

Opportunities for biomaterials

Economic, environmental and policy aspects

along their life cycle

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Opportunities for biomaterials

Economic, environmental and policy aspects along
their life cycle

Kansen voor biomaterialen

Economische, milieu- en beleidsaspecten in hun levenscyclus
(met een samenvatting in het Nederlands)

Chancen für Biomaterialien

Ökonomische, ökologische und politische Aspekte entlang ihres
Lebenszyklus
(mit einer Zusammenfassung auf Deutsch)

Proefschrift

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Für meine Eltern

Table of Contents

1	Introduction	1
1.1	The materials sector and resource use	1
1.1.1	Depletion of non-renewable resources	2
1.1.2	Import dependence	3
1.1.3	Climate change	3
1.2	Mitigation in the materials sector	4
1.3	Bio-based materials	5
1.3.1	Terminology	6
1.3.2	Production processes	7
1.3.3	Feedstocks	9
1.3.4	Sustainability of feedstock production	9
1.3.5	Biomass potentials	10
1.4	Aim and outline of this thesis	10
2	Today's and tomorrow's bio-based bulk chemicals from white biotechnology — a techno-economic analysis	13
2.1	Introduction	14
2.2	Methodology	15
2.2.1	Methodological background	15
2.2.2	Functional unit and product selection	16
2.2.3	Process design of WB routes	17
2.2.4	Technology assumptions for WB routes	18
2.2.5	Energy use	22

2.2.6	Process economics methodology	22
2.2.7	WB industrial process data	23
2.2.8	Prices of fermentable sugars	24
2.2.9	Petrochemical process data	25
2.2.10	Prices of utilities and auxiliaries	25
2.3	Case study on 1,3-propanediol	26
2.3.1	Production routes	26
2.3.2	Results for PDO production routes	26
2.3.3	Composition of production costs	27
2.4	Results	29
2.5	Discussion	32
2.5.1	Sensitivity analysis for high crude oil prices	32
2.5.2	Sensitivity to economic parameters	34
2.5.3	Sensitivity to key technical parameters	34
2.5.4	Comparison with earlier publications	34
2.5.5	Comparison with current developments in industry	35
2.6	Conclusions	36
3	Producing bio-based bulk chemicals using white biotechnology saves energy and combats climate change	39
3.1	Introduction	39
3.2	Methodology	40
3.2.1	Functional unit and product selection	41
3.2.2	System boundaries	41
3.2.3	Subsystem description	42
3.2.4	Allocation and system expansion	43
3.2.5	Environmental indicators	44
3.3	Life Cycle Inventory	45
3.3.1	Production of fermentable sugar	45
3.3.2	Assumptions for the fermentation process	46
3.3.3	Industry data for WB products	47
3.3.4	Petrochemical data for benchmarking	47
3.4	Results	47
3.5	Comparison of results	51
3.6	Sensitivity analysis and discussion	52
3.7	Conclusion	53
	Appendix A: Detailed calculation results	54
4	Twisting biomaterials around your little finger: environmental impacts of bio-based wrappings	59

4.1	Introduction	60
4.2	Methodology	61
4.2.1	Functional unit and system boundaries	61
4.2.2	System expansion	62
4.2.3	Environmental impact assessment	62
4.2.4	How to incorporate green electricity?	63
4.3	The laminates and their production	64
4.3.1	Laminates studied	64
4.3.2	Data sources	65
4.4	Results	70
4.4.1	Inner Packs	70
4.4.2	Outer Packs	72
4.5	Sensitivity analysis	75
4.5.1	Technology maturity	75
4.5.2	Wind credits for electricity use in film production and conversion	77
4.5.3	Waste management	77
4.6	Conclusions and Recommendations	78
5	To compost or not to compost: LCA of biodegradable materials' waste treatment	81
5.1	Introduction, Problem Setting and Goal	82
5.2	Background on biomaterials and footprinting methodology	83
5.2.1	Materials	83
5.2.2	Suitability of waste treatment options per material	84
5.2.3	Materials' biodegradation levels and certification of compostability	84
5.2.4	Functional unit and system boundaries	87
5.2.5	Data collection	87
5.2.6	Impact categories	88
5.3	Methodological particularities of biological waste treatment options	88
5.3.1	Long-term carbon storage in the soil	89
5.3.2	Carbon credits from replacing soil conditioners	91
5.3.3	Allocation of nitrous oxide emissions	92
5.3.4	Nitrogen credits	93
5.4	Waste treatment types	94
5.4.1	Industrial composting	95
5.4.2	Home composting	96
5.4.3	Anaerobic digestion	98
5.4.4	Incineration	99
5.5	Results	100
5.5.1	Comparison of waste management results	100

Table of Contents

5.5.2	Sensitivity analysis	100
5.5.3	Uncertainties and discussion	104
5.6	Conclusions	105
6	Policy efforts to increase market penetration of bio-based materials	107
6.1	Introduction	108
6.2	Inventory of existing policy measures	109
6.2.1	Method and scope	109
6.2.2	Normative mechanisms	109
6.2.3	Economic mechanisms	112
6.2.4	Communication mechanisms	115
6.2.5	Conclusions on existing policy measures	115
6.3	Comparison of biomaterial policies with bioenergy policies	117
6.4	Policy barriers	120
6.5	Case study of a potential policy measure: Carbon tax on materials	123
6.5.1	Economic data and calculations	124
6.5.2	Environmental data and calculations	125
6.5.3	Results	125
6.5.4	Sensitivity of results	126
6.5.5	Conclusions regarding the carbon tax case study	128
6.6	Conclusions	128
	Summary	131
	Samenvatting	139
	Zusammenfassung	147
	Abbreviations & Glossary	155
	List of Figures	159
	List of Tables	161
	Bibliography	163
	Acknowledgements	183
	Curriculum Vitae	185

1

Introduction

1.1 The materials sector and resource use

Global resource use, including ores and minerals, fossil energy carriers and biomass, has increased in weight by a factor of eight between 1900 and 2005 (Krausmann et al., 2009) and material consumption per capita doubled. With the world population growing from 6.5 billion in 2005 to approximately 9 billion by 2050 (UN, 2008) and economic activity expected to increase as well, the demand for resources is very likely to continue growing. Bulk material use (including cement, steel, bricks & tiles, paper & board, wood, plastics, glass and aluminium; see Figure 1.1) is in the order of 920 million tonnes per year in the EU-27 (Shen et al., 2009). The plastics and paper & board industries are important contributors at 60 and 100 million tonnes respectively and are therefore the focus of this thesis. Plastics, paper & board and other polymers can be used in the areas of packaging, building & construction, consumer goods, transportation, furniture and electronics; and the production of paper and of plastics is an important part of material use, as shown in Figure 1.1. Plastics production and consumption has increased significantly over the last years. Annual projected growth rates – before the economic crisis – were 3% in Western Europe, 7% in Eastern Europe and 4% per capita globally (projected for 2005–2015, PlasticsEurope, 2008), reaching a total of 245 million tonnes globally (PlasticsEurope, 2009). Both the paper & board and the plastics sectors are also important economically, contributing a significant part of the value added of the European manufacturing industry (Johansson, 2008).

The industrial manufacturing industry (incl. the paper & pulp and plastics industries, among others) currently uses roughly one third of both primary and final energy (IEA, 2007). Of this total, 9% are used for non-energy purposes globally, mostly for petrochemical feedstocks (IEA, 2009d) with less than 1% of this non-energy use stemming from renewable resources (IEA, 2009d). The predominant fossil feedstock is naphtha, an intermediate product from crude oil distillation; other feedstocks for petrochemicals are ethane (a by-product from natural gas processing), methane, liquefied petroleum gas

(propane and butane) and coal (Ren, 2009). Whichever fossil feedstock is used, its consumption is linked to a number of serious national, regional and global problems, such as the depletion of such non-renewable resources, the economic and possibly political dependence on imports of oil and gas or their derivatives, and the emission of greenhouse gases with their detrimental effects on the global climate.

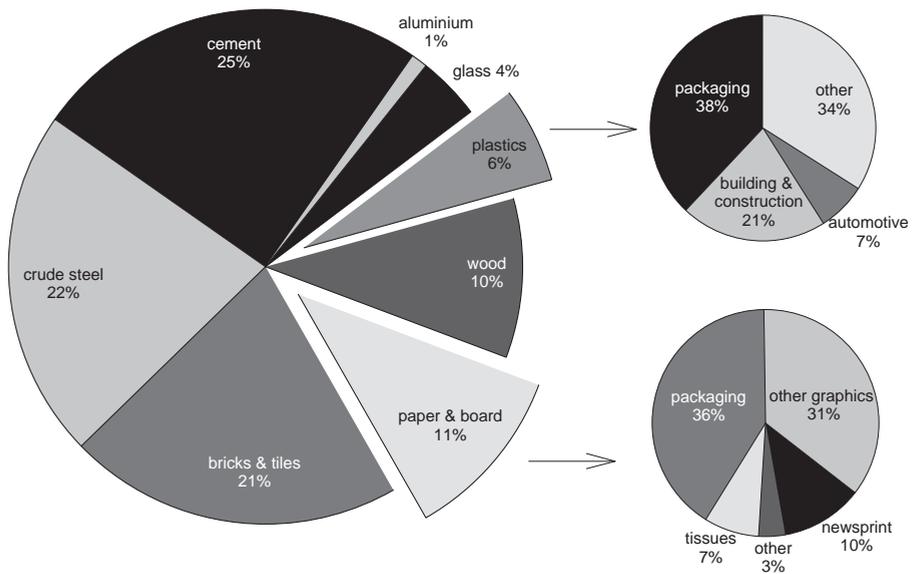


Figure 1.1: Market shares of bulk materials, including detailed uses of plastics and paper & board (derived from Shen et al., 2009; PlasticsEurope, 2009; CEPI, 2009)

1.1.1 Depletion of non-renewable resources

The natural resources of oil, coal and natural gas are finite and non-renewable. As early as 1956, Hubbert proposed that the typical pattern for the depletion of a finite resource is that production starts at zero, then increases exponentially to reach one or more peaks of production to finally decline exponentially to zero (not necessarily symmetrical to the increase). The peak of production for crude oil is also referred to as ‘peak oil’, and the same concept has been applied to natural gas and coal (Hubbert, 1956). Once production has started to decline, rising prices and continued demand will lead to the extraction of more ‘difficult’ reserves whose extraction would otherwise not have been economically attractive, such as tar sands. As a result, more energy will be invested in this extraction, leading to smaller energy returns on investment¹ and thus higher primary energy use per

¹Energy return on (energy) investment is the ratio of energy recovered over energy used in extraction, transportation and refining.

unit of useful, secondary energy delivered. The depletion of oil and natural gas reserves is likely to lead to a higher share of coal in the fossil fuel mix. As coal has a higher carbon to energy ratio than oil and natural gas, this will lead to higher CO₂ emissions per unit of energy. This trend as predicted by Hubble seems to have started already: carbon intensities of energy were decreasing globally since 1860 ('decarbonisation'; Nakicenovic, 1996), but recently, carbon intensities of energy in both developed and developing regions have been nearly constant or slightly increasing (Raupach et al., 2007).

1.1.2 Import dependence

Global reserves of oil and natural gas are unequally distributed around the globe, with generally few and relatively small reserves in Europe. The European Union therefore imports much of its fossil fuel consumption and the share of imports has risen significantly from 45% during the 1990s to 55% in 2008 (Eurostat, 2009). As the fossil fuel reserves in the European Union do not match its energy demand (Eurostat, 2009), this will lead to increased dependence on imports. Currently, the biggest producers are Saudi Arabia, the Russian Federation, the United States, Iran and China (IEA, 2009d) and the biggest exporting countries are the Middle East region, the Russian Federation and Nigeria and Angola (IEA, 2009d). Some of these areas are politically unstable, thereby jeopardising the security of supply of the importing countries which depend heavily on regular, calculable imports.

1.1.3 Climate change

Global surface temperatures have risen by 0.74°C ($\pm 0.18^\circ\text{C}$) between 1906 and 2005 with a much higher increase during the second half of that period (Trenberth et al., 2007). The emission of greenhouse gases, most importantly CO₂, CH₄ and N₂O, results in a higher concentration of these gases in the atmosphere, which leads to an increase of the average global temperatures and to changes in many other aspects of global climate (Forster et al., 2007), such as the geographic distribution, timing, frequency and severity of precipitation, floods, storms and droughts. Although the increase of CO₂ emissions from fossil fuel combustion and industrial processes has not been uniform (1.7% for 1971-1990, 0.8% for 1990-2000 (break-down of the Soviet bloc) and 4.1% for 2000-2004 derived from Bernstein et al. (2007) and Price et al. (2002)), the trend suggests that greenhouse gases emissions will continue to grow. On the basis of past and future emissions of greenhouse gases, the Intergovernmental Panel on Climate Change (IPCC) has projected a global temperature rise of 1.8-4°C (best estimate; full range: 1.1-6.4°C) across all scenarios by the year 2100 (Meehl et al., 2007). An increase of 2°C will already pose significant risks to many vulnerable ecosystems such as coral reefs, coastal areas or polar regions and heighten the threat on endangered species (Schneider et al., 2007). Larger rises in temperature will amplify adverse effects such as more extreme weather events.

The European Union has therefore set a target of limiting global warming to 2°C compared to pre-industrial times (EC, 2008d). This was also accepted in the Copenhagen

Accord (UNFCCC, 2009), which in addition suggests re-evaluating measures in the future, including the option of considering 1.5°C as a long-term goal. If this target is to be reached, then a reduction of global greenhouse gas emissions of at least 50% is needed by 2050 relative to current emissions and the peak of emissions must be reached by 2015 (Fisher et al., 2007). This obviously requires a major effort in terms of mitigation of emissions by all sectors. The combustion of fossil fuels for electricity, heat and transport is the largest contributor to total greenhouse gas emissions, and has received most attention so far in its transition from using traditional, non-renewable energy sources such as oil, coal and natural gas towards using more and more renewable energy sources such as wind, solar energy and biomass. The manufacturing industry also contributes to 40% of the total global direct CO₂ emissions (IEA, 2009c). The IEA estimated that the chemicals and the pulp & paper sectors could each reduce their CO₂ emissions by 20% of the current total by adopting best available technologies alone, leading to savings of almost 400 million t CO₂ annually (IEA, 2009c). As a consequence, if the goal is to reduce greenhouse gas emissions and to mitigate climate change, these two sectors and the manufacturing industry in general can and must play an important role.

1.2 Mitigation in the materials sector

There are several options for the materials sector to contribute to climate change mitigation, the reduction of the use of non-renewable resources and import dependence:

- by increasing energy efficiency during materials production, i.e. using less energy to produce the same amount and quality of material,
- by replacing the source of energy during materials production by a renewable one, e.g. using hydropower, wind or solar energy,
- by increasing the material efficiency during production, e.g. reducing material losses during production or providing the same service using less material such as thinner plastic bags,
- by replacing the source of materials by a renewable one, i.e. using renewable resources such as biomass,
- by recycling and/or re-using the material.

For four of these options, research efforts were well under way when I started the work presented in this thesis, but very little was known about the effects of replacing the material feedstock. Biomass feedstocks for use as a source of materials are renewable, by definition, and therefore avoid the depletion of non-renewable resources, provided that they are cultivated in a sustainable manner. Secondly, they allow to broaden the resource base, i.e. many countries can produce renewable resources for export, thereby reducing

import dependence and increasing security of supply. And last, but not least, they can reduce the emission of greenhouse gases and thus contribute to mitigate climate change.

With respect to the use of renewable resources, the energy sector has received most attention so far, predominantly regarding electricity production and transportation fuels. This is demonstrated by the numerous policy analyses which have been published (for example Kwant, 2004; Ericsson et al., 2004; van Rooijen and van Wees, 2006; Nilsson et al., 2006; Berndes and Hansson, 2007) and by the numerous policies on the regional and national levels which have been addressing the transition of this sector from using traditional, non-renewable energy sources such as oil, coal and natural gas, towards using more and more renewable energy sources such as wind, solar energy and biomass. Examples of such policies are the European Bioenergy and Biofuels Directive (2009), the German Renewable Energy Law (2004, 2008), the US Energy Policy Act (2005) and Brazil's bio-ethanol programme (1975). These policies have boosted the production and development of bioenergy and biofuels (see Thornley and Cooper, 2008; Rosillo-Calle and Cortez, 1998), leading to increases in production and consumption of biomass for energy purposes.

However, biomass can also be used as feedstock for the production of materials such as chemicals, plastics, fibres, lubricants or composite materials. Despite the publication of several studies showing that biomaterials may offer higher greenhouse gas savings per kilogramme or per hectare than bioenergy and biofuels (e.g. Dornburg et al., 2003; Kim and Dale, 2005c; Anex and Ogletree, 2006; Vink et al., 2007), this application has so far received comparatively little attention from policy-makers who have gone only so far as to identify bio-based material use as a very promising option in terms of environmental, economic and social benefits, e.g. by the European Climate Change Programme (European Commission, 2001), the EU's Lead Market Initiative (EC, 2003d) and the IPCC (Bernstein et al., 2007). With currently less than 1% of chemical feedstocks based on renewable resources, significant progress has to be made in starting up a bio-based materials economy. Scientific and technological breakthroughs that have been made in the last few years in the area of biotechnology and thermochemical processes using bio-based feedstocks, and which are expected for the coming years, can be the starting point in the production of bio-based materials.

1.3 Bio-based materials

In the following, we will discuss the research context of the bio-based material production chain, starting with the production of biomass feedstocks, followed by the sustainability and maximum potentials of such production and finally the conversion technologies used to produce materials from such biomass. Before entering this discussion, however, it is necessary to establish some terminology.

1.3.1 Terminology

Materials are called ‘bio-based’ when they are produced from material of biological origin, such as wood or agricultural crops, agricultural or coppice wastes, manure or microbial biomass. Paper & board are the most important bio-based materials in terms of annual mass of production (Shen et al., 2009). If materials are produced partly from bio-based and partly from conventional petrochemical sources then these are referred to as ‘partially bio-based’. It is important to distinguish bio-based materials from ‘bio-degradable’ ones, which can be decomposed by micro-organisms (or their enzymes) into smaller products such as organic compounds, carbon dioxide and water (OECD, 2002; EC, 2008a)². These biodegradable materials can be made from renewable or non-renewable resources.

Table 1.1: Classification of materials according to their biodegradability and bio-based content, not exhaustive

	biodegradable	non-degradable
bio-based	starch, cellulose, pulp & paper, polylactic acid (PLA), polyhydroxyalkanoates (PHA)	bio-based polyethylene (bio-PE), bio-based polypropylene (bio-PP)
partially bio-based	starch/polycaprolactone, polybutylene succinate-co-lactic acid (PBSL); any of the below polymers involving a bio-based monomer (e.g. PBSA, PBAT, PBST)	polytrimethylene terephthalate (PTT); any of the below polymers involving a bio-based monomer (e.g. PET, PBT)
petrochemical	polycaprolactone (PCL), polybutylene succinate (PBS), polybutylene succinate adipate (PBSA), polybutyrate-adipate-terephthalate (PBAT), polytetramethylene adipate/terephthalate (PTMAT), polybutylene-succinate-co-butylene-terephthalate (PBST), polyesteramides	polyethylene (PE), polypropylene (PP), polyethylene terephthalate (PET), polystyrene (PS), polybutylene-terephthalate (PBT), polyurethanes (PUR)

Table 1.1 gives some examples of materials, classified according to their biodegradability and bio-based content. Wood products, fibre board and similar, more conventional, bio-based products are not considered in this thesis and are thus not shown in Table 1.1. Throughout most of this thesis, the focus is on bio-based materials, both biodegradable and non-biodegradable, i.e. the materials in the first row of the table. These bio-based materials are compared to conventional petrochemical materials which are presented at the bottom right of the table, in chapters 2-4. In Chapter 5, the focus is on biodegradable materials, i.e. the materials in the first column of Table 1.1.

²There are also oxo-(bio)degradable materials: these are bio-based or petrochemical materials which are not biodegradable but break down through the reaction of sunlight with additives that have been introduced into the plastic formulation during production. As their break-down in the environment is incomplete and not a biodegradation process (European Bioplastics, 2009) and they cannot fulfill the requirements for biodegradation certification (e.g. EN 13432), oxo-degradables are not taken into account in this thesis.

1.3.2 Production processes

There are several technical options for converting biomass into useful materials (derived from Turkenburg et al., 2000; McKendry, 2002; Dornburg, 2004; de Jong, 2008; Zhao et al., 2009):

1. gasification, resulting in syngas or methanol, and subsequent conversion into materials,
2. pyrolysis, resulting in bio-oil, and subsequent conversion into materials,
3. catalytic conversion of biomass into materials, e.g. furanics,
4. pulping, resulting in paper and possibly lignin derivatives,
5. other physico-chemical synthesis of materials,
6. fermentation using micro-organisms, resulting in chemicals and intermediates,
7. enzymatic conversion, resulting in chemicals and intermediates,
8. extraction of feedstocks from traditional crops, resulting in e.g. vegetable oils for lubricants, and
9. extraction of materials that were produced *in planta*, e.g. PHAs, possibly enhanced by the use of green biotechnology³.

The starting point of this thesis and the focus of chapters 2 and 3 is on the production of bio-based bulk chemicals, which are often used as intermediates and subsequently converted into polymers. Chapters 4 and 5 then continue with polymers and plastics derived from such chemicals. Table 1.2 illustrates a number of polymers which are derived from bulk chemical intermediates analysed in this thesis.

Table 1.2: Examples of polymer production from bulk chemicals

chemical		polymer
acrylic acid	→	polyacrylamide
adipic acid	→	Nylon 6,6
caprolactam	→	Nylon 6
ethanol → ethylene	→	PE, PS, PET, etc.
lactic acid	→	PLA
1,3-PDO	→	PTT
succinic acid	→	PBS, PBSA

This thesis focusses on biochemical conversion processes which make use of micro-organisms or enzymes to convert a feedstock into a target material. As they are carried

³Green biotechnology is the application of biotechnology to agricultural processes. It can for example be used to increase PHA yields in potatoes.

out at lower temperatures and pressures than petrochemical conversion processes they offer potential savings in energy use and carbon emissions. Fermentation, which can be aerobic or anaerobic, and enzymatic conversion are the two most important biochemical conversion routes. Anaerobic digestion and composting also belong to this category, but these do not result in directly useful materials⁴. Of these, this thesis focusses on fermentation because this is the field to which white biotechnology (WB) can contribute most significantly. WB is an emerging technology which has been expected to progress and evolve quickly and to thereby contribute to a step-change in the fermentative production of bio-based materials (see for example *The Economist*, 2004). As with any emerging technology, the question was to what extent WB could improve the opportunities for bio-based materials and this thesis contributes to answering that question regarding the environmental benefits and economic opportunities.

‘White’ or ‘industrial’ biotechnology (resp. in Europe and the US) is the application of fermentation and enzymatic processes for the production of industrial products and fuels, potentially making use of breeding and genetic modifications of the micro-organisms used in fermentation. White biotechnology routes still offer room for significant improvement: one of the main research challenges of these routes to bio-based materials is to improve the efficiency of fermentation by increasing the productivities, the maximum concentration during the process and the yields of product per unit of input. This can be achieved by conventional breeding or by genetically modifying the organisms used for fermentation in such a way that they tolerate higher maximum concentrations, produce less (or no) by-products and increase their productivity; both methods of improving micro-organisms for use in the industrial production of materials is referred to as WB.

In white biotechnology, contrary to green biotechnology, if genetically modified organisms are used, these are contained in a closed system (an industrial production process). This means that for white biotechnology, the risks of unforeseen local, regional and possibly global effects as a result of proliferation of transgenic organisms are lower (van Overbeek, 2006) and public perception is more positive and less controversial than for green biotechnology (Patel et al., 2006, Terragni and Recchia in). The European Commission identified white (and green) biotechnology as one of the ‘Key Enabling Technologies’ that should be supported in order to enable a “shift to a low-carbon, knowledge-based economy, which is a precondition for ensuring welfare, prosperity and security of its citizens” and is thus indispensable (EC, 2009d). This thesis therefore focusses on white biotechnology to produce materials derived from biomass, and it also considers materials produced directly from biomass (such as starch) and from pulp (see Figure 1.1), which are well-established production processes. White biotechnology can also be used to produce biofuels such as ethanol. However, this thesis focusses on the material use of biomass, and ethanol for fuel use is therefore excluded.

⁴Other potential products from biochemical processes which are not considered in this thesis are solvents, lubricants and surfactants.

1.3.3 Feedstocks

Bio-based materials can be produced from several types of feedstocks: sugars, starches, oils and lignocellulosics. Sugars can be produced from sugar beet or cane, starches from wheat, maize or potatoes and oils from soybeans, sunflower seeds or rapeseed (canola). Agricultural cultivation of crops which supply sugars, starches and oils has been carried out for millennia in the context of food and feed production, and high crop yields are being achieved in developed countries. Much lower yields are generally achieved in developing countries and economies in transition, partly due to a lack of fertilisers (Larson and Frisvold, 1996).

Currently used feedstocks are readily fermentable sugars such as dextrose and starches, which can also be used for the production of food and feed ‘first generation’). Except when used for pulp & paper production, lignocellulosic biomass is a new feedstock. Examples of such feedstocks include perennial grasses (e.g. miscanthus), crop residues (e.g. stover) and short rotation woody crops (e.g. poplars). Lignocellulosic crops are generally not used for food, and thus only compete with these applications via the land they are cultivated on. The retrieval and conversion of lignocellulosics to target materials is the focus of on-going R&D efforts and is therefore considered future (or ‘second generation’) technology. The research challenge for the use of lignocellulosic feedstocks in biochemical production processes lies in improving feedstock pre-treatment as well as in manipulating the micro-organisms involved in the fermentation process in such a way that both types of sugars (C_5 and C_6) contained in lignocellulosics will be efficiently converted into target materials (Gavrilescu and Chisti, 2005). Oil feedstocks can also be used for the production of bio-based materials, most notably resins for coating, stabilisers and lubricants. However, the focus of this thesis is on products with larger market volumes (pulp & its derivatives, bulk chemicals, plastics); therefore oils are not considered in the following.

1.3.4 Sustainability of feedstock production

There are a number of aspects and potential research areas regarding the sustainable production of feedstocks for use in the production of bio-based materials, but as they are not the core subject of this thesis they will only be discussed briefly here: the production of feedstocks for bio-based materials is not necessarily sustainable. The sustainability of feedstock production, or the lack of it respectively, has been discussed from the perspective of bioenergy and biofuels production, most notably for palm oil imports (Koh and Wilcove, 2008; Tan et al., 2007; Wicke et al., 2008) and for the competition with food and feed production (e.g. Banse et al., 2008; Lewandowski and Faaij, 2006), but applies equally to materials. This discussion has resulted in a number of initiatives to set up certification criteria and schemes for biomass feedstocks (see e.g. Cramer et al., 2007; Elghali et al., 2007; Vis et al., 2008; NTA, 2009).

A detailed overview of recent developments in the area of biomass certification is given by Dam et al. (2008) and Palmujoki (2009): the most advanced and most important

national certification initiatives are those of the Dutch and UK governments, stipulating principles and criteria that have to be fulfilled not only in terms of greenhouse gas emission reductions but also including social and economic aspects. On the supra-national level, the European Commission included the requirement that biomass used for bio-fuels fulfill sustainability criteria in the Renewable Energy Directive (EC, 2009e) and is currently devising a sustainability scheme for other ways of using biomass for energy production.

1.3.5 Biomass potentials

The maximum *technical* potential⁵ of bio-based polymers to substitute their petrochemical counterparts has been calculated as amounting to 90% by Shen et al. (2009), indicating that the material properties are such as to enable a large-scale substitution of feedstocks for the production of bulk chemicals and plastics. The ensuing question therefore is whether enough biomass is available to achieve such a degree of substitution. In this context, numerous studies have been carried out to calculate the technical and economic potential of worldwide biomass feedstock production for non-food and non-feed uses. Although these studies have focussed on the use of such biomass for bioenergy and biofuel production, they are also valid for bio-based materials production because the very same crops can be used. The early studies concluded that the technical potential for biomass feedstock production is very large but that in order to realise this potential, advanced agricultural systems would have to be implemented and optimised such that food production can be satisfied with relatively lower land use than today, calling for an intensification of agricultural production e.g. in Africa and Eastern Europe (see e.g. Smeets et al., 2007; Hoogwijk, 2004; Berndes et al., 2003). The application of sustainability criteria for biomass production leads to a reduction of biomass potentials and more recent studies (e.g. Smeets et al., 2009a; Fischer et al., 2010; de Wit and Faaij, 2010) have provided a more cautious picture than earlier publications. This thesis will relate calculated potentials to the land requirements of bio-based materials.

1.4 Aim and outline of this thesis

So far, only very little quantitative information is available on the implications resulting from the potential use of white biotechnology for bio-based material production. This thesis therefore aims at closing a gap of knowledge with respect to the economic and

⁵The technical substitution potential is solely based on material properties and therefore amounts to 100% when producing chemically identical materials from bio-based resources, for example bio-based polyethylene can substitute 100% of petrochemical polyethylene. However, the technical substitution potential is less than 100% in many cases where a petrochemical is substituted by a functionally equivalent (but not identical) product, for example bio-based PLA replacing petrochemical PET, due to the differences between the respective material properties.

environmental benefits of replacing petrochemicals by bio-based materials which are produced using white biotechnology, as well as by bio-based materials which are produced using more established production processes, such as starch products and pulp & paper. We aim to do this by investigating the environmental benefits of and economic opportunities for bio-based materials throughout their life cycle and how these are affected by policy measures. The life cycle is covered by analyses of the *production* of bulk chemicals, the *application* of plastics produced from these chemicals and other polymers in food packaging and the *waste treatment* of such bio-based and/or biodegradable polymers. Along the life cycle, the environmental benefits as well as the economic opportunities are considered. Finally, we investigate the effects of existing and potential *policy measures* on the opportunities for bio-based materials.

A discussion of the research context as well as important terminology was given here in *Chapter 1*. *Chapters 2 and 3* answer the first two research questions to quantify the *extent to which bio-based bulk chemicals and intermediates offer economic opportunities and environmental benefits compared to current petrochemicals, today and in the future*. Both chapters evaluate 15 bio-based bulk chemicals and intermediates using a uniform methodology which allows a systematic and consistent modelling and evaluation of production processes for these products which can be produced from renewable resources. Renewable resources considered here are sugars and starches for current technology and additionally also lignocellulosics for future technology. The effects of white biotechnology regarding the improvement of conversion efficiencies are taken into account.

Chapter 2 focusses on the techno-economic prospects and opportunities of present and future production routes of bio-based bulk chemicals. This chapter investigates the influence of technological progress on the economic viability of bio-based products and analyses the effect of crude oil and fermentable sugar prices. *Chapter 3* analyses how the production of bio-based bulk chemicals and intermediates can contribute to solving two important environmental problems: depletion of fossil resources and climate change. We quantify the potential savings of CO₂ emissions and fossil energy use as well as the land-use that would be required when replacing current petrochemical bulk chemicals and intermediates by bio-based products. For this analysis we use the same product groups, assumptions and system boundaries as for Chapter 2.

