomassasystemen zijn de besparing op niet-hernieuwbaar energiegebruik, de vermindering van broeikasgas-emissies, het gebruik van (landbouw-)grond en de totale kosten van het systeem. Slechts weinig onderzoekers hebben multifunctionele biomassasystemen op deze punten kwantitatief geanalyseerd. Daarom is de centrale onderzoeksvraag van dit proefschrift: *wat is, kwantitatief het potentieel van multifunctionele biomassasystemen om de kosten en landgebruik-efficiency van de besparing op niet-hernieuwbare energie en de vermindering van broeikasgas-emissies te verbeteren?* Twee aspecten spelen een belangrijke rol in het beantwoorden van deze centrale vraag. Ten eerste moeten methodes om kosten, landgebruik, broeikasgas-emissies en niet-hernieuwbaar energiegebruik te berekenen, worden aangepast voor de evaluatie van multifunctionele biomassasystemen. In het bijzonder verdienen vraagstukken van allocatie van millieuimpacts en kosten en systeemuitbreiding, van het meenemen van de tijdsdimensie in berekeningen, en van het meenemen van marktprijsveranderingen door grootschalige introductie van biomassasystemen hierbij aandacht. Ten tweede hangen de potentiële voordelen van een multifunctioneel biomassasysteem af van het specifieke systeem in kwestie; om veelbelovende systemen te kunnen identificeren moeten de mechanismen van deze afhankelijkheid worden bestudeerd.

In dit proefschrift worden broeikasgas-emissies, niet-hernieuwbaar energiegebruik, gebruik van landbouwgrond en kosten van multifunctionele biomassasystemen gekwantificeerd. Hiervoor worden meerdere case studies van multifunctionele biomassasystemen die in eerste instantie veelbelovend lijken uitgevoerd. De case studies zijn gesitueerd in Europa en concentreren zich op Polen. Op deze manier kan het potentieel van biomassaproductie in Centraal-Oost-Europa worden onderzocht, waar staten (momenteel) over grote hoeveelheden landbouwgrond beschikken, potentieel gemiddelde tot hoge gewasopbrengsten kunnen bereiken, en relatief lage kosten van land en arbeid kennen.

Hoofdstuk 2 onderzoekt het concept van multiproductgebruik van landbouwgewassen, dat wil zeggen, het gedeeltelijk gebruiken van een gewas voor energieproductie en gedeeltelijk voor materiële toepassingen. Multiproductgebruik van gewassen kan wellicht tot toegevoegde inkomsten leiden, evenals tot additionele verminderingen van broeikasgas-emissies. In dit hoofdstuk worden de voordelen van multiproductsystemen boven enkelvoudige bio-energiesystemen onderzocht. De efficiëntie van de multiproductsystemen wordt uitgedrukt in kosten van primaire bio-brandstoffen en in verminderingen van broeikasgas-emissies per hectare biomassaproductie. Deze analyse wordt door middel van een case study uitgevoerd, en de gevoeligheid van de resultaten wordt middels een Monte-Carlo analyse onderzocht. De beschouwde gewassen zijn tarwe, hennep en korteomloop populier. De multiproductsystemen worden voor Nederland en voor Polen onderzocht. De vermindering van broeikasgas-emissies van deze systemen bedraagt tussen 0,2 en 2,4 Mg CO_{2eq} per hectare per jaar in Polen en tussen 0,9 en 7,8 Mg CO_{2eq} per hectare per jaar in Nederland. De kosten van primaire bio-brandstof voor deze systemen liggen tussen -4,1 en -1,7€/GJ in Nederland en tussen 0,1 en 9,8 €/GJ in Polen. De resultaten laten zien dat de economische aantrekkelijkheid van multiproductgewassen sterk van de marktprijzen van materialen, van de kosten van biomassaproductie en van de gewasopbrengsten afhangt. De netto vermindering van broeikasgas-emissies per hectare landgebruik per jaar wordt daarentegen sterk beïnvloed door de specifieke vermindering van broeikasgas-emissies van de materiaaltoepassing, van het referentie-energiesysteem en van de broeikasgas-emissies gedurende de biomassaproductie. Multiproductgebruik van gewassen kan de kosten van primaire bio-brandstoffen beduidend verminderen. Dit geldt echter niet algemeen, maar hangt af van het gewas en de materiaaltoepassing. De netto broeikasgas-emissie per hectare per jaar wordt door de toepassing van multiproductgebruik niet verminderd in de systemen die in dit hoofdstuk geanalyseerd zijn. Derhalve is multiproductgebruik van gewassen niet a-priori een mogelijkheid om de prestaties van bio-energiesystemen te verbeteren.

Hoofdstuk 3 heeft een tweeledig doel: (1) het ontwikkelen van een coherent methodologisch kader voor het vergelijken van verschillende biomassa-cascade-systemen in termen van kosten, landgebruik en CO_2-emissieverminderingen, en (2) het identificeren van belangrijke parameters en aspecten, die de efficiëntie van het cascadegebruik van biomassa beïnvloeden. Voor het tweede doeleinde wordt de in dit hoofdstuk ontwikkelde methode op een case study toegepast. Op basis van literatuurgegevens worden cascades voor hout van korte-omloop populier met elkaar vergeleken. De resultaten van deze cascades variëren sterk, namelijk, van een opbrengst voor de vermindering van CO_2-emissies van 200 €/Mg CO_2 tot kosten van 2200 €/Mg CO_2, en van netto verminderingen van CO_2-emissies van 28 Mg CO_2 tot netto toename van CO_2-emissies van 8 Mg CO_2 per hectare per jaar biomassaproductie. De toepassing van een netto-contante-waarde methode om CO_2-emissies en kosten te bepalen beïnvloedt de uitkomsten voor lange-termijn cascades aanzienlijk. In dit geval zijn de kosten en de CO_2-emissiereductie lager. Dikwijls zijn cascades geschikt om zowel de vermindering van CO_2-emissies per hectare gebruikte landbouwgrond van biomassagebruik te vergroten als de kosten van CO_2-emissiereducties te verlagen. Dit hangt echter sterk af van de biomassatoepassingen die in de cascades gecombineerd worden. Parameters die de resultaten sterk beïnvloeden zijn de marktprijzen en het gecumuleerde energiegebruik van de gesubstitueerde referentiematerialen en -energiedragers, en de efficiency van biomassaproductie. De onderzoeksmethode die in dit hoofdstuk wordt voorgesteld is geschikt om het landgebruik, de vermindering van CO_2-emissies en de economische prestaties van biomassa-cascades te kwantificeren. Bovendien kan deze methode worden gebruikt om de mogelijke effecten van de tijdsdimensie op de aantrekkelijkheid van specifieke cascades in beeld te brengen.

Hoofdstuk 4 vergelijkt besparingen op het niet-hernieuwbaar energiegebruik en verminderingen van broeikasgas-emissies voor de productie van kunststoffen uit biomassa met de productie van bio-energie voor hetzelfde doel op basis van het landgebruik. Gezien de politieke doelstelling om het gebruik van biomassa in de energievoorziening te verhogen en de huidige inspanningen om grootschalig kunststoffen uit biomassa te produceren, zou de hoeveelheid beschikbaar landbouwgrond voor de productie van non-food-gewassen een beperking kunnen gaan vormen. Vanwege het belang dat vraagstellingen rondom energie en broeikasgas-emissies in het huidige milieubeleid hebben, is het derhalve wenselijk om het gebruik van landbouwgrond in de vergelijking van verschillende biomassasystemen mee te nemen. In de afgelopen jaren zijn talrijke levenscyclus-analyses van kunststoffen uit biomassa uitgevoerd, maar slechts enkele van deze studies houden rekening met landgebruik. Daarom breidt dit hoofdstuk bestaande levencyclusanalyses uit door landgebruik in de analyse van de besparing van energie en de vermindering van broeikasgas-emissies mee te nemen. De resultaten tonen dat een vergelijking van energiebesparingen en vermindering van broeikasgas-emissies van kunststoffen uit biomassa op basis van de hoeveelheid gebruikte landbouwgrond tot een andere rangorde leidt dan een vergelijking op basis van de hoeveelheid geproduceerde kunststof. Als landgebruik de basis van de vergelijking is, presteren de productie van natuurvezel-composieten en thermoplastisch zetmeel beter dan de productie van bio-energie uit energiegewassen, terwijl de productie van polymelkzuur vergelijkbaar en de productie van polyhydroxyalkaonaten slechter presteren. Indien het gebruik van landbouwresiduen voor energieproductie in de analyse wordt meegenomen, verbeteren de energiebesparingen en de CO_2-emissiereducties van de kunststoffen uit biomassa sterk. Bovendien lijkt het waarschijnlijk dat de productie van kunststoffen uit biomassa op de middellange termijn nog efficiënter zal worden. De productie van kunststoffen uit biomassa biedt dus, gezien de mogelijk geringe beschikbaarheid van landbouwgrond, interessante mogelijkheden om aan de vermindering van het gebruik van niet-hernieuwbare energiedragers en de vermindering van CO_2-emissies bij te dragen.

Hoofdstuk 5 analyseert de potentiële kwantitatieve voordelen van multifunctionele bioraffinaderij-systemen met betrekking tot de vermindering van broeikasgas-emissies, besparingen op het niet-hernieuwbaar energiegebruik en de verlaging van kosten. Tevens wordt de afhankelijkheid van deze voordelen van de schaal van productie en van marktvolumes onderzocht. Voor dit onderzoek wordt een case study van bioraffinaderijsystemen voor polymelkzuurproductie uitgevoerd. De systemen behelzen een multifunctioneel gebruik van biomassa, dat wil zeggen, gebruik van landbouwresiduen voor energieopwekking, gebruik van bijproducten, recycling, en herwinning van energie uit afval. De prestaties van deze systemen worden per kilogram geproduceerd polymelkzuur en per hectare biomassaproductie geëvalueerd. De berekeningen zijn gebaseerd op gegevens van Polen, onder de aanname dat zowel biomassa- als polymelkzuurproductie in een Europese energie- en materiaalmarkt zijn ingebed. Eerst worden de prestaties van de verschillende bioraffinaderijsystemen middels een ketenanalyse bottom-up onderzocht. Vervolgens worden marktprijzen van polymelkzuurproducten, bijproducten en land met behulp van de prijselasticiteit afgeleid van de productiecapaciteit. Aldus worden de kosten van de bioraffinaderijsystemen bepaald, gerelateerd aan de vraag naar landbouwgrond en materialen. Alle in dit hoofdstuk beschouwde bioraffinaderijsystemen voor polymelkzuurproductie leiden tot netto besparingen op het gebruik van niet-hernieuwbare energiedragers van 70 tot 220 GJ per hectare per jaar en tot netto verminderingen van broeikasgasemissies van 3 tot 17 Mg CO_{2eq} per hectare per jaar. De meeste beschouwde bioraffinaderijsystemen resulteren in netto kosten van het totale systeem oplopend tot 4600 € per hectare per jaar. Daarentegen leidt polymelkzuurproductie uit korte-omloop hout tot een netto winst van ca. 1100 € per hectare per jaar mits een hoogwaardige product, in dit geval kunstvezel, een groot aandeel van de productie uitmaakt. Multifunctionaliteit is noodzakelijk om energiebesparing en vermindering van broeikasgas-emissies van bioraffinaderijsystemen voor polymelkzuurproductie te waarborgen. Echter, aan de vermindering van kosten draagt het multifunctioneel gebruik van biomassa nauwelijks bij. De prijselasticiteit van materialen beïnvloedt de kosten van bioraffinaderijsystemen sterk. De prijselasticiteit van land zou een belangrijke rol kunnen spelen als biomassasystemen op grote schaal worden geïntroduceerd.