As packaging is the biggest end-use segment of plastics representing almost 40% of all polymers (PlasticsEurope, 2009) and food packaging is the largest part of the plastic packaging market at 65% (Brocard and Durand, 2004), *Chapter 4* deals with food packaging as a specific application of bio-based materials in which their functionality, such as barrier properties for water and oxygen and mechanical properties, plays an important role. In this case study, we investigate whether *food packaging that partly or wholly consists of bio-based materials offers environmental advantages compared with the current material*, while considering two applications with differing requirements in terms of barrier properties. We analyse to what extent the potential extra material requirements due to material properties of bio-based materials as well as the lack of experience in processing them outweigh the environmental advantages of using them. This chapter presents results for a range of life cycle assessment impact categories, including not only global warming

potential, eutrophication and acidification but also ‘non-traditional’ ones such as land use and water use.

Waste treatment is an important factor in the life cycle of materials because during this phase the carbon which was sequestered during feedstock production and stored in the bio-based material can be partly or wholly released again in the form of CO₂ and/or CH₄, and thus contributes to the material’s overall global warming potential. The amount and type of such emissions per material vary, depending on the waste treatment type that it undergoes. *Chapter 5* therefore investigates what the *best waste treatment option for biodegradable materials* is. Biodegradable materials can not only be incinerated, but also composted or digested, thus offering more options for waste treatment than non-biodegradable materials. This chapter compiles data-sets for industrial and home composting, anaerobic digestion and incineration and shows the resulting inventory data, focussing on carbon and energy footprint calculations. Chapter 5 also highlights a number of methodological choices and challenges specific for biological waste treatment options (i.e. composting and digestion), such as long-term carbon storage in the soil and the allocation of nitrous oxide emissions, and suggests solutions for these issues.

As shown in Chapters 2–4, bio-based materials compete with conventional, petrochemical materials. Competition can occur between identical materials or those fulfilling the same function (i.e. are functionally equivalent materials). The technology for the production of petrochemicals has been improved and optimised for decades and although there is still room for some improvement (see Ren et al., 2006), the industry is considered mature. Bio-based materials, on the other hand, are still at the beginning of their development and there is the necessity of (and still much room for) optimisation. The bio-based materials market is considered an immature market, which can be supported and strengthened through policy measures. *Chapter 6* therefore analyses *how market penetration of bio-based materials is influenced by current and potential policy measures*. To this end we firstly compile an inventory of current world-wide policies which are relevant for bio-based materials. We then discuss the conflicts and disadvantages for biomaterials that arise from other current policy measures, e.g. on biofuels and bioenergy. Finally, we present a case study, modelling the potential effect of a carbon tax on bio-based materials and the reference petrochemicals.

In the final chapter of this thesis we draw overall conclusions from the individual chapters and reflect on the results of the overall thesis in relation to the main research aim (Chapter 7).

2

Today's and tomorrow's bio-based bulk chemicals from white biotechnology — a techno-economic analysis*

Abstract

Little information is yet available on the economic viability of the production of bio-based bulk chemicals and intermediates from white biotechnology (WB). This chapter details a methodology to systematically evaluate the techno-economic prospects of present and future production routes of bio-based bulk chemicals produced with WB. Current and future technology routes are evaluated for 15 products assuming prices of fermentable sugar between 70 €/t and 400 €/t and crude oil prices of US\$ 25/barrel and US\$ 50/barrel. The results are compared to current technology routes of petrochemical equivalents. For current state-of-the-art WB processes and a crude oil price of US\$ 25/barrel, WB-based ethanol, 1,3-propanediol, polytrimethylene terephthalate and succinic acid are economically viable. Only three WB products are economically not viable for future technology: acetic acid, ethylene and PLA. Future technology ethylene and PLA become economically viable for a higher crude oil price (US\$ 50/barrel). Production costs plus profits of WB products decrease by 20%–50% when changing from current to future technology for a crude oil price of US\$ 25/bbl and across all sugar prices. Technological progress in WB can thus contribute significantly to improved economic viability of WB products. A large-scale introduction of white-biotechnology-based production of economically viable bulk chemicals would therefore be desirable if the environmental impacts are smaller than those of current petrochemical production routes.

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2.1 Introduction

During the last few years, considerable progress has been made in biotechnology research and further major scientific and technological breakthroughs are expected for the future. The first large-scale industrial applications of modern biotechnology have been the areas of food and animal feed production (agricultural/green biotechnology) and of pharmaceuticals (medical/red biotechnology) (EuropaBio, 2005). In contrast, the production of organic compounds by means of fermentation or enzymatic conversion (so-called white biotechnology¹) on a large-scale is still in its infancy. Enzymatic conversion is excluded from this chapter because the number of bulk chemicals that can be produced by fermentation is much larger (Patel et al., 2006). In principle, white biotechnology (WB) can be applied to convert bio-based or petrochemical feedstocks such as methanol. Bio-based feedstocks are the default for WB and petrochemical feedstocks for WB are not covered in this chapter.

Most industrial activity has so far been centred on the production of bio-ethanol for fuel use, for example by Archer Daniel Midlands in the US and Iogen in Canada (Rogers, 2002) as well as by COPERSUCAR in Brazil (Zanin et al., 2000). NatureWorks is the first company producing a bio-based polyester, polylactic acid (PLA), in a large industrial plant (Vink et al., 2003) in Nebraska and DuPont is currently building a plant for the production of 1,3-propanediol (PDO) from maize-based glucose in Tennessee, expected to be operational in 2006. In Europe, practically all large chemical companies are exploring the possibilities for the production of chemicals by means of WB and they are partly developing industrial processes for selected, mostly higher-value products (EuropaBio, 2005). Moreover, the EU bio-fuels directive (EC, 2003a) is expected to boost the production volumes of bio-ethanol in the WB sector.

Very little data is publicly available on the techno-economic potentials of WB-based chemicals because of the sensitivity of this information among competitors and because WB is still an emerging technology. Some publications have analysed the potentials and barriers to the production of WB-based chemicals in general (Bachmann et al., 2000; Werpy and Petersen, 2004; EuropaBio, 2003) and many publications and reports deal with technical aspects of specific process steps such as fermentation and downstream processing. Some detailed techno-economic analyses have been produced by SRI Consulting (e.g. SRI, 2001) but these are multi-client studies and are therefore not publicly available. Of the few publications on the techno-economics of WB-based products almost all have focused on a single product: most predominantly on ethanol for fuel use (e.g. Hamelinck et al., 2005; O'Brien et al., 2000; Wooley et al., 1999), but also on ABE (Qureshi and Blascheck, 2001) and succinic acid (Zeikus et al., 1999; Landucci et al., 1994). Methodologies and background assumptions differ between these studies and the results are not easily compared. Additionally, most available analyses study current technologies and do not account for possible future technological progress. Against this background it is the

¹In the US, white biotechnology is referred to as industrial biotechnology.

aim of this chapter to assess the economic viability of current and future technologies for producing bio-based bulk chemicals using white biotechnology.

A general problem of process analyses — especially when concerning emerging technologies — is the confidentiality of process data. To overcome this problem this chapter presents and applies a generic approach that circumvents confidentiality issues by allowing to estimate the economic viability of WB production processes based on generic key data, a principle already proposed by Landucci et al. (1994); Lynd and Wang (2004). The methodology allows a systematic evaluation of present and future processing routes of WB-based bulk chemicals. Additionally, economic analyses were performed using confidential data provided by companies and institutes and by combining the basic information from the production cost estimates by SRI (e.g. SRI, 2001) with uniform assumptions as applied in this study.

After having discussed the generic approach in section 2.2, a detailed case study is presented on 1,3-propanediol in section 2.3. We proceed to calculate and discuss the current and future economics of 15 B chemicals and compare them to datasets from industry and petrochemical equivalents (section 2.4). These product-by-product analyses were carried out for variable sugar prices and a crude oil price of US\$ 25/barrel, with a sensitivity analysis for US\$ 50/barrel (section 2.5).

This chapter focuses on the economics of WB-based bulk chemicals, drawing on detailed data as presented in the EU project “BREW” (Patel et al., 2006). Based on this project and as an extension to this chapter, Dornburg et al. (2008) perform an evaluation of the technical, economic and market potentials of these WB products including scenarios with changing crude oil prices until 2050 (Dornburg et al., 2008). As a complement to this chapter, we moreover apply the generic approach to prepare a systematic comparison of environmental impacts across products and for different sources of fermentable sugar (see Chapter 3).

2.2 Methodology

2.2.1 Methodological background

A generic approach is a method allowing standardised comparisons between different processes based on a small number of components. Many calculations can then be carried out based on a small number of input data. Due to the very limited availability of process data for WB processes a specific generic approach has been developed. This generic approach allows an *ex ante* estimation of the economic viability of biotechnological processes for which pilot plant or lab scale data do not yet exist or for which process data are not publicly available. We applied this method to processes representing the current state-of-the-art as well as future technology.

The results from this generic approach were compared to results for WB products calculated using industry data and they were also compared to petrochemical equivalents.

The start of the generic approach was the preparation of a process flow diagram of the bioprocess, which converts fermentable sugar to the target WB chemical. These process flow diagrams contained standard modules (e.g. fermentation, ultrafiltration, evaporation). For each process flow diagram representing one production route, the mass balance containing the quantities of all inputs and outputs at the level of unit processes was determined (see sections 2.2.3 and 2.2.4). On this basis, the costs related to all inputs and the investment costs were estimated and the overall production costs calculated (see section 2.2.6). As an alternative, the information from the mass balance can be used to perform an environmental assessment, which is the topic of a separate chapter (Chapter 3). In order to ensure the comparability of the results, a common database for process inputs was used for all calculations. This database consists of market prices for chemicals, auxiliaries and utilities as well as prices for fermentable sugar (see sections 2.2.8 and 2.2.10).

2.2.2 Functional unit and product selection

The purpose of this chapter is to estimate overall production costs for organic bulk chemicals at the factory gate. Therefore, the functional unit was one tonne of organic chemical at the factory gate. Factories were assumed to be dedicated large-scale installations because of the advantage of economies of scale of large processes, which outweigh the biomass collection costs (Wyman, 2003). We make the simplifying assumption that there is only one main product: the WB chemical. This simplification allows a consistent comparison between different chemicals and an overview of the current and future techno-economic potential for WB chemicals. The selection of products in this chapter was based on three criteria:

1. the fermentation process was considered feasible either in literature (see Table 2.1) or by a panel of experts²,
2. basic data on the stoichiometry of the fermentation process was available and
3. the chemicals or intermediates studied had the potential to be used in bulk quantities, i.e. a production of at least 200 kilotonnes³ (kt) per year in Western Europe was envisaged for the medium or long term.

As shown in Table 2.1, current production capacities of the petrochemical equivalents of the selected WB chemicals exceed this value by far. The application of these criteria has resulted in the following selection of products: 1,3-propanediol (PDO), acetic acid, acrylic acid, adipic acid, butanol (from the ABE process), ethanol, citric acid, lysine, lactic acid, polyhydroxyalkanoates (PHA) and succinic acid. Additionally, several products were considered that are formed by chemical conversion of a product included in the previous list: polycondensation of lactic acid results in polylactic acid (PLA), that

²The panel consisted of representatives from A&F, BP Chemicals, Degussa, DSM Research, DuPont, NatureWorks, Novozymes, Roquette Frères, Shell International Chemicals and Uniqema. These are the industry partners of the BREW project (Patel et al., 2006).

³All tonnes referred to in this thesis are metric tonnes (ca. 1.102 short tonnes).

of PDO and purified terephthalic acid in polytrimethylene terephthalate (PTT); ethanol is dehydrated to yield ethylene; ethanol and lactic acid are converted to ethyl lactate; and lysine is transformed into caprolactam via chemical cyclisation. Table 2.1 shows the production volumes of the petrochemical equivalents of the WB products. Lysine and lactic acid were excluded because they have no petrochemical equivalents; both are used as intermediates, e.g. for caprolactam and PLA, but may also have stand-alone applications such as poly-(ϵ)-lysine (see Shih et al., 2006). Total petrochemical production volume in Western Europe amounted to 46 million tonnes in 1999 (derived from Weissermel and Arpe, 2003; EC, 2001), showing that a significant portion of the market was included in this study.

Table 2.1: Production capacities of the petrochemical equivalents of the 15 WB products in Western Europe, installed capacity taken from Weissermel and Arpe (2003) except PET (from Glenz (2004))

Product	Installed capacity (kt/yr)	Year	Equivalent for WB-based
Acetic acid	820	1999	Acetic acid
Acrylic acid	1,400	2000	Acrylic acid
Adipic acid	1,000	1999	Adipic acid
Butanol	930	1999	Butanol
Caprolactam	1,100	1999	Caprolactam
Ethanol ^a	580	1998	Ethanol
Ethyl acetate	310	1999	Ethyl lactate
Ethylene	22,200	2000	Ethylene
Maleic anhydride	380	1999	Succinic acid
PDO/PTT	no data		PDO/PTT
Polyethylene	12,900	2000	PHA
Polyethylene terephthalate	2,170	2004	PLA

^a The installed capacity of bio-based production of ethanol is 560 kt/yr (Weissermel and Arpe, 2003). This is not included in the table because bio-based ethanol is mostly used as fuel, whereas the ethanol used in the chemical industry predominantly stems from petrochemical production processes.

2.2.3 Process design of WB routes

Separate process flow diagrams were made for current and future technologies, both with respect to the fermentation processes and downstream processing for product separation and purification. All process flow diagrams (prepared at the level of unit processes) consisted of the following sections (see Figure 2.1): seed & inoculum trains (provision of micro-organism), fermentation (conversion of sugar to the target product and by-products), filtration (removal of solid by-products) and downstream processing (several steps, to purify the target product). The material inputs to the system were fermentable sugar, water, nutrients, auxiliary substances and utilities such as electricity and steam. The outputs were the target product as well as solid waste and waste-water.

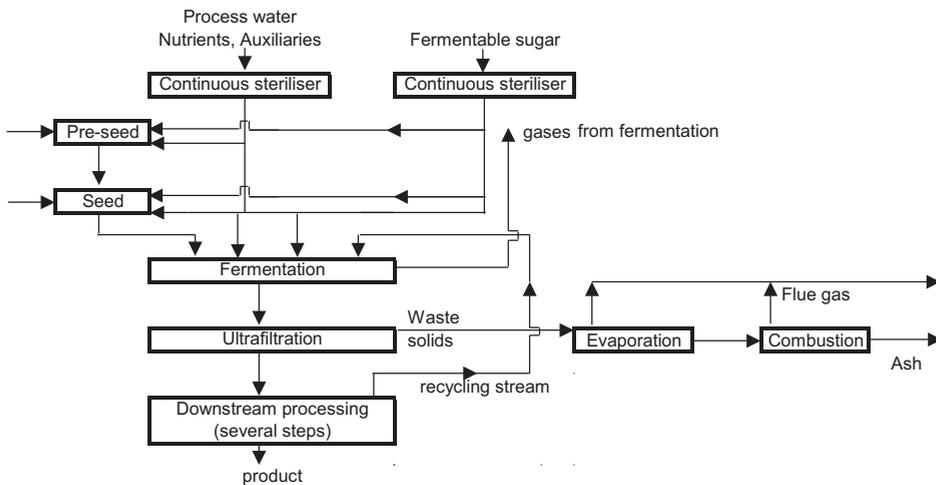


Figure 2.1: Simplified flowsheet as used in the generic approach

2.2.4 Technology assumptions for WB routes

When preparing the mass balance, the mass flows of all compounds were estimated based on the following key parameters: yields, productivity and broth concentration of the fermentation step (see Table 2.2). The broth concentration⁴ determines the amount of water in the broth and therefore influences the energy required in downstream processing. Together with productivity⁵ it determines the residence time as well as the size of the fermentation equipment. The yield⁶ influences not only the required input of fermentable sugar but also the amount of waste biomass produced. Waste biomass is separated from the product stream by means of an ultrafiltration step that immediately follows fermentation. The waste biomass is dried and is then burnt for steam production. Process water is recycled wherever possible in order to avoid excessive consumption.

As part of the generic approach, assumptions were made for the key parameters of current and future technology regarding fermentation and downstream processing (Table 2.2). While both continuous and batch processes were assumed for current technology, only continuous processes were assumed for the future. Current technology calculations rely on data from industrial units, pilot plants or laboratory experiments, future technology calculations rely on two to three decades of successful research and development (R&D, see Table 2.2). The resulting future process data represent a possible upper level of technological feasibility. For current fermentation technology, the values of yields, broth concentration and productivity in Table 2.2 were based on published data. Any ferment-

⁴Broth concentration is defined as the mass of product relative to the volume of water in the fermentation broth (in g/L).

⁵Productivity is defined as the mass of the product divided by the volume of fermentation broth per unit of time needed to produce this amount (in g/(L*h)).

⁶Yield is defined as the mass of the product divided by the mass of the fermentable sugar (in g/g).

able sugar not converted to the target product was split between by-products, waste biomass and CO₂ emissions according to published data (SRI, 2001; Qureshi and Blascheck, 2001; Lynd and Wang, 2004; Campos et al., 2002; Ezeji et al., 2003; Lee et al., 2001; Huang et al., 1998; Niu et al., 2002; Gryta et al., 2000; Bayrock and Ingledew, 2005; SRI, 2002a, 1999a; Akiyama et al., 2003; Lee et al., 2000b). Regarding lactic acid and acrylic acid, generic calculations were only carried out for future technology because industrial data for producing lactic acid according to today's technology were available (Vink et al., 2003) and because the production of acrylic acid is still in a very early stage of R&D.

For future fermentation processes, we assumed a yield of 90 mol-% of the maximum theoretical yield. The remaining fermentable sugar (10 mol-%) is converted into waste biomass and CO₂ only, with an assumed carbon ratio of 1:2 for aerobic and 1:1 for anaerobic processes⁷. This implies that fermentation of by-products can be suppressed, which will most likely require genetic modification of the micro-organism. Future productivities were based on citric acid and ethanol as two representatives of advanced aerobic and anaerobic fermentation processes that have improved faster than other substances because of higher R&D efforts. Their future upper productivity levels were estimated by experts to reach 10 g/(L*h) for citric acid⁸ and 50 g/(L*h) for ethanol⁹. Productivity of the other chemicals was assumed to approach the horizon values of citric acid for aerobic and ethanol for anaerobic fermentation, representing a comparable level of ambition. Future broth concentrations of continuous processes were estimated to be in the range of today's end-of-batch values due to interactions between productivity and concentration:

- Future technology will rely on continuous fermentation at the point of maximum productivity of the micro-organism with in situ removal of the product. The maximum productivity for current processes occurs at approximately half the maximum batch concentration.
- For future technology, the broth concentration corresponding to this maximum productivity was assumed to be increased by a factor of two, thereby resulting in broth concentrations in the range of current batch fermentation. The values assumed for these key technical parameters were critically reviewed by project partners and represent the technical potential after 20–30 years of R&D. However, predicting future key technical parameters always involves uncertainty. We have therefore carried out an exemplary sensitivity analysis for ethanol with a significantly lower productivity to assess its influence on economic viability (see section 2.5.3).

⁷This estimate is based on detailed calculations on the carbon splitting of 4 anaerobic and 4 aerobic fermentation processes. The aerobic processes have ratios around 1:2. For anaerobic processes the spread is much larger; since less metabolic CO₂ should be formed than for aerobic processes we assume a ratio of 1:1.

⁸Productivities as high as 5 g/(L*h) have been reported (Reismann, 1988), so an increase by a factor of two resulting in 10 g/(L*h) appears feasible within 20–30 years.

⁹Some authors (Reddy Kunduru and Pometto, 1996; Lee et al., 1983) reported productivities even higher than 50 g/(L*h), but with current productivities around 2 g/(L*h), an increase beyond 50 g/(L*h) on an industrial scale appeared unlikely within 20–30 years.

Table 2.2: *Type of fermentation and key data on concentration, productivity and yields of fermentation for current and future technology*

Product	Type of fermentation		Concentration g/l	Productivity g/(L* ^h)	Yield g product/ g glucose	Reference
	today/ future	aerobic/ anaer.				
Acetic acid	today	anaer.	18	0.15	0.50	Lee et al. (2001); Huang et al. (1998)
	future	anaer.	50	15	0.90	
Acrylic acid	future	anaer.	50	10	0.72	Niu et al. (2002)
	today	aer.	20	0.42	0.17	
Adipic acid	future	aer.	40	10	0.47	Qureshi and Blaschek (2001); Campos et al. (2002); Ezeji et al. (2003)
	today	anaer.	20	0.36	0.42	
Ethanol	future	anaer.	45	15	0.50	Lynd and Wang (2004); Gryta et al. (2000); Bayrock and Ingledew (2005)
	today	anaer.	100	2.20	0.46	
Lactic acid	future	anaer.	130	50	0.47	SRI (2002a)
	future	anaer.	180	20	0.95	
Lysine	today	aer.	100	1.70	0.34	SRI (1999a)
	future	aer.	140	10	0.63	
PDO	today	aer.	100	1.67–6	0.41	Akiyama et al. (2003)
	future	aer.	100	15	0.54	
PHA	today	aer.	150	3.00	0.35	SRI (2001); Lee et al. (2000b)
	future	aer.	150	10	0.43	
Succinic acid	today	anaer.	80	1.8	0.88	SRI (2001); Lee et al. (2000b)
	future	anaer.	150	15	1.01	

Table 2.3: *Types of downstream processing used for separation in today's and future technology for the WB products*

	Today's Technology	Future Technology
Adsorption	–	Lysine
Crystallisation	Succinic (SRI, 2001), Adipic (Wibowo et al., 2001)	Succinic, Adipic
Distillation	PDO (SRI, 1999a), Ethanol (Gryta et al., 2000), ABE (Qureshi and Blascheck, 2001)	Ethanol, ABE
Electrodialysis	Succinic (Zeikus et al., 1999), Acetic (Fidaleo and Moresi, 2005)	Succinic, Lactic, Acetic, Adipic
Enzymes	PHA (SRI, 2002b)	PHA
Extraction	Acetic (Wisniewski and Pierzchalska, 2005), PHA (SRI, 2002b)	Acetic ^a , Acrylic
Gas stripping	ABE (Ezeji et al., 2003)	ABE
Ion-exchange	Lysine (SRI, 2002a)	–
Pervaporation	Ethanol (Lee et al., 2000a)	PDO, Ethanol, ABE

^a For acetic acid future technology, there are not only flowsheets on extraction and electrodialysis but also one that combines both of these separation technologies.

Table 2.3 presents an overview of the types of downstream processes that were used to separate the products of current and future technology. Polylactic acid, polytrimethylene terephthalate, ethylene, ethyl lactate and caprolactam do not appear in Table 2.3 because they are derived by chemical conversion from one of the WB products listed in this table. The type of downstream processing for today's technology was derived from literature (see Table 2.3). Assumptions for future technology separation processes were based on a number of considerations:

- Precipitation was not considered viable for large-scale production because it involves the use of large amounts of chemicals and leads to low-value by-products such as gypsum and/or waste-water with high salt loads.
- Extraction and adsorption were considered acceptable future options because of the potential use of “green solvents” with clearly lower environmental impacts (e.g. in terms of carcinogenic and toxic effects) compared to current solvents.
- Membrane processes such as pervaporation, electrodialysis and ultrafiltration were taken into account due to their (expected) low energy use. However, a significant amount of R&D will often still be necessary to put these membrane processes into use on an industrial scale.
- Today's as well as future technology assumed single step evaporation up to a water-product ratio of 5 : 1 for evaporation processes; for larger proportions of water, double-effect evaporation was assumed and increased investment costs were accounted for.

2.2.5 Energy use

The process energy for the system covered in the generic approach was determined by multiplying the mass and volume throughputs by the estimated specific energy use for each process step. The specific process energy as shown in Table 2.4 was estimated based on literature, then calibrated (for a detailed description of the procedure see Patel et al. (2006)).

Table 2.4: Key data on specific energy use of unit processes in fermentation and downstream processing

Unit process	Amount	Unit
<i>Fermentation</i>		
Sterilisation	0.100	kg steam/kg medium _{fermentation}
Agitation	0.500	kW power/m ³ volume _{fermentation}
Agitation and Aeration	3.000	kW power/m ³ volume _{fermentation}
<i>Downstream processing</i>		
Membrane Filtration		
- Microfiltration	2.000	kWh power /m ³ permeate
- Ultrafiltration	5.000	kWh power /m ³ permeate
- Diafiltration	5.000	kWh power /m ³ permeate
- Nanofiltration	7.000	kWh power /m ³ permeate
- Reverse osmosis	9.000	kWh power /m ³ permeate
Electrodialysis	0.100	kWh power / mol-equivalent
Evaporation of water		
- single stage	1.200	kg steam/kg evaporated
	0.040	kWh power/kg evaporated
- multi-stage	0.500	kg steam/kg evaporated
	0.005	kWh power /kg evaporated
Distillation	1.3*product's heat of evap.	kg steam/kg evaporated

2.2.6 Process economics methodology

The economic analysis for WB chemicals was performed for an assumed plant capacity of 100 kt/yr, which was a compromise in view of economies of scale on the one hand and transport costs for the bio-feedstocks on the other. This scale was considered representative for a WB plant, but larger scales are also possible and a sensitivity analysis was carried out to assess the influence of plant size on economic viability (see section 2.5.2). Petrochemical processes were calculated for current plant sizes, which can be clearly larger than 100 kt depending on the product (see section 2.2.9).

All cost calculations were based on investments for building a new plant in Western Europe, with calculations carried out in €₂₀₀₀. The investment (Total Fixed Capital, TFC) and labour requirements were estimated by DSM by applying their so-called Functional Unit Method (Simons and Nossin, 2005). These calculations were carried out for each

generic WB route based on the individual product flow sheet, mass and energy balance (see Patel et al., 2006, for individual numbers). We used market prices for petrochemical feedstocks and auxiliaries, the prices of fermentable sugar were set exogenously (see section 2.2.8). The procedure for the economic assessment (see Figure 2.2) is in line with standard business economics: Firstly, variable costs (feedstock, auxiliaries/catalysts, by-products, utilities, waste treatment) and fixed costs (supplies, labour) were added to obtain the total direct operating costs. Secondly, taxes, insurance fees and plant overhead were added to this figure as well as an allowance for marketing, administration, and R&D. And finally, the so-called capital charge, representing the total of depreciation and profits, was added. The final result is the production cost plus profits (PCPP; also known as profited production cost) which is a proxy for the market price. The capital charge was calculated by multiplying the total fixed capital with a fixed percentage. In consultation with industry experts, a capital charge of 30% was used, partially accounting for contingency (see section 2.5.2 for a sensitivity analysis). Value added tax was not included in the calculations.

A given WB product was considered economically viable if its PCPP was lower than the market price or the PCPP of its petrochemical counterpart. The real market price of the WB product may be higher or lower than its PCPP depending on demand and supply:

- The PCPP of a WB product is usually substantially lower than its market price if the WB product is new on the market and if it is used for niche applications. Possible reasons are that the profit made is higher, the real capacity of the production facilities is clearly lower than 100 kt/yr, the process is not optimised and/or continuous operation cannot be ensured.
- The PCPP of the WB product can also be higher than the market price. This is in particular the case if the WB product is chemically identical with a petrochemical product that has been manufactured for decades via an established production route. Such petrochemical processes can be economically superior due to the advantageous economies of scale and/or production in depreciated plants.

2.2.7 WB industrial process data

For some bio-based processes, confidential information was made available to us by companies and research institutes for the biotechnological plant as a whole. This confidential data consisted of aggregated information on the physical quantities of inputs and outputs per tonne of WB chemical (fermentable sugar, auxiliaries and final energy use by types) as well as total investment costs and labour requirements. For PDO and lactic acid, industry data stemmed from industrial plants of DuPont (Alles, 2003) and NatureWorks (Vink, 2005). For ethanol, lysine, and succinic acid, it originated from pilot plants (SRI, 2001, 2002a, 1999b). This data is referred to as industry data.

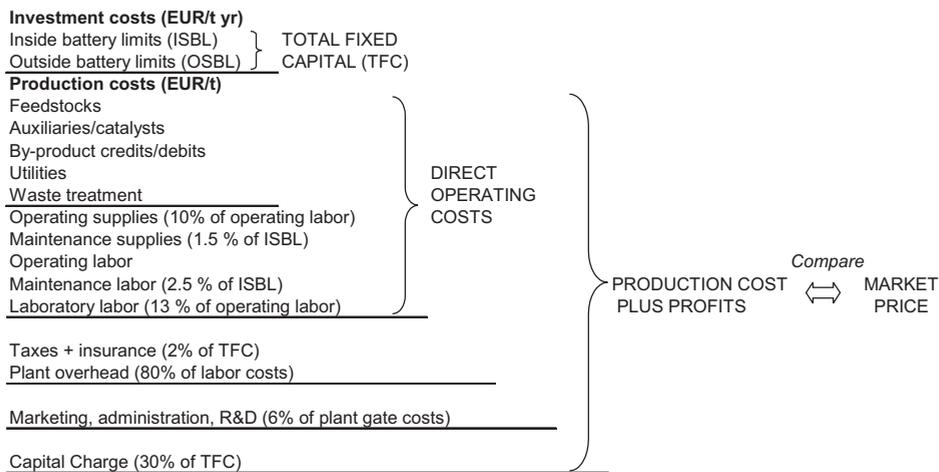


Figure 2.2: Procedure to calculate production costs plus profits (PCPP)

2.2.8 Prices of fermentable sugars

Fermentable sugar was the feedstock of WB processes and its price influenced the economics of WB products. Fermentable sugar may be raw or refined and consists of biomass-derived readily fermentable carbohydrates such as sucrose, hydrolysed starches or pre-treated and hydrolysed lignocellulose, which is still an emerging technology. In order to account for variations in fermentable sugar prices both in the near and longer-term future and for world regions, calculations were carried out for four price levels of fermentable sugar. The lowest price of 70 €/t represents local sugar prices in Brazil¹⁰: due to good climatic conditions and the availability of very cheap labour, the production cost of fermentable sugar from sugar cane was at its lower boundary. The high sugar price of 200 €/t represents a 10 year average of world raw sugar (contract 11) as traded at the New York Board of Trade (NYBOT, 2005a). An intermediate price was chosen at 135 €/t. An extreme level at 400 €/t represents a 10 year average of US domestic raw sugar (contract 14) as traded at the New York Board of Trade (NYBOT, 2005b). These price levels were based on raw sugar prices. However, if the micro-organism is sensitive to impurities then refined sugar will be used for fermentation. World refined sugar as traded at the London Stock Exchange has been traded for an average price of ca. 250 €/t during the last 10 years (Rupp-Dahlem, 2005) and thus within the range of prices considered. The price ranges for a tonne of fermentable sugar were exogenous inputs to our calculations, i.e. we do not perform economic analyses for different combinations of feedstock types with technologies for producing fermentable sugar.

¹⁰Based on an average sugar price of R\$ 200 for the 2003/2004 season (ORPLANA, 2005) and an exchange rate of 1 R\$ = 0.29 € for the same time frame.