Hoofdstuk 6 evalueert de mogelijke invloed van grootschalige introductie van biomassamateriaal- en -energiesystemen en hun marktvolumes op marktprijzen van land, materiaal en energie. Ook wordt de terugkoppeling van deze effecten naar de kosten van broeikasgas-emissiereductie beoordeeld. In dit hoofdstuk worden daartoe kosten-aanbod-curves voor broeikasgas-emissiereductie opgezet voor biomassagebruik op grote schaal. Dit wordt gedaan middels een methode die een bottom-up analyse van biomassatoepassingen, biomassa-aanbod-curves, en marktprijzen van land, biomaterialen en bio-energiedragers combineert. Deze marktprijzen hangen af van de schaal van het biomassagebruik en van het marktvolume van materialen en energiedragers en ze worden geschat met behulp van prijselasticiteiten. De methode wordt gedemonstreerd door middel van een case study gericht op Polen met als zichtjaar 2015. In deze case study wordt gebruik gemaakt van verschillende scenario's voor economische ontwikkeling en handel in Europa die het aanbod van biomassa en de markten van land, materialen en energiedragers beïnvloeden. De beschouwde technologieën zijn de productie van vezelplaat, polymelkzuur, elektriciteit en methanol. Voor deze technologieën geldt dat in de scenario's de kosten van broeikasgas-emissiereducties sterk met de schaal van biomassaproductie stijgen. Er is gevonden, dat de invloed van een grootschalige introductie van het gebruik van biomassa op de aanbodkosten van biomassa en de marktprijzen van land, materiaal en energiedragers het potentieel van de vermindering van broeikasgas-emissies bij kosten onder 50 €/Mg CO_{2eq} met ca. 13-70% verlagen, afhankelijk van de verschillende scenario's. De productie van biomateriaal blijkt slechts weinig bij te dragen tot goedkope vermindering van broeikasgas-emissies. Dit is toe te schrijven aan de tamelijk kleine materiaalmarkten, die zorgen voor een sterke prijsdaling van biomaterialen bij grootschalige productie. De kosten van broeikasgas-emissiereductie hangen sterk van de biomassa-aanbod-curves, de prijselasticiteit van land en de marktvolumes van bio-energiedragers af. Onze analyse toont aan dat met dergelijke effecten rekening moet worden gehouden bij het ontwikkelen van strategiën voor implementatie van biomassagebruik. De schattingen van prijselasticiteiten in de wetenschappelijke literatuur zijn echter onzeker, en marktvolumes van biomassatoepassingen hangen af van hun concurerend vermogen. Om deze onzekerheden het hoofd te bieden wordt een combinatie van een bottom-up analyse met een analyse van markten geadviseerd.

Samenvattend: we hebben diverse multifunctionele biomassasystemen en hun potentiële voordelen met betrekking tot vermindering van broeikasgas-emissie, besparingen op het gebruik van niet-hernieuwbare energiedragers, landgebruik en systeemkosten geanalyseerd. De resultaten voor de verschillende systemen variëren aanzienlijk, van geen tot duidelijke voordelen van multifunctioneel biomassagebruik. De efficiëntie van multifunctionele biomassasystemen wordt door vele factoren beïnvloed. Enerzijds hangen deze factoren af van de structuur van het biomassasysteem, zoals het soort geproduceerde materiaal en energiedrager, of de efficiëntie en de kosten van de productie. Anderzijds hangen deze factoren ook af van externe omstandigheden, zoals marktvolumes en prijzen van materialen, of de CO_2-intensiteit van het referentiesysteem.

De meeste multifunctionele biomassasystemen die wij beschouwd hebben verhogen echter de efficiëntie van biomassagebruik voor wat betreft kosten, vermindering van broeikasgas-emissies en landbouwgrondgebruik. In vergelijking met bio-energieproductie verminderen de multiproductsystemen die onderzocht zijn in hoofdstuk 2 de primaire kosten van bio-brandstof met ongeveer 5 tot ruim 50 €/GJ_{LHV}, waarbij de grootste vermindering alleen bereikt wordt met een zeer hoogwaardige materiële toepassing (ter vergelijking: de primaire kosten van bio-brandstof in de bio-energiesystemen zijn ca. 2—15 €/GJ_{LHV}, terwijl de prijzen voor steenkool ca. 2 €/GJ_{LHV} zijn). Voor wat betreft broeikasgas-emissies verlagen deze multiproductsystemen de reductie daarvan met 3—10 Mg CO_{2eq} per hectare per jaar biomassateelt. De resultaten betreffende de cascades van korte-omloop hout die in hoofdstuk 3 onderzocht zijn hebben een grote bandbreedte. Door cascadering veranderen, in vergelijking met bio-energieproductie, de vermeden broeikasgas-emissies met ca. -13 tot 23 Mg CO_2 per hectare per jaar, en de kosten van broeikasgas-emissiereducties met ca. -300 tot 2000 €/Mg CO_2. Ter vergelijking: het gebruik van korte-omloop hout voor bio-elektriciteitsproductie resulteert in vermeden emissies van ca. 5 Mg CO_2 per hectare per jaar en in kosten van ongeveer 100 €/Mg CO_2. Het gebruik van landbouwresiduen voor energieproductie, zoals onderzocht in hoofdstuk 4, verhoogt de efficiëntie van de productie van kunststof uit biomassa. Dat wil zeggen, de vermindering van broeikasgas-emissies stijgt met waarden tot 15 Mg CO_{2eq} per hectare per jaar. Tot slot leidt multifunctioneel biomassagebruik in de bioraffinaderijsystemen voor polymelkzuurproductie, zoals geanalyseerd in hoofdstuk 5, tot een efficiëntieverhoging van 4-12 MgCO_{2eq} per hectare per jaar en 0-200 €/Mg biomassa input.

Daarnaast kan geconcludeerd worden dat binnen de structuur van biomassasystemen de materiële hoofdtoepassing de grootste invloed op de prestaties van multifunctionele biomassasystemen heeft. Bovendien kan het gebruik van landbouwresiduen voor energieproductie de prestaties van biomassasystemen beduidend verbeteren. De waardering van multifunctionele biomassasystemen hangt uiteraard sterk af van de referentiesystemen. Het soort materiaal en energiedrager dat wordt gesubstitueerd en het afvalverwerkingssysteem blijken essentieel, zowel voor de kosten als voor de vermindering van broeikasgas-emissies van biomassasystemen.

In het geval van biomaterialen met een relatief lange levensduur kan de tijdsdimensie van de koolstofopslag in deze materialen een belangrijke rol spelen. Wanneer bijvoorbeeld tijd middels een netto-contante-waardemethode in de berekeningen meegenomen wordt, verminderen de netto broeikasgas-emissiereducties van biomassacascades. Ook beïnvloeden de marktprijzen van land, materialen en energiedragers de economische prestaties van multifunctionele biomassasystemen sterk. Het onderzoek in hoofdstuk 6 wijst aan dat met een toenemend gebruik van biomassa voor materialen en energie, de kosten van deze opties voor broeikasgas-emissiereductie kunnen stijgen. Omdat veel multifunctionele biomassasystemen toch al tamelijk hoge kosten voor broeikasgas-emissiereductie met zich meebrengen, zou daardoor het economische potentieel van deze systemen om broeikasgas-emissies te verminderen verder worden beperkt. Dit geldt vooral voor biomaterialen met relatief kleine markten. Voorts moet worden opgemerkt dat marktmechanismen de soorten referentietoepassingen die worden gesubstitueerd kunnen beïnvloeden (bijvoorbeeld, de soort fossiele energiedrager).

Uit de analyses in dit proefschrift kunnen ook methodologische lessen worden getrokken. Er kan worden geconcludeerd dat het meenemen van landbouwgrondgebruik in de vergelijking van biomassasystemen waardevolle inzichten in de rangschikking van deze systemen op hun effectiviteit kan verschaffen. Omdat vaak biomaterialen, waarvoor voor de productie land gebruikt wordt, gesubstitueerd worden in het referentiesysteem, is het noodzakelijk ook in het referentiesysteem dit landgebruik in rekening te brengen. De benadering die in dit proefschrift wordt gehanteerd, dat wil zeggen, de aanname dat op dat land anders biomassa voor energie-opwekking geproduceerd zou worden, is geschikt om een vergelijking van biomassasystemen mogelijk te maken. Evenwel zou overeenstemming in de wetenschap over een standaardmethodologie voor het omgaan met dit probleem wenselijk zijn. Ook zou er een standaardmethodologie ontwikkeld moeten worden voor het meenemen van de tijdsdimensie in de berekening van broeikasgas-emissiebalansen, omdat biomateriaalsystemen koolstof relatief lang kunnen opslaan. Onze analyse, die een netto-contante-waardemethode gebruikt, toont de potentieel grote invloed van de tijd op de resultaten aan. Tenslotte beïnvloedt de schaal van biomassaproductie en -gebruik zowel de kosten als de broeikasgas-emissiebalans van biomassasystemen. Dit is toe te schrijven aan veranderingen van prijzen en van gesubstitueerde referentietoepassingen. De combinatie van een bottom-up analyse en een eenvoudige analyse van de vraag naar land-, materiaal- en energie die in dit proefschrift wordt voorgesteld, toont belangrijke strekkingen van de afhankelijkheid van biomassasysteemkosten van de schaal van biomassagebruik.

Ter conclusie: mits zorgvuldig ontworpen, rekening houdend met referentiesystemen en met land-, materiaal- en energiemarkten, zijn multifunctionele biomassasystemen een goede keuze om biomassa efficiënt te gebruiken voor wat betreft de vermindering van broeikasgas-emissies, gebruik van (landbouw-)grond en totale systeemkosten. De beste multifunctionele biomassasystemen die in dit proefschrift zijn onderzocht, verhogen de broeikasgas-emissiereductie per eenheid gebruikte landbouwgrond met een factor vijf in vergelijking met enkelvoudig gebruik van biomassa, en verlagen de totale systeemkosten eveneens met een factor vijf. Nochtans, voor de evaluatie van (multifunctionele) biomassasystemen bij grootschalig biomassagebruik moeten de interacties tussen biomassagebruik en land-, materiaal- en energiemarkten beter begrepen worden. Daarom zou verder onderzoek naar optimale biomassasystemen voor vermindering van broeikasgas-emissies bottom-up informatie over complexe biomassasysteem moeten combineren met kennis van marktmechanismen uit top-down analyses.

.

Zusammenfassung und Schlussfolgerungen

Biomasse kann einen Beitrag zur Vermeidung von Treibhausgas (THG) -Emissionen leisten, indem entweder herkömmliche Materialien ersetzt oder Biobrennstoffe zur Verfügung gestellt werden. In industrialisierten Ländern wird gegenwärtig jedoch nur wenig Biomasse für derartige Zwecke genutzt. Eine Hauptursache für diesen geringen Anteil von Biomassenutzungen in Europa sind ihre oft hohen Produktionskosten, welche u.a. auf die verhältnismäßig geringe Verfügbarkeit landwirtschaftlicher Flächen zurückzuführen ist. Kurz- bis mittelfristig werden daher effizientere und kostengünstigere Alternativen zur Einführung der Biomassenutzung benötigt. Solche Alternativen könnten die weitere Entwicklung multifunktionaler Biomassesysteme sowie die Verlagerung der Biomasseproduktion in vorteilhaftere Gebiete, z.B. Osteuropa, sein. Multifunktionale Biomassesysteme umfassen die Konzepte der "Mehrfachproduktnutzung" und der "Kaskadierung". Die Definition von Mehrfachproduktnutzung ist die Verwendung einer Biomasseressource für verschiedene Zwecke, während Kaskadierung die aufeinander folgende Nutzung von Biomasse für mehrere Zwecke, d.h. Materialien, Materialrecycling und Energierückgewinnung, beinhalten. Wichtige Indikatoren für die Effizienz multifunktionaler Biomassesysteme sind die Einsparung nicht erneuerbarer Energie, die Vermeidung von THG-Emissionen, der Bedarf an (landwirtschaftlicher) Fläche und die Gesamtkosten des Systems. Bisher haben jedoch nur wenige Studien zu multifunktionalen Biomassesystemen diese Parameter quantitativ analysiert. Die zentrale Frage dieser Dissertation lautet daher: *Wie groß ist das Potential multifunktionaler Biomassesysteme zur Verringerung der Kosten und zur Verbesserung der Flächennutzungseffizienz in Bezug auf sowohl die Einsparung nicht erneuerbarer Energien als auch die Verringerung von THG-Emissionen.* Zwei Aspekte spielen eine wichtige Rolle bei der Beantwortung dieser Frage. Erstens ist es nötig die Methoden zur Berechnung von Kosten, Flächennutzung, THG-Emissionen und dem Verbrauch nicht erneuerbaren Energien den Erfordernissen der Bewertung multifunktionaler Biomassesysteme anzupassen. Insbesondere Herangehensweisen zur Allokation von Umwelteinflüsse und Kosten oder der Systemerweiterung, zur Einbeziehung der Zeitdimension in die Berechnungen und zur Integration von Marktpreisveränderungen auf Grund der Einführung von Biomassesystemen in großem Maßstab verdienen besondere Aufmerksamkeit. Zweitens hängt der potenzielle Vorteil von der Art des betrachteten Biomassesystems ab. Um viel versprechende multifunktionale Biomassesysteme identifizieren zu können, müssen daher die Mechanismen dieser Abhängigkeit untersucht werden.

In dieser Dissertation wird die Effizienz multifunktionaler Biomassesyteme hinsichtlich der THG-Emissionen, des Verbrauchs nicht erneuerbarer Energien, des Bedarfs an landwirtschaftlichen Flächen und der Kosten quantitativ bestimmt. Hierfür werden mehrere Fallstudien zu multifunktionalen Biomassesystemen, die in einer ersten Analyse viel versprechend erschienen, ausgeführt. Die Fallstudien sind in Europa angesiedelt, wobei insbesondere die Verhältnisse in Polen näher untersucht wurden. Auf diese Weise kann das Biomasseproduktionspotenzial in Osteuropa erforscht werden, wo die Staaten über (gegenwärtig) große landwirtschaftliche Gebiete, potenziell durchschnittliche bis hohe landwirtschaftliche Erträge und vergleichsweise geringe Landnutzungs- und Arbeitskosten verfügen.

Kapitel 2 untersucht das Konzept der Mehrfachproduktnutzung landwirtschaftlicher Pflanzen, d.h. der teilweisen Nutzung einer Pflanze als Energieträger und als Rohstoff für Materialien. Mehrfachproduktnutzung kann womöglich sowohl eine zusätzliche Einkommensquelle darstellen als auch für weitere Verminderungen von THG-Emissionen sorgen. In diesem Kapitel werden die Vorteile von Mehrfachproduktsystemen gegenüber der ausschließlichen Produktion von Bioenergie analysiert. Die Effizienzen der Mehrfachproduktsysteme werden dargestellt als Kosten primärer Biobrennstoffe und THG-Emissionsreduktionen pro Hektar landwirtschaftlicher Nutzfläche. Für diese Analyse wird eine Fallstudie durchgeführt und mit einer Monte-Carlo-Analyse die Sensitivität der Ergebnisse untersucht. Die betrachteten Pflanzen sind Weizen, Hanf und Pappeln in Kurzumtriebsplantagen. Die Mehrfachproduktsysteme werden für die Niederlande und Polen untersucht. Die Reduktion von THG-Emissionen dieser Mehrfachproduktnutzungssysteme betragen ca. 0,2 bis 2,4 Mg $CO_{2\text{-Äq}}$ per Hektar und Jahr in Polen und 0,9 bis 7,8 Mg $CO_{2\text{-Äq}}$ per Hektar und Jahr in den Niederlanden. Die Bereitstellungskosten primärer Biobrennstoffe liegen zwischen 0,1 und 9,8 €/GJ in Polen und –4,1 und –1,7 €/GJ in den Niederlanden. Die Ergebnisse zeigen, dass der wirtschaftliche Vorteil von Mehrfachproduktsystemen stark von den Marktpreisen für Biomaterialien, den Biomasseproduktionskosten und den Biomasseerträgen abhängt. Die Nettoreduktion von THG-Emissionen per Hektar und Jahr wird dagegen durch die spezifische THG-Emissionsreduktion der Materialanwendung, das Referenzenergiesystem und die THG-Emissionen der landwirtschaftlichen Produktion stark beeinflusst. Die Mehrfachproduktnutzung landwirtschaftlicher Pflanzen kann die Kosten für primäre Biobrennstoffe deutlich verringern. Dies gilt jedoch nicht im Allgemeinen, sondern hängt von der Art der Pflanze und der Materialnutzung ab. Durch die Mehrfachproduktnutzung ergibt sich keine Nettoreduktion der THG-Emissionen für die Biomassesysteme, die in diesem Kapitel betrachtet wurden. Demnach ist die Mehrfachproduktnutzung nicht von vorneherein geeignet um die Effizienz von Bioenergiesystemen zu verbessern.

Kapitel 3 hat eine zweifache Zielsetzung: (1) die Auswahl und Entwicklung einer kohärenten Methode zum Vergleich von Biomassekaskaden im Hinblick auf den Flächenbedarf, die Potenziale zur Verminderung von CO_2-Emissionen und die Gesamtkosten des Systems und (2) die Identifizierung wichtiger Parameter und Aspekte, die die Effizienz von Biomassekaskaden beeinflussen. Für letzteres wird die hier entwickelte Methode in einer Fallstudie angewendet, in der verschiedene Kaskaden von Pappelholz aus Kurzumtriebsplantagen auf der Grundlage von Literaturdaten miteinander verglichen werden. Die Ergebnisse für die betrachteten Kaskaden variieren stark und reichen von Gewinnen für die CO_2-Emissionsreduktion von 200 €/Mg CO_2 zu Kosten für die CO_2-Emissionsreduktion von 2200 €/Mg CO_2, sowie von einer Nettoreduktion von CO_2-Emissionen pro Hektar und Jahr Pappelanbau von 28 Mg CO_2 zu Netto-CO_2-Emissionen von 8 Mg CO_2 pro Hektar und Jahr. Die Berechnung von CO_2-Emissionen und Kosten mittels Kapitalwertmethode verändert die Ergebnisse für Langzeitkaskaden deutlich. In diesem Fall verringern sich die Kosten und nehmen die CO_2-Emissionen zu. Im Allgemeinen kann Kaskadierung sowohl die CO_2-Emissionsreduktion pro Hektar und Jahr Biomasseproduktion als auch die Kosten für diese CO_2-Emissionsreduktion verbessern. Dies hängt jedoch von den Biomassenutzungen, die in der Kaskade kombiniert werden, ab. Parameter, die diese Ergebnisse stark beeinflussen, sind die Marktpreise und der kumulierte Energiebedarf der substituierten Referenzmaterialien und –energieträger und die Effizienz der Biomasseproduktion. Die Methode, die in diesem Kapitel präsentiert wird, ist geeignet, um den Bedarf an landwirtschaftlicher Fläche, die CO_2-Emissionsreduktion und die Gesamtkosten bzw. -gewinne von Biomassekaskaden zu quantifizieren und hebt den möglichen Einfluss der Zeitdimension auf die Attraktivität einzelner Kaskaden hervor.

Kapitel 4 vergleicht die Einsparung nicht erneuerbaren Energien und die Verminderung von THG-Emissionen der Produktion von Kunststoffen aus Biomasse mit der Produktion von Bioenergie auf der Basis des Flächenbedarfs. Angesichts der politischen Zielsetzungen zur Erhöhung des Einsatzes von Biomasse für die Energieversorgung und den gegenwärtigen Bemühungen Kunststoffe aus nachwachsenden Rohstoffen en gros zu produzieren, könnte die Verfügbarkeit landwirtschaftlicher Flächen für die Produktion von industriellen Nutzpflanzen zu einem limitierendem Faktor werden. Im Hinblick auf die Bedeutung, die der Energie- und THG-Problematik in der gegenwärtigen Umweltpolitik beigemessen wird, ist es wünschenswert, den Bedarf an landwirtschaftlicher Fläche in den Vergleich verschiedener Biomassenutzungen mit einzubeziehen. In den letzten Jahren sind zahlreiche Ökobilanzen für Kunststoffe aus Biomasse erstellt worden, von denen jedoch nur wenige die landwirtschaftliche Flächennutzung berücksichtigen. Deswegen werden in diesem Kapitel bestehende Ökobilanzen durch eine Analyse des Bedarfs an landwirtschaftlicher Fläche, bezogen auf die Einsparungen nicht erneuerbarer Energien und THG-

Emissionsreduktionen, erweitert. Die Ergebnisse zeigen, dass die Vorzüglichkeit verschiedener Optionen der Produktion von Biokunststoffen bezüglich der Höhe der Energieeinsparungen und THG-Emissionsreduktionen eine andere Rangfolge einnimmt wenn sie auf Basis des Bedarfs an landwirtschaftlicher Fläche anstatt auf Basis einer Einheit produzierten Kunststoffes verglichen werden. Mit dem Bedarf an landwirtschaftlicher Fläche als Grundlage des Vergleiches schneiden naturfaserverstärkte Kunststoffe und thermoplastische Stärke besser, Polymilchsäure vergleichbar und Polyhydroxyalkaonate schlechter ab als die Produktion von Bioenergie. Wenn zusätzlich die bioenergetische Nutzung von Ernterückständen in die Analyse miteinbezogen wird, verbessern sich die Energieeinsparungen und THG-Emissionsreduktionen der Biokunststoffe erheblich. Auch erscheint es wahrscheinlich, dass mittelfristig die Produktion von Kunststoffen aus Biomasse effizienter wird. Daher bietet die Produktion von Kunststoffen aus Biomasse angesichts einer möglicherweise geringen Verfügbarkeit landwirtschaftlicher Flächen interessante Möglichkeiten, um zur Verminderung des Verbrauchs nicht erneuerbarer Energien und zur Reduktion von THG-Emissionen beizutragen.

Kapitel 5 analysiert die möglichen quantitativen Vorteile multifunktionaler Bioraffineriesysteme hinsichtlich der Einsparung an Energie, der Reduktion von THG-Emissionen und der Kosten. Ebenfalls wird die Abhängigkeit dieser Vorteile vom Produktionsumfang und Marktvolumen untersucht. Für diese Analyse wird eine Fallstudie von Bioraffineriesystemen zur Produktion von Polymilchsäure durchgeführt. Die Systeme beinhalten multifunktionale Nutzungen von Biomasseresourcen, d.h. die energetische Nutzung von Ernterückständen, die Nutzung von Nebenprodukten, Materialrecycling und Energierückgewinnung aus Abfall. Die Leistung dieser Systeme wird sowohl per Kilogramm Kunststoffproduktion als auch per Hektar Biomasseproduktion betrachtet. Die Berechnungen werden auf der Grundlage von polnischen Daten und unter der Annahme, dass Biomasse- und Kunststoffproduktion in einem europäischen Energie- und Materialmarkt eingebettet sind, durchgeführt. Zuerst werden die Leistungen verschiedener Bioraffineriesysteme mit Hilfe einer Bottom-up-Analyse der einzelnen Prozessschritte untersucht. Darauf folgend werden die Marktpreise der Polymilchsäureprodukte, der Nebenprodukte und der landwirtschaftlichen Flächen, ausgehend von der Produktionskapazität, an Hand der Preiselastizität der Nachfrage bestimmt. Auf diese Weise werden die Kosten der Bioraffineriesysteme in Abhängigkeit von der Nachfrage nach landwirtschaftlicher Fläche und Materialien berechnet. Sämtliche Bioraffineriesysteme zur Produktion von Polymilchsäure führen zu Nettoeinsparungen des nicht erneuerbaren Energieverbrauchs von 70 bis 220 GJ pro Hektar und Jahr und zu Nettoreduktionen von THG-Emissionen von 3 bis 17 Mg $CO_{2\text{-Äq}}$ pro Hektar und Jahr. Die meisten der untersuchten Bioraffineriesysteme resultieren in Nettokosten des Gesamtsystems von bis zu 4600 € pro Hektar und Jahr. Dahingegen hat die Polymilchsäureproduktion aus Kurzumtriebsholz Nettogewinne des Gesamtsystems von ca. 1100 € pro Hektar und Jahr zur Folge, wenn eine relativ hochwertiges Produkt, in diesem Fall Kunstfasern, einen großen Anteil der Produktion einnimmt. Multifunktionalität ist notwendig, um Energieeinsparungen und THG-Emissionsreduktionen von Bioraffineriesystemen zur Produktion von Polymilchsäure zu gewährleisten. An der Verringerung der Kosten hat die multifunktionale Nutzung von Biomasseresourcen jedoch nur einen kleinen Anteil. Die Preiselastizität der Nachfrage nach Materialien beeinflusst die Kosten der Bioraffineriesysteme stark. Die Preiselastizität der Nachfrage nach Land bekäme eine große Bedeutung, wenn Biomassesysteme in einem großen Umfang eingeführt würden.