2.2.9 Petrochemical process data

In this chapter we differentiate between WB industry data (see section 2.2.7) and petrochemical data. With petrochemical data we mean PCPP values of bulk petrochemicals which have been calculated according to the method described in section 2.2.6. The required process data on inputs, outputs, labour requirements and investment costs stems from SRI (SRI, 2000). Prices of petrochemical products were based on a crude oil price of US\$ 25 per barrel, but a sensitivity analysis was carried out with twice this price (see section 2.5.1). Although petrochemical process technologies may improve in the future, we did not take into account any technological progress and always used current petrochemical technology for comparisons (see section 2.5.3 for a discussion).

Typical plant sizes range between 50 kt/yr for maleic anhydride and 700 kt/yr for ethylene and acetic acid. We kept plant sizes larger than 100 kt/yr unchanged but increased production capacities of pre-market products such as PDO and PTT to 100 kt/yr if they were smaller in SRI (2000), using scaling factors of 0.75 for investment costs and 0.25 for labour. Similar to WB products, petrochemical products may also show discrepancies between market prices and PCPPs. By analogy to the explanation given in section 2.2.6, the PCPP of a petrochemical product is usually substantially lower than its market price if the product is new on the market and/or if it is used for niche applications. If the PCPP of the petrochemical product is higher than the market price this is typically the consequence of fierce competition accompanied by overcapacities. The difference between the market price and the PCPP can be substantial for both WB and petrochemical products. Since this difference was largely unpredictable we based our economic comparisons for current and future technology on the PCPPs of the WB products and their petrochemical counterparts to ensure a level playing field.¹¹

2.2.10 Prices of utilities and auxiliaries

For the economic assessment we assume a default stock market crude oil price of US\$ 25 per barrel (bbl; 4.1 €/GJ) and a natural gas price of 4 €/GJ. Prices were valid for the year 2000 for large West European industry. This corresponds to an estimated electricity price of 15 €/GJ_e and a steam price of 12 €/t (5.7 €/GJ). The sensitivity analysis in 2.5.1 studies the effect of higher energy prices, with a crude oil price of US\$ 50/bbl (8.2 €/GJ), a natural gas price of 6 €/GJ and corresponding steam and power prices of 17 €/t (8.3 €/GJ) and 16 €/GJ_e respectively. Economic credits were introduced in order to account for the avoided production of heat and power in case energy was recovered. Enzyme and membrane prices were assumed to decrease in the future due to their large-scale production. Enzyme prices for current technology were set to 100 €/kg and were assumed to decrease by a factor of 10 for future technology, the prices of high quality membranes were set to 100 €/tonne of product and were assumed to decrease by a factor of two.

¹¹Market prices of petrochemicals may be used if the distorted situation is expected to continue or if a “snapshot” of the current situation is required.

2.3 Case study on 1,3-propanediol

In order to provide insight into the type of analysis performed, this section presents the findings of the techno-economic assessment in a detailed case study on 1,3-propanediol (PDO). PDO is used to produce polytrimethylene terephthalate (PTT), a polymer with the potential to replace nylon and PET fibres. We select PDO for this case study, because both the petrochemical and the WB-based routes are currently being pursued by industry: Shell currently produces PDO from petrochemical feedstocks, while DuPont is building a plant for PDO production based on fermentable sugars from maize starch. The same type of analysis was performed for the other products, the results of which will be presented in condensed form in section 2.4.

2.3.1 Production routes

Four different production routes of WB-based PDO were calculated according to the generic approach and were then compared to industry data on the production of PDO by means of WB as well as petrochemistry. The four WB routes studied with the generic approach all consist of aerobic fermentation of fermentable sugar to PDO, with seed train, inoculum train, fermentation and ultrafiltration steps, followed by different types of downstream processing. Routes 1 and 2 represent current technology, with downstream processing by distillation. Routes 3 and 4 represent future technology, where separation of PDO occurs by pervaporation, which has been successful on the laboratory scale (Li et al., 2001). DuPont's route consists of aerobic fermentation with undisclosed downstream processing. The petrochemical route results in PDO via hydroformylation of ethylene oxide. Based on the data for concentrations, productivities and yields from Table 2.2, mass and energy balances were established. The energy balance was derived from energy requirements of unit processes (see Table 2.4). Table 2.5 shows the resulting in- and outputs for the four WB routes according to the generic approach.

2.3.2 Results for PDO production routes

Following the methodology, the inputs and outputs according to the generic approach as well as from industry and petrochemical process data were used to calculate the production cost plus profits (PCPP) of the different routes to PDO. Figure 2.3 shows the resulting PCPPs for routes 1 and 4 according to the generic approach, for DuPont's WB route and for the petrochemical pathway as well as the current market price of PDO. The values for routes 2 and 3 remain between those of routes 1 and 4 (see also Figure 2.3). The PCPP values of route 1 and industry coincide well, thereby corroborating the generic approach. As shown in Figure 2.3, substantial savings are possible with current state-of-the-art technology relative to the market price. However, PDO is a relatively new product, which is currently manufactured in small quantities and sold at high prices for niche applications. Instead of using the current market price as a benchmark, it was therefore better to choose the PCPP of petrochemical PDO. This comparison shows that

Table 2.5: Process inputs and outputs for the production of 1 tonne of PDO (at the plant gate) with current and future technology as well as related investment costs and labour requirements for a 100 kt plant

		Route 1	Route 2	Route 3	Route 4
Technology		today	today	future	future
Fermentation		batch	continuous	continuous	continuous
Downstream		distillation	distillation	pervaporation of water	pervaporation of PDO
	unit				
<i>Inputs</i>					
Dextrose	t	2.4	2.4	1.9	1.9
Nutrients	t	0.1	0.1	0.1	0.1
Process water	t	0.7	0.7	0.7	0.7
Electricity	GJ _e	9.1	2.9	1.6	1.4
Steam	t	6.6	6.6	10.2	0.9
Membranes	€			50.0	50.0
<i>Outputs</i>					
PDO	t	1.0	1.0	1.0	1.0
Waste biomass	t	0.2	0.2	0.1	0.1
<i>Fixed Costs</i>					
ISBL ^a	M €	75	67	42	39
OSBL	M €	28	25	20	18
Labour	fte	28	15	11	11

^a Costs for inside battery limits (ISBL), outside battery limits (OSBL) and labour are given for an entire plant, based on a capacity of 100 kt. ISBL deals with the core process equipment, piping, instrumentation etc. OSBL deals with steam and power generation and supply, wastewater treatment, cooling towers, etc. 'fte' is short for full time equivalents.

for future technology (route 4), PDO is competitive with petrochemical PDO for a sugar price of up to 400 €/t and crude oil prices of US\$ 25/bbl.

2.3.3 Composition of production costs

In order to better understand the main factors determining the PCPP, we discuss the contribution of feedstock costs, utility costs, capital charge and the remainder (other costs). Feedstock costs depend on both the amount of fermentable sugar required and its price level. Utility costs can differ considerably between current and future technology. The investment costs depend on the number of fermentation vessels needed (determined by the productivity level) and the type of downstream processing, e.g. the number of extraction columns.

Figure 2.4 shows that for high fermentable sugar prices (400 €/t), feedstock costs alone may be higher than the total PCPP for low sugar prices (70 €/t). For low and medium sugar prices, dependence on feedstock prices of WB-based PDO was much weaker

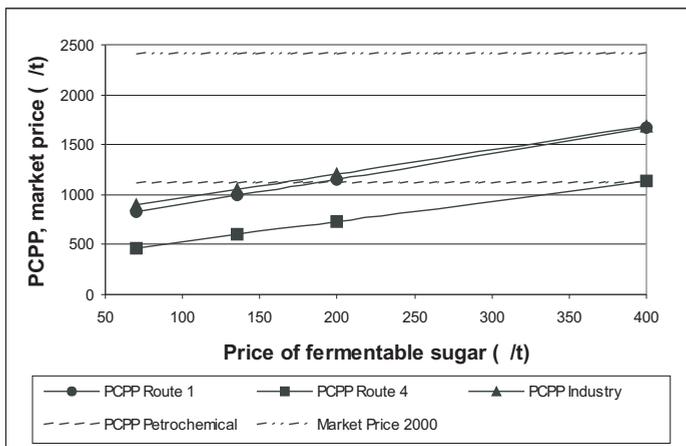


Figure 2.3: PDO production cost plus profits (PCPP) for today's and future WB processes compared to both petrochemical PCPP and current market price for PDO (energy price level: US\$25/bbl)

compared to petrochemical PDO.¹² For very high sugar prices, the contribution of feedstock costs was higher for WB processes compared to the petrochemical processes. Finally, a doubling of the oil price leads to feedstock costs that are almost equal to the PCPP of the low oil price.

¹²Note that there is a methodological difference between the two petrochemical PDO cases: while the first case (crude oil price of US\$ 25/bbl) uses market prices for the inputs to production, the second case uses PCPPs for these inputs, because market prices at a crude oil price of US\$ 50/bbl were not available (see also section 2.5.1).

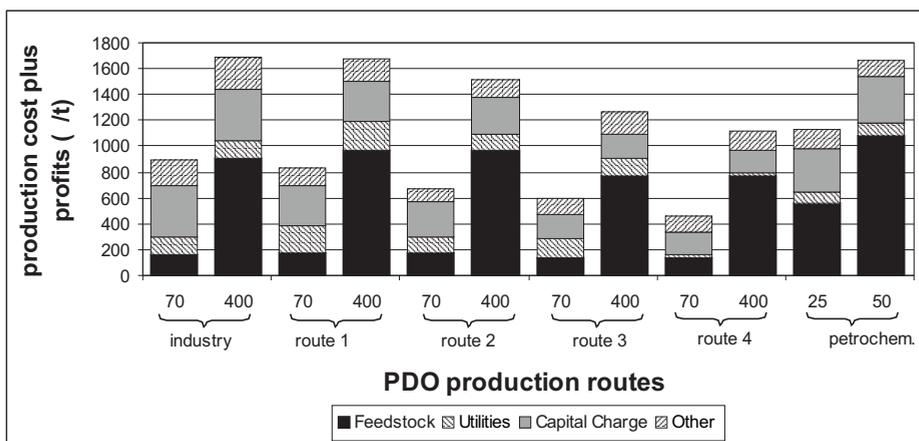


Figure 2.4: Cost composition for PDO production for the industry and 4 generic production routes of PDO for low (70 €/t) and high (400 €/t) prices of fermentable sugar (crude oil price: US\$25/bbl) as well as the petrochemical production route for crude oil prices of US\$25/bbl and US\$50/bbl

2.4 Results

Production costs plus profits (PCPPs) were calculated for all WB products using the methodology described above (see Figure 2.2). Table 2.6 shows the results not only for the generic approach (see rows labelled “Today” and “Future”) but also for industry data. For lactic acid, acrylic acid and caprolactam, no generic approach calculations were available for today’s technology (see section 2.2.4). The ranges specify PCPPs of at least two different separation processes (see Table 2.3) in the generic approach or different datasets in the case of industry data and show the whole range of PCPPs calculated. The petrochemical PCPPs in Table 2.6 refer to benchmark substances that are chemically identical with the bio-based compounds, unless indicated otherwise. In the cases of lactic acid and lysine no petrochemical benchmarks can be given because these products are produced from bio-based feedstocks via WB already today; in these cases the comparison was made with current industrial practice.

Comparing the results of the generic approach for today’s technology to industry data (for those products where both were available), we conclude that the results correlate well for lysine, PDO and PTT, where the PCPPs are in the same range. In the case of succinic acid, the calculations according to the generic approach are somewhat lower than the industry data: investment costs for downstream processing are higher for the latter. In general, the PCPPs of current technology according to the generic approach and those of industry are in the same range, with a small spread of 5–10% for a given price of fermentable sugar. An exception is the large spread in industry data on lysine, which is due to differences in salt composition and degrees of purity. We therefore conclude that the generic approach yields reliable results.

Table 2.6: Production cost plus profits (€/t) of WB products for varying sugar prices and current and future technology according to the generic approach (rows entitled 'Today' and 'Future'), as well as industry, of the petrochemical benchmark and the market price (for US\$ 25/barrel crude oil)

Product	Techno-logy	Production Cost Plus Profits (€/t)				Petro-chemical	Market Price (€/t)
		Bio-based, as function of fermentable sugar price	70 €/t	135 €/t	200 €/t		
ABE	Today	1160–1230	1370–1430	1570–1640	2210–2270	700 n-butanol	500
	Future	390–480	570–660	740–830	1270–1360		
Acetic acid	Today	2070–2080	2210–2220	2350–2360	2770–2790	370	400
	Future	620–750	700–820	780–900	1020–1140		
Acrylic	Future	960	1050	1150	1440	1050	880
	Today	2580	2980	3380	4600		
Adipic acid	Future	980–1050	1120–1200	1270–1350	1720–1810	1100	1090
	Today	510	660	800	1260		
Ethanol	Today	500–570	650–730	800–880	1260–1350	870	440
	Future	360–370	510–520	660–670	1110–1130		
Lactic acid	Industry	690–900	770–980	850–1060	1090–1300	–	1390
	Future	390–410	460–485	540–560	770–800		
Lysine	Industry	1230–2110	1450–2300	1670–2490	2350–3080	–	1440
	Today	1580	1890	2200	3160		
PDO	Future	810	1030	1250	1920	1120	2410
	Industry	900–960	1050–1070	1190–1210	1550–1690		
PHA	Today	660–820	830–990	1000–1160	1510–1670	930	1100
	Future	470–600	600–730	730–860	1130–1260		
Succinic acid	Today	1090–2220	1310–2430	1530–2640	2220–3280	860 PE	700
	Future	1090	1250	1420	1930		
Caprolactam	Industry	1110	1180	1250	1460	1860	1320
	Today	750–870	830–950	910–1020	1150–1260		
Ethyl lactate	Future	470–570	540–640	610–710	820–920	860	650
	Today	1200	1380	1560	2100		
Ethylene	Future	790–890	920–1020	1040–1140	1420–1530	610	720
	Today	1050	1320	1590	2430		
PLA	Future	820	1080	1340	2140	1050	1200
	Industry	1310–1420	1410–1520	1510–1630	1830–1940		
PTT	Future	1180–1280	1270–1380	1370–1480	1670–1770	1180	–
	Industry	1090–1110	1150–1160	1200–1210	1340–1400		
PTT	Today	1000–1060	1060–1130	1130–1190	1330–1390	1180	–
	Future	750	750	750	760		

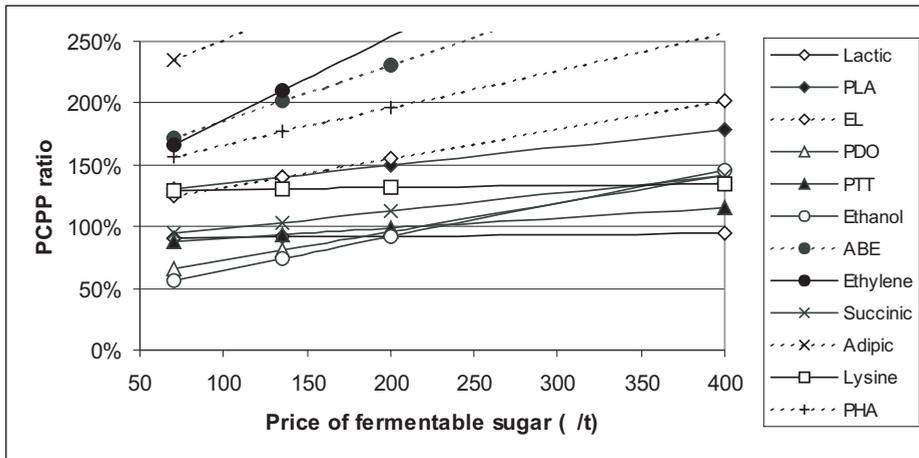


Figure 2.5: Economic viability of today's WB technology: ratio of production cost plus profits (PCPP) of the WB product to its petrochemical counterpart for today's technology as a function of the sugar price level (for 25US\$/barrel crude oil)

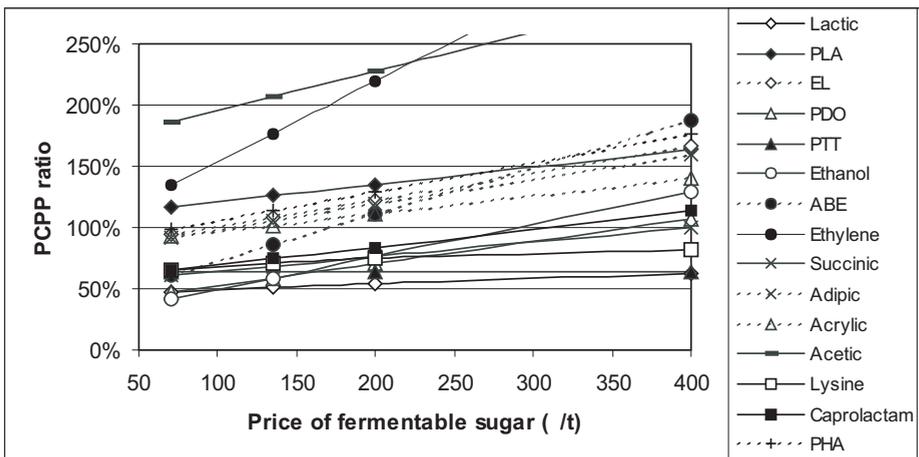


Figure 2.6: Economic viability of future WB technology: ratio of production cost plus profits (PCPP) of the WB product compared to its petrochemical counterpart for future technology as a function of the sugar price (for 25US\$/barrel crude oil)

Most importantly, Table 2.6 shows that for current technology, 30% of WB products are economically viable for low sugar prices. PDO, PTT and ethanol are economically viable even for high sugar prices (200 €/t). These findings are more directly visible from Figure 2.5, which shows the ratio of PCPPs of the current technology WB products compared to their petrochemical counterparts. Values below 100% indicate that the production

costs are lower for the WB product, while values above 100% represent cases in which the production of the WB product is more expensive than its petrochemical counterpart. The PCPP ratio of acetic acid ranges from 550% to 740% and therefore lies outside of the range of Figure 2.5. By analogy, Figure 2.6 shows the ratios of PCPP for future technology WB products to the (current technology) petrochemical. Almost all products researched offer economic savings for a sugar price of 70 €/t, with the exception of acetic acid, ethylene and PLA.

In conclusion, technological progress can contribute significantly to improve economic viability of WB products: on average, the PCPPs of products that are directly obtainable from fermentation (e.g. lactic acid) are 40%–50% lower than for today’s technology; for products which require a chemical conversion after fermentation (e.g. PLA), technological progress reduces the PCPP by approximately 20%. This results in many more products becoming economically viable in the future, even at rather high sugar prices. Table 2.7 summarises the viability of current and future WB chemicals for four sugar prices.

Table 2.7: Viability of WB-chemicals for four price levels of fermentable sugar for current and future technology (crude oil price: US\$25/bbl); Note: ‘Possibly’ indicates that the PCPP of the WB-chemical is higher than the PCPP of its petrochemical equivalent, but lower than the current market price of the product.

Sugar price	Today	Future
400 €/t	PDO (possibly)	Lactic acid, PTT, lysine, PDO (possibly)
200 €/t	PDO, PTT, ethanol	Lactic acid, PTT, lysine, PDO, caprolactam, ethanol, succinic acid
135 €/t	PDO, PTT, ethanol	Lactic acid, PTT, lysine, PDO, caprolactam, ethanol, succinic acid, ABE
70 €/t	PDO, PTT, ethanol, succinic acid, PHA (possibly)	Lactic acid, PTT, lysine, PDO, caprolactam, ethanol, succinic acid, ABE, acrylic acid, adipic acid, ethyl lactate, PHA (possibly), PLA (possibly)

2.5 Discussion

2.5.1 Sensitivity analysis for high crude oil prices

This sensitivity analysis investigates the effect of the crude oil price on the competitiveness of the WB products compared to their petrochemical counterparts. The crude oil price is set to US\$ 50/bbl (8.9 €/GJ) and the natural gas price for end users to 6 €/GJ. Our analysis accounts for increased prices of feedstock and auxiliaries as well as utilities but not for increased investment costs as a consequence of higher energy costs. In order to perform the calculations for a crude oil price of US\$ 50/bbl, the methodology has to be adapted: while, for the default case (US\$ 25/bbl) we use market prices for feedstocks

and auxiliaries, these are not available to us for an oil price level of US\$ 50/bbl. For this reason, we use PCPPs for these process inputs instead of market prices. In the cases of ethyl acetate and PET, some petrochemical process inputs have higher PCPPs than market prices even for the low crude oil price (US\$ 25/bbl), leading to PCPP values that are significantly higher (by 30%–70%) than those presented in Table 2.6. This indicates that their production is not economically viable.

For all other products and their petrochemical counterparts, the PCPPs derived from this changed method differ only to a small extent (by 4% on average) from the PCPPs presented earlier. The sensitivity analysis is performed for a selection of WB-based products that have a high chance of gaining a large market share, i.e. ABE, acetic acid, adipic acid, ethanol, ethyl lactate, ethylene, PDO, PHA, PLA, PTT and succinic acid (we therefore exclude acrylic acid, caprolactam, lactic acid and lysine). In general, the PCPPs of the petrochemical products are found to increase by 23% on average as a result of higher feedstock and energy costs, whereas the PCPPs of the WB products only increase by 4%–6% due to higher utility costs. Some WB products become economically competitive even at the highest sugar price, both with current and future technology, for example PDO, PTT and succinic acid. The values for acetic acid are outside the range of Figure 2.7 (the economic viability deteriorates compared to lower crude oil prices due to the large amount of energy necessary in downstream processing).

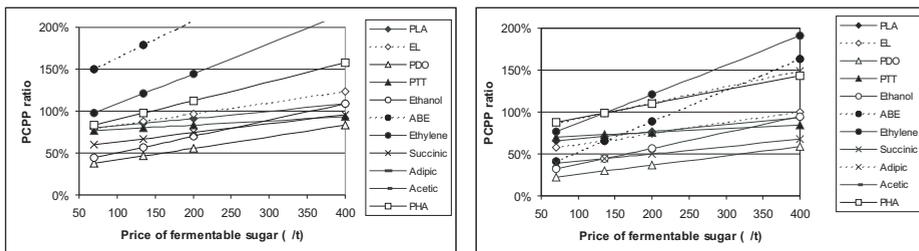


Figure 2.7: Selected production cost plus profits (PCPP) ratios as a function of the sugar price for current (left) and future (right) technology at a high crude oil price (US\$ 50/bbl)

In the case of future technology ethylene, the WB product is economically competitive with the petrochemical product for the high crude oil price (US\$ 50/bbl) up to a sugar price of 135 €/t. For PLA, the effect of higher oil and gas prices is even larger: it is economically viable for the high crude oil price (US\$ 50/bbl) up to the highest sugar price for future technology. On average, relative economic viability of WB products increases by 25% (ethylene is an exception and increases by 70%). This indicates that the development of the oil price will have a significant effect on the economic viability of WB products. Comparing Figures 2.5 and 2.6 to Figure 2.7 it becomes clear that at the higher oil price more WB products have a PCPP ratio of less than 100%, i.e. are economically viable. A more detailed scenario analysis dealing with a larger variety of oil prices for a selection of products is carried out by Dornburg et al. (2008).

2.5.2 Sensitivity to economic parameters

All WB calculation results presented refer to a plant capacity of 100 kt per year, but for most of the products studied, sensitivity analyses have been performed for larger plant sizes (200 kt to 400 kt per year) for future technology. The results of the sensitivity analysis show that larger scale production plants reduce the PCPP of the WB routes by on average 10% (range: 5%–20%) for a high sugar price of 400 €/t and 16% (range: 10%–30%) for a low sugar price of 70 €/t. A sensitivity analysis was also carried out for a lower capital charge of 10% of the TFC instead of 30% for current and future technology. This reduced the PCPP on average by 15% (range: 5%–25%) for the high sugar price of 400 €/t and by 25% (range: 10%–35%) for the low sugar price of 70 €/t. The average sensitivities of the petrochemical equivalents are 20% both to scale and to capital charge. Therefore, both scale and capital charge are found not to significantly influence the economic competitiveness of the WB product compared to the petrochemical equivalent. The results are therefore rather robust both to scale and capital charge.

2.5.3 Sensitivity to key technical parameters

To quantify the effect of improved biotechnology, data on production of electricity, fertiliser, and biomass yields in agriculture were kept constant throughout the analysis. However, it is likely that these technologies will also improve in the future. For all comparisons, current technology was assumed for petrochemical production routes: it has not been taken into account that petrochemical processes will also improve in the future. Including technological progress in petrochemical processes would lead to smaller benefits by WB because savings in process energy of 20% and beyond are possible for future technology petrochemicals (Ren et al., 2006). However, utilities only make up 3% of the PCPP of petrochemical ethylene, and savings in the range of 20% will therefore not significantly change the relative PCPPs of WB chemicals compared with petrochemicals. Additionally, the economic calculations of future WB technology routes strongly rely on the key assumptions on productivities, yields and concentrations (Table 2.2). The values assumed for these key parameters were critically reviewed by project partners and are considered to represent the technical potential that may be reached after two to three decades of R&D. A sensitivity analysis is performed for future ethanol production to assess the influence of the values assumed for future productivity: a significantly lower productivity of 10 g/(L*h) instead of 50 g/(L*h) increases the PCPP only by 5% for the whole range of sugar prices. In general, the real technical data are expected to remain below the values in Table 2.2 within the next 10 years and the real economic viability for this time-frame may therefore fall short of our calculations. More details of the sensitivity analyses are also available in Patel et al. (2006).

2.5.4 Comparison with earlier publications

Production of succinic acid from maize starch (Zeikus et al., 1999) and glucose (Landucci et al., 1994) results in succinic acid prices in ranges of US\$ 550–2200 per tonne. These

values are in line with the PCPPs from the generic approach (760–920 €/t for current technology). Production of ethanol from maize starch (O'Brien et al., 2000) and lignocellulosics (Hamelinck et al., 2005) results in ethanol prices in the range of US\$ 650–960 per tonne. These values correlate well with the PCPPs from the generic approach (500–1350 €/t for current technology). Production of ABE from maize starch (Qureshi and Blascheck, 2001) results in butanol prices in the range of US\$ 340–1070 per tonne. These values differ from the PCPPs from the generic approach (1170–2270 €/t for current technology). There are three factors contributing to this difference: the price of maize, the choice of location and the credits from by-products. The price of (whole) maize is set to US\$ 79/t and US\$ 197/t, which translates into US\$ 56/t glucose and US\$ 140/t glucose. Both values are at the lower end of the sugar prices used here. The type of plant has a significant influence, Qureshi and Blaschek 2001 assume that the plant will be built as an extension to an existing corn milling plant, with proportionately lower investment costs compared to a grassroots plant as assumed in the generic approach. The credits from by-products are decisive: in this research we only consider economic credits for acetone and ethanol produced, but not for gases, cell mass, remaining sugars, etc. The credits for acetone and ethanol only make up 17% of the by-product credits in Qureshi and Blaschek (2001), with total by-product credits much larger than the revenues from butanol. Recalculating their data for a new plant, with by-product credits only for acetone and butanol results in a butanol price of US\$ 1319/t (for a maize price of US\$ 197/t), well within the range of our results.

2.5.5 Comparison with current developments in industry

The results from the generic approach show that the production of PDO/PTT is economically viable at the low crude oil price (US\$ 25/bbl). This is in line with current developments in industry, which show DuPont building a plant in Tennessee to produce bio-based PDO using fermentable sugars from maize. The generic approach also shows that bio-based ethanol is economically viable using current technology but although it is produced on a large scale for fuel use, it has not entered the chemical industry sector in West Europe. This may be due to the presence of already depreciated plants that continue to be used. Although the generic approach shows that PLA is not economically viable at the low crude oil price (US\$ 25/bbl), calculations for the high crude oil price (US\$ 50/bbl) show that PLA is viable for current technology up to a sugar price of 200 €/t and all sugar prices for future technology. High crude oil prices in recent years have validated Nature-Works' building of a large industrial PLA plant in Nebraska with production running since 2001. The results from the generic approach show that the production of PHA is economically viable at the high crude oil price (US\$ 50/bbl) for low and medium sugar prices. This is in line with current developments in industry, which show Archer Daniels Midland and Metabolix announcing plans to build a plant in Iowa to produce bio-based PHA from maize starch (ADM, 2006).

2.6 Conclusions

In this chapter we presented and applied a generic approach which allows the systematic evaluation of present and future production routes of bio-based chemicals from white biotechnology, based on available data and consistent assumptions on future (bio)technology developments. The production costs plus profits of current technology according to the generic approach and those of industry are in the same range, implying that the generic approach yields reliable results.

In general, a large number of white biotechnology chemicals are economically viable compared to their petrochemical equivalents. This economic competitiveness depends to a large extent on the prices of oil and sugar. For a crude oil price of US\$ 25/bbl the following products are economically viable for current technology: 1,3-propanediol, polytrimethylene terephthalate, succinic acid and ethanol. Comparing current to future technology, production cost plus profits of products directly obtained from the fermentation step are 40%–50% lower and 20% lower for products that require a chemical conversion step after fermentation for a crude oil price of US\$ 25/bbl and across all sugar prices. This shows that technological progress can contribute significantly to improved economic viability of white biotechnology chemicals. For future technology, all studied products except for acetic acid, polylactic acid and ethylene are economically viable at fermentable sugar prices of 70 €/t. The sensitivity analysis shows that at a crude oil price of US\$ 50 per barrel, future technology ethylene will be economically viable for a sugar price of up to 135 €/t and polylactic acid will be viable up to the highest sugar price. All other products improve in economic competitiveness.

A large-scale introduction of white-biotechnology-based production of economically viable bulk chemicals would therefore be desirable if the environmental impacts are smaller than those of current petrochemical production routes (this is discussed in Chapter 3). Under these conditions, white biotechnology could become the centre of attention for the chemical industry as well as for policy-makers.

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3

Producing bio-based bulk chemicals using white biotechnology saves energy and combats climate change*

Abstract

The production of bulk chemicals from biomass can make a significant contribution to solving two of the most urgent environmental problems: climate change and depletion of fossil energy. We analysed current and future technology routes leading to 15 bulk chemicals using white biotechnology and calculated their CO₂ emissions and fossil energy use. Savings of more than 100% in non-renewable energy use and greenhouse gas emissions are already possible with current state of the art biotechnology. Substantial further savings are possible for the future by improved fermentation and downstream processing. Worldwide CO₂ savings in the range of 500–1,000 million tonnes per year are possible using future technology. White biotechnology hence offers excellent opportunities for mitigating greenhouse gas emissions and decreasing dependence on fossil energy sources and therefore has the potential to make inroads into the existing chemical industry.

3.1 Introduction

Bulk chemicals are currently produced almost exclusively from petrochemical feedstocks derived from crude oil and natural gas. Producing the same or functionally equivalent

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chemicals from renewable resources can decrease the environmental impact, e.g. emissions of greenhouse gases. In the light of long-term emission targets such as the EU target of 15–30% reduction of CO₂ emissions by 2020 as well as increasingly high oil prices, bio-based chemicals may play an important role by contributing towards achieving emission targets as well as by reducing dependence on expensive fossil resources. New perspectives and opportunities for the production of chemical bulk materials and intermediates from renewable resources have been opened by recent progress in white biotechnology (WB), due primarily to increased productivities and yields of fermentation. Further substantial progress in this area is expected, especially related to genetically modified microorganisms. Nevertheless the application of WB for the production of bulk bio-chemicals has so far received much less attention than bio-ethanol for fuel use and energy production from biomass.

WB comprises the production of organic compounds by means of fermentation or enzymatic conversions. In principle, both bio-based and petrochemical feedstocks can be used as raw materials for WB processes, but bio-based feedstocks are most commonly used in WB and petrochemical feedstocks for WB are therefore not considered here. Ragauskas et al. (2006) have stated that converting biomass sugars into biomaterials and fuels will be one key feature of future biorefineries. However, environmental assessments related to WB-based bulk chemicals produced from renewable resources are scarce: a number of publications on the environmental impacts of WB-based products are available, but almost all have focused on a single product: predominantly on fuel ethanol (Kim and Dale, 2005a; Sheehan et al., 2003) but also on (poly)lactic acid (Vink et al., 2003; Bohlmann, 2004), and polyhydroxyalkanoates (Kim and Dale, 2005c; Akiyama et al., 2003). Background assumptions, system boundaries and methodologies differ among these studies and the results are not easy to compare. Furthermore, future technology improvement is disregarded although it may substantially reduce the environmental impacts of bulk chemicals from WB. To overcome this problem, we have developed and applied a generic approach that allows estimating the inputs of feedstock and energy and the associated emissions and costs (see Chapter 2).

Earlier analysis based on this generic approach has shown that WB-based chemicals have *economic* advantages over their petrochemical counterparts for fermentable sugar prices even above current market prices (see Chapter 2). Here we quantify *environmental* impacts of producing one tonne of chemical using current and future technology WB. We do this according to the principles of life-cycle assessment, i.e. covering the entire process chain starting from the extraction of resources.

3.2 Methodology

The goal of this study was to analyse the environmental performance of producing bulk chemicals from biomass using industrial biotechnology (WB) considering current and future (2030) technology and to compare it with bulk petrochemicals. We focus exclusively on the use of the selected products as chemicals and exclude their use as fuels and animal

feed. Following the principle proposed by Landucci and Lynd (Landucci et al., 1994; Lynd and Wang, 2004), we simulate large-scale industrial processes based on WB by modelling the conversion of fermentable sugar to the target WB chemical using standard modules such as fermentation, filtration, and distillation (see Chapter 2 for a more detailed description). For each production route, mass and energy balances were determined at the level of unit processes and a common database for all process inputs was used. Our generic approach therefore allows standardised comparisons between different processes based on a small number of input data.

3.2.1 Functional unit and product selection

The functional unit is one tonne of chemical. The selection of products in this chapter was based on three criteria: 1) a feasible fermentation process, 2) available information on the stoichiometry of this process and 3) the potential to be sold in bulk quantities in the medium or long term (see Chapter 2 and Patel et al. (2006) for details). The following 10 products were studied: 1,3-propanediol (PDO), acetic acid, acrylic acid, adipic acid, butanol (from the ABE process), ethanol, lysine, lactic acid, polyhydroxyalkanoates (PHA), and succinic acid. Additionally, five products were included that are formed by subsequent chemical conversion of these products: caprolactam, ethyl lactate, ethylene, polylactic acid (PLA), and polytrimethylene terephthalate (PTT).

3.2.2 System boundaries

In this study, environmental assessments were carried out for the systems cradle-to-factory gate and cradle-to-grave. We only present results for the latter because it is comprehensive as it includes waste management. To evaluate the environmental effects of these chemicals, an assessment across the whole life cycle (cradle to grave) of the products is necessary. Five subsystems were distinguished when modelling the life cycle (see Figure 3.1):

1. extraction of non-renewable energy resources such as crude oil,
2. agricultural production and biomass pre-treatment,
3. bio-process (the actual production process),
4. process waste management, and
5. post-consumer waste management.

The use phase was excluded because it is usually identical for comparable bio-based and petrochemical products (e.g. a given plastic component in a passenger car); as a further argument, bulk chemicals usually do not lead to emissions during this phase. A more detailed description of the subsystems can be found in the Supporting Information.

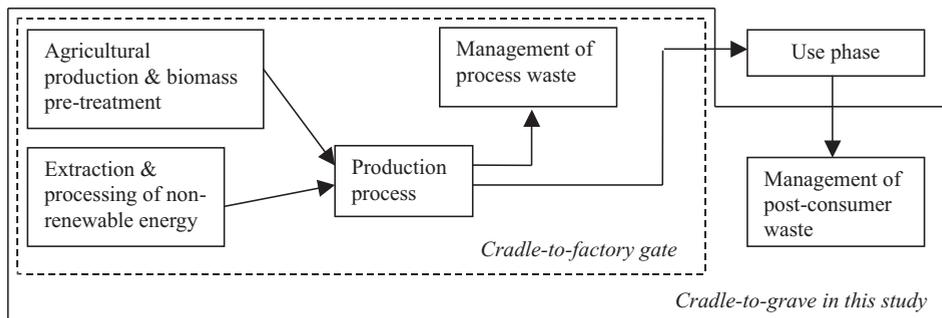


Figure 3.1: Subsystems and system boundaries considered within the environmental assessment for bio-based and petro-based chemicals

3.2.3 Subsystem description

Within the subsystem *agricultural production and biomass pre-treatment* we consider three feedstocks as sources of fermentable sugar: maize starch, lignocellulosics, and sugar cane. Corn was chosen as a representative for starch crops and maize stover as representative for lignocellulosic biomass; sugar cane was included because the world sugar production relies on sugar cane (for details see section 3.3.1). The production of fermentable sugar from sugar cane and starch crops has been operated on large scale for several decades and this technology is widely available and mature. In contrast, production of sugars from lignocellulose still is a technology that is not yet mature or cost-effective (see also Ragauskas et al., 2006).

The subsystem *extraction and processing of non-renewable energy* particularly concerns oil, gas and coal for utilities and as feedstocks. All data related to electricity and heat production draw on conversion factors for steam and electricity production in Europe according to the year 2000. The production of one tonne of steam requires 2.73 GJ_p of primary energy and leads to 170 kg CO₂ emissions. For electricity we assumed a mix of electricity from the grid as well as on-site production by industry using combined heat and power (CHP) systems. The data for grid electricity represents the EU-15 average (IEA, 2003; Loesoenen, 2003) with a generating efficiency of 33% and primary energy for electricity stemming mainly from nuclear (46%), coal (30%), and gas (11%). CHP data was based on average electricity and heat production of the EU chemical industry (Loesoenen, 2003). This results in a weighted average electricity production efficiency of 40% and weighted average CO₂ emissions of 117 kg/GJ_e. All values include the energy requirements for extraction and transportation of the primary resources.

The subsystem *production process* represents the conversion of feedstock to organic chemicals. For WB production this includes inoculum and seed train preparation, fermentation of the bio-based feedstock, filtration and downstream processing. The generic approach modelled this step through mass flow and energy use calculations (see Chapter

2). For petrochemical equivalents and industry data, the inputs to and outputs from the system were readily available.

The subsystem *management of process waste* deals with the treatment of solid biomass and waste-water originating from the bioprocess subsystem. It was assumed that all solid biomass waste (gross calorific value of 19 GJ_{HHV}/t dry matter) generated within the production process is incinerated with energy recovery for steam production with a thermal efficiency of 82% and that all waste-water is treated on-site by a standard process similar to municipal waste-water treatment.

For the subsystem *management of post-consumer waste* we assumed that the WB chemicals are transported to waste-to-energy-facilities after their end of life and that they are incinerated without energy recovery. No energy recovery from incineration means that the energy use is the same for both system boundaries and that greenhouse gas (GHG) emissions for the system cradle-to-grave consist of those for the system cradle-to-factory gate plus the CO₂ emissions from carbon embodied in the product. A sensitivity analysis was carried out for waste treatment consisting of incineration with energy recovery. For chemical intermediates (e.g. lactic acid), post-consumer waste treatment is irrelevant because only end products (e.g. PLA) that reach the consumer and are discarded can be treated in waste management plants. However, waste management calculations were performed for all products studied in order to understand the importance of this stage in the process chain as well as to take the carbon stored in the product into account.

Indirect environmental impacts related to the production of the necessary capital goods were not taken into account because they are not important for the production of energy-intensive bulk products (Boustead, 1999).

3.2.4 Allocation and system expansion

For all multi-functional processes resulting in more than one product, a method must be chosen that allows expressing the environmental impact relative to the functional unit chosen. The two most commonly used methods to this end are system expansion and allocation. Typical system expansion accounts for a co-product by expanding the system analysed to include also the production of the co-product by alternative means. This approach changes the functional unit to one tonne of desired chemical plus amount x of co-product. In order to carry out system expansion but limit the functional unit to one tonne of desired chemical we introduce credits representing the avoided impacts related to the manufacture of co-products (see Weidema, 2000). In general, system expansion is used when the co-product (e.g. electricity from bagasse) can also be produced by a stand-alone process (e.g. electricity from coal). Allocation on the other hand has to be used when there is no such production process, such as maize stover (which can only be produced in conjunction with maize). Allocation is mostly done through mass or economic allocation, i.e. by partitioning the overall environmental impact either according to the mass ratio or the price ratio of co-products.

For this study, allocation and system expansion were relevant for the production of fermentable sugar, the joint production of materials and energy, and the joint production

of co-product chemicals in the fermentation step. We used system expansion whenever energy was produced from co-products within any of the subsystems. Energy credits were assigned to these systems equal to the amount of non-renewable energy that would have been used for the production of the respective amount of electricity or steam. For example, the credit for the energy produced from burning waste biomass originating from the bioprocess resulted in an energy credit. This was deducted from the inputs of non-renewable energy because this energy displaced the use of non-renewable energy, e.g. electricity from the grid. System expansion was also used to deal with co-products of fermentation: all inputs to fermentation were added up and credits were introduced for the co-products (e.g. acetone and ethanol in the case of ABE). These credits were equal to the petrochemical production of these co-products because WB co-products were assumed to displace chemically identical petrochemicals. Allocation was only used in the production of fermentable sugars (see section 3.3.1).

3.2.5 Environmental indicators

A full life cycle assessment (LCA) includes calculating a range of environmental impacts including acidification, eutrophication, particulate emissions, human toxicity, and environmental toxicity. However, the study presented here is a prospective environmental assessment dealing with future processing routes and this study uses proxies for the overall environmental impact: non-renewable energy use (NREU), greenhouse gas emissions (GHG), and land use (LU) which may become scarce resources. We calculated non-renewable energy use, greenhouse gas emissions, and land use for all WB products. These indicators are good proxies for the overall environmental impact.

NREU represents a straightforward and practical approach because many environmental impacts are related to energy use (Huijbregts et al., 2006). NREU encompasses fossil and nuclear energy and was expressed as higher heating value (HHV), also called the gross calorific value. In line with LCA methodology, the NREU values reported here represent the cumulative energy demand for the system cradle-to-grave.

Greenhouse gas emissions are of growing importance due to the increasing attention paid to the greenhouse effect in the policy arena, by companies, and the public. GHG emissions were calculated in CO₂ equivalents and consist of GHG emissions from the system in the form of CO₂ or CH₄ as well as nitrous oxide (N₂O) from fertiliser use in biomass production. CO₂ emissions from renewable carbon extracted from the atmosphere during plant growth were excluded.

Land use refers to agricultural land use only and will be of increasing importance in the future due to the growth of land requirements for bio-based energy, liquid bio-fuels, bio-based chemicals, and food and feed production. We neglected the land requirements for industrial plants, for transportation infrastructure, and for waste management because they are small compared with agricultural land use and are comparable for bio-based and petrochemical products.

3.3 Life Cycle Inventory

Availability of process data for assessing WB processes is often very limited and therefore a generic approach has been developed. A uniform methodology and a common data-set were used to systematically evaluate the environmental aspects of producing bio-based bulk chemicals according to the principles of life cycle assessment (see ISO, 1997-2000). We refer to Chapter 2 for a general description of the generic approach and will focus on specific features of the methodology that are especially relevant to the environmental analysis. Environmental impacts were quantified starting from mass and energy inputs to all sub-systems considered for the system boundaries. Inputs to the bioprocess system include fermentable sugar, auxiliaries, (bio)catalysts, and utilities. Outputs include the product, co-products, waste biomass, and waste-water. For the generic approach, the amount of inputs and outputs of bioprocesses were calculated using mass balances derived from process flow diagrams (see Chapter 2). Each input and output was characterised by its calorific value (HHV in GJ/t product), the cumulative energy use for its production, its embodied carbon, the inputs' cumulative GHG emissions, and the land use required for agricultural production. The Excel database on inputs and outputs included 20 petrochemical feedstocks/intermediates, 7 bio-based feedstocks and intermediates, approximately 30 auxiliaries, 10 catalysts, 10 utilities and fuels, 5 types of waste, its management and energy recovery, and 5 polymers. The data on petrochemicals used as auxiliaries represents current technology and stems from industry (SRI, 2000) and from own calculations based on Patel et al. (1999). These datasets were then used to calculate the cumulative GHG emissions, non-renewable energy, and land use by adding up the respective data of all process inputs.

3.3.1 Production of fermentable sugar

Three types of fermentable sugar were considered for the environmental analysis: glucose from maize starch, sucrose from sugar cane and fermentable sugars from lignocellulosics. A sensitivity analysis on the effect of default and sensitivity cases for fermentable sugars is included in the Supporting Information.

Glucose $C_6H_{12}O_6$ from maize is produced using enzymes to hydrolyse maize starch. The data-set was derived from Vink et al. (unpublished a,b). For agricultural production, the environmental impacts were split between maize and maize stover removed from the field through an economic allocation using a price ratio of 4 : 1 for maize compared to stover. Fermentable sugars from maize starch were assumed to stem from a modern maize wet mill by hydrolysis of starch to dextrose. A detailed mass-based co-product allocation took into account the sub-processes that were required for the co-products of fermentable sugar from maize (see also Vink et al. unpublished b). After accounting for co-products the net input is 1.06 kg of dry maize for 1 kg of glucose. The sensitivity case requires higher non-renewable energy use for maize production (derived from Boustead and Panvalkar, 2001).

Milling of sugar cane results in sucrose ($C_{12}H_{22}O_{11}$) and bagasse. The data-set was calculated from Damen (2001) and Macedo (1998). Bagasse is burnt to generate energy because it is a low-value product with limited use¹. Therefore, we used the total factory input of sugar cane as starting point in the default allocation and assigned credits for energy produced from bagasse. 8.7 kg of wet sugar cane are required to produce 1 kg of sucrose. For a sensitivity case we assumed that bagasse is used as animal feed and we assigned 20% of the overall environmental burden to bagasse based on the ratio of the economic value of sugar to bagasse.

Fermentable sugars from woody biomass, also referred to as C_5/C_6 sugars, are typically a mixture of glucose, xylose and smaller quantities of other sugars obtained through depolymerisation of cellulose and hemicellulose. We assumed the production of C_5/C_6 sugars from maize stover and derived the data-set from Aden et al. (2002). The main purpose of the cultivation of maize has been and will be the production of starch from maize kernels. The default case for allocating inputs for the agricultural production to maize stover is therefore an economic allocation using a price ratio of 4 : 1 for maize compared to stover. The production of 1 tonne of fermentable sugar requires 1.79 tonne of maize stover. The sensitivity case consists of fermentable sugars from miscanthus. We assumed an energy credit approach in which environmental credits from surplus energy production were subtracted from total inputs for the conversion of maize stover to fermentable sugar.

Table 3.1: Summary of background data for 1 tonne of fermentable sugar. Note: Negative values imply net NREU or GHG savings, because the credits for energy production were larger than the NREU or GHG emissions for sugar production.

Sugar Source		NREU GJ/t	REU GJ/t	GHG t CO ₂ /t	LU ha/t
Corn Starch	Default	6.2	17.3	0.40	0.13
	Sensitivity	10.3	17.3	0.63	0.13
Sugar Cane	Default	-12.8	41.8	-0.54	0.13
	Sensitivity	1.4	33.4	0.11	0.10
Lignocellulosics	Default	-4.4	29.2	0.16	0.05
	Sensitivity	-4.0	34.3	0.04	0.13

3.3.2 Assumptions for the fermentation process

When estimating the mass balance of the fermentation process, the flows of all compounds are estimated based on yields, productivity and broth concentration of the fermentation step (see Chapter 2). The three types of fermentable sugar are assumed to be interchangeable as feedstock (1 t glucose = 1 t sucrose = 1 t fermentable sugar from maize stover), implying that the yield for all bioprocesses studied is independent of the type of sugar. However, this is a simplification, because the microorganisms involved in the biotechnological conversion of fermentable sugar to the desired chemical are most often engineered

¹This represents current industrial practice, see Berndt and Hodzic (2007) and is therefore the basis for this attributional LCA.

for a specific feedstock such as glucose.² The direct conversion of mixtures of pentoses and hexoses (C₅/C₆) sugars to target molecules will therefore require additional efforts in R&D before it can be implemented for a wide range of WB products and with the same yields and productivities as from pentoses or hexoses only. We thus assumed that the conversion of C₅/C₆ sugars produced from lignocellulosic feedstocks such as maize stover to desired products will be possible on an industrial scale in the future, but not today.

3.3.3 Industry data for WB products

For some products, it was possible to compare the results obtained from the generic approach with confidential information on industrial WB processes. Such industry data was provided by DuPont for PDO (Alles, 2003), by NatureWorks (Vink, 2005) and Shell (Linton and Nisbet, 2000) for lactic acid, and by A&F for PHA (Dielissen and Weusthuis, 2004). Ethanol, lysine, and succinic acid were derived from process analyses (SRI, 1999b, 2001, 2002a). We used aggregated information on inputs to and outputs from the bioprocess to carry out the environmental assessment, following the same methodology and system boundaries as for generic WB routes.

3.3.4 Petrochemical data for benchmarking

WB chemicals were assessed by comparing them with chemically identical or functionally equivalent petrochemical products. Data on the production of petrochemical equivalents of the WB products analysed represent conventional current technology. Petrochemical data was available for acetic acid, acrylic acid, adipic acid, n-butanol, caprolactam, ethyl acetate (benchmark for ethyl lactate), ethylene, maleic anhydride (benchmark for succinic acid), 1,3-propanediol, high density polyethylene (HDPE; benchmark for PHA), polytrimethylene terephthalate (PTT) and polyethylene terephthalate (PET; benchmark for PLA). The respective data stems from SRI (2000); Patel et al. (1999); Alles (2003); Boustead (1999-2005). We applied the same system boundaries and methodology to carry out the environmental assessment of the petrochemical benchmark data as for the WB routes.

3.4 Results

The environmental impacts of the processes studied depend to a large extent on the productivities, yields and concentrations assumed for the fermentation stage (see Chapter 2). In order to assess the long-term potential of WB, we chose future parameter values that are

²Ethanol has already been produced from C₅/C₆ on laboratory scale (Kuyper et al., 2005), but with today's technology only C₆ sugars can be converted to the target chemicals on an industrial scale. These C₆ sugars are nowadays commonly produced a) from starch crops such as maize and b) from sugar crops such as sugar cane.

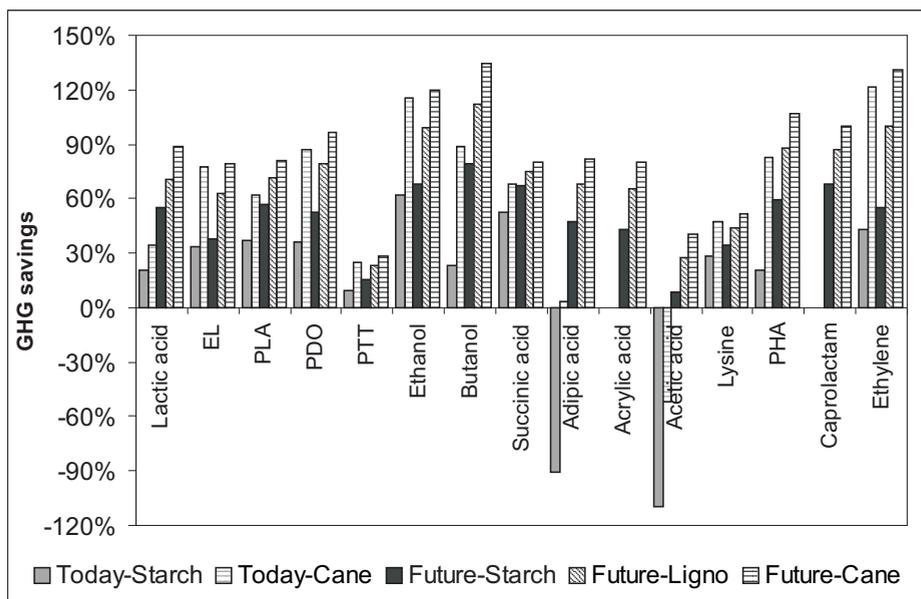


Figure 3.2: Greenhouse gas emission savings per tonne of WB chemical compared with their petrochemical counterparts for current and future technology, system cradle-to-grave

expected to be reached after 20–30 years of successful research and development (horizon values). In this chapter we present results only for GHG emissions, but results for NREU show the same pattern (see Patel et al., 2006). Figure 3.2 shows a product-by-product analysis of average GHG savings in WB products compared with their petrochemical equivalents.

Each bar in Figure 3.2 represents the arithmetic mean across several WB production routes for the same chemical. Figure 3.2 shows that the products with the highest relative savings are ethanol, butanol and ethylene, and acetic acid and PTT have lowest savings. Differences between best cases and arithmetic means were 7%–20% in GHG savings. Figure 3.2 shows that almost all products promise GHG savings for current technology. For PHA and adipic acid this depends on the source of fermentable sugar. Acetic acid offers no savings using current technology because of low broth concentration and low productivity in fermentation as well as high utility use in downstream processing due to the difficulty of separating acetic acid from water (azeotropic mixture). GHG savings for PTT are low because this polymer is made from PDO and purified terephthalic acid, with the latter being produced from petrochemical feedstocks. GHG savings for sugar cane as the source of fermentable sugar are clearly higher than for maizestarch due to the co-production of significant amounts of electricity which can be exported (see section 3.2). Figure 3.2 shows that in order to maximise savings in greenhouse gas emissions for WB products sugar cane is favoured over lignocellulosics, which in turn is preferable to maizestarch as source of fermentable sugar. In temperate climates such as Europe and

North America where sugar cane is not available from domestic production, lignocellulose should be the preferred future feedstock.

In some of the WB lignocellulose and sugar cane cases the savings are larger than 100% (Figure 3.2) because the energy credits from co-combustion of waste biomass or from side-streams of agricultural production were larger than the NREU for the WB process chain (see section 3.2). On average, GHG savings for future WB technology are 25%–35% higher than for current WB technology. This shows that technological progress can further enhance the environmental advantage of WB products over their petrochemical equivalents. Low GHG emissions of a process may be due to 1) high product yield from fermentation or 2) low product yield from fermentation combined with large energy credits from subsequent combustion of co-produced biomass. In the second case, the inefficient fermentation processes require considerably more land for biomass production than efficient fermentation does. If land availability becomes limited, GHG savings should be maximised for a given amount of land or, alternatively, land use should be minimised for a certain amount of GHG to be saved. Figure 3.3 therefore simultaneously analyses GHG savings in all WB-routes relative to the petrochemical route and land use per tonne of chemical.

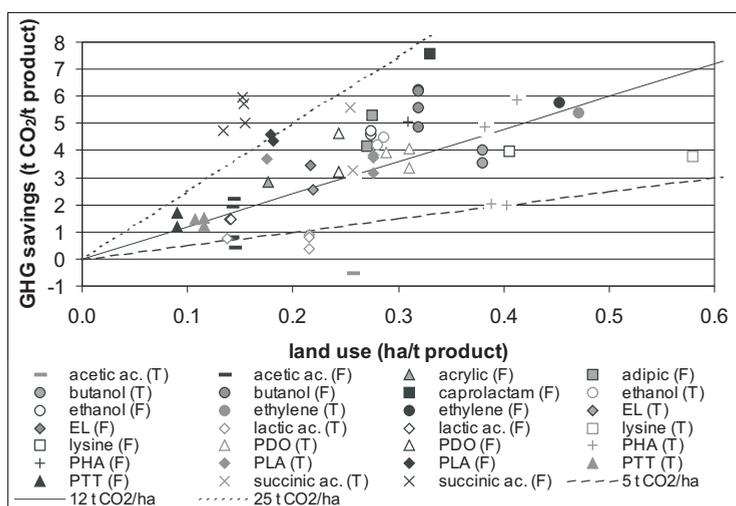


Figure 3.3: Fermentable sugar from sugar cane — Greenhouse gas (GHG) savings versus land use for current (T) and future (F) technology WB production routes per tonne of chemical (system cradle-to-grave). Straight lines are iso-lines representing CO₂ savings per hectare.

Figure 3.3 shows that there is a relationship between the type of chemical and the amount of land use for its production from sugar cane. For the production of one tonne carboxylic acids 0.1–0.2 ha land are required, whereas the alcohols are in the range of 0.25–0.35 ha/t. For PTT, land use and GHG savings are low because only a part of this polymer is produced from bio-based feedstocks. Putting GHG savings and low land-use first, succinic acid, caprolactam, PLA and butanol are the most attractive. Our results

compare well with data from publications on individual products (Kim and Dale, 2005a; Sheehan et al., 2003; Vink et al., 2003; Bohlmann, 2004; Kim and Dale, 2005c; Akiyama et al., 2003) when accounting for differences in allocation. Producing fuel ethanol from sugar cane results in savings of 10–16 t CO₂-eq/ha (Patel et al., 2006). Several WB chemicals show CO₂ savings per hectare above 16 t CO₂-eq and are therefore preferable from the point of view of CO₂ mitigation.

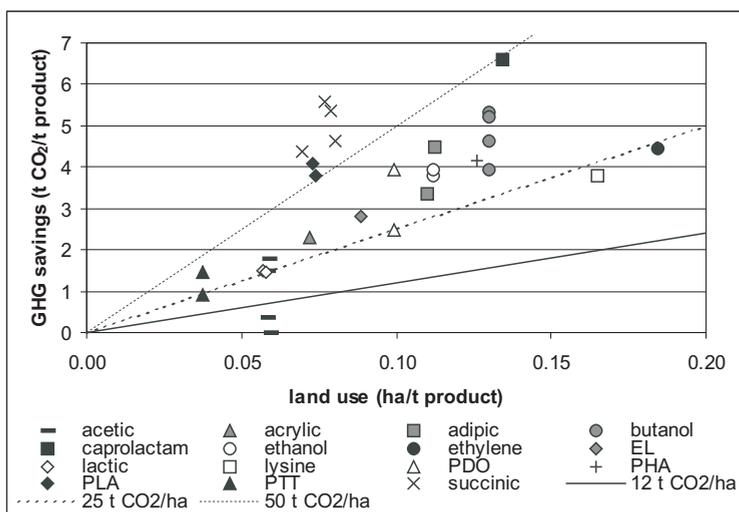


Figure 3.4: Fermentable sugar from maize stover (lignocellulosics): Greenhouse gas (GHG) savings versus land use for future technology WB production routes (system cradle-to-grave). Straight lines are iso-lines representing CO₂ savings per hectare.

Figure 3.4 only shows future technology because the commercial production of fermentable sugars from lignocellulosics is not yet possible on a large scale. The comparison of Figures 2 and 3 shows that land-use efficiency in terms of CO₂ savings per hectare is much better for maize stover than for sugar cane. Converting maize stover to chemicals using future technology almost always results in CO₂ savings above 25 t/ha. Biomass for electricity use saves approximately 12 t CO₂-eq/ha for whole crop wheat (Dornburg et al., 2005) and using lignocellulosics for fuel ethanol production saves 2–7 t CO₂-eq/ha (Quirin et al., 2004)³. Putting CO₂ savings first, this implies that most chemicals are preferred over bioenergy if using sugar cane as feedstock and almost all chemicals are preferred if using maize stover.

Table 3.2 quantifies GHG savings potential assuming full substitution of the petrochemical equivalents and based on world production capacities in the years 1999/2000 (Weissermel and Arpe, 2003). The total saving potential for the future according to Table 3.2 (510 million tonnes CO₂-eq for maize starch) disregards growth of the chem-

³GHG savings from ethanol here are ca. 33 t CO₂-eq/ha because of the reference system: chemical ethanol replaces ethanol produced via ethylene, whereas fuel ethanol replaces gasoline.

Table 3.2: Potential worldwide annual production and best case GHG savings of the 15 WB products, using maize starch as feedstock, system cradle to grave; installed capacity according to Weissermel and Arpe (2003)

Product		GHG savings (t CO ₂ /t)	Installed world capacity (kt/yr)	Annual GHG savings (kt CO ₂ /yr)
Acetic acid	Today	-2.4	8,300	N/A
	Future	1.2		9,570
Acrylic acid	Future	1.5	2,900	4,380
Adipic acid	Today	-5.2	2,400	N/A
	Future	3.3		7,880
Butanol	Today	1.2	2,460	3,040
	Future	3.9		9,610
Caprolactam	Future	5.2	3,900	20,100
Ethanol ⁴	Today	2.7	2,600	6,970
	Future	2.7		7,080
Ethyl lactate	Today	1.3	1,200	1,580
	Future	1.9	Ethyl acetate	2,220
Ethylene	Today	1.9	100,000	191,050
	Future	2.5		245,710
Lysine	Today	2.1	640	1,370
	Future	3.6		2,280
Succinic acid	Today	4.5	1,350	6,070
	Future	5.0	Maleic anhydride	6,780
1,3-Propanediol	Today	1.8	No data	N/A
	Future	2.9		N/A
Polyhydroxy-alkanoates	Today	2.9	57,000	162,730
	Future	2.8	Polyethylene	159,640
Polylactic acid	Today	2.3	11,100	25,150
	Future	3.3	PET	36,500
Total	Today			397,960
	Future			511,780

ical industry. The future saving potential is even higher if lignocellulosics (820 million tonnes CO₂-eq) or sugar cane (1,030 million tonnes CO₂-eq) are used as feedstock. For comparison: GHG emissions from current technology production of the petrochemical equivalents lead to 880 million tonnes CO₂-eq for the same installed capacity and system boundaries. This shows that the potential GHG savings for current technology and maize starch as feedstock already reach 45%.

3.5 Comparison of results

Production of ethanol from maize starch results in CO₂-eq emissions in the range of 0.3–0.9 t/t according to Kim and Dale (2005a). Allocation choices (system expansion vs.

mass allocation) lead to the difference with our results (-0.7–0.2 t CO₂-eq/t). Sheehan et al. (2003) provides a “conservative view of the life-cycle impacts of stover-derived ethanol”, due to the assumption that an acre of maize and an acre of soybeans are functional equivalents. Not surprisingly, their values are higher (0.1 t CO₂-eq/t) than those from the generic approach (-2– -1.7 t CO₂-eq/t) for maize stover for current technology.

Production of PHA using glucose from maize starch for the system cradle-to-factory gate results in emissions of 0.5 and 1.4 t CO₂-eq per tonne product according to Akiyama et al. (2003) and 1.6–4.1 t CO₂-eq/t according to Kim and Dale (2005c). The former only deals with one type, the latter does not specify types of downstream processing. This study covers a broad range of CO₂ emissions (-0.7–4.0 t CO₂-eq per tonne product for current technology) because of large differences in downstream processing.