Kapitel 6 evaluiert den möglichen Einfluss der Einführung von Biomasse zur stofflichen und energetischen Nutzung in großem Umfang und ihrer Marktvolumen auf die Marktpreise von Land, Material und Energie. Weiterhin wird die Rückkoppelung dieser Einflüsse auf die Kosten der Verminderung von THG-Emissionen betrachtet. In diesem Kapitel werden Angebotskurven von Kosten der Verminderung von THG-Emissionen für eine Nutzung von Biomasse im großem Maßstab aufgestellt Dies geschieht mit Hilfe einer Methode, die eine Bottom-up-Analyse von Biomassenutzungen, Angebotskurven von Biomassekosten und Marktpreise von landwirtschaftlichen Flächen, Biomaterialien und Bioenergie kombiniert. Letztere Marktpreise hängen von der Größenordung der Biomassenutzung und vom Marktvolumen der Materialien und Energieträger ab und werden mit Hilfe Preiselastizitäten geschätzt. Die Methode wird für eine Fallstudie in Polen im Jahre 2015 demonstriert. In dieser Fallstudie werden unterschiedliche Szenarien für die ökonomische Entwicklung und den Handel in Europa, die sowohl die Angebotskurven von Biomassekosten als auch die Land-, Material- und Energiemärkte beeinflussen, untersucht. Die Technologien, die in diesem Kapitel betrachtet werden, sind die Produktion von Holzfaserplatten, Polymilchsäure, Elektrizität und Methanol. Für diese Technologien gilt, dass innerhalb der untersuchten Szenarien die Kosten von THG-Emissionsreduktionen stark mit dem Umfang der Biomasseproduktion ansteigen. Der Einfluss einer Einführung von Biomassenutzungen en gros, auf die Kosten des Biomasseangebots und die Marktpreise von Land, Material und Energieträgern, verringern das Potenzial zur Verminderung von THG-Emissionen unter Kosten von 50 €/Mg $CO_{2\text{-Äq}}$ um ca. 13-70% in den verschiedenen Szenarien. Die Produktion von Biomaterialien trägt nur geringfügig zum kostengünstigen Reduktionspotenzial von THG-Emissionen bei. Die Ursache hierfür ist der verhältnismäßig kleine Umfang von Materialmärkten und die daraus folgende starke Abnahme der Marktpreise von Biomaterialien mit steigendem Produktionsumfang. Die Kosten der Reduktion von Treibhausgasemissionen hängen stark von den Angebotskurven von Biomasse, der Preiselastizität der Landnachfrage und den Marktvolumen von Bioenergieträgern ab. Unsere Analyse zeigt, dass diese Einflüsse bei der Entwicklung von Strategien zur Implementierung von Biomassenutzungen berücksichtigt werden sollten. Jedoch sind die Schätzwerte für Preiselastizitäten, welche in der wissenschaftlichen Literatur vorhanden sind, unsicher und die Marktvolumen von Biomasseprodukten hängen von ihrer jeweiligen Wettbewerbsfähigkeit ab. Um diesen Ungewissheiten entgegenzuwirken, ist eine Kombination von einer Bottom-up Analyse mit einer Analyse von Märkten zu empfehlen.

Zusammenfassend haben wir verschiedene multifunktionale Biomassesysteme und ihre potenziellen Vorteile hinsichtlich der Reduktion von THG-Emissionen, der Energieeinsparungen, des Flächenbedarfs und der Kosten analysiert. Die Ergebnisse für die unterschiedlichen Systeme variieren beträchtlich und reichen von keinen bis zu deutlichen Vorteilen multifunktioneller Biomassesysteme. Die Effizienz multifunktionaler Biomassesysteme wird durch viele Faktoren beeinflusst. Diese Faktoren hängen einerseits von der Struktur des Biomassesystems, z. B. von der Art der produzierten Materialien und Energieträger oder deren Effizienz und Kosten Produktion, und andererseits von äußeren Umständen, z.B. den Marktvolumen und Preisen von Materialien oder CO_2-Intensitäten der Referenzsysteme, ab.

Die meisten multifunktionellen Biomassesysteme, die wir untersucht haben, erhöhen jedoch die mögliche Effizienz der Nutzung von Biomasse in Bezug auf die Kosten, die Reduktion von THG-Emissionen und den Bedarf an landwirtschaftlicher Fläche. Im Vergleich zur Produktion von Bioenergie verringern die Mehrfachproduktsysteme, die in Kapitel 2 betrachtet werden, die Kosten von Primärbrennstoffen um 5 bis zu mehr als 50 €/GJ_{LHV}, wobei die größten Kostenreduktionen nur mit einer sehr hochwertigen Materialanwendung erreicht werden. (Die Kosten von Primärbrennstoffen in Bioenergiesystemen liegen vergleichsweise bei ca. 2—15 €/GJ_{LHV}, während die Preise von Kohle bei ca. 2 €/GJ_{LHV} liegen.) Hinsichtlich der THG-Emissionen vermindern diese Mehrfachproduktsysteme deren Reduktion um ca. 3-10 Mg $CO_{2\text{-Äq}}$ pro Hektar und Jahr Biomasseanbau. Die Resultate für die Kaskaden von Holz aus Kurzumtriebsplantagen, die in Kapitel 3 untersucht werden, zeigen eine sehr große Bandbreite. Im Vergleich zur alleinigen Nutzung von Biomasse für die Energieproduktion verändern sich durch die Kaskadierung die verminderten THG-Emissionen um ca. -13 bis 23 Mg CO_2 pro Hektar und Jahr und die Reduktionskosten von THG-Emissionen um ca. -300 bis 2000 €/Mg CO_2. Im Vergleich dazu führt die Nutzung von Holz aus Kurzumtriebsplantagen zur Elektrizitätsproduktion zu einer Vermeidung von THG-Emissionen von ca. 5 Mg $CO_{2\text{-Äq}}$ pro Hektar und Jahr bei Kosten von ca. 100 €/Mg CO_2. Die energetische Nutzung landwirtschaftlicher Reststoffe, die in Kapitel 4 betrachtet wird, erhöht den Nutzen der Herstellung von Kunststoffen aus Biomasse, so erhöht sich die Reduktion von THG-Emissionen um ca. 15 Mg $CO_{2\text{-Äq}}$

pro Hektar und Jahr. Schließlich führt die multifunktionale Biomassenutzung in Bioraffineriesystemen zur Polymilchsäureproduktion, die in Kapitel 5 analysiert werden, zu Vorteilen von ca. 4-12 Mg $CO_{2\text{-Äq}}$ pro Hektar und Jahr und 0-200 €/Mg Biomasseinput gegenüber der nicht multifunktionalen Biomassenutzung.

Weiterhin kann geschlossen werden, dass ausgehend von der Struktur der Biomassesysteme, die materielle Hauptanwendung den größten Einfluss auf die Effizienz multifunktionaler Biomassesysteme hat. Außerdem kann die energetische Nutzung landwirtschaftlicher Reststoffe zur Energieproduktion die Effizienz der Biomassesysteme erheblich verbessern. Die Bewertung multifunktionale Biomassesysteme hängt natürlich stark von den alternativen Referenzsystemen ab. Sowohl die Art der Materialien und Energieträger, die substituiert werden, als auch das Abfallwirtschaftssystem sind entscheidend für die durch Biomassesysteme entstandenen Kosten und erzielten THG-Emissionsreduktionen.

Bei Biomaterialien mit einer verhältnismäßig langen Lebensdauer kann die Zeitdimension der temporären Speicherung von Kohlenstoff in diesen Materialien eine wichtige Rolle spielen. Wenn die Zeitdimension zum Beispiel durch die Anwendung einer Kapitalwertmethode berücksichtigt wird, sinken die Nettoreduktionen von THG-Emissionen durch die Kaskadierung. Marktpreise von Land, Materialien und Energieträgern beeinflussen die ökonomische Effizienz von multifunktionalen Biomassesystemen ebenfalls stark. Die Analyse in Kapitel 6 deutet darauf hin, dass sich mit einer zunehmenden Verwendung von Biomasse für Materialien und Energie die Kosten von THG-Emissionsreduktionen erhöhen können. Während viele multifunktionale Biomassesysteme bereits recht hohe Reduktionskosten von THG-Emissionen mit sich mitbringen, könnte ihr ökonomisches Potenzial zur Reduktion von THG-Emissionen hierdurch zusätzlich eingeschränkt werden. Dies gilt insbesondere für Biomaterialien, die durch vergleichsweise kleine Märkte charakterisiert sind. Außerdem sollte zur Kenntnis genommen werden, dass Marktmechanismen auch die Art der substituierten Referenzmaterialien und –energieträger, z.B. die Art fossiler Brennstoffe, beeinflussen.

Auch methodische Lektionen können aus den Analysen in dieser wissenschaftlichen Arbeit gelernt werden. Es kann geschlossen werden, dass die Einbeziehung der Nutzung landwirtschaflicher Flächen in den Vergleich von Biomassesystemen wertvolle Erkenntnisse über deren relative Vorzüglichkeit verschaffen kann. Da häufig Biomaterialien im Referenzsystem substituiert werden, ist es notwendig, auch die Landnutzung für deren Produktion im Referenzsystem zu berücksichtigen. Die Methode, die in dieser Dissertation verwendet wird, nämlich die Annahme, dass alternativ Biomasse für die Energieproduktion auf diesen Flächen produziert wird, ist für den Vergleich von Biomassesystemen geeignet. Jedoch wäre ein Einvernehmen über eine standardisierte Methode, für den Umgang mit dieser Problemstellung, in der wissenschaftlichen Forschung wünschenswert. Auch ist die Entwicklung einer Standardmethode für die Einbeziehung der Zeitdimension in die Berechnung von THG-Emissionsbilanzen erforderlich, weil Biomaterialsysteme Kohlenstoff für verhältnismäßig lange Zeiträume speichern können. Unsere Analyse, die eine Kapitalwertmethode verwendet, zeigt den potenziell großen Einfluss der Zeitdimension auf die Ergebnisse. Schließlich beeinflusst die Größenordung der Biomasseproduktion und –nutzung sowohl die Systemkosten als auch die THG-Emissionsbilanzen durch die Veränderung von Marktpreisen und Referenzsystemen. Die Kombination einer Bottom-up-Analyse mit einer einfachen Analyse der Nachfrage nach Land, Material und Energieträgern, wie sie in dieser Dissertation vorgestellt wird, zeigt bedeutende Tendenzen der Abhängigkeit von Biomassesystemen von der Größenordung der Biomassenutzung.

Abschließend kann geschlossen werden, dass multifunktionale Biomassesysteme eine gute Alternative sind um Biomasse effizient im Hinblick auf die Reduktion von THG-Emissionen, den Bedarf an (landwirtschaftlicher) Fläche und die Gesamtkosten des Systems zu nutzen, sofern sie sorgfältig unter Berücksichtigung der Referenzsysteme und der Land-, Material- und Energiemärkten ausgewählt werden. Die besten der hier untersuchten multifunktionalen Biomassesysteme erhöhen die Reduktion von THG-Emissionen pro Einheit landwirtschaftlicher Fläche bis zu einem Faktor fünf verglichen mit einfachen Biomassenutzungen. Ebenfalls verringeren sie die Gesamtkosten der Biomassesystem um bis das Fünffache. Um jedoch (multifunktionale) Biomassesysteme für großangelegte Biomassenutzungen bewerten zu können, müssten die Wechselwirkungen zwischen der Biomassenutzung und den Land-, Material- und Energiemärkten besser verstanden werden. Weitergehende Untersuchungen über optimale Biomassesysteme für die Verminderung von THG-Emissionen sollten daher Informationen aus Bottom-up-Analysen komplexer Biomassesysteme mit Erkenntnissen über Marktmechanismen aus Top-down-Analysen kombinieren.

References

AAc: 2001, *Eco-profile of starch – Summary report*. Association des Amidonneries de Céréales de l'Union Européenne, Brussels.

Aden, A., Ruth, M., Ibsen, K, Jechura, J., Neeves, K., Sheehan, J., Wallace, B., Montague, L., Slayton, A. and Lukas, J.: 2002, *Lignocellulosic Biomass to Ethanol Process Design and Economic Utilizing Co-Current Dilute Acid Prehydrolysis ad Enzymatic Hydrolysis for Corn Stover*. NREL, Golden.

Annetts, J.E. and Audsley, E.: 2003, Modelling the value of a rural biorefinery - part I: the model description, *Agricultural Systems*, **76**, 39—59.

Anonymous: 2001, Ethylene Glycol Market Remains Weak Globally as Mideast Producers Target Asia for Exports, *Chemical Market Reporter*, **2001** (July 2).

AOO: 2002, *Mileueffectrapport Landelijk Afvalbeheerplan - Achtergronddocument A 14 Uitwerking "GFT-afval"*. Afval Overleg Orgaan, Utrecht.

APME: 1999, *Eco-profiles of polymers and related intermediates*. Association of Polymers Manufacturers in Europe, Brussels.

APME: 2003a, *Plastics – An analysis of plastics consumption and recovery 2001&2002*, Association of Polymers Manufacturers in Europe, Brussels.

Atherton, B.G., Ruth, M.F., Lechura, J.L. and Ibsen, K. N.: 2002, *Overall energy balance for the corn stover to ethanol process*. Paper presented at the 24th Symposium on Biotechnology for Fuels and Chemicals, 28 April–1 May, Gatlinburg.

Azar, C. and Berndes, G.: 1999, The implication of carbon dioxide abatement policies on food prices. In: Dragun, A.K. and Tisdell, C. (Eds.), *Sustainable Agriculture and Environment - Globalisation and the impact of trade liberalisation*, Edward Elgar, Cheltenham, 153—170.