Production of polylactic acid (PLA) from maize starch results in CO₂ emissions in the range of 1.8 t CO₂-eq/t according to Vink et al. (2003) for the system cradle-to-factory gate and 0.7–2.3 t CO₂-eq/t according to Bohlmann (2004) for the system cradle-to-grave. For the system cradle-to-grave, the values from the generic approach are higher (3.0–4.2 t CO₂-eq/t) because Bohlmann (2004) assumes that the carbon stored in the product is not released for the lower value of 0.7 and that for the higher value of 2.3 energy is recovered from landfill methane to generate electricity. Comparing this upper value with generic approach data for cradle-to-grave including energy recovery results in CO₂ emissions in the range of 2.8–3.7 t CO₂-eq/t, which are comparable. Our results compare well for the system cradle-to-factory gate: 1.2–2.4 t CO₂-eq/t.

3.6 Sensitivity analysis and discussion

To quantify the effect of improved biotechnology, data on production of electricity, fertiliser, and biomass yields in agriculture were kept constant throughout the analysis. However, it is likely that these technologies will also improve in the future. Including technological progress in petrochemical processes would lead to smaller benefits by WB. Up to 20% savings in process energy are possible for future petrochemicals production (Ren et al., 2006); however, in the case of ethylene, this translates to only 5% lower NREU of the petrochemical and only 0.1% lower NREU savings of bio-based ethylene. The overall evaluation and conclusions would therefore remain unchanged.

The type of allocation in the production of fermentable sugars was found to have little influence on the environmental results for chemicals from maize starch and lignocellulose but a large effect on the results for chemicals from sugar cane: the results for NREU were higher for the sensitivity case of sugar cane (economic allocation, see methodology) than those for maize stover. In this case, lignocellulosics would be preferred not only to maize starch but also to sugar cane as source of fermentable sugar.

All calculations for the system cradle-to-grave were carried out for incineration without energy recovery as the type of post-consumer waste treatment. We compared this default case to incineration with energy recovery. For energy recovery, environmental

credits were introduced in order to account for the avoided production of heat and power, based on estimated average conversion efficiencies for energy recovery in European incineration plants of 12% power and 12% heat (Phylipsen et al., 2002). Compared to the base case, incineration with energy recovery resulted in GHG savings that were 2%–15% higher for current technology and 10%–20% higher for future technology when also taking energy recovery for the equivalent petrochemicals into account. Other post-consumer waste treatment types (e.g. digestion of biodegradable products) were also analysed (Patel et al., 2006) but do not yield different conclusions.

Several environmental and health effects were excluded but are potentially important: allergic effects to humans, the large-scale use of process water for current technology, possible biodiversity loss in the natural environment in the case of large-scale biomass production, and environmental risks from the use of genetically modified organisms in the bioprocess step.

3.7 Conclusion

In summary, even at present, bio-based bulk chemicals from white biotechnology offer clear savings in non-renewable energy use and GHG emissions with current technology compared to conventional petrochemical production. Substantial further savings are possible for the future by improved fermentation and downstream processing. Of all feedstocks, sugar cane is to be favoured over lignocellulosics, which in turn is preferable to maizestarch as source of fermentable sugar in order to maximise savings. The products with the highest savings are acrylic acid, butanol (from ABE process), ethanol, ethylene, PDO, and PHA.

From a policy perspective, environmental advantages make the production of bio-based bulk chemicals using industrial biotechnology desirable on a large-scale, because savings of more than 100% in non-renewable energy use and greenhouse gas emissions are already possible at the current level of biotechnology. This builds a strong case for the production of bio-based bulk chemicals using white biotechnology considering the economic advantages of 1,3-propanediol, polytrimethylene terephthalate, succinic acid and ethanol for current technology and of all products except acetic acid for future technology (see Chapter 2). As a consequence, using industrial biotechnology to produce bio-based chemicals can contribute significantly to reducing climate change and the depletion of fossil energy. It is therefore a key strategy for sustainable development of the chemical industry.

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Appendix A: Detailed calculation results

Table 3.3 shows the results for NREU, Table 3.4 for GHG emissions and Table 3.5 for land use, not only for current and future technology in the generic approach but also for WB industry data and petrochemical benchmark data. Ranges specify different types of fermentation (batch or continuous) and/or downstream processing and show the whole range of calculated values. The petrochemical compounds refer to chemicals that were identical to the bio-based chemicals, unless indicated otherwise in that column. In the cases of lactic acid and lysine no petrochemical benchmarks can be given and they were compared with current industrial practice based on continuous fermentation and filtration for lactic acid and batch fermentation and ion exchange for lysine. Comparing WB products to their petrochemical equivalents shows that NREU savings are possible already for current technology for all products studied except for adipic acid and acetic acid irrespective of the source of fermentable sugar (see Table 3.3). Comparing current technology to industry data for those products where both were available, we conclude that the results from the generic approach are in line with industry data for ethanol, succinic acid and PDO/PTT.

Table 3.3: *Non-renewable energy use (NREU) of WB products based on current and future technology in the generic approach as well as industry data and petrochemical equivalents, calculated for three sources of fermentable sugar and the system cradle-to-grave; petrochemical equivalents are identical petrochemicals unless otherwise stated.*

Product		Non-renewable energy use (GJ _p /t)			
		Corn starch	Ligno-cellulosics	Sugar cane	Petro-chemical
Acetic acid	Today	109–145	–	71–106	56
	Future	39–65	27–53	18–43	
Acrylic acid	Future	31	16	4	47
Adipic acid	Today	195	–	86	86
	Future	44–60	21–37	3–19	
Butanol	Today	57–64	–	1–7	69
	Future	7–29	-20–3	-41–(-18)	n-butanol
Ethanol	Industry	27	–	-14	54
	Today	19–24	–	-23–(-19)	
	Future	18–20	-5–(-2)	-23–(-21)	
Lactic acid	Industry	28–37	–	7–16	–
	Future	20–23	8–11	-1–2	
Lysine ⁵	Industry	66–189	–	5–137	–
	Today	169	–	83	
	Future	131	97	71	
PDO	Industry	41–46	–	-2–14	69
	Today	38–53	–	-9–7	
	Future	20–43	0–23	-17–7	
PHA	Industry	43	–	-14	77
	Today	38–112	–	-24–52	HDPE
	Future	33	8	-13	
Succinic acid	Industry	46	–	27	68
	Today	27–67	–	5–45	Maleic
	Future	28–47	18–38	9–31	anhydride
Ethylene	Today	40	–	-30	66
	Future	31	-6	-36	
Ethyl lactate	Today	41	–	7	59
	Future	36–43	18–34	4–10	Ethyl acetate
Caprolactam	Future	43	16	-5	117
PLA	Industry	49–61	–	21–33	77
	Future	40–41	25–26	13–14	PET
PTT	Industry	63–68	–	47–53	74
	Today	62–68	–	45–50	
	Future	54–64	47–56	42–50	

⁵Lysine can be produced and sold as salt containing different ions and in varying degrees of purity. Energy use, emissions and land use are therefore standardized to 1 tonne lysine content.

Table 3.4: Greenhouse gas emissions (GHG) of WB products based on current and future technology in the generic approach as well as industry data and petrochemical equivalents, calculated for three sources of fermentable sugar and the system cradle-to-grave; petrochemical equivalents are identical petrochemicals unless otherwise stated.

Product		Greenhouse gas emissions (t CO ₂ -eq/t)			
		Corn starch	Ligno-cellulosics	Sugar cane	Petro-chemical
Acetic acid	Today	5.7–8.1	–	3.8–6.2	3.3
	Future	2.1–3.9	1.5–3.3	1.1–2.9	
Acrylic acid	Future	2.0	1.2	0.7	3.5
Adipic acid	Today	11.0	–	5.6	5.8
	Future	2.5–3.6	1.3–2.4	0.5–1.6	
Butanol	Today	3.0–3.5	–	0.2–0.7	4.3
	Future	0.4–1.8	-1.0–0.3	-2.0–(-0.6)	n-butanol
Ethanol	Industry	1.7	–	-0.3	3.2
	Today	1.2–1.5	–	-0.8–(-0.6)	
	Future	1.2–1.3	0–0.1	-0.9–(-0.7)	
Lactic acid	Industry	1.8–2.7	–	0.8–1.6	–
	Future	1.2	0.6	0.2	
Lysine ⁵	Industry	3.7–10.3	–	0.7–7.7	–
	Today	8.2	–	3.9	
	Future	6.7	4.9	3.7	
PDO	Industry	2.3–2.9	–	0.2–1.2	4.1
	Today	2.3–3.0	–	0–0.7	
	Future	1.2–2.6	0.1–1.6	-0.6–0.9	
PHA	Industry	4.6	–	2.2	4.7
	Today	1.9–6.4	–	-1.1–3.4	HDPE
	Future	1.8	0.5	-0.5	
Succinic acid	Industry	2.8	–	1.8	6.8
	Today	2.3–4.6	–	1.3–3.6	Maleic
	Future	1.8–2.9	1.3–2.4	0.9–2.1	anhydride
Ethylene	Today	3.1	–	-0.4	4.4
	Future	2.0	0.0	-1.4	
Ethyl lactate	Today	2.7	–	1.0	3.9
	Future	2.0–3.0	1.1–2.0	0.4–1.5	Ethyl acetate
Caprolactam	Future	2.4	1.0	0.0	7.6
PLA	Industry	3.0–4.2	–	1.7–2.8	5.5
	Future	2.2–2.5	1.4–1.7	0.9–1.2	PET
PTT	Industry	4.5–4.9	–	3.7–4.2	5.2
	Today	4.5–4.8	–	3.7–3.9	
	Future	4.1–4.7	3.7–4.3	3.4–4.0	

Table 3.5: Land use of WB products based on current and future technology in the generic approach as well as industry, calculated for three sources of fermentable sugar

Product		Land use (ha/t)		
		Corn starch	Lignocellulosics	Sugar cane
Acetic acid	Today	0.26	–	0.26
	Future	0.14	0.06	0.15
Acrylic acid	Future	0.18	0.07	0.18
Adipic acid	Today	0.74	–	0.75
	Future	0.27–0.28	0.11	0.28
Butanol	Today	0.38	–	0.39
	Future	0.32	0.13	0.33
Ethanol	Industry	0.28	–	0.29
	Today	0.28–0.29	–	0.29
	Future	0.27	0.11	0.28
Lactic acid	Industry	0.14–0.22	–	0.14–0.22
	Future	0.14	0.07	0.14
Lysine ⁵	Industry	0.36–0.41	–	0.37–0.42
	Today	0.58	–	0.59
	Future	0.41	0.17	0.41
PDO	Industry	0.22–0.29	–	0.22–0.29
	Today	0.31	–	0.32
	Future	0.24	0.10	0.25
PHA	Industry	0.38	–	0.39
	Today	0.40	–	0.41
	Future	0.31	0.13	0.32
Succinic acid	Industry	0.15	–	0.16
	Today	0.25–0.26	–	0.26
	Future	0.13–0.15	0.07–0.08	0.14–0.16
Ethylene	Today	0.47	–	0.48
	Future	0.45	0.18	0.46
Ethyl lactate	Today	0.22	–	0.22
	Future	0.22–0.27	0.09–0.14	0.22–0.28
Caprolactam	Future	0.33	0.13	0.34
PLA	Industry	0.18–0.28	–	0.18–0.28
	Future	0.18	0.07	0.18
PTT	Industry	0.08–0.11	–	0.08–0.11
	Today	0.12	–	0.12
	Future	0.09	0.04	0.09

4

Twisting biomaterials around your little finger: environmental impacts of bio-based wrappings*

Abstract

Background, aim, and scope Packaging uses nearly 40% of all polymers, a substantial share of which is used for sensitive merchandise such as moisture-sensitive food. To find out if bio-based materials are environmentally advantageous for this demanding application, we compared laminated, printed film across the whole life cycle.

Materials and methods We compared bio-based materials (paper, polylactic acid, bio-based polyethylene, and a bio-based polyester) as well as conventional ones (polypropylene, polyethylene). Data stemmed from 13 companies that produce raw materials, films and/or laminates and which co-operated with us in a project commissioned by a large food producer. The functional unit chosen for this study is 1 m² of packaging film. This is (mostly) laminated, printed film that is delivered on reels to the food industry, where the laminate is cut, sealed and filled. The impact assessment is presented for non-renewable energy use, total energy use, global warming potential, depletion of abiotic resources, photo-oxidant formation, acidification, eutrophication, water use, and land use.

Results For Inner Packs that get in direct contact with food and therefore require certain barrier properties, the environmental performance of many laminates is not better than the reference, petrochemical material. However, our study shows that paper/polypropylene laminates perform equally well as the current material (polypropylene) if the material is landfilled, and better if incinerated with energy recovery. For Outer Packs, bio-based polyethylene film shows a particularly low environmental impact. Paper/bio-based polyester laminates also offer significant savings compared with the current material. For Inner as well as Outer Packs, laminates including polylactic acid offer environmental advantages when accounting for wind credits or when assuming a future technology level for polymer

*With kind permission from Springer Science & Business Media: International Journal of LCA (2010) 15(4), 346–358. Co-authors: Kornelis Blok and Martin Patel (both Utrecht University)

or film production.

Discussion Increased technology maturity of PLA and cellulose in the film production stage offers significant environmental improvement with respect to global warming potential compared with today's technology. Though large, the uncertainty regarding the degree of degradation of paper, cellulose, PLA and bio-based polyester, is not decisive for the conclusions.

Conclusions and recommendations Generally, laminates and films (partly) consisting of bio-based polymers offer opportunities for significantly reducing environmental impacts of food packaging. Large variations in land-use are possible depending on the type of bio-based material that is used. The environmental advantages differ depending on the polymer and the final product (Inner vs. Outer Pack). Lack of experience and investment in converting bio-based polymers into final products and comparatively unfavourable material properties result in lower environmental advantages for some novel bio-based materials than one may expect. However, a) already today, the options with the lowest global warming potential are partly or fully bio-based and b) bio-based materials will benefit more from technological progress than conventional materials, potentially making certain bio-based laminates highly attractive options for the future. Overall, Outer Packs are more promising than Inner Packs when introducing bio-based wrappings to replace the current petrochemical material because a) the opportunities are clearer for this application and b) the product specifications (required barrier properties) are less demanding. Starting with the Outer Packs would also allow bio-based polymer producers and processors to invest and learn, thus offering the opportunity to reduce the environmental impact even further.

4.1 Introduction

The first man-made polymers were derived from biomass resources, but since the 1930s petrochemical polymers have gradually displaced them during the growth of the petrochemical industry. Since the 1980s and especially during the 1990s, bio-based polymers have experienced a comeback, with the main drivers being the limited volume of landfill capacity and the bad general image of plastics and other packaging materials as well as the high oil price, the rapid progress in biomass-based processes (e.g. white biotechnology) and the outstandingly good public perception of bio-based polymers (Käb et al., 2002; Patel et al., 2006). In the EU 25 plus Norway and Switzerland, 37% of all plastics or 18.3 million tonnes went into packaging in 2006 (PlasticsEurope, 2008), thus indicating this sector's importance. It is a market that has been dominated by petrochemical plastics, most notably by polyethylene (PE) and polypropylene (PP). However, in light of the comeback of bio-based polymers in general, the packaging market and the producers of food and consumables are increasingly discovering the opportunities of bio-based packaging materials. Their use can offer new waste management strategies for packaging, e.g. through compostable wrappings that may reduce the pressure of household waste on landfills¹ and may improve public perception of the product.

¹In the EU 25, two-thirds of all municipal waste is still being landfilled (Eurostat, 2008).

Key bio-based materials with potential applications in the packaging sector are paper, starch, cellulose, polylactic acid (PLA) and other bio-based polyesters. Some of these materials have been used and produced for a long time (e.g. paper), others only for a short time on industrial scales (e.g. PLA). In the context of long-term emission targets, one of the important issues is to what extent bio-based materials score better in environmental terms compared to petrochemical polymers. Several studies have already been carried out regarding the environmental advantages or disadvantages of using novel biopolymers such as PLA (Vink et al., 2003, 2007), bio-based polyethylene (Chapter 3) or polyhydroxyalkanoates (Kim and Dale, 2005c) concluding that there are environmental advantages for some bio-based materials and that other materials could become advantageous if technologies used in their production improve and progress. So far, these studies have focussed on comparing quantities of materials (e.g. 1 kg of PLA with 1 kg of conventional plastics). This study goes one step further and considers the functionality of these materials for a specific application: we focus on the production of a film or laminate for snack food packaging as a case study. No public literature is yet available on an environmental assessment of this specific application.

Some of the bio-based materials are known to have lower barrier properties for water and oxygen, and may therefore require thicker material layers or additional layers supplying barrier functions when used for packaging purposes. But do extra material requirements and less experience in processing these materials outweigh the environmental advantages of using bio-based materials in packaging and if so, to what extent? Which combination of materials, consisting at least partly of bio-based materials, can substitute synthetic polymers to a considerable degree in the short term future? To answer these questions, we carried out a life cycle assessment (LCA) in co-operation with industrial film producers and suppliers.

4.2 Methodology

4.2.1 Functional unit and system boundaries

The functional unit chosen for this study is 1 m² of packaging film. This is (mostly) laminated, printed film that is delivered on reels to the food industry, where the laminate is cut, sealed and filled. Cutting of the sheets and sealing and filling of the bags are excluded from the analysis because across all packs of the same size and function, this step can be assumed to be identical, and to produce the same level of waste.

The results of the environmental assessment are reported both for the system ‘cradle-to-factory gate’ (CF) and for the system ‘cradle-to-grave’ (CG): The system CF includes all activities in the process chain starting from the extraction & processing of non-renewable resources (e.g. oil and gas) or agricultural & silvicultural production (e.g. maize from seeds, including fertiliser and machinery use) up to and including film production, lamination and printing; it also covers all transportation activities and treatment of any

process waste up and until the laminated film is delivered on reels to the food producer. The system CG includes the system CF plus waste management of the post-consumer packaging waste² where all key options are studied, i.e. incineration (with and without energy recovery), landfilling, composting, and digestion. We calculate GWP for the system cradle-to-factory gate for all bio-based products by adding all emissions of fossil greenhouse gas emissions and subtracting the biogenic carbon that is physically embedded in the product. As a consequence, both fossil and biogenic emissions of greenhouse gases from the waste treatment stages are considered.

4.2.2 System expansion

In this study, we applied system expansion (also referred to as ‘avoided burdens’) to account for the co-generation of electricity and heat. To determine the credit, electricity that is co-produced, e.g. during the incineration of waste, is assumed to replace electricity produced according to the average power generation in Europe (see section 4.2.3) and heat is assumed to replace average production in a gas-fired boiler.

4.2.3 Environmental impact assessment

In this study, we used the CML 2 baseline 2000 method (Guinée et al., 2001) for calculating the mid-point results, adding water use and land use as impact categories. For CF, the so-called LCA mid-point results are presented for the following impact categories: Non-renewable energy use, Total energy use (total of non-renewable and renewable energy use), Global warming potential, Depletion of abiotic resources, Photo-oxidant formation, Acidification, Eutrophication, Water use, and Land use³; for CG, only results for the categories non-renewable energy use and global warming potential are shown because of the large uncertainties related to estimating individual process emissions other than CO₂, CH₄ and water during the waste treatment phase, especially of novel materials such as polylactic acid (PLA).

So far, water use as an impact category has not been very common in LCA studies but is receiving more and more attention (Mila i Canals et al., 2009; Pfister et al., 2009). Within the category of water use, we consider process water, cooling water and irrigation water. Process water includes all the water used during the production of raw materials, films and laminates but excludes the water used for electricity production from hydropower. The subcategory cooling water includes water used for cooling in any of the process steps. Irrigation water means water fed to the agricultural system during crop growth; it excludes rainfall because there is no generally accepted methodology

²The use phase is not taken into account because the direct impacts of packaging in the use phase are negligible (no significant release of compounds from the packaging to the environment in retail and in households).

³We exclude results for toxicity primarily because of limitations of available data. Other reasons are doubts about the quality of toxicity calculations in LCA assessment tools, caused by the lack of reliable toxicity assessment models (see Dreyer et al., 2003; Guinée et al., 2001; Gustafsson and Börjesson, 2007). We exclude results for stratospheric ozone depletion, which is not relevant any more since the phasing out of chlorofluorocarbons as a result of the Montreal Protocol.

that would allow a consistent comparison of rainfall quantities across agricultural as well as silvicultural crops and across geographical regions of production. (For example, the FAO's CROPWAT model only applies to agricultural crops.) Energy consumption for irrigation is excluded because it only contributes to a small extent to the total energy consumption of agricultural crops (Mila i Canals, 2003).

Land for agriculture and forestry will be increasingly important in the future because of increasing land requirements not only for the production of food and feed but also of bio-energy, bio-fuels, and biomaterials. A growing number of environmental assessments of biomaterials include land use in their analyses (see Dornburg et al., 2003; Kim and Dale, 2005b, Chapter 2) and methodological work is under way (see e.g. Jolliet et al., 2003; Mila i Canals et al., 2007; Kløverpris et al., 2008) but has not yet established one single accepted methodology in terms of how different types of land use should be compared. In this study, we focus on land use for agriculture and for (sustainable) forestry. The different types of land use (e.g. agriculture and forestry) are aggregated 1:1. The rotation period of forestry is taken into account. Land use for industrial plants, transportation infrastructure and waste management is comparable for different types of material and thus not taken into account.

4.2.4 How to incorporate green electricity?

As can be observed among private consumers, companies are also increasingly purchasing 'green electricity' in order to reduce their environmental footprint; one such company is NatureWorks, the most important producer of PLA. Purchasing green electricity is an option for any producer – and this raises the question if such environmental credits should be considered when conducting an LCA. In order to describe the different views, we distinguish between the company perspective and the technology perspective. An LCA carried out from a company perspective includes not only the processes operated by the company itself but also the purchasing decisions the company makes regarding materials and energy inputs. As a consequence, a study using the company perspective gives credits for the avoided environmental burden related to green electricity. We argue that the company perspective should be applied for decisions concerning business relations. A comparison from a technology perspective of materials production and processing strives to eliminate the effect of whether the company uses power generated by using wind energy, natural gas or coal, focussing instead on the core technology⁴. The technology perspective is the adequate choice when making decisions concerning material choice or R&D strategies. In this paper, we focus on the technology perspective because we are mainly interested in material choice; but we include a case of company perspective to show the difference between the two.

In the following, we describe the effect of a company's decision to purchase 'green electricity', using the example of wind energy. The accounting practice adopted by the International Energy Agency (IEA, 2003) for their energy balance tables is that the primary

⁴Here, core technology refers to the production of materials and their subsequent processing to produce packaging film, but excludes the generation of green power as an optimisation strategy.

energy form should be the first energy form downstream in the production process for which multiple energy uses are practical. The application of this principle leads to the following primary energy forms: 1) Heat for nuclear electricity, for geothermal heat or electricity and for solar heat production 2) Electricity for hydro, wind, wave/ocean and photovoltaic electricity production. This convention has been agreed upon internationally for energy balances. As there is no such agreement for LCAs in general, we follow the IEA convention. As a consequence, the primary energy consumption related to the production of 1 kWh of electricity is lower if it is generated from hydropower, wind, wave/ocean and/or photovoltaics compared to its generation from fossil resources (oil, gas, coal) or nuclear, geothermal or solar heat.

PLA is the only material in this LCA for which renewable energy credits were bought (starting in 2006). As shown in Table 4.1, moving towards wind energy in 2006 meant replacing 23 MJ/kg PLA of primary fossil energy by 6 MJ/kg PLA primary renewable energy. The total primary energy use decreased from 77.3 MJ/kg PLA to 60.3 MJ/kg PLA without any change in technology or secondary energy use and therefore solely as a consequence of the accounting practice for wind energy. The wind credits also have a large influence on impact categories other than energy use, most notably global warming potential, abiotic depletion and acidification.

Table 4.1: Renewable energy use (REU) and non-renewable energy use (NREU) for the production of 1 kg of polylactic acid (PLA) with and without wind energy credits derived from Vink et al. (2007)

	perspective	PLA ₂₀₀₅	PLA ₂₀₀₆
	wind credits	technology	company
		no	yes
NREU	MJ/kg	52.0	29.0
REU	MJ/kg	25.3	31.3
Total	MJ/kg	77.3	60.3

4.3 The laminates and their production

4.3.1 Laminates studied

This study got the producers of the raw materials and semi-finished products as well as the producers of the packs in their final form (printed laminates) involved for the first time in order to provide data and review the results⁵. The films and laminates that are considered in this study were selected in collaboration with a multinational food producer and its film suppliers and converters. Criteria for selection were 1) that films and laminates consisted

⁵The names of the companies involved are not disclosed and only generic names are used for the various components of laminates.

at least in part of bio-based materials and 2) that they were proven or expected to have comparable barrier properties as the currently used materials.

In total, 32 alternatives were studied; of these 32 options, 17 represent ‘Inner Packs’ and 15 are ‘Outer Packs’. Inner Packs are in direct contact with the food and need to provide water and oxygen barriers. The targets are $< 0.3 \text{ gm}^{-2} \text{ day}^{-1}$ moisture vapour transmission rate (MVTR) and $< 30 \text{ cm}^3 \text{ m}^{-2} \text{ day}^{-1}$ oxygen transmission rate (OTR) respectively⁶. Outer Packs serve as containers (bags) for the Inner Packs, therefore have no direct food contact and require no barrier function. The options studied are shown in Table 4.2 and the distribution of weight across the different types of materials in Figure 4.1. The majority of the films and laminates are at various stages of development, from conceptual development to small-scale test production and are compared with the currently used reference material.

Throughout the text we distinguish between films, which consist of one material (e.g. only PP), and laminates, which consist of multiple layers of materials (e.g. PP and paper). In order to make best use of the properties of the various materials, most of the packaging films are multi-material laminates. We consider laminates consisting practically only of bio-based materials, only of petrochemical materials or hybrid films consisting of both petrochemical and bio-based materials. As a consequence, only some of the novel material composites studied are biodegradable (marked with an asterisk in Table 4.2). The influence of the metallized layer (aluminium, aluminium oxide or silicon oxide) is minimal in composting: the weight of such a layer is usually much less than 1%, thereby fulfilling requirements of composting certification (EN 13432) and there are several metallized biodegradable films on the market that are certified compostable.

4.3.2 Data sources

The companies involved were supplied with surveys on the inputs and outputs regarding that part of the production process which takes place at their respective production sites. The data they provided was on the level of process inputs (materials and energy) and outputs (materials, waste and emissions). We then went on to perform a plausibility check and to benchmark the received data against other company data as well as database data (where available). When large discrepancies were found, feedback was provided to the company. In some instances this led to a correction of the material inputs or outputs, but in other cases there was a technical explanation for certain differences. This resulted in a consistent data-set which was then incorporated in our calculations. At the end of the project and before submitting the final results, all contributors to the data-sets received a copy of the preliminary report and an opportunity to give feedback.

It is important to note that despite all these efforts, there can still be substantial differences between individual production sites, which are caused by differences in the quality of data or the level of technology. In the first case, data quality varied because only some of the suppliers had individual meters installed to measure energy consumption by

⁶Testing carried out subsequent to this analysis has shown that films 3b and 3c do not fulfill the required barrier properties.

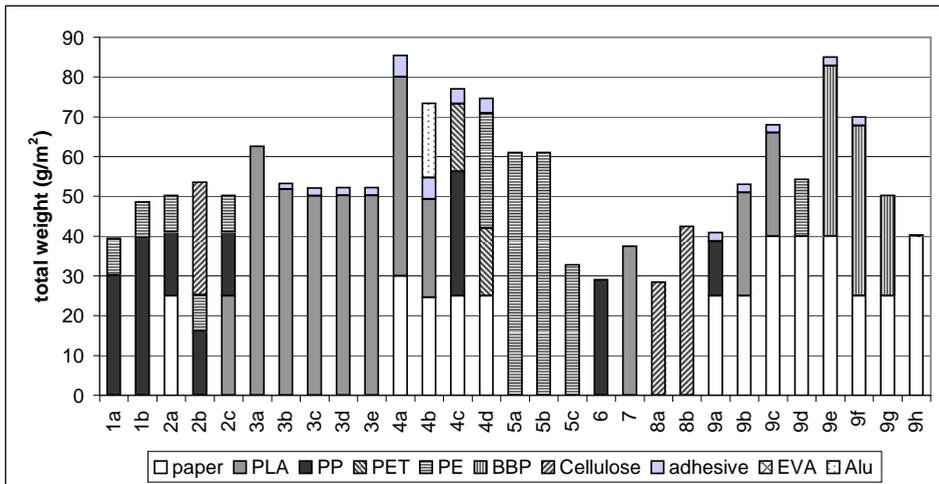


Figure 4.1: Total weight of films and laminates (inner and outer Packs), compare list of abbreviations

individual process steps of the entire production process. Most suppliers broke down data from an entire production site or from the average yearly use of an installation to individual films or laminates. In the second case, the level of technology can lead to higher energy use for novel materials compared with ‘conventional’ films for the following reasons:

1. old production lines may be used for small scale production of these films and, in general, these production lines are less energy-efficient than state-of-the-art production lines as used for current conventional materials, and
2. production processes for conventional materials have been fine-tuned over decades, leading to higher throughput, yields and/or energy efficiency of production. Both cases apply especially to novel materials that have not yet been converted into films on a large scale and where, as a consequence, fine-tuning has not yet occurred.

We took these aspects into account in a sensitivity analysis (see section 4.5.1).

4.3.2.1 Location of production

We distinguished two levels of specificity, i.e. producer-specific⁷ data and regional (supra-national) data. Producer-specific data was collected from production sites and can be grouped into information on materials production and processing. Producer-specific data

⁷sometimes also referred to as ‘site-specific data’

Table 4.2: Laminates included in this study, biodegradability is indicated by an asterisk. Reference materials are 1a and 1b for Inner Packs and 5a and 6 for Outer Packs. For Inner Packs: PP laminates are 1, PP hybrids are 2, PLA laminates are 3, paper laminates are 4. For Outer Packs: PE films are 5, PP film is 6, PLA film is 7, cellulose films are 8 and paper laminates are 9.

No.	Material type ^a	No.	Material type
	Inner Packs		Outer Packs
1a	OPP / PE / MOPP	5a	PE
1b	OPP / PE / MOPP	5b	bio-based PE
2a	Paper / PE / MOPP	5c	bio-based PE
2b	Cellulose / PE / MOPP	6	OPP
2c	PLA / PE / MOPP	7*	PLA
3a	MPLA / PLA / PLA	8a*	Cellulose
3b*	PLA / AlO _x coated PLA	8b*	Cellulose
3c*	PLA / SiO _x coated PLA	9a	Paper / OPP
3d*	PLA SiO _x coated / SiO _x coated PLA	9b*	Paper / PLA
3e*	MPLA / MPLA	9c*	Paper / PLA
4a*	Paper / SiO _x coated PLA / PLA	9d	Paper / PE
4b	Paper / Aluminium / PLA	9e*	Paper / BBP ^b
4c	Paper / MPET / peelable ^c PP	9f*	Paper / BBP
4d	Paper / MPET / peelable PE	9g*	Paper / BBP
		9h	Paper / EVA heat-seal

^a Compare list of abbreviations

^b A bio-based polyester, material not further specified upon request of the producer.