Baitz, M., Kreissig, J. and Wolf, M.: 2000, Methode zur Integration der Naturraum-Inanspruchnahme in Ökobilanzen, *Forstwissenschaftliches Zentralblatt*, **119**, 128—149.

Balatinecz, J., Kretschmann, D.E. and Leclercq, A.: 2001, Achievements in the utilisation of poplar wood - guideposts for the future', *The Forestry Chronicle* **77**, 265—269.

Benjamin, Y. and van Weenen, H.: 2000, *Design for Sustainable Development - Crops for Sustainable Enterprise*, Environmental Design for Ecological Need (EDEN), International Design and Environmental Activities (IDEA), Dublin.

Berndes, G., Hoogwijk, M. and van den Broek, R.: 2003, The contribution of biomass in the future global energy supply: a review of 17 studies, *Biomass and Bioenergy*, **25**, 1—28.

Biewinga, E.E., and van der Bijl, G.: 1996, *Sustainability of energy crops in Europe*. Centre for Agriculture & Environment, Utrecht.

Bollen, J., Manders, T. and Mulder, M: 2004. *Four Futures for Energy Markets and Climate Change*, CPB Netherlands Bureau for Economic Policy Analysis, The Hague.

Börjesson, P.: 1999, Environmental effects of energy crop cultivation in Sweden - I: Identification and quantification, *Biomass and bioenergy*, **16**, 137—154.

Börjesson, P. and Gustavsson, L.: 2000, Greenhouse gas balances in building construction: wood versus concrete from life-cycle and forest land-use perspectives, *Biomass and Bioenergy* **28,** 575—588.

Boogardt, V.C.A.: 2000, *Hout in de herkansing - Onderzoek naar cascades in het houtgebruik*. Stichting Bos en Hout, Wageningen.

Brandrup, J., Immergut, E.H., Grulke, E.A., Abe, A., and Bloch, D.R.: 1999, *Polymer Handbook*. 4th edition, Wiley, New York.

Carlsson-Kanyama, A. and Faist, M.: 2000, *Energy use in the Food Sector: A data survey*. Stockholm University, Stockholm.

van den Broek, R.C.A., van den Burg, T., van Wijk, A.J.M. and Turkenburg, W.C.: 2000, Electricity generation from eucalyptus and bagasse by sugar mills in Nicaragua: A comparison with fuel oil electricity generation on the basis of costs, macro-economic impacts and environmental emissions, *Biomass & Bioenergy* **19**, 311—335.

van den Broek, R., Faaij, A., and van Wijk, A.: 1995, Biomass Combustion Power Generation Technologies. In: European Commission (Ed.), *Energy from Biomass – an Assessment of two promising Systems for Energy Production - JOU2-CT93-0397*, EC, Brussels.

van den Broek, R.C.A., Treffers, D.J., Meeusen, M., van Wijk, A., Nieuwlaar, E., Turkenburg, W.C. and Londo, M.: 2001, Green energy or organic food: a Life Cycle Assessment comparing two uses of set-aside land', *Journal of Industrial Ecology*, **5**, 65—87.

van den Broek, R.C.A., van Wijk, A., Turkenburg, W.C.: 2002, Electricity from energy crops in different settings—a country comparison between Nicaragua, Ireland and the Netherlands, *Biomass and Bioenergy*, **22** (2), 79—98.

CBS: 2002, *Statline databank*. http://www.cbs.nl/nl/cijfers/statline/index.htm (accessed 2002), Central Bureau voor de Statistiek, Heerlen.

CEFIC: 2000, *Petrochemistry Activity Review*. European Chemical Industry Council, Brussels.

CEFIC: 2002, *A panorama of the petrochemical industry's performance in 2000 – 2001*, European Chemical Industry Council - petrochemistry programme, Brussels.

CEFIC: 2003, *Petrochemistry in Western Europe – facts 2003*, http:www.petrochemistry.net (accessed 2004), European Chemical Industry Council - petrochemistry programme, Brussels.

Ciaian, P., Swinnen, W. and Münch, W.: 2002. *The impact of the common agricultural policy on income distribution and welfare in Central and Eastern European countries*. Paper presented at the 5th International Workshop on European Economy, Lisbon, 22—23 November 2002.

CIRFS: 2002, *Man-made fibre production*, Comité International de la Rayonne et des Fibres Synthétiques. Brussel.

Clarinval, A.-M.: 2002, Classification and comparison of thermal and mechanical properties of commercialised biodegradable polymers. In: IENICA (Ed.), *Proceedings of Bioplastics Work*, York.

CML: 2001, *LCA; An operational guide to the ISO standards*. University Leiden, Leiden.

Coelman, B.T. and Pijanowska, B.: 1996, *Mogelijkheden voor kleinschalige energie-opwekking met geteelde biomassa*, CPV, Wageningen.

Corbière -Nicollier, T., Gfeller Laban, B., Lundquist, L., Leterrier, Y., Månson, J.-A.E. and Jolliet, O.: 2001, Life cycle assessment of biofibres replacing glass fibres as reinforcement in polymers. *Resources, Conservation, and Recycling* **33** (4), 267–287.

Crank, M., Hüsing, B., Marscheider-Weidemann, F., Patel, M. and Schleich, J.: 2004, *Techno-economic feasibility of large-scale production of biobased polymers in Europe (PRO-BIP)*, Utrecht University, Fraunhofer ISI, Utrecht/Karlsruhe.

CSO: 2003, Concise statistical yearbook of Poland 2003, Central Statistical Office Poland, Warsaw.

Dale, B.E.: 2002, *Integrated biorefining systems: A sustainable future seen through a glass, darkly*. Paper presented at 2002 Fall Annual Meeting, American Institute of Chemical Engineers, 3—8 November, Indianapolis.

Van Dam, J., Lewandowski, I. and Faaij, A.: 2004, *Regional biomass potential assessment in Eastern European Accession Countries*. Paper presented at the 2nd World Conference for Biomass for Energy, Industry and Climate Protection, 10—14 May, Rome.

DCO: 1999, *Sustainable Technological Development in Chemistry - Improving the quality of life through chemistry and agriculture*, Stichting DCO, Wageningen.

De la Torre Ugarte, D.D.G. and Ray, D.E.: 2000, Biomass and bioenergy applications of the POLYSYS modeling framework, *Biomass and Bioenergy*, **18**, 291—308.

De La Torre Ugarte, D., Walsh, M.E., Shapouri, H. and Slinsky, S.P.: 2003, *The economic impacts of bioenergy crop production on US agriculture*, USDA, Washington, DC.

DEFR: 2003, *Growing SRC: best practical guidelines*, UK Department for Environment, Food and Rural Affairs, London.

DHV: 2001, *Waste management policies in Central and Eastern European countries: Current Policies and Trends*. DHV, Prague.

Diener, J. and Siehler, U.: 1999, Ökologischer Vergleich von NMT- und GMT-Bauteilen. *Die Angewandte Makromolekulare Chemie*, **272**, 1–4.

DiGITIP: 2002, *Composite Materials Drive and Innovation. Le 4 Pages de statistques industrielles*, **158** (february 2002).

Dinkel, F., Pohl, C., Ros, M. and Waldeck, B.: 1996, *Ökobilanz stärkehaltiger Kunststoffe*. BUWAL (Bundesamt für Umwelt, Wald und Landschaft. Bern.

Dinkelbach, L., van Doorn, J., Jansma, R., den Raad, A., Jager, J., van Onna, M.J.G.M., Huisman, W., Heineman, A., Annevelink, E., Kasper, G.J.: 1999, *Geteelde biomassa voor energieopwekking in Nederland; identificatie van de meest veelbelovende mogelijkheden tot kostenreductie in vier ketens*. Energy Research Foundation ECN, Petten.

Dix, B., Roffael, E., and Buchholzer,P.: 1993, Industrieholz fuer die Spanplatten- und Zellstoffherstellung. In: Hüttermann, A. E. (ed), *Anbau von Pappel bei mittlerer Umtriebszeit - Aspekte der Produktionsbiologie, Nutzungstechnologien und Ökonomie*. J.D. Sauerländer's Verlag, Frankfurt a. M., 113—135.

DOE: 1997, *Renewable energy technology characterizations; biomass technologies*. US Department of energy, DOE/EPRI, Washingtonne D.C.

Dornburg, V. and Faaij, A.: 2001a, Cost and CO_2-emission reduction by applying multi-product and cascading systems of biomass. Paper presented at *Bioenergy for Rural Area Development*, Polish Academy of Science, 26—29 september 2001, Warsaw.

Dornburg, V. and Faaij, A.: 2001b, Efficiency and economy of wood-fired biomass energy systems in relation to scale regarding heat and power generation using combustion and gasification technologies, *Biomass and Bioenergy*, **21** (2), 91—108.

Dornburg, V. and Faaij, A.: 2001c, *Scenario-ontwikkeling voor de optimalisatie van energieproductie uit afvalstoffen in Nederland.* Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Van Driel, H. Nadall, V. and Zeelenberg, K.: 1997, The demand for food in the United States and the Netherlands: a systems approach with the CBS model. *Journal of applied econometrics*, **12**, 509—532.

DTO: 1997, Geintegreerde plantconversie. In: DTO(Ed.) *Duurzame Technologische Ontwikkeling Chemie; vijf technologie ontwikkelingsvelden*, Willems & Van den Wildenberg, Den Hague.

EC: 1997, *Energy for the future: Renewable sources of energy.* White Paper for a Community Strategy and Action Plan, European Commission, COM(97)599, Brussels.

EC: 1999, *Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste*. European Commission, Brussels.

EC: 2000, *Green Paper: Towards a European strategy for the security of energy supply*. European Commission, COM(2000)769, Brussels.

EC: 2002, *Atlas - Buildings - Envelope (and Passive Solar components)*, http://europa.eu.int/comm/ energy_transport atlas/htmlu/envelope.html (accessed 2004), European Commission, Brussels.

EC-BREC: 2003, *Biomass energy developments in Poland*. http://btgs1.ct.utwente.nl/pl-bec/index.html (accessed 2003), Polish Biomass Energy Centre, Warsaw.

ECCP: 2001, *European Climate Change Programme, Long Report 2001*. European Climate Change Programme, Brussels.

ECN: 2001, *Energie verslag Nederland 2000*, Energy Research Foundation ECN, Petten.

ECN: 2003, *PHYLLIS - database for biomass and waste*, http://www.ecn.nl/Phyllis (accessed 2003), Energy Research Foundation ECN, Petten.

Edwards, R.: 2004, *What's important in WTW analyses.* Paper presented at: Biofuels in transportation workshop at the 2nd World Conference for Biomass for Energy, Industry and Climate Protection, 11 May, Rome.

Eibl, M., Mangeng, B., and Alber, S.: 1996. *Ökobilanz von Lenzing Lyocell,* Lenzing Lyocell, Heiligenkreuz.

Eggels, P.G., Ansems, A.M.M. and van der Ven, B.L.: 2001, *Eco-efficiency of recovery scenarios of plastics packaging*, TNO, Apeldoorn.

Elliot, D.C.: 2004, Chemicals from Biomass. In: Cleveland, C.J. (Ed.), *Encyclopedia of energy*, Elsevier, Oxford.

Espey, M.: 1998, Gasoline demand revisited: an international meta-analysis of elasticities. *Energy Economics,* **20** (3), 295—319.

Estermann, R., Schwarzwälder, B. and Gysin, B.: 2000, *Life cycle assessment of Mater-Bi and EPS loose fills*. Composto, Olten.

Eurostat: 1997, *Panorama of EU Industry*. Office for Official Publications of the European Communities, Brussels/Luxembourg.

Eurostat: 2000, *Agricultural land prices and rents in the UE, Data 1977 – 1998.* Office for Official Publications of the European Communities, Brussels/Luxembourg.

Eurostat: 2000, *Panorama of European business*. Office for Official Publications of the European Communities, Brussels/Luxembourg.

Eurostat: 2003, *Agriculture in the European Union*. Office for Official Publications of the European Communities, Brussels/Luxembourg.

Eurostat: 2004, *CODED. European Concepts and Definition database*. Office for Official Publications of the European Communities, Brussels/Luxembourg.

Faaij, A., Hamelinck, C. and Tijmensen, M.: 2000, Long term perspectives for production of fuels from biomass; integrated assessment and RD&D priorities - preliminary results. In: Kyritsis, S., Beenackers, A.A.C.M., Helm, P., Grassi, A., and Chiaramonti, D. (eds.), *Proceedings of 1st World Conference on Biomass for Energy and Industry*, James & James, London, 687—691.

Faaij, A., Meuleman, B., and Van Ree, R.:1998: *Long term perspectives of BIG/CC technology*. NOVEM, Utrecht

FAO: 2004, *FAO Statistical Database - Forestry Data*, http://apps.fao.org/page/collections?subset=forestry (accessed 2003—2004), FAO, Rome.

FAO: 2003, Fertiliser use by crop in Poland, FAO, Rome.

Fan, Y., Nishida, H., Shirai, Y. and Endo, T.: 2003' Racemization on thermal degradation of poly(l-lactide) with calcium salt en structure, *Polymer Degradation and Stability*, **80**, 503—511.

Finke, R., Lu, W.-H., Theil, H.: 1984, A cross-country tabulation of own-price elasticities of demand, *Economics Letters,* **14**, 137—142.

Finnforest: 2003, *Interim Report for the Finnforest Corporation 1 – 6/2003.* Finnforest Corporation, Espoo.

Finnveden, G. and Ekvall, T.: 1998, Life-cycle assessment as a decision-support tool - the case of recycling versus incineration of paper, *Resources, conservation and Recycling* **24**, 235—256.

Forintek: 1993, *Raw material balances, energy profiles & environmental unit factor estimates for structural wood products*. Forintek Canada Corp., Vancouver.

Fraanje, P.J.: 1997, Cascading of pine wood, *Resource, Conservation and Recycling* **19**, 21—28.

Franssen, S.: 1999, *MATTER 3.0: Relaxing the Demand in the MATTER model*. ECN, Petten.

Frischknecht, R., Hofstetter, P., Knöpfel, I., Dones, R. and Zollinger, E.: 1994, *Ökoinventare für Energiesysteme*. ETH, Zürich.

Frühwald A., Scharai-Rad M., Hasch J., Wegener G. and Zimmer B.: 1997, *Erstellung von Őkobilanzen für die Forst- und Holzwirtschaft*. Informationsdienst Holz, Deutsche Gesellschaft für Holzforschung, München.

Gardner, W.D., Ximenes, F., Cowie, A. Marchant, J.F., Mann, S. and Dods, K.: 2002 *Decomposition of wood products in the lucas Heights landfill facility,* Paper presented at the Third Australian conference on life cycle assessment, 17—19 July 2002, Gold coast.

Garlotta, D.: 2001, A literature review of Poly(lactic acid), *Journal of Polymers and the Environment*, **9**, 63—84.

Gärtner, S.O., Müller-Sämann, K., Reinhardt, G.A., and Vetter, R.: 2002, Corn to Plastics: A comprehensive Environmental Assessment. In: *Proceedings of the 12th European Biomass Conference, 17 – 21 June 2002,* London: James & James, 1324—1326.

Gerngross , T.U. and Slater, S.: 2000, How green are green plastics? *Scientific American*, August: **282** (8), 37–41.

Gielen, D.J., Bos, A.J.M., de Feber, M.A.P.C. and Gerlagh, T.: 2000, *Biomass For Greenhouse Gas Emission Reduction Task 8: Optimal emission reduction strategies*. ECN, Petten.

Gielen, D., Fujino, J., Hashimoto, S. and Moriguchi, Y.: 2003, Modeling of global biomass policies, *Biomass and Bioenergy*, **25**, 177—195.

Goerten, J. and Beranek, M.: 2004a, *Electricity prices for EU households on 1 January 2004*, Eurostat, Brussels.

Goerten, J. and Beranek, M.: 2004b, *Electricity prices for EU industries on 1 January 2004*, Eurostat, Brussels.

Goldemberg, J. (ed): *World Energy Assessment of the United Nations*. UNDP, New York.

Goverse, T., Hekkert, M.P., Groenewegen P,. Worrell E., and Smits, R.E.H.M.: 2001, Wood innovation in the residential construction sector; opportunities and constraints, *Resource, Conservation and Recycling* **34,** 53—74.

Green, C.: 2000, Potential scale-related problems in estimating the costs of CO2 mitigation policies, *Climatic Change,* **44**, 331—349.

Groenenberg, H.: 2002, *Convergence and Development*, Thesis, Utrecht University, Utrecht.

Gruber, P. and O'Brien, M.: 2002, Polylactides "NatureWorks PLA", In: Y. Doi and A. Steinbüchel (Ed.), *Biopolymers Vol. 4*, Wiley-VCH, Weinheim, 235—249.

Guo, L.B., and Gifford, R.M.: 2000, Soill carbon stocks and land use change: a meta analysis, *Global Change Biology,* **8,** 345—360.

Hallam, A., Anderson, I.C. and Buzton, D.R.: 2001, Comparative economic analysis of perennial, annual, and intercrops for biomass production, *Biomass and Bioenergy*, **21**, 407—424.

Hamelinck, C.N. and Faaij, A.P.C.: 2002, Future prospects for production of methanol and hydrogen from biomass. *Journal of Power Sources* **111**, 1—22.

Hamelinck, C.H., Suurs, R. and Faaij, A.PC.: 2003, *International bioenergy transport cost and energy balance*. Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Hamelinck, C.: 2004. *Outlook for advanced biofuels.* PhD thesis, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Hansen, E.A.: 1993, Soil carbon sequestration beneath hybridpoplar plantations in the North Central United States, *Biomass & Bioenergy*, **5,** 431—436.

Haygreen, J.G., and Bowyer, J.L.: 1996*, Forest Products and Wood Science.* Iowa State University Press, Ames.

Heyde, M.: 1998, Ecological considerations on the use and production of biosynthetic and synthetic biodegradable polymers, *Polymer Degradation Stability,* **59**, 3–6.

van Heijningen, R.J.J., de Castro, J.F.M., Worrell, E., and Hazewinkel, J.H.O.: 1992, *Meer energiekentallen in relatie tot preventie en hergebruik van afvalstromen*. Castro Consulting Engineer, Amersfoort.

Hekkert, M.P., and Worrell, E.: 1998, *Technology Characterisation of Natural Organic Materials, Input data for Western European MARKAL*. Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Hekkert, M.P., Joosten L.A.J., and Worrell, E.: 2000, Reduction of CO2 emissions by management of material and product use, the case of transport packaging. *Resources, Conservation and Recycling,* **30**, 1—27.

Heller, M.C., Keoleian, G.A. and Volk, T.A.: 2003, Life cycle assessment of a willow bioenergy cropping system*, Biomass and Bioenergy*, **25**, 147—165.

Hoogwijk, M., Faaij, A., van den Broek, R., Berndes, G., Gielen, D. and Turkenburg W.: 2003, Exploration of the ranges of the global potential of biomass for energy, *Biomass and Bioenergy,* **25**, 119–133.

Hoogwijk, M., Faaij, A., Eickhout, B., de Vries, B. and Turkenburg, W.: 2004a, Potential of biomass energy under four land-use scenarios - Part A: The geographical and technical potential. In: Hoogwijk, M.: *On the global and regional potential of renewable energy sources*, PhD thesis, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Hoogwijk, M., Faaij, A., de Vries, B. and Turkenburg, W.: 2004b. *Potential of biomass energy under four land-use scenarios - Part B: Exploration of regional and global cost-supply curves*. In: Hoogwijk, M.: *On the global and regional potential of renewable energy sources*, PhD thesis, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Hydro Agri Dülmen GmbH, 1993. *Faustzahlen für Landwirtschaft und Gartenbau*, Verlagsunion Agrar, Münster.

IEA: 2002a, *Energy Balances of OECD Countries, 1999 – 2000*, International Energy Agency, Paris.

IEA: 2002b, *End-user petroleum product prices and average crude oil import costs - April 2002*. International Energy Agency, Paris.

IEA: 2002c, *CO_2 emissions from fuel combustion. 1971 – 2000*. International Energy Agency, Paris.

IEA: 2002d, *Energy Prices and* Taxes 2002. International Energy Agency, Paris.

IEA: 2003a. *Energy balances of non-OECD countries 2000 – 2001*, International Energy Agency, Paris.

IEA: 2003b. *Energy balances of OECD countries 2000 – 2001*, International Energy Agency, Paris.

IEA: 2003c. *Key world energy statistics 2003*, International Energy Agency, Paris.

IEA: 2004a. *IEA Bioenergy Task 38: Greenhouse Gas Balance of Biomass and Bioenergy systems*, http://www.joanneum.ac.at/iea-bioenergy-task38/, (accessed 2004), International Energy Agency, Paris.

IEA: 2004b, *End-user petroleum product prices and average crude oil import costs - March 2004*. International Energy Agency, Paris.

ILO: 2001, *Yearly statistics of Employment, Unemployment, Hours of Work, Wages, Labour Cost, Consumer Price Indices, Occupational Injuries, Strikes and Lockouts: 1969 – 2001*. International Labour Office, Bureau of Statistic, Geneva.

IPCC: 1995, *IPCC Second Assessment, Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses, ,* Cambridge University Press, Cambridge.

IPCC: 2000, *Special Report on Emission Scenarios,* Cambridge University Press, Cambridge.

IPCC: 2001a, *Climate Change 2001, The Scientific Basis.* Contribution of Working Group 1 to the Third Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.

IPCC: 2001b, *Climate Change 2001, Climate Mitigation 2001,* Contribution of Working Group 3 to the Third Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.

IPCC: 2001c, *Climate Change 2001, Impacts, Adaptations and Vulnerability,* Contribution of Working Group 2 to the Third Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.

Jansen, J.J., Sevenster, J. and Faber, P.J. *Opbrengst tabellen voor belangrijke boomsoorten in Nederland*. IBN-DLO, University Wageningen, Wageningen.

Johansson, D.: 2000, *Renewable Raw Materials - a way to reduced greenhouse gas emissions for the EU industry?* European Commission - DG Enterprise, Brussels.

Johansson, T.B. and Turkenburg, W.: 2004, Policies for renewable energy in the European Union and its member states: an overview, *Energy for Sustainable development*, 2004, **VIII** (1), 5 – 24.

Joosten, L.A.J. (2001). *The Industrial Metabolism of Plastics; Analysis of Material Flows, Energy Consumption and CO2 Emissions in the Lifecycle of Plastics*, PhD thesis, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

JRC: 2003, *Well-towheels analysis of future automotive fuels and powertrains in the European context*, European Comission - Joint Research Centre, Brussels.

Kaenzig, J., Houillon, G., Rocher, M., Bewa, H., Bodineau, L., Orphelin, M., Poitrat, E. and Jolliet, O.: 2004, *Comparison of the environmental impacts of bio-based products.* Paper presented at the 2nd World Conference for Biomass for Energy, Industry and Climate Protection, 10 – 14 May 2004, Rome.

Kaltschmitt, M. and Reinhardt, G.: 1997 *Nachwachsende Energietrager: Grundlagen, Verfahren, ökologische Bilanzierung*. Vieweg, Braunschweig.

Kaltschmitt, M. and Hartmann, H.: 2001, *Energie aus Biomasse – Grundlagen, Techniken und Verfahren*. Springer, Berlin.

Kamerschen, D.R. and Porter, D.V.: 2004, The demand for residential, industrial and total electricity, 1973—1998. *Energy Economics*, **26**, 87—100.

Karus, M., Kaup, M. and Lohmeyer, D.: 2000, *Studie zur Markt- und Preissituation bei Naturfasern (Deutschland und EU)*. Fachagentur Nachwachsende Rohstoffe, Gülzow.

Katofsky, R.E.: 1993, *The production of fluid fuels from biomass*. Centre for Energy and Environmental Studies, Princeton University, Princeton.

Karus, M., Lohmeyer, D., Leson, G.: 1996, Das Hanfproduktlinienprojekt (HPLP) - Erarbeitung vn Produktlinien auf Basis von einheimischem Hanf - aus technischer, ökonomischer und ökologischer Sicht, nova Institut Institut für angewandte Forschung, (IAF)/FH Reutlingen, ifeu-Institut für Energie- und Umweltforschung, Hürth/Koeln.

Keith, D.W.: 2001, Sinks, energy crops and land use: coherent climate policy demands and integrated analysis of biomass - An Editorial Comment, *Climatic Change*, **49**, 1—10.

Kirschbaum, M.U.F.: 2003, To sink or burn? A discussion of the potential contributions of forests to greenhouse gas balances through storing carbon or providing biofuels, *Biomass and Bioenergy*, **24**, 297—310.

Klass, D.L.: 1998, *Biomass for Renewable Energy, Fuels and Chemicals,* Academic Press, San Diego.

Kok, D.: 2001, *Costs, energy use and CO2 emissions of hemp pulp and comparison with wood pulp.* Student report, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Kozlowski, R.: 2001, Personal communication, Director general Institute of Natural Fibres, Poznan.

Kozlowski, R., Baraniecki, P., Mackiewicz-Talarczyk, M.: 2001, Non-Food Applications of Crops in Poland. In: IENICA (Ed), *Newsletter No 14*, Interactive European Network for Industrial Crops and their Applications, York.