^c Peelable means that this layer can easily be removed, manually, from the laminate.

was used for some polymers in primary form (e.g. PLA) and regional data for others (e.g. PP) and similarly for polymer processing such as the production of films from these materials. For example, regional data was used for OPP film but producer-specific data for PLA film because worldwide, there is only one large-scale plant to produce PLA. Similarly, also cellulose films for food packaging (using a novel technology) and a specific bio-based polyester (BBP) represent very specific products that are currently being produced at one or very few locations only. All these materials are novel and unique, the technology is not widely available and considerable progress can be made from one year to another. For paper we also used producer-specific data because of the multitude of different types of paper and their related LCA profiles, which make it difficult to decide which of the existing data-sets are representative. On the other hand, bulk materials such as PP granulate are generally purchased from a wide range of sources, nationally and internationally, and therefore no close link exists to a specific producer or production site.

Producer-specific data were used for:

- production of polylactic acid (PLA, Vink et al., 2007) and bio-based polyesters (BBP; these are complete life cycle inventories and therefore include producer-

- specific data on process inputs and outputs, including electricity⁸)
- generation of electricity and heat for the production of PLA and bio-based polyesters, for which there is only one producer each and therefore the respective local electricity profile was used (e.g. producer-specific power production in the USA was assumed for PLA)
 - process inputs and outputs for the production of paper (includes producer-specific consumption of energy but draws on regional data for the electricity mix, see below)
 - process inputs and outputs for the production of silicon oxide (SiO_x) from silica sand
 - process inputs and outputs for the metallisation of films (with aluminium, aluminium oxide or silicon oxide)
 - process inputs and outputs for the polymer processing to produce films and adhesive layers from polymer granules
 - process inputs and outputs for the lamination and printing processes.

Regional data were used for:

- generation of electricity for which we assumed the average European mix⁹ of power generation for fuel types as well as efficiency (SimaPro, 2007)
- production of process heat for which we assumed an estimated average efficiency and fuel mix for Europe (with the exceptions just explained for power SimaPro, 2007)
- production of standard petrochemical polymers, namely polypropylene (PP, as granulate and oriented film), polyethylene (PE), polyethylene terephthalate (PET), polyvinylidene chloride (PVdC) and polyurethane (PUR, used as adhesive) as well as ethyl vinyl acetate (EVA) for all of which we made use of data-sets that represent a European average (Boustead, 1999-2005)
- production of inorganic compounds, in particular aluminium (Al), and aluminium oxide (AlO_x) for which we used European averages (SimaPro, 2007)
- transportation by road, rail and ship for which we assumed European averages (SimaPro, 2007); for rail additionally also a US average for the transportation of PLA to the harbour was included.

⁸Grid electricity in the USA and Italy has comparable or larger environmental impacts than the European average, so the fact that these inventories do not use the European average does not benefit these materials.

⁹Mix of UCTE (65%, including Austria, Belgium, Bosnia-Herzegovina, Croatia, France, Germany, Greece, Italy, Luxembourg, Macedonia, Netherlands, Portugal, Serbia & Montenegro, Slovenia, Spain and Switzerland), NORDEL (13%, including Denmark, Finland, Norway and Sweden, excluding Iceland), CENTREL (9%, including Czech Republic, Hungary, Poland and Slovakia) and UK (12%) and Ireland (1%). Percentage contributions of countries are proportional to its relative total production.

- waste management by composting, digestion, landfilling, and incineration (see section 4.3.2.3)

All regional data (such as electricity, heat, plastic granulate production) were assumed for a European setting, with the exception of PLA raw material produced in the USA and transported to Europe. Technical specifications such as tie layer thickness in extrusion lamination also vary among regions. Results were therefore only calculated for packs used and produced in Europe and may not be identical, nor lead to the same conclusions for the USA, Asia or other regions.

4.3.2.2 Transport of materials

Transportation was considered during all stages of production, i.e. from the transportation of raw materials (such as wood for paper) to the delivery of laminates to the food producers. All manufacturing activities (production of materials, films and laminates) were assumed to occur within Europe, with the exception of PLA granulate production, for which the only large-scale industrial process is located in the U.S. As a consequence, the transportation distances and therefore also the environmental impacts from transportation are much larger for PLA than for the other materials¹⁰. Generic, regional data-sets were used for transport by lorry, rail and ship. Lorries were assumed to have a total maximum weight of 40 tonnes and a conservative load factor of approximately 50%. Ships were also considered to transport loads in large quantities and over long distances. Trains were assumed to run predominantly (70%) on electricity in Europe. A separate data-set was used for the transportation of PLA by train in the USA with higher primary energy use to account for lower energy efficiency.

4.3.2.3 Post-consumer waste collection and treatment

Compared to the impacts of material production, transportation generally causes comparatively small environmental impacts. Due to the short collection distances this is particularly the case for collection & transportation of mixed household waste (including Inner and/or Outer Packs) in municipalities and their surroundings. We therefore excluded post-consumer waste collection from the LCA calculations. For all post-consumer waste treatment options, readily available data from databases and publications were compiled per type of material embodied in the waste. Based on this available information and additional input by experts from the field, we estimated the water and carbon released (as carbon dioxide or methane) from the material during the waste phase for all materials (Hermann and Patel, 2007). For landfilling, composting and digestion, values in literature diverge significantly with respect to the carbon released over time (see e.g. 0-100% for

¹⁰Exact transportation routes as well as exact load factors are difficult to model because they frequently change: all suppliers involved in the project use a hauling system, where the haulier determines the exact route depending on other goods that may be transported in the same container. We have therefore assumed direct routes but low load factors (i.e. an empty return trip), based on the understanding that a higher load factor would lead to a more indirect route, compensating each other.

PLA in landfill in Bohlmann (2004), 55% for composting of PLA in Iovino et al. (2008), 80% for composting of PLA in Kale et al. (2007a)). We take this uncertainty into account through uncertainty ranges for the degradation levels of these materials, i.e. high and low carbon storage (resp. low and high level of degradation, see Figures 4.3 and 4.7).

4.4 Results

4.4.1 Inner Packs

As Figure 4.2 shows, the reference material OPP films (No. 1a and 1b) are among the best or the best for total energy use (TEU), photochemical oxidant formation (POF), acidification (AP) and eutrophication (EP). The paper/OPP film (No. 2a) is a more favourable option than the reference materials: it scores at least equally well as the reference material for all environmental indicators except for water use (H2O) and eutrophication. In terms

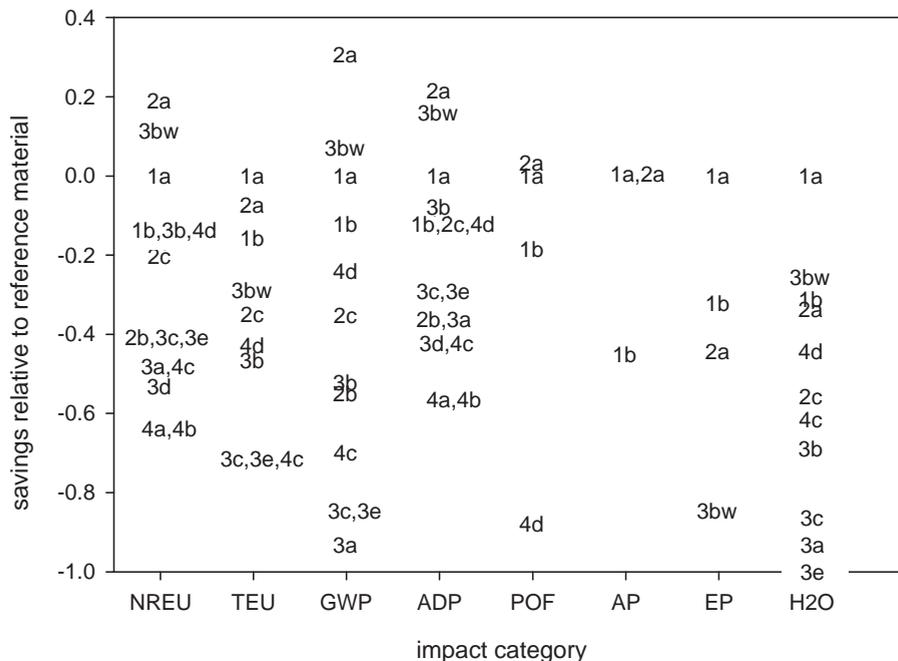


Figure 4.2: Savings of inner packs relative to reference material (No. 1a) for eight impact categories and including one case of PLA with wind credits (No. 3bw); system cradle-to-factory gate, including transportation. Note: negative values represent cases with higher environmental impacts than the reference material.

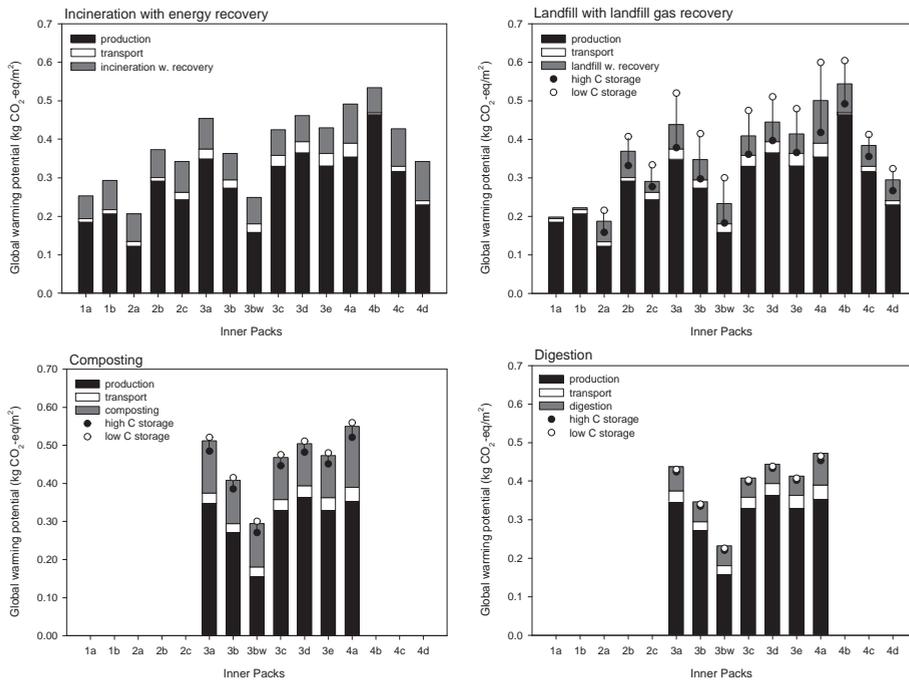


Figure 4.3: Cradle-to-grave: Global warming potential for inner packs for four waste treatment types: incineration with energy recovery, landfilling with landfill gas recovery, composting and digestion

of global warming (GWP) and non-renewable energy use (NREU), Figure 4.2 shows that a PLA-based laminate with wind credits (No. 3bw) is comparable with the reference cases (No. 1a and 1b) and somewhat less attractive than the paper hybrid (No. 2a). The results for land-use are shown in Figure 4.4. Laminates which only partially consist of bio-based films (No. 2a-c, 4c,d) have lower land-use than completely bio-based ones because of biomass cultivation. Films 3a-4b are in the same range, with the exception of 4a, which is the Inner Pack with the highest material input (i.e. heaviest laminate, see Figure 4.1). The environmental advantages of laminates containing PLA (e.g. No. 3b) or cellulose (e.g. No. 2b) can be further enhanced by future energy and material efficiency improvement (see section 4.5.1). As shown in Figure 4.3, the paper/OPP film (No. 2a) scores better (30% for GWP) than the reference materials (films No. 1a and 1b) for incineration with energy recovery and approximately comparable for landfilling with landfill gas recovery. One double-layer PLA film (No. 3b) scores best for composting and digestion among the biodegradable laminates.

Across all materials and waste management options, the global warming impacts are lowest (close to 0.2 kg CO₂/m²) for landfilling of the reference materials (No. 1a and

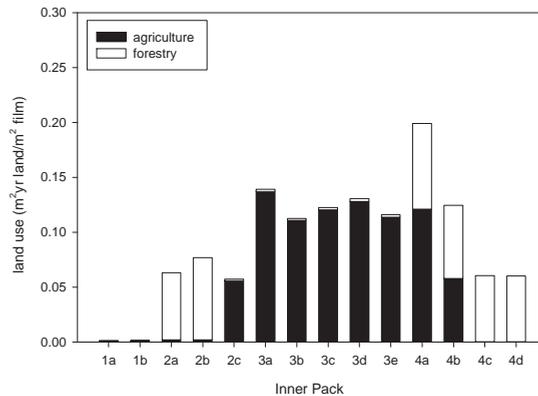


Figure 4.4: Agricultural and silvicultural land use of inner packs; system cradle-to-factory gate. Note: reference materials have little or no such land use.

1b)¹¹ and of the paper/OPP film (No. 2a), they are slightly higher for incineration with energy recovery of the paper/OPP film (No. 2a). The paper/OPP film (No. 2a) and the current OPP film (No. 1a) are the preferred options in terms of GWP in Europe.

4.4.2 Outer Packs

There are two films that are currently being used as Outer Packs: PE film (No. 5a) and OPP film (No. 6). As shown in Figure 4.5, the PE film scores much worse (10-70%) than the OPP film in all impact categories. Significant environmental improvements could thus be achieved by just replacing the current PE film by the current OPP film. However, the current OPP film (No. 6) is not the best option. Focussing on the partially bio-based laminates, the paper/OPP laminate (No. 9a), the paper/PE laminate (No. 9d) and especially the paper/EVA laminate (No. 9h) represent alternatives with a clearly improved environmental performance. In the group of totally bio-based laminates, the thinner bio-based PE film (No. 5c) is an interesting alternative for the current PE film (No. 5a) because it scores better than the reference material (No. 6) in most impact categories (worse only for AP and EP). The bio-based polyesters (No. 9e-9g) are a promising option, especially if the impacts regarding photochemical oxidant formation (POF) and eutrophication (EP) can be reduced. As Figure 4.5 shows for Outer Packs, the pure PLA film becomes attractive compared to the reference material (No. 6) if wind credits are taken into account (No. 7w), though not without wind credits (No. 7). In terms of land-use, completely bio-based laminates (e.g. No. 9b) show higher land-use than partially bio-based laminates (e.g. No. 9a) because of biomass cultivation. Those laminates that include bio-based PE (No. 5b,c) or BBP (No. 9e-g) show relatively high land-use, also when comparing with Inner Packs.

¹¹These materials consist of PP, which does not degrade in landfills and therefore releases no emissions during this phase.

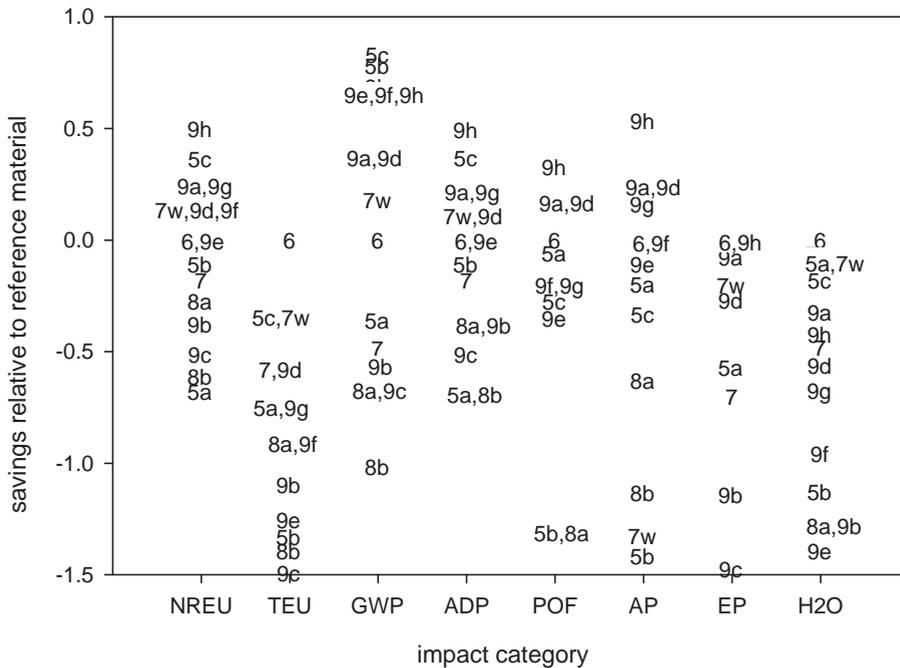


Figure 4.5: Savings of outer packs relative to reference material (No. 6) for eight impact categories and including one case of PLA with wind credits (No. 7w); system cradle-to-factory gate, including transportation. Note: negative values represent cases with higher environmental impacts than the reference material.

The cradle-to-grave analysis shows that the reference OPP film (No. 6) does not score best for any waste treatment type. For all waste treatment types, there are several laminates that perform better than the reference material: No. 5c, 9a, 9d–h (see Figure 4.7). For incineration with energy recovery, the films that score best are the bio-based PE film (No. 5c) and the paper/EVA laminate (No. 9h), followed by the bio-based polyester (No. 9g) and the paper/OPP laminate (No. 9a). For landfilling with landfill gas recovery, the bio-based PE films (No. 5b and 5c) score significantly better than the reference material (No. 5a and 6). The paper/OPP laminate (No. 9a) and the paper/EVA laminate (No. 9h) score as well as the current OPP film (No. 6). None of the reference materials are biodegradable, so composting and digestion cannot be compared relative to the current materials. For composting and digestion, the bio-based polyesters (No. 9e–9g) score best. When also taking land-use into account, the partially bio-based laminates (No. 9a, 9h, 9d) score best. Overall, the environmentally most attractive Outer Packs

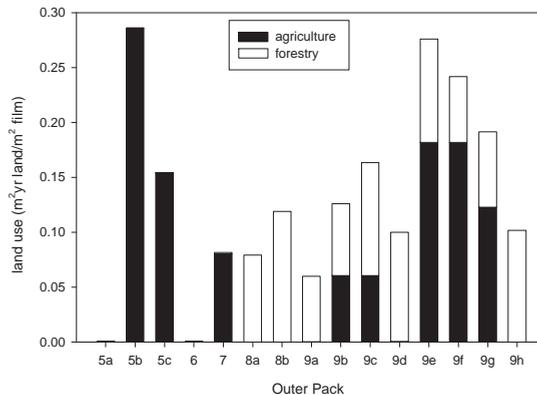


Figure 4.6: Agricultural and silvicultural land use of outer packs; system cradle-to-factory gate. Note: reference materials have little or no such land use.

are bio-based PE¹² (No. 5c), paper/PP laminate (No. 9a), paper/EVA (No. 9h), paper/bio-based polyester (No.9g) and to a somewhat lesser extent also paper/petrochemical PE (No.9d). Incineration with energy recovery is the best waste treatment option for non-degradable materials. For biodegradable materials, digestion is a better waste management option than composting in terms of global warming potential and non-renewable energy use and it is slightly better than incineration with energy recovery of the same materials.

¹²Bio-based polyethylene (and bio-based ethylene) is currently not on the market, but Braskem and Dow are building plants for commercial production in 2009 and 2011 respectively (Schut, 2009).

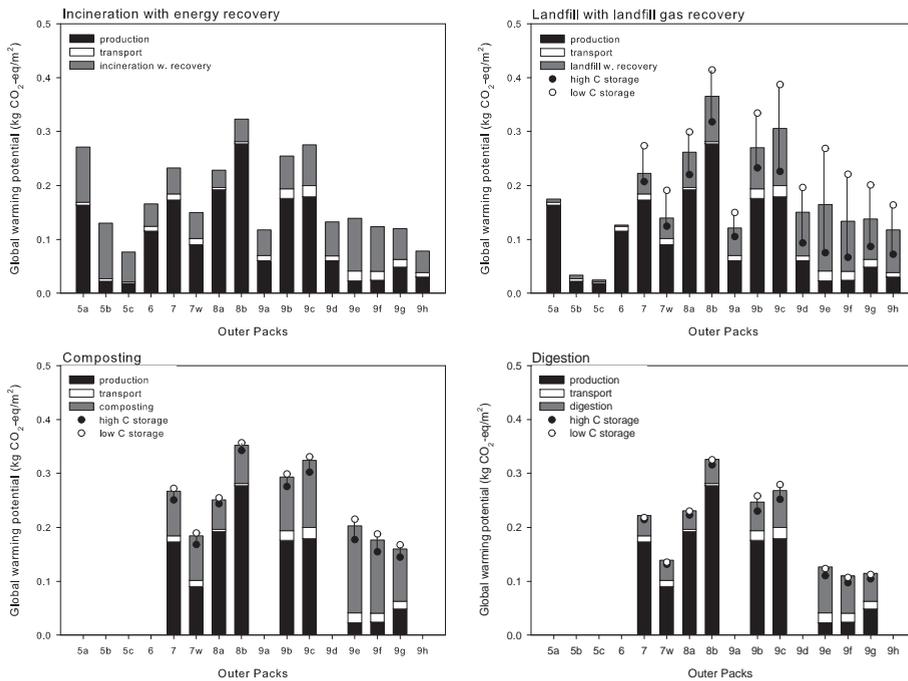


Figure 4.7: Cradle-to-grave: Global warming potential for outer packs for four waste treatment types: incineration with energy recovery, landfilling with landfill gas recovery, composting and digestion

4.5 Sensitivity analysis

4.5.1 Technology maturity

The production of petrochemical materials has made significant progress over the past 100 years, but the production of some bio-based materials (e.g. PLA and BBP) is relatively new. For the production of bio-based materials, there is a significant potential to reduce the environmental impacts in the future with increasing technology maturity both at the material production (granulate) stage and at the processing (film production). We show this improvement potential by considering future PLA granulate production (PLA-NG¹³, see Figure 4.8) and improved film production for PLA and cellulose. For PLA film production, the future case entails that the energy use for making film from polymer granules as well as the material efficiency (process waste) are assumed to equal that of today's OPP film production from PP granules. For cellulose film production, the potential future energy use for making the cellulose film from pulp was estimated on the basis of that of cellulose fibre production.

¹³Future PLA granulate production is also referred to as 'next generation' (NG).

According to Schmidtbauer (1997), non-renewable energy use in Viscose fibre production is approximately 19 MJ/kg in a large-scale plant with a total energy use of 75 MJ/kg, thus including a large share (75%) of renewable energy. To account for differences in scale between cellulose fibre and cellulose film production, for a higher share of non-renewable energy and for important differences in processing technology, we conservatively assume that the non-renewable energy use for cellulose film production in the medium to long term is four times as high as for viscose fibres, i.e. 76 MJ/kg.

Figure 4.8 shows that for PLA and cellulose, future film production significantly improves the global warming potential compared with today's technology. In the case of PLA, when considering wind credits for granulate production (see section 4.2.4) and/or future raw material production (PLA-NG), the PLA film (No. 3b) becomes advantageous compared with the reference material (No. 1a). In the case of cellulose, the future cases are almost as good as the reference materials but the future cellulose film cannot compete with the paper hybrid laminate in this specific application.

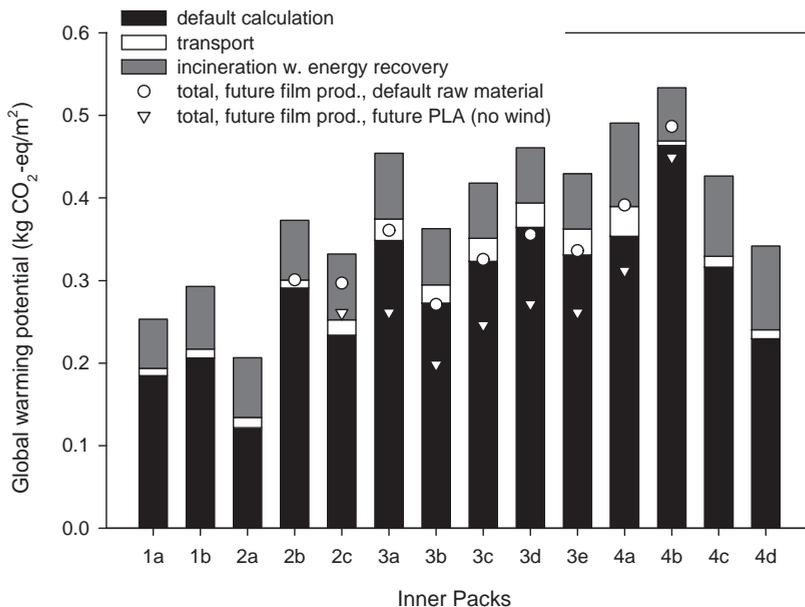


Figure 4.8: Global warming potential of laminates assuming future technology levels for film-making of PLA and cellulose: 1) today's technology (all laminates), 2) current raw material production and future film production (PLA and cellulose, ○) and 3) future granulate & future film production (only PLA, ▽); system boundary: cradle-to-grave

4.5.2 Wind credits for electricity use in film production and conversion

In section 4.2.4, we discussed the methodology of accounting for wind credits and showed the effect of buying wind credits at the raw material production stage (PLA). But wind energy credits can be bought by any company, also by film producers. In this section we therefore show the effect of compensating for energy use during film production by means of wind credits.¹⁴ As film producers are closer to the final consumers, they have more potential interest in improving the environmental profile of their films; wind energy credits are therefore more likely to be bought in by companies on the level of film producers and converters (producing e.g. PP film from granulate), and less so by the large-scale producers of petrochemical materials (producing e.g. PP granulate). Figure 4.9 shows that for the reference material Inner Packs (No. 1a and 1b), the environmental score can be improved by wind energy credits for the energy used in film production and lamination, but the hybrid paper film (No. 2a) still performs better. For the PLA laminates, wind credits for the film production and lamination & printing stages significantly improve the environmental profile and make it environmentally comparable with the reference material (No. 1a). If in addition, wind credits for raw material production are added, the PLA film (No. 3b) also outperforms the paper hybrid laminate (No. 2a)¹⁵. Wind energy credits to offset electricity use by film producers or suppliers therefore offer environmental advantages as an additional measure but by themselves are not enough for any film or laminate to overtake the best environmental profile of a film or laminate without such credits.

4.5.3 Waste management

The uncertainties related to the assessment of the waste management stage are very large, given the very wide range of the values reported in literature. These uncertainties concern both carbon storage in the solid phase (i.e. in compost, digestate or stored inside a landfill) and the composition of CO₂ and CH₄ in the gaseous phase (landfill gas, biogas from digestion and CH₄ emissions to the atmosphere). The overall difference between different types of waste-treatment with respect to GWP is rather small for biodegradable laminates and larger for non-degradable laminates where essentially no GHGs are released during landfilling. These differences are entirely due to the degradability (or lack thereof) of the material and as such cannot be reduced. Although the uncertainty regarding the degree of degradation of paper, cellulose, PLA and BBP is large, this uncertainty is not decisive for the conclusions: the difference between the materials in terms of environmental advantages is large enough for the conclusions to still hold (compare Figures 4.3 and 4.7).

¹⁴In principle, it is even possible for a company to compensate beyond the non-renewable energy use of its own processes through wind credits. However, this case is not considered here.

¹⁵It should be noted that the results presented in Figure 4.9 exclude PLA raw material production with future technology (PLA-NG).

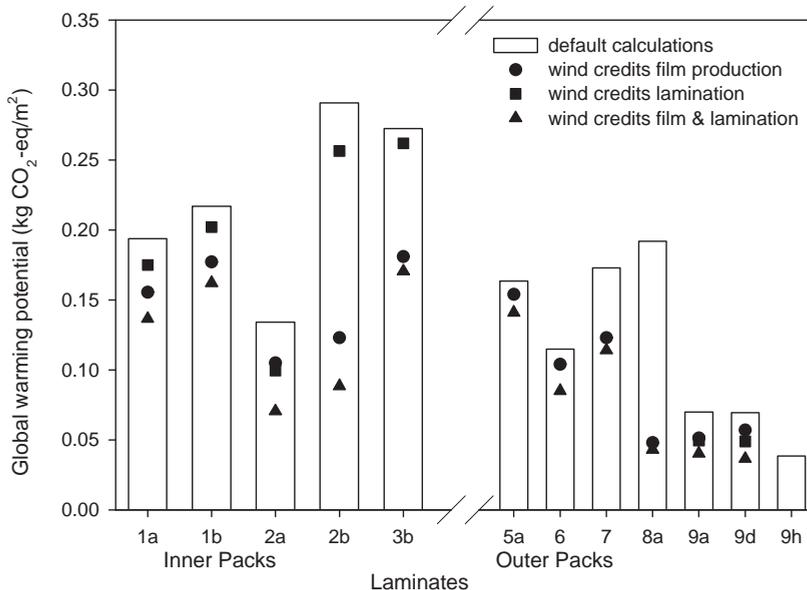


Figure 4.9: Global warming potential of wind credits during film production from granulate and/or during lamination; system boundary: cradle-to-factory gate, excluding transportation. Note: films 5a, 6 – 8a only consist of one material and thus no wind credits for lamination can be assigned.

4.6 Conclusions and Recommendations

The environmental advantages of novel bio-based materials are sometimes lower than one may expect because of comparatively high energy consumption in one or more of the production stages. This is partly due to less favourable barrier and mechanical properties that result in higher material inputs. The lack of experience and investment in converting these polymers into final products (higher percentage losses during processing than comparable petrochemical polymer gauges, long down-times, old and inefficient machines being used for the current testing phase) also plays an important role. This lack of investment is easily resolved under suitable market conditions. Some of the novel bio-based materials score remarkably well: the paper hybrid Inner Pack (No. 2a) offers GHG savings of 30% for cradle-to-factory gate and small advantages for cradle-to-grave relative to the reference material (OPP film, see Figure 4.2). Outer Packs consisting of paper and a bio-based polyester (BBP) offer GHG savings in the order of 60%-80% for cradle-to-factory gate and of 15%-30% for cradle-to-grave (incineration with recovery) relative to the reference material (OPP film, see Figures 4.3 and 4.7).

We recommend investigating to which extent options that score well for Outer Packs (bio-based polyethylene to replace petrochemical PE, paper/EVA film (No. 9h) and the

paper/bio-based polyester laminates (No. 9e-9g)) can be applied in suitable combinations also for Inner Packs. Inner and Outer Packs containing PLA film produced using today's technology and excluding wind energy credits (technology perspective) offer no significant environmental advantages, but when future technology for PLA is considered or if wind credits are assigned, PLA laminates become environmentally comparable with the reference material. It is therefore important a) that all decisions fully account for the consequence of choosing the technology perspective or the company perspective and b) that more R&D is carried out towards optimising PLA granulate and film production. Films and laminates containing cellulose produced using today's technology do not show any environmental advantages. This may change in the medium to long-term future, if producing cellulose film on a large scale and/or by using novel technologies. Bio-based polyester (BBP) offers environmental advantages, but leads to higher land-use than other bio-based materials.