Kreutz, T.G., Larson, E.D. and Consonni, S.: 1998, Performance and preliminary economics of black liquor gasification combined cycles for a range of kraft pulp mill sizes, In: Proceedings of the TAPPI International Chemical Recovery Conference, 1—4 June 1998, Tampa.

Kuhn, E.: 1995, *Ko-fermentation: Arbeitspapier 219*. KTBL (Kuratorium für Technik und Bauwesen in der Landwirtschaft), Darmstadt.

Kumar, A., Cameron, J.B. and Flynn, P.C.: 2003. Biomass power cost and optimum plant size in western Canada, *Biomass and Bioenergy*, **24**, 445—464.

Kurdikar, D., Paster, M., Gruys, K. J., Fournet, L., Gerngross, T.U., Slater, S.C. and Coulon, R.: 2001, Greenhouse gas profile of a polymer derived from a genetically modified plant, *Journal of Industrial Ecology*, **4** (3), 107–122.

van der Lans, R.P., Pedersen, L.T., Jensen, A., Glarborg, P. and Dam-Johansen, K.: 2000. Modelling and experiments of straw combustion in a grate furnace, *Biomass and Bioenergy*, **19**, 199—208.

Larsson, S. and Lindegaard, K.: 2003, *Full scale implementation of short rotation willow coppice SRC in Sweden*. Agrobränsle AB, Örebro.

Leaversuch, R.: 2002, Renewable PLA Polymer Gets 'Green Light' for packaging use, *Plastictechnology*, **march 2002.**

Ledin, S.: 1996, Willow wood properties, production and economy, *Biomass and Bioenergy,* **11,** 75—83.

Leemans, R. van Amstel, A., Battjes, C. Kreileman, E. and Toet, S.: 1996, The land cover and carbon cycle consequences of large-scale utilisations of biomass as an energy source, *Global Environmental Change*, **6** (4), 335—357.

Lewandowski, I.: 2001, Energiepflanzenproduktion, in Kaltschmitt, M. and Hartmann, H. (eds.), *Energie aus Biomasse - Grundlagen, Techniken und Verfahren*. Springer, Berlin, Heidelberg, New York., pp. 57—95.

Lewandowski, I., Clifton-Brown, J.C., Scurlock, J.M. and Huisman, W.: 2000, Miscanthus: European experience with a novel energy crop, *Biomass and Bioenergy*, **19**, 209–227.

Lewandowski , I. and Heinz, A.: 2003, Delayed harvest of miscanthus - influences on biomass quantity and quality and environmental impacts of energy production. *European Journal of Agronomy* **19**, 45-63.

Lewandowski, I., Londo, M., Schmidt, U., and Faaij, A.P.C.: 2003, Energiepflanzen und Landnutzungsfunktionen - Ein Überblick zu Kombinationsmöglichkeiten, biophysikalischen Effekten und ökonomischen Ansätzen. In: Kauter, D., Kämpf, A., Claupein, W. and Diepenbrock, W.: *Mitteilungen der Gesellschaft für Pflanzenbauwissenschaften Band 15*, Verlag Günter Heimbach, Stuttgart, 110—113.

Lettens, S., Muys, B., Ceulemans, R., Moons, E., Garcia, J. and Coppin, P.: 2003 Energy budget and greenhouse gas balance evaluation of sustainable coppice systems for electricity production, *Biomass & Bioenergy* **24**, 179—197.

Li, S. and McCarthy, M.: 1999, Further investigations on the hydrolytic degradation of poly (DL-lactide), *Biomaterials*, **20**, 35—44.

Lindeijer, E.: 2000, Review of land use impact methodologies, *Journal of Cleaner Production*, **8**, 273—281.

Linden, D.R., Clapp, C.E., and Dowdy, R.H.: 2000, Long-term corn grain and stover yields as a function of tillage and residue removal in east central Minnesota, *Soil and Tillage Research*, **56**, 167–174.

Londo, M.: 2002, *Energy farming in multiple land use - An opportunity for energy crop introduction in the Netherlands*, PhD thesis, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

LWK: 2002, *Grunddüngung mit Phosphat, Kali, Magnesium*. www.lkwl.de/rlp/landbau/duenger/gr_naehr/gr_dueng.htm (accessed 2002), Landwirschaftskammer Westfalen-Lippe, Münster.

Lynd, L.R., Elander, R.T., and Wyman, C.E.: 1996, Likely features and costs of mature biomass ethanol technology, *Applied Biochemistry and Biotechnology* **57/58**, 741—761.

Marland, G., Fruit, K. and Sedjo, R.: 2001, Accounting for sequestered carbon: the question of permanence *Environmental Science & Policy* **4**, 259—268.

Mann, M. and Spath, P.: 1997, *Life cycle assessment of a biomass gasification combined-cycle power system.* NREL, Golden.

Marland, G., Schlamadinger, B., and Leiby, P.: 1997, Forest/Biomass Based Mitigation Strategies: Does the Timing of Carbon Reductions Matter? *Critical Reviews in Environmental Science and Technology* **27**, 213—226.

Marrison, C.I. and Larson, E.D.: 1996, A preliminary analysis of the biomass energy production potential in Africa in 2025 considering projected land needs for food production, *Biomass and Bioenergy*, **10**, 337—351.

McAloon, A., Taylor, F., Yee, W., Ibsen, K. and Wooley, R.: 2000, *Determining the costs of producing ethanol from corn starch and lignocellulosic feedstocks.* NREL, Golden.

Ney, R.A. and Schnoor, J.L.: 2002, Incremental life cycle analysis: using uncertainty analysis to frame greenhouse gas balances from bioenergy systems for emission trading, *Biomass & Bioenergy*, **22**, 257—269.

Nonhebel, S.: 2002, Energy yields in intensive and extensive biomass production systems, *Biomass & Bioenergy*, **22**, 159-167.

Nossin, P., Joosten, J. and Bruggink, A.: 2002, *Future Feedstock for Commodity Polymers: Ecothene TM Sustainability in the 21st Century*. DSM Research BV, Geleen.

NRC: 1999, *Biobased Industrial Products - Priorities for Research and Commercialization*, National Academy Press, Washington D.C.

OECD and IEA: 2001, *Forestry projects: Permanence, Credit accounting and Lifetime*. OECD Environment Directorate, International Energy Agency, Paris.

Otto, D. and Gallagher, P.: 2001, *The effects of expanding ethanol markets on ethanol production, feed markets and the Iowa Economy*, Iowa State Univerity, Ames.

Overend, R.: 2004, *Life Cycle Assessments – Where are we going?* Paper presented at the 2nd World Conference for Biomass for Energy, Industry and Climate Protection, 10—14 May 2004, Rome.

PAGV: 1994, *Kwantitatieve informatie voor de Akkerbouw en de Groenteteelt in de Vollegrond. Bedrijfssynthese 1995*, Proefstation voor de Akkerbouw en de Groenteteelt in de Vollegrond, Lelystad.

Paster, M., Pellegrino, J. and Carole, T.: 2003, *Industrial Bioproducts: Today and Tomorrow*, Department of Energy, Washington, D.C.

Patel, M.: 1994, *Analyse der Herstellung von Chemifasern (unpublished)*.currently at STS, Utrecht University.

Patel, M: 1999, *Closing Carbon Cycles - Carbon Use for Materials in the Context of Resource Efficiency and Climate Change*, Thesis Dissertation, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Patel, M.: 2000, Biomass for ethylene or for electricity generatio', *Agro-Industry HI-Tech*, **Sept/Oct 2000**, 9—13.

Patel, M., Bastioli, C., Marini, L., and Würdinger, E.: 2003, Environmental assessment of biobased polymers and natural fibres. In: Doi, Y. and Steinbüchel, A. (Eds), *Biopolymers Vol. 10*, Wiley-VCH, Weinheim, 409—452.

Patel, M., Jochem, E., Marscheider-Weidemann, F., Radgen, P. and von Thienen, N.: 1999, *C-Ströme: Abschätzung der Material-, Energie- und CO2-Ströme für Modellsysteme im Zusammenhang mit dem nichtenergetischen Verbrauch, orientiert am Lebensweg – Stand und Szenarienbetrachtung - Part 1*. Fraunhofer ISI, Karslruhe.

PAV: 2000, *Kwantitatieve Informatie Akkerbouw en Vollegrondsgroenteteelt 2000/2001*. Praktijkonderzoek voor de Akkerbouw en de Vollegronds groenteteelt, Lelystad.

Paustian K, Cole, C.V., Sauerbeck, D., and Sampson, N.: 1998, CO_2 mitigation by agriculture: an overview, *Climatic Change* **40**, 135—162.

Peereboom, E.C., Kleijn, R. , Lemkowitz, S. and Lundie, S.: 1998, Influence of inventory data sets on lifecycle assessment results: A case study on PVC. *Journal of Industrial Ecology,* **2** (3), 109–130.

Pervaiz, M. and Sain, M.M.: 2003, Carbon storage potential in natural fiber composites, *Resources, Conservation, and Recycling*, **39** (4), 325–340.

Phylipsen, D, de Beer, J., Kerssemeeckers, M., Timmers, G., Patel, M., Byers, C., Hendriks, C. and Joosen, S.: 2002, *Clean technologies in the materials sector - Current and future environmental performance of material technologies.* Ecofys, Utrecht.

Portney, P.R. and J.P. Weyant (Eds): 1999, *Discounting and intergenerational equity.* Resources for Future, Washington D.C.

Realff, M.J. and Abbas, C.: 2003, Industrial symbiosis - refining the biorefinery, *Journal of Industrial Ecology*, **7** (3/4), 5—9.

REU and FAL: 1996, *Renewable Energy - Potential energy crops for Europe and the Mediterranean region*, REU Technical Series Nr. 46, FAO Regional Office for Europe, Federal Agricultural Research Centre Germany, Braunschweig.

Richards, K.R. and Stokes, C.: 2004, A review of forest carbon sequestration cost studies: A dozen years of research, *Climatic Change*, **63**, 1—48.

Richter, K., Fischer, M., Gahlmann, H., Huser, A.,and Menard, M.: 1995, *Energie- und Stoffbilanzen bei der Herstellung von Wärmedämmstoffen*. EMPA - Abteilung Holz, Dübendorf.

RIVM: 2001. *The Image 2.2 implementation of the SRES scenarios*, CD-ROM Publication 481508018, RIVM, Bilthoven.

Roffael, E., Dix, B., Khoo, K.C., Ong, C.L. and Lee, T.W.: 1992, Mitteldichte Faserplatten (MDF) aus jungem Pappelholz unterschiedlicher Eigenschaften, *Holzforschung* **46**, 163—170.

Rösch, C. and Wintzer, D.: 1997, *Monitoring "Nachwachsende Rohstoffe": Vergasung und Pyrolyse von Biomasse – Zweiter Sachstandsbericht*. Büro für Technikfolgen-Abschätzung beim Deutschen Bundestag, Bonn.

Samson, R., Girourd, P., Zan, C., Marin. B., Mehdi, R., and Hennin, J.: 1999, *The Implications of Growing Short-Rotation Tree Species for Carbon Sequestration in Canada*. R.E.A.P., Quebec.

Scheer-Triebel, M., Heyland, K.-U., and Leon, J.: 2000, Einfluss des Erntetermins auf Morphologie, Ertrag und Qualität verschiedener Leingenotypen, *Pflanzenbauwissenschaften,* **4** (2), 91–102.

Schlamadinger, B., Apps, M., Bohlin, F., Gustavsson, L., Jungmeier, G., Marland, G., Pingoud, K. and Savolainen, I.: 1997, Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems, *Biomass & Bioenergy* **13**, 359—375.

Scott, D.S., Lamarre, P., Tsang, W., Piskorz, J. and Radlein, D.: 1997, Feasibility study for a biomass refinery concept. In: Overend, R.P. and Chornet, E. (Eds), *Making a business from biomass in energy, environment, chemicals, fibers, and materials - Proceeding of the third biomass conference of the Americas*, Pergamon, Oxford, 933—945.

Sedjo, R., Sohngen, B. and Mendelsohn, R.: 2001. *Estimating Carbon Supply Curves for Global Forests and Other Land Uses*, Resources for the Future, Washington DC.

SEO: 1998, *De Prijselasticiteit van de energievraag; Stand van Zaken 1998*. Stichting voor Economisch Onderzoek der Universiteit van Amsterdam, SEO-rapport 483, Amsterdam.

Shapouri, H., Duffield, J.A. and Wang, M.: 2002, *The energy balance of corn ethanol: An update*. USDA, Washington, DC.