Generally, laminates and films (partly) consisting of bio-based polymers offer opportunities for significantly reducing environmental impacts of food packaging. Large variations in land-use are possible depending on the type of bio-based material that is used. The environmental advantages differ depending on the polymer and the final product (Inner vs. Outer Pack). Overall, when introducing bio-based wrappings to replace the current petrochemical material, Outer Packs are more promising than Inner Packs because a) the opportunities are much clearer for this application and b) the product specifications (required barrier properties) are less demanding. Starting with the Outer Packs would also allow producers and converters of bio-based polymers to invest and learn, and therefore offer them the opportunity to reduce the environmental impact even further.

5

To compost or not to compost: LCA of biodegradable materials' waste treatment*

Abstract

Many life cycle assessments of bio-based and biodegradable materials neglect the post-consumer waste treatment phase because of a lack of consistent data, even though this stage of the life cycle may strongly influence the conclusions. The aim of this Chapter is to approximate carbon and energy footprints of the waste treatment phase and to find out what the best waste treatment option for biodegradable materials is by modelling home and industrial composting, anaerobic digestion and incineration. We have compiled data-sets for the following biodegradable materials: paper, cellulose, starch, polylactic acid (PLA), starch/polycaprolactone (MaterBi), polybutyrate-adipate-terephthalate (PBAT, Ecoflex) and polyhydroxyalkanoates (PHA) on the basis of an extensive literature search, experiments and analogies with materials for which significant experience has been made. During biological waste treatment, the materials are metabolised so a part of their embodied carbon is emitted into air and the remainder is stored as compost or digestate. The compost or digestate can replace soil conditioners supporting humus formation, which is a benefit that cannot be achieved artificially. Experimental data on biodegradable materials shows a range across the amount of carbon stored of these materials, and more trials will be required in the future to reduce these uncertainties. Experimental data has also shown that home and industrial composting differ in their emissions of nitrous oxide and methane, but it should be noted that data availability on home composting is limited. The results show that anaerobic digestion has the lowest footprint for the current level of technology, but incineration may become better in the future if energy efficiency in waste incineration plants improves significantly. Home composting is roughly equal to incineration with energy recovery in terms of carbon and energy footprint when carbon credits

*Under review at Polymer Degradation & Stability. Co-authors: Lies Debeer and Bruno De Wilde (both Organic Waste Systems, Gent), Kornelis Blok and Martin K. Patel (both Utrecht University)

are considered. The same applies to industrial composting if carbon credits are assigned for compost to replace straw. Carbon credits can therefore considerably affect the results, but there are significant uncertainties in how they are calculated. When soil carbon is a limiting factor, biological waste treatment options should be chosen to safeguard soil carbon despite incineration with energy recovery possibly performing better in the future due to efficiency improvements.

5.1 Introduction, Problem Setting and Goal

Polymers today are predominantly made from non-renewable resources and disposed of in landfills or waste incineration plants. In light of international efforts to mitigate greenhouse gas emissions and to divert waste materials from landfills which are approaching their maximum capacity, bio-based and biodegradable materials are becoming increasingly important. In the past, this has led to efforts, investments and progress made in this area of materials science. The importance of these materials is likely to keep increasing in the future, and further scientific and technological breakthroughs will make these materials easier, and therefore cheaper, to produce (see Chapter 2). Many of these materials can be used for packaging of food and as food service items and will therefore often become available mixed with wet food waste. Treating this waste by composting or digestion eliminates the need for separation of this waste and avoids a lowering of the calorific value (from wet food waste) in an incineration plant. In addition, consumers and industry are paying more and more attention to the environmental aspects of material use, e.g. through carbon footprints of products. The environmental advantages of these new, bio-based and (often) biodegradable materials have already been compared with standard materials such as petrochemical plastics through life cycle assessments (LCAs).

Incineration is a logical waste treatment option for conventional plastics, but for biodegradable plastics (which often are bio-based) also biological treatment options such as composting and anaerobic digestion are suitable. However, most of the life cycle studies on bio-based and (often) biodegradable materials only consider incineration (e.g. Dornburg et al., 2003, and Chapter 3), or they limit themselves to the production phase (system boundary cradle-to-factory gate), neglecting the post-consumer waste treatment stage completely (e.g. Kim and Dale, 2005c; Vink et al., 2007). Much information is therefore available on the production of bio-based materials, but relatively little about their waste treatment.

One reason for neglecting post-consumer waste treatment has been the lack of consistent data for this stage of their life cycle: many of these materials are new, have not been produced on a large scale yet or if so only for a short time. Therefore comprehensive, consistent and long-term testing of their behaviour in various types of waste treatment has not yet been carried out. This lack of experiments calls for an approximation and derivation of data, especially when considering that the waste treatment stage of the life cycle may influence the conclusions as they would be drawn when only considering the production and transportation of the materials (system boundary cradle-to-factory gate). A consistent

approximation of the behaviour of and emissions from biodegradable materials undergoing biological waste treatment also allows comparative environmental assessments of these waste treatment options.

The aim of this chapter is to find out what the best waste treatment option for biodegradable materials is. In order to do so, we compare the results for the waste treatment of biodegradable materials, modelling home and industrial composting, digestion and incineration. In section 5.2, we provide some background information on the selected bio-based and petrochemical reference materials and describe the methodology used for the carbon and energy footprint calculations. Next, we detail the specific particularities of the biological waste treatment options (composting and digestion, section 5.3). In section 5.4, we compile the data-sets for industrial and home composting, anaerobic digestion and incineration and show the resulting inventory data. Section 5.5 then presents a comparison of results for the waste treatment of bio-based and petrochemical reference materials and a detailed sensitivity analysis. The conclusions we draw are shown in section 5.6.

5.2 Background on biomaterials and footprinting methodology

In this section, we provide background information and data on biodegradable materials (sections 5.2.1–5.2.3) as well as a description of the methodology for carbon and energy footprinting that we applied.

5.2.1 Materials

We selected the following bio-based and/or biodegradable materials because these have received the most attention from research and industry (see Shen et al., 2010): chemical and mechanical pulp for paper and cellulose production, starch, polylactic acid (PLA), starch/polycaprolactone (starch/PCL, MaterBi™), polybutyrate-adipate-terephthalate (PBAT, Ecoflex™) and polyhydroxyalkanoates (PHA).

Paper, cellulose and starch have been used for a long time, others such as PLA and starch/PCL are relatively new. MaterBi™ is a polymer that usually is produced from a bio-based (starch or vegetable oil) and a biodegradable petrochemical compound; here we assume that it consists of 50% starch and 50% poly-ε-caprolactone (PCL) and therefore abbreviate it as starch/PCL. There are several other types of MaterBi™ as well, which can be significantly different in composition and thus also in biodegradation. PBAT is a petrochemical, biodegradable material. In Table 5.1, we show the key properties of the biodegradable materials that are considered in this chapter.

Table 5.1: Key properties of biodegradable materials

Material	HHV (MJ/kg)	C content (weight %)	share of renewable C (weight % on C)	Source
Paper	19.7	44.4	100	ECN (2009)
Cellulose	18.8	44.4	100	ECN (2009)
Starch	17.4	44.4	100	ECN (2009)
PLA	18.7	50.0	100	Vink et al. (2007)
starch/PCL	22.8	53.7	50	derived from constituents
PHA	36.6	70.4	100	Patel et al. (2006)
PHB	23.8	55.6	100	Patel et al. (2006)
PHV	27.4	60.0	100	Patel et al. (2006)
PCL	28.2	64.0	0	own calculations
PBAT	29.2	62.4	0	own calculations

5.2.2 Suitability of waste treatment options per material

Throughout this chapter we consider four types of waste treatment: home composting, industrial composting, anaerobic digestion and incineration. We exclude mechanical and chemical recycling because these waste treatment types would merit a separate publication and because most of today's controversies around the end-of-life stage of biomaterials have focussed on the environmental performance of biological waste treatment options (i.e. composting and anaerobic digestion) as opposed to municipal solid waste incineration are still in their infancy for biomaterials and results would be very unreliable. The biological waste treatment options provide differing conditions in terms of temperature, oxygen and micro-organisms for the breakdown of biomaterials and biomass in general. Figure 5.1 summarises these conditions and shows which biomaterials biodegrade in which type of biological waste treatment (technically, all materials can be landfilled or incinerated).

5.2.3 Biodegradation levels of materials and certification of compostability

Biodegradation is defined as the decomposition of organic material by micro-organisms or their enzymes into smaller products such as carbon compounds (including CO₂, CH₄ and solids) and water (OECD, 2002; EC, 2008a)¹. The term focuses solely on the way the degradation process occurs, the origin of the carbon in the biodegradable material is unimportant, so petrochemical as well as bio-based materials can be biodegradable. Biodegradation can occur in the natural environment, e.g. in soils, or in a controlled environment such as during biological waste treatment. Composting and digestion are

¹Metabolisation in this specific case includes breaking down the biomaterials (catabolism) and producing cell components (anabolism).

	Anaerobic Bacteria, no fungi	Aerobic Bacteria & fungi
50-60°C	Chemical pulp Starch PLA starch/PCL PHA	Chemical pulp Mechanical pulp Starch PLA starch/PCL PHA PBAT
≤35°C	Chemical pulp Starch starch/PCL PHA	Chemical pulp Mechan. pulp Starch starch/PCL PHA PBAT

Figure 5.1: Materials which are biodegradable in four types of biological waste treatment

examples of aerobic and anaerobic biodegradation processes respectively, resulting in the formation of gases (e.g. CO₂, CH₄ and N₂O), water and compost² or digestate³.

There are several standards that can be used to certify that a material is compostable. The European standard EN 13432 can be used to certify *industrial* compostability of packaging materials. To achieve certification, a material has to undergo testing at maximum 58°C for 180 days and has to biodegrade by 90% in absolute terms or relative to cellulose within that time frame. The definition of the 180 day maximum was intended to cover both biodegradation during the composting process as well as the biodegradation afterwards, during application of compost to soil. Other requirements that the material has to fulfil are disintegration (i.e. no visible remains of the material after 12 weeks), lack of toxicity and lack of negative effects on the composting process. The US American equivalent (ASTM D6400⁴ and D6868⁵) is more lenient, stipulating only 60% degradation after 180 days and allowing for higher levels of heavy metals (see Kijchavengkul and Auras, 2008). There is no home composting certification scheme on the level of ISO or ASTM yet, but the Belgian ‘OK Compost Home’ scheme, managed by Vinçotte, is widely used. It requires materials that are to be certified for *home* composting to degrade at ambient temperatures (20–30°C) by 90% relative to a reference material (cellulose) within 12 months (Vinçotte, 2007).

For the quantification of carbon storage within the framework of a life cycle assessment, it is important to note that industrial composting processes are usually much shorter than the 180 days stipulated in the standards, namely 2-8 weeks on average for fresh

²Compost is the solid residue of the process of composting, consisting of stable organic matter.

³Digestate can also be referred to as digester solids and is a solid substance similar to compost but a residue of the process of digestion.

⁴ASTM D6400 covers plastic films and bags.

⁵ASTM D6868 covers packaging designed to be composted, such as plastic coated paper and board.

Table 5.2: Extent of biodegradation of materials under typical conditions in an operating industrial composting plant, results of the literature survey and selected values for this analysis; all materials satisfy the requirements set out in section 5.2.3

Material	Carbon degradation (%)	Source	Selected
Newspaper (mech.)	43	Lopez Alvarez et al. (2009)	
Newspaper (mech.)	45	Haug & Bidlingmaier (in Krogmann, 1994)	Mech. Pulp
Pulp, mech.	50	Haug & Bidlingmaier (in Krogmann, 1994)	55
Newspaper (mech.)	79	De Wilde and Boelens (1998)	
Paper, mech.	<70	Venelampi et al. (2003)	
Writing Paper (chem.)	65	Lopez Alvarez et al. (2009)	
Paper, chem.	>70	Venelampi et al. (2003)	Chem. Pulp
Kraft Paper (chem.)	80	Pagga et al. (1995)	80
Pulp, chem.	90	Haug & Bidlingmaier (in Krogmann, 1994)	
Hemicellulose	70	Haug & Bidlingmaier (in Krogmann, 1994)	
Cellophane	68	Narayan (1989)	
Cellulose	74	Du et al. (2008)	
Cellulose	79	Lopez Alvarez et al. (2009)	80
Cellulose	84	Pagga et al. (1995)	
Cellulose	ca.85	De Wilde and Boelens (1998)	
Cellulose	>90	Avella et al. (2000)	
Cellulose	100	Mohee et al. (2008)	
Corn Starch	73	Kale et al. (2007b)	
Starch (TPS)	73	Du et al. (2008)	
Starch (TPS)	87	Iovino et al. (2008)	80
Starch	89	Fritz et al. (2001)	
Starch	97	Degli Innocenti et al. (1998)	
PLA	55	Iovino et al. (2008)	
PLA	65	Kale et al. (2007b)	
PLA	71	Fritz et al. (2001)	80
PLA	ca.80	Kale et al. (2007a)	
PLA	>90	Tyler (1997)	
PLA	95	Kish (2008)	
MaterBi ⁶	27	Mohee et al. (2008)	
MaterBi	60	Novamont (2001)	
MaterBi	75	Scandola et al. (1998)	70

continued on next page

⁶We use the tradename MaterBi™ on purpose here, because in many cases the exact formulation (e.g. share of starch) is not clearly stated in the source and this may explain a large part of the variation of degradation levels.

Table 5.2 – continued from previous page

MaterBi	>90	Tyler (1997)	
PBAT	60	Kijchavengkul et al. (2008)	
PBAT	70 ⁷	Reimer et al. (2008)	70
PBAT	100	Witt et al. (2001)	
PHBV	65	Avella et al. (2000)	
PHBV	70	Mergaert et al. (1994)	PHAs
PHB	78	Fritz et al. (2001)	80
PHBV	88	Pagga et al. (1995)	
PHBV	90	De Wilde and Boelens (1998)	

compost (VAB, 2008), 12 weeks in total including maturing. The level of biodegradation achieved during composting is also very much determined by the particle size of the product. The thicker a material, the smaller the level of biodegradation reached. In laboratory testing, biodegradation is tested on samples after milling in order to reach high levels of biodegradation as fast as possible. This means that in reality, biodegradation levels reached at the end of the composting cycle will be (much) lower. In order to work with ‘realistic’ biodegradation levels of bio-based materials, we have therefore conducted a literature survey on the biodegradation of selected bio-based materials in industrial composting (summarised in Table 5.2) and selected average values for these materials based on this table as well as on information from experts.

5.2.4 Functional unit and system boundaries

The functional unit selected for this study is 1 kg of material. In terms of system boundary, we focussed on the waste treatment of the products only, excluding the use phase and transportation. This functional unit has one caveat: the behaviour of materials in most waste treatment types (except incineration) also depends on the final function of the material, such as the material thickness and/or the combination with another material (see section 5.5.3). Additionally, the mechanical and other material properties also determine how much material is required for a given application; these material properties are not considered here.

5.2.5 Data collection

For all post-consumer waste treatment options, we compiled readily available data from databases and publications per type of material (see Tables 5.6 and 5.7 and references in sections 5.4.3 and 5.4.4). The types of materials for which this data was collected

⁷after 50-60 days

were various types of paper, various types of organic, biodegradable material including kitchen and green waste. We translated the material inputs and outputs for each type of waste treatment into inputs and outputs per kg of (bone) dry carbon input entering the treatment process and checked the mass, carbon and energy balances. Based on this preparatory work, we developed a set of chosen data for each type of waste treatment. By making analogies with materials for which more details were known, we generated data-sets for mechanical and chemical pulp, starch, PLA, starch/PCL, PHAs and PBAT. Cellulose and Kraft paper are both made from chemical pulp and are therefore assumed to behave like chemical pulp in the waste treatment phase and we therefore only show results for chemical pulp. The data-set for mechanical pulp also applies to paper made from mechanical pulp. The resulting data-sets are presented in section 5.4.

5.2.6 Impact categories

We calculated the carbon and energy footprints. The carbon footprint is based on the characterisation factors for global warming potential (GWP) as published by the IPCC 2007 and takes into account renewable and non-renewable carbon assuming that any biogenic carbon which is stored in the materials is deducted during the production phase (cradle-to-factory gate) of a carbon footprint/life cycle assessment. The energy footprint represents the cumulative fossil and nuclear energy demand in terms of primary, non-renewable energy use (NREU), i.e. the energy content of a material as well as the non-renewable energy spent on its extraction. NREU was expressed as higher heating value (HHV), also called 'gross calorific value'. We discuss the effect of leaving out other potential impact categories in section 5.5.3.

5.3 Methodological particularities of biological waste treatment options

The biological waste treatment options considered here (see Figure 5.1) lead to a metabolisation of the biodegradable materials during the waste treatment process. As a result of this metabolisation, part of the carbon from the material is emitted into air, mainly as carbon dioxide (CO_2) and some of it as methane (CH_4). The rest of the carbon is also metabolised by the microorganisms but remains stored in the form of a biodegradation/metabolisation product such as compost or digestate⁸. This compost or digestate degrades further over time, for example after its application to soil, but at a much slower rate than during the biological waste treatment. As mentioned in section 5.2.6, one of the benefits of compost, and analogously also digestate, is this long-term carbon storage. The

⁸This is true for all materials which are composted or digested, whether they are food and garden waste or biodegradable polymers. The rate of metabolisation depends on polymer characteristics such as crystallinity, morphology, molecular weight and crosslinking (see Kijchavengkul and Auras, 2008). The production of compost and digestate leads to build-up of a humus-like substance (see below Kale et al., 2007b; Cuhls et al., 2008; Kijchavengkul and Auras, 2008).

so-called Rothamsted field trials have shown that continually using land for agriculture leads to carbon losses from soil (Woburn and Foster fields, Rothamsted 2006). Bellamy et al. (2005) found that UK soils have lost 13 million tonnes of carbon annually between 1978 and 2003, equalling 13% of the carbon stock. Long-term carbon storage in soils is important because soils store much more carbon than biomass (1550 Gt C soil organic carbon vs. 760 Gt C in the atmosphere, Lal (2004)) and the UNFCCC acknowledged agricultural soils as valid carbon sinks (UNFCCC, 2008). Therefore the value of compost & digestate as soil conditioners to counteract these carbon losses is obvious. Three specific benefits that are considered in this analysis are:

- Compost stores carbon in the soil,
- Nitrogen contained in compost is an organic fertiliser,
- Compost can be applied instead of peat or straw as a soil conditioner in order to improve the soil structure, thus avoiding fossil emissions from the use of peat

In the following, we describe the methodologies used to quantify these benefits.

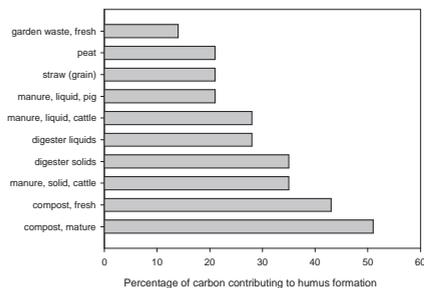
5.3.1 Long-term carbon storage in the soil

Soil organic carbon can be divided into three groups: 1) 'active' or easily degradable carbon, 2) slowly degradable carbon and 3) 'passive' carbon (see Favoino and Hogg, 2008). Mature compost (as it is initially applied to the land) belongs to the slow fraction, humus belongs to the passive fraction (Favoino and Hogg, 2008). Part of this compost is converted into humus, by a process called humification. Humus as considered in soil science is a portion of soil organic matter in a stable, equilibrium state that has already undergone decomposition (Whitehead and Tinsley, 1963; Encyclopaedia Britannica, 2009). The process of humification, i.e. the formation of humus, is complex and not fully understood, involving decomposition of biomass, metabolisation by microbes, cycling of carbon between soil organic matter and microbial biomass and finally formation of humic compounds (Encyclopaedia Britannica, 2009; Whitehead and Tinsley, 1963).

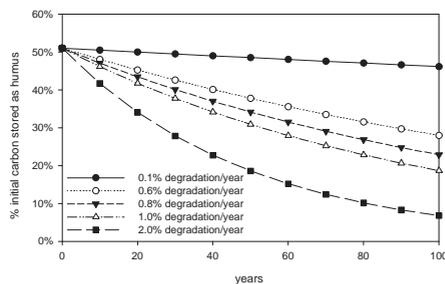
When compost leaves the composting facility, 50%–80% of the biodegradable material has degraded (see Table 5.2), but long-term carbon storage over a period of 100 years is much lower than 20%-50%. The long-term carbon sequestration from compost (resp. digestate) in soil can in principle be quantified by taking into account the respective share of carbon that contributes to humus formation (i.e. of type 2 or 3) and that carbon's degradation over time. However, the quantification of this type 2 and 3 carbon is also complex because the exact composition of humus is uncertain and the turnover rates of those compounds also can depend on local conditions. The modelling of humus degradation is therefore difficult. Humus degradation can also be measured through experiments, but their results are also uncertain because extraction of humic substances from soil is difficult (Whitehead and Tinsley, 1963) and data therefore varies. Humus degradation was measured to be 0.1%-0.8% per year in agricultural soils (Bellamy et al., 2005) depending on

the initial carbon content of that soil; this was one of few long-term carbon experiments but there were many factors that potentially influenced humus production/degradation, and the overall change was attributed to climate change by the authors. Favoino and Hogg (2008) suggested that the degradation rate is 1% per year, but the data on which they based this rate is not given.

Of the carbon originally stored in mature compost, 51% contribute to humus formation (see Figure 5.2a)⁹. A part of that humus is mineralised, but the rate of mineralisation (humus degradation) is subject to considerable uncertainties (see Figure 5.2b). Calculating with the upper value from Bellamy et al. (0.8%/yr), this means that after 100 years, 23% of the humus from that carbon still remains in the soil (see Figure 5.2; in section 5.5.2.4 we show that the results are not sensitive to the chosen degradation rate). If we choose the end-point of carbon storage (i.e. 23% of the original C for composting), it would not make a difference whether the carbon was released immediately (year 1) or after being sequestered for 99 years. We therefore calculate a weighted average over 100 years for long-term carbon storage in the soil, resulting in a figure for ‘integrated’ storage of 35.1%, which is significantly higher than the 22.8% that represent the end-point status. The difference between 22.8% and 35.1% seems quite large, but the intermediate results (see Table 5.8) show that the difference in resulting carbon footprint is small; therefore the results section only considers the integrated values. We apply the same concept to carbon storage in digester solids, where the relative carbon contributing to humus formation is 35% of what is originally stored in the digester solids (see Figure 5.2a).



(a) Carbon contributing to humus formation as share of total carbon in soil conditioners (Reinhold, 2007)



(b) Evolution of long-term carbon storage from compost in soil over time (Bellamy et al., 2005; Favoino and Hogg, 2008)

Figure 5.2: Humus formation for various soil conditioners and humus degradation over time

⁹This is possibly a conservative assumption because Luxhøi et al. (2007) estimated the inert fractions of compost as 78% and of digestate as 48%.

5.3.2 Carbon credits from replacing soil conditioners

When compost is used as a soil conditioner to substitute other soil conditioners, this saves carbon and/or nitrogen from other sources. We quantify the carbon credits from replacing such soil conditioners in this section, nitrogen credits are discussed in section 5.3.4. Soil conditioners are valuable because they can counteract carbon losses from agriculture and forestry (see section 5.3). The most common ones are peat and straw¹⁰. Peat is used either directly or as a constituent of potting soils by private gardeners and the horticultural industry, straw is used in agriculture. This means that in reality, compost from home composting replaces only peat¹¹, whereas compost from industrial composting replaces peat and straw at a ratio of 1:3 based on industrial compost use in the UK, Germany, the Netherlands and Switzerland (Nikitas et al., 2008; Thelen-Jüngling, 2009; VAB, 2008; Schleiss, 2008).

We use the method developed by Fuchs and Schleiss (2006) and combine the carbon content of soil conditioners with the shares of carbon that contribute to humus production (see Figure 5.2) to calculate the carbon credits. Substitution factors can then be derived for compost in comparison to other soil conditioners through equating the materials' humus-C and then standardising to compost (see Table 5.3). For example, the factor for peat is $61.2/77.7 = 79\%$, therefore 1 kg of compost (fresh matter) can replace 0.79 kg peat, 0.73 kg straw or 1.17 kg digester solids. As stated earlier, the easily degradable carbon that is contained in the soil conditioners degrades and is oxidised to biogenic or, in the case of peat, fossil carbon dioxide. Table 5.3 shows that carbon dioxide emissions per unit of humus-carbon are lower for compost (by weight) compared to peat and straw by almost a factor of four. This means that compost is more efficient than peat and straw in building up humus-C.

In order to take into account that the carbon dioxide emissions from the application of straw are biogenic and thus are cycled through agriculture on the short term, we set these biogenic CO₂ emissions to zero, valuing only the fossil carbon dioxide emissions from the use of peat¹². We can easily quantify fossil CO₂ savings per unit of compost which replaces peat: using the compost replacement factor for peat (0.79) and the emissions from the oxidation of easily degradable C (1.07 kg CO₂/kg FM) the savings amount to 3.7 kg CO₂ per kg C in compost and 45 MJ/kg C in saved NREU. For straw replacement, there are no direct (fossil) CO₂ savings but as part of the sensitivity analysis we quantify benefits from straw which could be used for electricity production when replaced by compost on the agricultural fields (see section 5.5.2.1).

¹⁰Solid animal manure can be used as a soil conditioner as well but can be disregarded based on the low quantities used. Liquid animal manure is frequently used but is a source of nutrients only and does not contribute to soil improvement.

¹¹In the case of home composting, 70% of the peat in potting soils can be replaced by compost without loss of quality of the potting soil according to Blok and Verhagen (2009).

¹²The CO₂ emissions from compost are already considered through its inverse: long-term carbon storage.

Table 5.3: Carbon sequestration properties of various soil conditioners according to Fuchs and Schleiss (2006)

unit	Dry matter (DM) in fresh matter (FM) [kg/kg]	C in FM [kg/kg]	humus factor [%]	humus-C in FM [g/kg]	emissions from oxidation of easily degradable carbon [kg CO ₂ /kg h-C]	emissions from oxidation of easily degradable, non-renewable carbon [kg CO ₂ /kg FM]
Peat	0.70	0.37	21	77.7	13.8	1.07
Straw	0.80	0.40	21	84.0	13.8	0
Compost	0.56	0.12	51	61.2	3.5	0
Digestate	0.53	0.15	35	52.5	6.8	0

5.3.3 Allocation of nitrous oxide emissions

During home and industrial composting processes, quantities of nitrous oxide are released that impact the total GWP from these processes. Whether or not these emissions are taken into account depends on the methodological perspective chosen. Three perspectives can be used: the individual, the systemic and the combined perspective.

The *individual* perspective entails considering each of the inputs to the composting process as an independent entity. As a consequence, each material input to the composting process is only 'held accountable' for the environmental impacts or benefits it individually causes. When working from an individual perspective, nitrous oxide emissions from composting are not allocated to the biomaterials studied here, based on the fact that they do not contain nitrogen. At the same time, no fertilisation credits are allocated either because the main parameter for such a credit would be the nitrogen content of the compost, to which the biomaterials do not contribute. However, for the home or industrial composting process to work, the recommended range of carbon to nitrogen ratio of the input material is 20–40 g/g (Bruno De Wilde)¹³.

The *systemic* perspective considers the composting process to be a system that would not work if the input materials were considered individually, i.e. the optimal carbon to nitrogen ratio is only achieved by the right combination of input materials. Based on this reasoning, the effects of nitrogen and carbon should be distributed across the entire system and thus all material inputs. This means that a share of nitrous oxide emissions is assigned to the biomaterials, based on the fact that composting would not work without a source of nitrogen.

Finally, the *combined* perspective is based on the systemic perspective, i.e. it considers nitrogen-related emissions of nitrous oxide but uses individual degradation rates of the

¹³The C/N ratios of individual material inputs are often outside the optimal range, with for example regular kitchen waste being too rich in nitrogen and too dense, biomaterials on the other hand being poor in nitrogen. Biomaterials can therefore be used to increase the C/N ratio of the mixture and to decrease its density, thereby replacing common bulking agents in composting such as wood chips or sometimes paper.

materials. This perspective thus takes into account that without nitrogen, the composting process would not work, and at the same time that a uniform degradation rate (as used in the systemic perspective) is too far from reality for most biomaterials.

We show composting results for all three perspectives, but use the individual perspective as a default for our calculations. The results can differ between perspectives because the share of carbon initially stored in the compost differs. The following table summarises the methodological differences between the three perspectives:

Table 5.4: Overview of the methodological differences between the three perspectives

Perspective	N ₂ O emissions	N credits	C credits	biodegradation
Individual	no	no	yes	material-specific
Systemic	yes	yes	yes	average of all inputs
Combined	yes	yes	yes	material-specific

5.3.4 Nitrogen credits

The systemic and combined perspectives take into account emissions of nitrous oxide from composting due to the necessity of nitrogen in the composting system (see section 5.3.3). This is done based on the ratio of carbon to nitrogen in the composting input. In order to remain consistent, the fertilising effect of compost (through N in the compost) also has to be considered. We assume that the nitrogen in the compost can replace nitrogen from synthetic fertilisers¹⁴, the most important of which are urea, ammonium nitrate, calcium ammonium nitrate and ammonium sulphate. Different types of fertiliser vary considerably in their greenhouse gas emissions and energy use. In order to correctly quantify the credits assigned to compost, we calculated a weighted average based on the nitrogen content of these fertilisers (Ramirez and Worrell, 2006) as well as the consumption of these fertilisers in 2007 (IFA, 2008) in Western and Central Europe (see Table 5.5).

When allocating the nitrogen fertilisation credits to the biomaterials, we used the same method which was applied to allocate nitrous oxide emissions: the carbon to nitrogen ratio of the compost. This means that carbon content of the compost (and thus the carbon from the biodegradable material that does not degrade during waste treatment, see Table 5.2) is considered to be in a fixed ratio to the the nitrogen content of the compost which determines the quantity of synthetic fertiliser replaced. The nitrogen content in compost

¹⁴Note: there are indications that not all of the nitrogen stored in compost is available to plants; this is particularly true for compost and other (solid) fertilisers that also lead to humus reproduction (Luyten-Naujoks, 2009). According to this publication, 70% of the N in liquid manure and as little as 10-35% of the N in compost may be available for plants, Hansen et al. (2006) gives similar numbers. In our calculations, this is not relevant for the standard results (based on individual perspective which does not take nitrogen emissions and credits into account), and has little influence on the GWP for materials when considering the systemic or combined perspective because the effect of nitrogen credits is small (see Table 5.11).

is on average around 10 kg per tonne of fresh matter (Fuchs and Schleiss, 2006; van Ewijk, 2008; EPEA, 2008), which roughly equals 62 kg of nitrogen per tonne of carbon in compost. The emissions of nitrous oxide resulting from the application of fertilisers to land are excluded due to the fact that these emissions are dependent only on the nitrogen content of the fertiliser and not on the type of fertiliser¹⁵ and we quantify credits according to nitrogen content. Therefore, the emissions of N₂O per unit of nitrogen in the fertiliser during this phase can be considered equal regardless of whether the nitrogen stems from compost or from synthetic fertilisers. Savings for replacing the production of synthetic nitrogen fertilisers result in 1.3 kg CO₂ and 11.9 MJ NREU per kg C in compost.