Sheehan, J., Aden, A., Riley, C., Paustian, K., Brenner, J. Killian, K., Cushman, J., Walsh, M., and Nelson, R.: 2002. *Is biomass part of a sustainable energy future? Adventures in cyber-farming*. Paper presented at 2002 Fall Annual Meeting, American Institute of Chemical Engineers, 3—8 November, Indianapolis.

Skarzynska, A. and Augustynska-Grzymek, I.: 2001a, *Produkcja, Koszty Economiczne I Dochodowość Wybranych Produktów Rolniczych W Latach 1999 – 2000*. IER, IGZ, Warsaw.

Skarzynska, A. and Augustynska-Grzymek, I.: 2001b, *Koszty Jednostkowe I Dochodowosc Produkcji Rolniczej W Gospodarstwach Indywidualnych W 2000 Roku*. IER, IGZ, Warsaw.

Smeets, E., van Dam, J, Faaij, A. and Lewandowski, I.: Bottom-up methodologies for assessing the technical and economic bioenergy production potentials. In: Brouwer, F. and McCarl, B. (eds): *Rural Lands, Agriculture and Climate beyond 2015: Usage and Management Responses,* Kluwer, Dordrecht (in press).

Södergård, A. and Stolt, M.: 2002, Properties of lactic acid based polymers and their correlation with composition, *Progress in Polymer Science*, **27**, 1123—1163.

Solantausta, Y., Bridgewater, T., and Beckman, D.: 1997. *Electricity production by advanced biomass power systems*. VTT, Espoo.

Stańczyk, K. and Ludwik, M. 2003, *Energy forest development on areas in Central Eastern Europe, where the agricultural production is uneconomical.* Central Mining Institute, Katowice.

Stavins, R.N.: 1999, The costs of carbon sequestration: A revealed-preference approach, *The American Economic Review*, **89**, 994—1009.

Stolp, J., Tjeedsma, B. and Sierra-Alvarez, R.: 1996, *Kansen voor juveniel populierehout: technische en economische analyse.* CPV, Wageningen

Suurs, R.: 2002, *Long distance bioenergy logistics: An assessment of costs and energy consumption for various biomass energy transport chains*. Student report, Dept. of Science, Technology and Society, Utrecht University.

Szczukowski, S., Kisiel, R., Tworkowski, J., Piechocki, J. and Stolarski, M.: 2000a, *Costs and Profitability of short rotation willow coppice cultivation in case of different harvest cycles*. University of Warmia and Mazury, Olsztyn.

Szczukowski, S., Kisiel, R., Piechocki, J., Tworkowski, J., and Stolarski, M.: 2000b, *Energetic efficiency of production of short rotation willow coppice*, University of Warmia and Mazurin, Olsztyn.

Szczukowski, S., Tworkowski, J. and Kwiatkowski, J.: 1998, Możliwości wykorzystania biomasy Salix sp. pozyskiwanej z gruntów ornych jako ekologicznego paliwa oraz surowca do produkcji celulozy i płyt wiórowych. Postępy Nauk Rol., 2, 53—63.

Technical Insights: 1999, Synthetic Lubricants: Potential Applications and Markets in Western Europe, North america, and Asia-Pacific, http://www/the-infoshop.com/study/ti4663_synthetic_lubricants_ potential.html, Technical Insights.

Teeuwissen, S.K.: 1999, Co-firing Biomass in the Europeat Power Station in Ireland - An assessment of pure energy crops versus multi-product plantations and sawmill residues, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Termeer, G.: 2002, Analysis of bioenergy costs and CO2-emission reduction by the use of multi-product crops in Europe; A comparison of three possible multi-product crops in a Western and Eastern European country. Student report, Dept. of Science, Technology and Society, Utrecht University, Utrecht.

Tsuji, H., Daimon, H., and Fujie, K.: 2003, A new strategy for recycling and preparation of poly(l-lactic acid): Hydrolysis in the Melt, Biomacromolecules, 4, 835—840.

Turkenburg, W.C., Beurskens, J., Faaij, A., Fraenkel, P., Fridleifsson, I., Lysen, E., Mills, D., Moreira, J.R., Nilsson, L.J., Schaap, A., and Sinke, W.C.: 2000, Section 7: Renewable energy technologies. In: Goldemberg, J. (ed): World Energy Assessment of the United Nations. UNDP, New York.

UBA: 2003: Basisdaten Umweltmanagement, http://www.umweltbundesamt.de/uba-info-daten/baum, (accessed 2003), Umweltbundesamt, Berlin.
UN: 2002, Industry Commodity Production Statistic Database 1950—2000. UN, New York.
UN/ECE and FAO: 2001, Forest Products Annual Market Review, 1999—2000, FAO, Rome.
UN/ECE and FAO: 2002, Forest Products Annual Market Review, 2000—2001, FAO, Rome.
UNFCC: 1997, Kyoto Protocol to the United Nations Framework Convention on Climate Change. UNFCC, Bonn.
USDA: 2003. Agricultural land values and cash rents, USDA, Washington DC.
Venturi, P. and Venturi, G.: 2003, Analysis of energy comparison for crops in European agricultural systems, Biomass and Bioenergy, 25, 235—255.
Vikman, P.-A., Klang, A., Gustavsson, L.: 2004, Evaluating greenhouse gas balances and mitigation costs of bioenergy sytems - a review of methodologies, Mid Sweden University, Östersund.
Vink, E.T.H., Rábago, K.R., Glassner, D.A. and Gruber, P.R.: 2003, Applications of life cycle assessment to NatureWorks(TM) polylactide (PLA) production, Polymer Degradation and Stability, **80** (3), 403—419.
Vöhringer, F.: 2004, Personal communication. Assistant Professor, Environmental Economics and Natural Resources, Wageningen University, Wageningen.
VVAV: 1994, Handboek Afvallogistiek. Drukkerij Peter Szulc, Utrecht.
Walsh, M.E.: 2000, Method to estimate bioenergy crop feedstock supply curves, Biomass & Bioenergy, **18**, 283—289.
Wiley: 1985, *Encyclopedia of polymer science and engineering,* Wiley-Interscience, New York, Chichester, Brisbane, Toronto, Singapore.
Williams, R.H., Larson, E.D., Katofsky, R.E. and Chen, J.: 1995, Methanol and hydrogen from biomassa for transportation, *Energy for sustainable development,* **5** 18—34.
Wiltsee, G.: 2000, *Lessons learned form existing biomass power plants*. NREL, Golden.
Wooley, R., Ruth, M., Sheehan, J., Ibsen, K., Majdeski, H. and Galvez, A.: 1999, *Lignocellulosic biomass to ethanol - Process design and economics utilizing co-current dilute acid prehydrolysis and enzymatic hyrolysis - Current and futuristic scenarios*. NREL, Golden.
Wolfram, C.D.: 1999, Measuring duopoly power in the British electricity spot market. *The American Economic Review*, **89**(4), 805—826.
Wötzel, K., Wirth, R. and Flake, R.: 1999, Life cycle studies on hemp fibre reinforced components and ABS for automotive parts. *Angewandte Makromolekulare Chemie,* **272**, 121–127.
WSRC: 2004, *Harvester Directory Contents, Coppice Harvesting Decision Support System*, Wood Supply Research Group, University of Aberdeen, Aberdeen.
Wright, L.L. and Cushman, J.H.: 1997, A national Research & Development strategy for biomass crop feedstock. In: Overend, R.P. and Chornet, E. (Eds), *Making a business from biomass in energy, environment, chemicals, fibers, and materials - Proceeding of the third biomass conference of the Americas*, Pergamon, Oxford, 933—945.
Würdinger, E., Roth, U., Wegener, A., Peche, R., Borken, J., Detzel, A., Fehrenbach, H., Giegrich, J., Möhler, S., Patyk, A., Reinhardt, G.A., Vogt, R., Mühlberger, D., and Wante, J.: 2002, *Kunststoffe aus nachwachsenden Rohstoffen: Vergleichende Ökobilanz für Loose-fill-Packmittel aus Stärke bzw. aus Polystyrol*. Bayrisches Institut für angewandte Umweltforschung und-technik, Augsburg.
Wyman, C.E.: 2003, Potentials synergies and challenges in refining cellulosic biomass to fuels, chemicals and power, *Biotechnology* Progress, **19**, 254—262.
Yamamoto, H., Fujino, J. and Yamaji, K.: 2001, Evaluation of bioenergy potential with a multi-regional global-land-use-and-energy model, *Biomass and Bioenergy*, **21**, 185—203.
Zeevalkink, J.A., and van Ree, R.: 2000, Conversietechnologieen voor de productie van elektriciteit en warmte uit biomassa en afval, Novem, Utrecht.

Dankwoord

Een proefschrift schrijf je alleen en ook weer niet. *Ten eerste zijn er natuurlijk mijn promotor en co-promotor…* Wim Turkenburg wil ik bedanken voor zijn goede suggesties, zijn heldere en kritische manier van commentaar geven en zijn betrokkenheid bij de afronding van dit proefschrift. André Faaij bedank ik voor het sturen in de goede richting en het openen van deuren. In het geval dat ik behoefte aan ideeën, denkbeelden of waardering voor mijn werk had, kon ik altijd bij hem terecht *… en daarnaast mijn 'mede-begeleiders'…* Iris Lewandowski was een sprankelende bron van informatie over energiegewassen en de Duitse taal, en heeft me soms weer met beide benen op de landbouwgrond gezet. Martin Patel heeft me op een plezierige manier geïnspireerd om nieuwe inzichten op het gebied van biomaterialen te verwerven. *… en andere co-auteurs…* Gijs Termeer heeft veel gegevens uit Polen weten op te duiken en heeft me met zijn vele vragen gescherpt. Jinke van Dam heeft met haar kennis van biomassapotentialen behulpzaam een belangrijke fundering voor mijn laatste hoofdstuk gelegd*… en Biopush projectpartners…* De samenwerking met Simona Negro, Marko Hekkert, Ruud Smits, Adriana Ignaciuk en Ekko van Ierland heeft me waardevolle inzichten in andere vakgebieden verschaft. In het bijzonder waren de discussies over economie met Frank Vöhringer voor mij leerzaam en plezierig. Richard van den Broek is bedankt voor het schrijven van het onderzoeksvoorstel voor het project. *… and the persons around the world who answered my questions, involved me in interesting discussions and gave me advice…*I am grateful to all of them, in particular to Robert Anex (Iowa State University), Ben Dronkers (Hempflax), Michael Karus (Nova Institute), Ryszard Kosłowski (Institute of Natural Fibres), Gregg Marland (Oak Ridge National Laboratory), Magdalena Rogulska (EC Baltic Renewable Energy Center), Bernhard Schlamadinger (Joanneum Research), Martin Snijder (ATO/DLO) and Virginia Tolbert (Oak Ridge National Laboratory).*… en collega's bij NW&S…* Carlo Hamelinck en Monique Hoogwijk hebben al mijn biomassavragen proberen te beantwoorden en zijn er meestal nog in geslaagd ook. De mensen van het secretariaat bedank ik voor hun goede ondersteuning. Monica Zimmerman, Andrea Ramirez and Manuela Crank were vivid colours in our brown-grey office. Ook heb ik erg genoten van het klimmen en kletsen met Jethro Betcke. Maar wat zou NW&S zijn zonder al die wijsneuzige, onzinnige en roddelige gesprekken tijdens de pauzes? Ad, Bas, Dirk-Jan, Edward, Eric, Erik, Esther, Evert, Günther, Heleen, Jeroen, Jos, Kay, Maarten, Marc, Martin, Martijn, Nico, Nils, Penny, Pita, Tao, Themy, Vlasis, Wilfried en Willem bedankt*… en vrienden.* Dear friends, thanks to all of you the last four years have been fun!

Curriculum vitae

Veronika Dornburg was born on the 21th of December 1971 in Wülfrath, West Germany. From 1991 to 1999 she studied environmental technology at the Technical University of Berlin, specialising in waste management. During this time she worked as teaching assistant and was student representative in the educational committee of her faculty. In 1998 she did a 4-month traineeship in development cooperation at the GTZ (Deutsche Gesellschaft für Technologische Zusammenarbeit), working on solid waste manage ment issues in Aqaba, Jordan. Afterwards, she carried out her graduation research at the Department of Science, Technology and Society (STS), currently part of the Copernicus Institute of Utrecht University. She graduated in August 1999 on the subject of 'System analysis of biomass energy systems in relation to scale'. In December 1999 she became junior researcher at STS, developing a waste management optimisation tool for the Nether lands. Since September 2000 she has been working at STS as PhD-student on the subject of multi-functional biomass systems. During that time she was PhD student representative at the board of the Copernicus Institute.