Table 5.5: Consumption of common fertilisers in Western and Central Europe, their N-content and resulting weighted GWP

unit	Consumption of fertiliser in Western and Central Europe [kt N/yr]	N content of fertiliser [%]	Consumption of fertiliser in Western and Central Europe [kt/yr]	Weighted average [%]	GWP of fertiliser production and application ^a [t CO ₂ eq/t]
Urea	4,898	46	10,648	30.3	4.10
Ammonium nitrate	3,202	34	9,418	26.8	8.64
Calcium ammonium nitrate	2,863	25	11,452	32.6	8.76
Ammonium sulphate	761	21	3,624	10.3	2.78
Total	11,724	-	35,141	100.0	avg 6.70

^a This GWP contains the emissions of greenhouse gases from the production phase (cradle-to-factory gate) as well as the CO₂ emissions from the release of carbon from the disintegration of the urea complex (which is not applicable to the other fertilisers).

5.4 Waste treatment types

In the EU-27 countries, an average of 41% of the municipal waste was landfilled, 20% incinerated and 39% recycled or composted in 2007 (Eurostat, 2007). Landfilling can be below 10% (Denmark, the Netherlands) and incineration higher than 50% (Denmark). The EU landfill directive (EC, 1999) calls for a 50% reduction of landfilled biodegradable municipal waste by 2009 compared to 1995 quantities and of 65% by 2016. In contrast to ‘standard’ materials, the biodegradable materials are in general suitable for a larger variety of waste treatment options and alternative waste treatment types for organic, biodegradable waste thus have to be evaluated. In this section we present the inventory data

¹⁵Bouwman et al. (2002) still differentiated between fertiliser types in their nitrogen model in 2002, but discontinued this differentiation in their most recent version, see Smeets et al. (2009b).

for four types of waste treatment: industrial composting, home composting, anaerobic digestion and waste incineration. All of the data presented here take into account only the waste management stage, but not any environmental effects from the production of the biodegradable materials. We calculate long-term carbon storage for each material via biodegradation of the material during composting and the degradation rate of humus.

5.4.1 Industrial composting

As shown in Figure 5.1, industrial composting occurs when bacteria and fungi degrade biomass under aerobic conditions and at high temperatures (50-60°C). Compared with home composting, industrial composting not only reaches higher temperatures, but the biomass involved is also mixed more frequently, thereby ensuring a higher homogeneity and thus faster degradation of biomass. The data for industrial composting used here is based on eight data-sets on industrial composting, which all show similar levels of biodegradation (see Table 5.6). Tables 5.6 and 5.7 also illustrate that the carbon balance is not always closed for industrial composting¹⁶. The data chosen for this study are in line with the latest publications on industrial composting of vegetable, fruit and green waste (VFG) in terms of methane and nitrous oxide emissions (see van Ewijk, 2008; VAB, 2008; EPEA, 2008)¹⁷. Many more publications are available on industrial composting (see e.g. Boldrin et al., 2009, for an overview), but most of these do not specify carbon input and biogenic CO₂ emissions from the composting stage and therefore do not allow calculating a carbon balance.

The GWP from industrial composting as calculated is dominated by emissions of carbon dioxide. Emissions of methane are rather exceptional and are small when they do occur and compared with home composting, nitrous oxide emissions are much lower. We use the results for process emissions from industrial composting of VFG (see Table 5.6) and material-specific biodegradation levels (see Table 5.2) to derive data for the biodegradable materials. The results per kg material are shown in Table 5.11.

¹⁶In the case of Andersen et al. (2010), this gap is probably the quantity of carbon stored in materials rejected after composting such as large pieces of wood which have not broken down sufficiently.

¹⁷The comparison of literature data on industrial composting (Table 5.6) shows that the variation of emissions of CH₄ per kg carbon is much higher than for N₂O. The large methane variation may be due to the moisture content of the composting material, its density and the efficiency of aeration; additionally, methane emissions are known to occur discontinuously (Edelmann and Schleiss, 2001) and can therefore vary significantly depending on the set-up of the experiment. Brinkmann et al. (2004) and van Ewijk (2008) criticise that the data from AOO (2002) was based on data now considered outdated because of 'new insights, technology improvement and the availability of higher quality data'. Specifically, Brinkmann et al. (2004) state that the methane emissions in AOO (2002) were measured at an installation at a time when the aeration and air cleaning system were not working properly. This suggests working with the more recent publications on industrial composting regarding air emissions, because the older data (Edelmann and Schleiss, 2001) are worse than the AOO (2002) data. EPEA (2008) data do not appear to be based on measurements. VAB (2008) suggest much lower emission factors for methane and slightly lower ones for nitrous oxide from the industrial composting processes based on measurements at several composting installations in the Netherlands. We use these factors, considering them to be industry averages.

Table 5.6: Comparison of literature data on carbon and nitrogen emissions for industrial composting per kg carbon input, assuming short-term carbon storage in compost; in all cases, fruit, vegetable and green waste is the main material input to the composting process.

Source	technology	C in CO ₂ [%]	C in CH ₄ [%]	C stored short-term [%]	C inaccuracy [%]	N ₂ O emitted [g/kg C]
Edelmann and Schleiss (2001)	closed	45	5.08	50	0	0.93
Edelmann and Schleiss (2001)	open	47	2.53	50	0	0.93
AOO (2002)	closed	60	1.55	38	0	0.83
Brinkmann et al. (2004)	avg	57	0.13	43	0	0.87
den Boer et al. (2005)	closed	53	1.54	38	7	0.66
VAB (2008)	avg	58	0.11	42	0	0.60
EPEA (2008)	avg	56	0.57	44	0	0.67
van Ewijk (2008)	closed	57	0.11	43	0	0.60
Andersen et al. (2010)	open	56	1.16	37	7	0.96
Selected for this study ¹⁷		57	0.11	43	0	0.60

5.4.2 Home composting

As shown in Figure 5.1, home composting occurs when bacteria and fungi degrade biomass under aerobic conditions at ambient temperatures ($\leq 35^\circ\text{C}$). Compared with industrial composting, here the biomass remains at lower temperatures, and is mixed less frequently. As a result, biomass degrades more slowly. In this article, we focus on home composting in temperate climates. Smith and Jasim (2009) have collected extensive measurement data of home compost temperatures over a period of two years in the UK. They showed that compost temperatures are usually above ambient temperature, roughly between 20 and 30°C in summer and 5 to 20°C in winter. Their data shows relatively large variations in terms of temperature between individual composts, suggesting variations in metabolic activity.

The data for home composting is based on a set of experiments carried out at OWS in Belgium (Debeer et al., 2008), the goal of which was to quantify gaseous emissions from the system. The experiments showed a large variation in nitrous oxide emissions at temperatures of 23°C (see Table 5.7). In some of the 16 experimental vessels, the formation of N₂O seemed to have levelled off already at the end of the experiment (12 weeks), whereas in other vessels the emissions were still rising; at this point approximately 50% of the carbon originally embedded in the input material had been degraded and the rate of carbon dioxide and methane emissions had slowed down. Debeer et al. (2008) conducted composting experiments at 23°C and at 38°C. These two temperatures represent the wide range of home composting conditions which can be found between North and South Europe. Generally, the higher the temperature is, the faster the materials will break down. We focus on the results for low temperature home composting because these are considered representative for passive home composting in a temperate climate, i.e. home composting where interference with the compost pile is kept to a minimum

Table 5.7: Comparison of literature data on carbon and nitrogen emissions for home composting per kg carbon input assuming short-term carbon storage in compost; in all cases, fruit, vegetable and green waste is the main material input to the composting process

Source	C in CO ₂ [%]	C in CH ₄ [%]	C stored short-term [%]	C inacc- uracy [%]	N ₂ O emitted [g/kg C]
Amlinger and Peyr (2002); Amlinger et al. (2008)	47	1.32	44	7.7	1.16
Amlinger and Peyr (2002); Amlinger et al. (2008)	30	0.47	35	34.5	1.53
Debeer et al. (2008), avg, 23°C	47	0.02	53	0	3.86
Debeer et al. (2008), best, 23°C	35	0.01	65	0	1.32
Debeer et al. (2008), worst, 23°C	60	0.04	40	0	6.92
Debeer et al. (2008), avg, 38°C	49	0.21	51	0	0.83
Debeer et al. (2008), best, 38°C	38	0.11	62	0	0.05
Debeer et al. (2008), worst, 38°C	61	0.44	39	0	2.98

and where there are smaller volumes of compost. We use 23°C as an approximation for home composting conditions in temperate climates because it is in the lower half of the temperature range for certified home compostability (Vinçotte, 2007) and is roughly 10°C above average annual temperatures in European temperate climates.

A number of data-sets were available to draw up a carbon balance for industrial composting, but very few specifically for home composting; we compare these in Table 5.7. The data match in terms of CO₂ emissions and short-term carbon storage, but there are significant differences in the amount of methane and nitrous oxide emissions. Methane emissions are low, which is in line with earlier publications (e.g. Smith and Jasim, 2009). The nitrous oxide emissions have a significant effect on the GWP from home composting and the large differences between experimental vessels call for an analysis that takes this large variation into account. We have therefore analysed three cases, one average, one best case (lowest stabilised emissions of N₂O) and a worst case (extrapolated emissions of N₂O to take into account the large range of measured N₂O emissions as well as the potential development of emissions after the end of the experiment). Table 5.7 shows the home composting results per kg C for VFG for the short-term. We use the experimental data (short-term, Table 5.7) for home composting together with the humus degradation rates (see section 5.3.1) to derive data for the biomaterials. For all perspectives, we also take into account that some materials do not biodegrade at all under home composting conditions, such as PLA (see Figure 5.1).

For the systemic perspective, the data-set is equal to the average data-set for home composting at 23°C by Debeer et al. (2008), rounding off the CO₂ emissions. For the individual and combined perspectives, we work under the assumption that the biomaterials will biodegrade to the same extent (level) as under industrial composting conditions, just taking longer to do so due to much lower temperatures (i.e. a lower degradation rate). We therefore use the biodegradation levels from Table 5.2. We combine this data with the

Table 5.8: Data on carbon and nitrogen emissions for home composting of biomaterials per kg carbon input. The last two columns show results for end-point and integrated carbon storage.

Material	Perspective unit	C as CH ₄ [%]	N ₂ O emitted [g N ₂ O]	C stored short-term [%]	C stored long-term [%]	GWP (no credits) [kg CO ₂]
Temperate climate (23°C)						
All	Systemic	0.02	3.86	50	11.4/17.6	4.40/4.17
Mech. pulp	Individual	0.02	0	45	10.3/15.8	3.29/3.09
	Combined	0.02	3.86	45	10.3/15.8	4.43/4.23
Chem. pulp &Starch &PHA	Individual	0.02	0	20	4.6/7.0	3.50/3.42
	Combined	0.02	3.86	20	4.6/7.0	4.64/4.56
starch/PCL &PBAT	Individual	0.02	0	30	6.8/10.5	3.42/3.29
	Combined	0.02	3.86	30	6.8/10.5	4.56/4.43
PLA	does not degrade under home composting conditions.					
(Sub)Tropical climate (38°C)						
All	Systemic	0.21	0.83	50	11.4/17.6	3.46/3.23
Mech. pulp	Individual	0.21	0	45	10.3/15.8	3.32/3.12
	Combined	0.21	0.83	45	10.3/15.8	3.52/3.32
Chem. pulp &Starch &PHA	Individual	0.21	0	20	4.6/7.0	3.52/3.44
	Combined	0.21	0.83	20	4.6/7.0	3.70/3.62
starch/PCL &PBAT	Individual	0.21	0	30	6.8/10.5	3.44/3.31
	Combined	0.21	0.83	30	6.8/10.5	3.62/3.49
PLA	does not degrade under home composting conditions.					

emission data for methane and nitrous oxide from the home composting experiment and disregard or allocate nitrous oxide emissions for the individual and combined perspectives respectively. This leads to the inventory presented in Table 5.8. Additionally, we assign carbon credits for replacing peat (based on section 5.3.2). The results per kg of material are shown in Table 5.11.

5.4.3 Anaerobic digestion

As shown in Figure 5.1, digestion occurs when bacteria only (no fungi) degrade biomass under anaerobic conditions. Depending on the temperature, one distinguishes between thermophilic (50–60°C) and mesophilic ($\leq 35^\circ\text{C}$) digestion. We do not distinguish between these two types of digestion for our data-sets due to the fact that this difference in temperature mostly influences the speed of degradation but this speed is not relevant for a LCA time-scale of 100 years. (Note, however, that PLA will only degrade in a thermophilic environment.) In terms of data-sets, several sources are available but they are essentially limited to organic green and kitchen waste (den Boer et al., 2005; EPA, 2006; Edelmann and Schleiss, 2001; Vagron, 2008; van Ewijk, 2008). No data-sets are available specifically for paper, cellulose, starch or biodegradable polymers. The data-sets therefore had to be estimated through analogies with composting and digestion of green waste. The levels of biodegradation of the materials were assumed to be equal to those

of industrial composting; the amounts of methane and carbon dioxide produced during digestion are shown in Table 5.9. The biogas (methane and carbon dioxide) is then burnt for electricity production. We assign credits to the net export of electricity using the same regional data for electricity as for incineration with energy recovery (see section 5.4.4). The efficiency of electricity production from digestion is approximately 36% (van Ewijk, 2008), with roughly a quarter of that electricity used internally for running the digester (Vagron, 2008); the electricity exported is therefore 28%. Digestion also leads to the production of digestate, which we took into account using the same methodology as for compost (see sections 5.3.1 and 5.3.2).

Table 5.9: Carbon balance for anaerobic digestion of materials per kg C input to digestion

Material unit	C in CO ₂ (biogas) [%]	C in CH ₄ (biogas) [%]	C in digestate short-term [%]	Net power exported [MJ _{el}]	GWP short-term [kg CO ₂]
Chem. pulp & PLA & Starch & PHA	40	40	20	5.11	1.42
starch/PCL	35	35	30	4.46	1.24
Mech. Pulp & PBAT are not proven digestable					

5.4.4 Incineration

We calculated results for incineration with energy recovery modelling materials according to their carbon and energy contents. For incineration with energy recovery we chose the average of all municipal solid waste incineration (MSWI) plants in Europe (EU-25): according to Reimann (2006) the net average export of power represents approximately 11% of the total gross calorific value (higher heating value) of the waste fed to the plant and the respective percentage for the net export of heat is approximately 22%. These figures are clearly below potential efficiencies of an optimised installation for heat and power (>50% according to Ragossnig et al., 2008) but represent industry averages for waste incineration for which efficiencies are always lower due to lower temperatures to prevent corrosion (Smith et al., 2001). We show the results of a sensitivity analysis for the efficiency of electricity production from waste in section 5.5.2.7. Using the heating values in Table 5.1, we then assigned credits to the net export of electricity and heat for each material. Electricity from incineration of waste replaces average electricity from the grid in Europe (see section 5.3.2) and the recovered heat replaces heat from natural gas which is the most common source of heat in industry. Table 5.10 shows the results for the biodegradable materials considered throughout this chapter.

Table 5.10: Global warming potential for incineration of materials with and without energy recovery, per kg material input

Material unit	Gross GWP [kg CO ₂]	Net heat out [MJ]	Net power out [MJ _{el}]	Net GWP [kg CO ₂]
Mech. Pulp	1.63	4.20	1.99	1.03
Chem. Pulp	1.63	4.20	1.99	1.03
Starch	1.63	3.89	1.84	1.08
PLA	1.83	4.18	1.98	1.24
starch/PCL	1.99	5.09	2.42	1.26
PHBV	2.12	5.72	2.71	1.31
PBAT	2.29	6.53	3.10	1.36

5.5 Results

5.5.1 Comparison of waste management results

Figure 5.3 shows the results for all waste management types, i.e. home composting, industrial composting, digestion and incineration. For all materials, digestion is the preferred waste treatment type, as it combines energy recovery with the production of digestate, which can be used as a soil conditioner. Composting and incineration are comparable in terms of GWP, but NREU credits are higher for incineration. Figure 5.3 also shows that although home and industrial composting results are practically identical for GWP when considering the individual perspective, there are significant differences between the two in terms of NREU. The main reason for these differences is that NREU credits are higher for industrial composting, because the resulting compost is assumed to replace straw on agricultural fields, whereas compost from home composting is assumed to replace peat (see Table 5.11).

5.5.2 Sensitivity analysis

5.5.2.1 Sensitivity to the effect of carbon credits

By default, we considered carbon credits for the use of compost or digestate to replace peat as a soil conditioner but did not quantify those for straw replacement (see section 5.3.2). One possibility of quantifying the carbon credits for straw is to switch from an attributional to a consequential LCA method by assuming that in a future, bio-based economy, the straw which is replaced by compost on the agricultural fields can then be used for energy production (heat and power) by co-firing. Using an energy content of straw of 18 MJ/kg (dry; ECN, 2009), a (conservative) power conversion efficiency of 30% as well as the compost replacement factor, the compost replaces straw which in turn replaces 1.1 kWh of electricity that would otherwise have been produced predominantly from fossil sources. We then assign credits for these 1.1 kWh using the European average

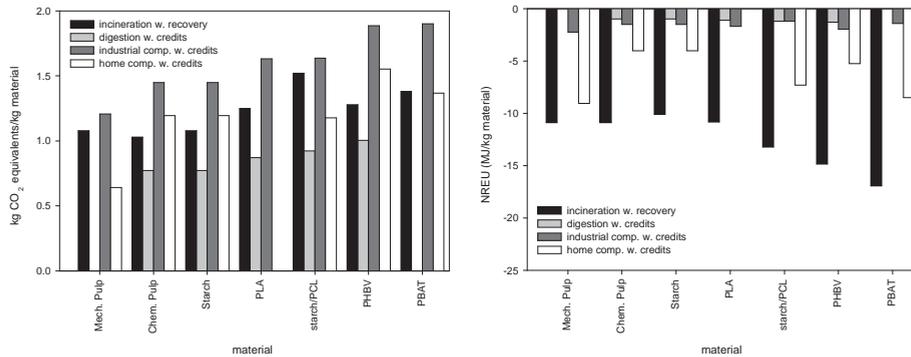


Figure 5.3: Comparison of four waste treatment types for biomaterials and reference materials, results for GWP (left) and NREU (right), individual perspective; note: LCA results only for the waste phase; NREU values of home composting are very small

electricity profile¹⁸, resulting in savings per kg C in compost of 3.5 kg CO₂ and 45 MJ NREU. As a result, the carbon credits shown in Table 5.11 for industrial composting¹⁹ and digestion would increase by a factor of three.

5.5.2.2 Sensitivity to the effect of nitrogen credits

As discussed before (section 5.3.4), there are indications that only a fraction of the nitrogen in compost is available to plants. We did not take this into account in our calculations and may have therefore over-estimated nitrogen credits. However, this is not relevant for our default calculations (individual perspective) and for the systemic and combined perspectives, the effect of nitrogen credits on the carbon and energy footprints is relatively small (see Table 5.11) and the over-estimation is therefore not significant. For the systemic and combined perspectives, we assigned credits for the nitrogen contained in compost as a replacement for synthetic N fertilisers, but no credits for the potassium and phosphorus contained in the compost. Both P and K are seldom the limiting factor for growth in current agricultural systems which have been fertilised for long times (Hansen et al., 2006). The results are not sensitive to this omission: the energy use for and the resulting CO₂ emissions from the production of both potassium and phosphorus fertilisers is much lower than for nitrogen fertilisers. When considering the potassium and phosphorus content of compost, the amount of CO₂ credits for fertiliser substitution (N credits) would increase by roughly 15% per kg C, decreasing the results for the waste treatment stages (com-

¹⁸65% UCTE, 13% NORDEL, 12% UK, 9% CENTREL, 1% Ireland. Percentages represent relative contribution to overall electricity in Europe, see abbreviations for which countries are encompassed. The share of primary renewable energy is only 8%, consisting of mostly hydropower and biomass in Scandinavian countries. The footprints amount to 0.15 kg CO₂/MJ_{el} for carbon and to 2.82 MJ_p/MJ_{el}.

¹⁹Home compost is only assumed to replace peat as a soil conditioner, the credits from straw replacement are therefore irrelevant.

Table 5.11: Overview of contribution of processes to overall GWP, expressed as CO₂-equivalents per kg material and totals for the individual (Ind), systemic (Sys) and combined (Com) perspectives (see section 5.3.3 for the methodology); for the systemic data-set we assumed a material with 50% C content

Home Composting									
material	Process			humus	C	N	sum	sum	sum
	CO ₂	CH ₄	N ₂ O	deg.	credits	credits	Ind	Sys	Com
Mech. pulp	0.90	0.003	0.51	0.48	-0.73	-0.26	0.64		0.89
Chem. pulp	1.30	0.003	0.51	0.21	-0.33	-0.12	1.19		1.59
Starch	1.30	0.003	0.51	0.21	-0.33	-0.12	1.19		1.59
PLA	does not degrade								
starch/PCL	1.38	0.003	0.62	0.38	-0.59	-0.21	1.19		1.58
PHBV	1.69	0.003	0.66	0.28	-0.42	-0.15	1.55		2.07
PBAT	1.60	0.004	0.72	0.45	-0.69	-0.24	1.37		1.84
systemic	0.92	0.003	0.58	0.59	-0.92	-0.33		0.85	

Industrial Composting									
material	Process			humus	C	N	sum	sum	sum
	CO ₂	CH ₄	N ₂ O	deg.	credits	credits	Ind	Sys	Com
Mech. pulp	0.90	0.016	0.09	0.48	-0.18	-0.26	1.21		1.04
Chem. pulp	1.30	0.016	0.09	0.21	-0.08	-0.12	1.45		1.42
Starch	1.30	0.016	0.09	0.21	-0.08	-0.12	1.45		1.42
PLA	1.47	0.018	0.10	0.24	-0.09	-0.13	1.63		1.60
starch/PCL	1.38	0.020	0.11	0.38	-0.15	-0.21	1.64		1.54
PHBV	1.69	0.021	0.12	0.28	-0.11	-0.15	1.89		1.85
PBAT	1.60	0.023	0.13	0.45	-0.17	-0.24	1.90		1.79
systemic	1.10	0.018	0.10	0.48	-0.18	-0.26		1.25	

Digestion									
material	Process			humus	C	N	E _{el}	sum	sum
	CO ₂	CH ₄	N ₂ O	deg.	credits	credits	credits	Ind	Com
Chem. pulp	1.30	0	0	0.21	-0.07	-0.12	-0.67	0.77	0.66
Starch	1.30	0	0	0.21	-0.07	-0.12	-0.67	0.77	0.66
PLA	1.47	0	0	0.24	-0.08	-0.13	-0.76	0.87	0.74
starch/PCL	1.38	0	0	0.38	-0.13	-0.21	-0.71	0.92	0.71
PHBV	1.70	0	0	0.28	-0.09	-0.15	-0.87	1.01	0.85

Incineration									
material	Process			humus	C	N	E _{el+h}	sum	
	CO ₂	CH ₄	N ₂ O	deg.	credits	credits	credits		
Mech. pulp	1.63	0	0	0	0	0	-0.60	1.03	
Chem. pulp	1.63	0	0	0	0	0	-0.60	1.03	
Starch	1.63	0	0	0	0	0	-0.55	1.08	
PLA	1.83	0	0	0	0	0	-0.59	1.24	
starch/PCL	1.99	0	0	0	0	0	-0.72	1.26	
PHBV	2.12	0	0	0	0	0	-0.81	1.31	
PBAT	2.29	0	0	0	0	0	-0.93	1.36	

posting and digestion) by not more than 0.1 kg CO₂-equivalents per kg of material and is therefore negligible. However, the phosphorus recycling by composting and digestion may become a significant advantage in the light of the depletion of phosphorus reserves (Cordell et al., 2009).

5.5.2.3 Sensitivity to choice of perspective

The choice of perspective determines whether nitrous oxide emissions and nitrogen credits are considered (see section 5.3.3). The individual perspective is the default that we used for the composting calculations. As Table 5.11 shows, nitrous oxide emissions and nitrogen credits affect the results but to a lower extent than material degradation, carbon credits or humus degradation. In the case of home composting, nitrous oxide emissions are not compensated by nitrogen credits, whereas for industrial composting credits are larger than emissions. Figure 5.3 showed that the differences in carbon footprint between home and industrial composting is very small when applying the individual perspective; this is due to the fact that nitrous oxide emissions, which differ significantly between the two types of composting, are disregarded.

5.5.2.4 Sensitivity to humus degradation rates

We carried out all calculations using a humus degradation rate of 0.8% per year. However, Figure 5.2 shows that a range of values can be used (0.1%–2%); this illustrates the large uncertainties regarding soil carbon processes in general and humus degradation rates in particular. The results are not sensitive to this range of values: the GWP changes by only 5% per kg carbon when using either of the extreme values of this range (integrated, not end-point storage). This translates to an average change of less than 2% per kg material.

5.5.2.5 Sensitivity to higher temperatures during home composting

The home composting experiment was carried out at 23°C, which is representative for temperate climates in Central and Northern Europe. Experiments were also carried out at 38°C, which is representative for warmer climates, e.g. in Southern Europe; they showed the same carbon degradation (order of 50%), but higher methane emissions and much lower nitrous oxide emissions (compare Table 5.8). This is consistent with the findings of Amlinger et al. (2008), who also found methane peaks at higher (thermophilic) temperatures and higher nitrous oxide emissions at mesophilic rotting stages. When calculating a GWP for home composting at 38°C it is approximately 20% lower than at 23°C and thus much more favourable than at lower temperatures. This suggests that the climatic conditions have a significant effect on the desirability of home composting from the point of view of greenhouse gas emissions.

5.5.2.6 Sensitivity to variation in experimental data for home composting

The home composting experiments showed some variation in biodegradation levels, and in emissions of methane and nitrous oxide. The effect of this variation is very small for the individual perspective, because emissions of carbon dioxide are based on material-specific degradation levels, the variation in methane emissions is relatively small and nitrous oxide emissions are not considered. The effect of this variation is largest for

the systemic perspective (a change of 25%–30% of the GWP per kg carbon), because degradation levels are set according to the experiment, and both methane and nitrous oxide emission variations are considered (compare data in Table 5.7). Variation is considerable for the combined perspective as well (20% of the GWP per kg carbon), but lower than for the systemic perspective. This is due to fixed emissions of carbon dioxide based on material-specific degradation levels.

5.5.2.7 Sensitivity to efficiency of waste incineration

We used the average of all MSWI plants in Europe to calculate the net average export of heat and power from waste incineration, amounting to resp. 22% and 11% of the total gross calorific value (higher heating value) of the waste input. However, higher efficiencies can be reached and we therefore discuss the results from a sensitivity analysis regarding these efficiencies here. To do so, we calculated two scenarios: In the first we kept the export of heat constant but almost tripled the net power exported to 30%, which represents the optimal (not average) efficiency from a state-of-the-art plant in Amsterdam (Simoës, 2008). In the second scenario we used the efficiencies from Smith et al. (2001) where heat recovery is much higher (50%). In both cases, the improved efficiencies greatly improve the carbon and energy footprints for incineration with energy recovery: on average, the carbon footprint for incineration is reduced to 50% of the default value. This means that if the average efficiency of electricity production from waste is increased to state of the art levels or if above-average quantities of heat are exported, then incineration may become the preferred waste treatment type.

5.5.3 Uncertainties and discussion

The approximation of data for waste management carried out in this study serves as a first assessment and is limited to two indicators, energy (NREU) and carbon (GWP), because of the lack of detailed and consistent data regarding process emissions from waste treatment in the cases of ‘new’, bio-based materials. As Huijbregts et al. (2006) showed, NREU is a good approximation for overall environmental impacts, but results still need to be interpreted carefully. For instance, we did not consider the effect of toxic compounds (e.g. from inks on printed paper). However, for a material application to be put on the market as certified compostable, no toxic compounds may be present (see section 5.2.3 on composting standards). We also did not consider ammonia emissions from composting, which would influence acidification potential. In practice, ammonia emissions can be reduced by working at the right C/N ratio. This ratio is increased by biodegradable materials, which is favourable because the bio-waste stream often contains much nitrogen and thus a low C/N ratio. Additionally, the results presented here refer to pure materials, but the direct bonding of several materials in, for example, a laminate can change the rate of biodegradation and possibly also the extent of biodegradation at the end of the biological waste treatment process. The same holds for material thickness, which is a material parameter that has to be specified when applying for certification according to

EN 13432 and for which material-specific maximum values apply (see DIN CERTCO, 2009). The uncertainties related to these two issues are not included in the ranges reported and can only be eliminated through detailed experimental analysis in the future. Further experimental analysis would also be required in order to understand the reasons for the variation in experimental home composting data (see section 5.5.2.6). Another uncertainty regards the effect of management practice on emissions from home composting: the frequency of mixing and the water content of the home composting pile affect home composting process, e.g. through the temperatures in the pile. These in turn affect the rate of biodegradation and the emissions of methane and nitrous oxide (see section 5.5.2.5) and therefore the results for global warming. However, management practice is difficult to influence.

5.6 Conclusions

In this chapter we compared the carbon and energy footprints from four types of waste treatment of biodegradable materials. Not all materials are suitable for all types of biological treatment, PLA for example does not degrade in home composting in temperate climates and PBAT and mechanical pulp do not degrade in anaerobic digestion. In terms of waste treatment types, we focussed on industrial composting, home composting in temperate climates, anaerobic digestion and waste incineration. Of these waste treatment options, digestion is the most favourable for biodegradable materials for the time being because it combines energy recovery with the production of digestate, which can be used as a soil conditioner. Home and industrial composting are roughly equal to incineration with energy recovery, with small differences across materials. They differ in terms of methane and especially nitrous oxide emissions and the choice of perspective, i.e. if nitrous oxide emissions and nitrogen credits are allocated or not, therefore has an effect on the results for industrial and especially home composting. Carbon credits for the use of compost as a soil conditioner significantly improve the carbon and energy footprints for both types of composting. The temperature under which home composting is carried out also has a large effect on methane and nitrous oxide emissions and thus on the carbon footprint.

Anaerobic digestion has the lowest footprint for the current level of technology, but incineration may become better in the future if the average energy efficiency in waste incineration plants improves significantly. The biological waste treatment options are at least equal to incineration in terms of carbon and energy footprint, but have the additional benefit of producing a soil conditioner that supports humus formation, which cannot be achieved artificially. This also signifies that if soil carbon becomes a limiting factor in the future, then despite incineration with energy recovery possibly performing better due to efficiency improvements, biological waste treatment options should be chosen to safeguard soil carbon.

6

Policy efforts to increase market penetration of bio-based materials*

Abstract

The use of renewable resources for the production of materials offers higher greenhouse gas savings than bioenergy per kg feedstock as well as per hectare and has received some attention from policy-makers, but analyses of either existing or potential policy measures are almost entirely missing in this field. This chapter investigates how current and potential policy measures affect market penetration of bio-based materials. In general, market penetration of biomaterials remains low and few policies seem to have had a significant effect. Our inventory of existing measures reveals a lack of consistent and comprehensive policy measures to support the introduction of renewable resources for material use. Furthermore, most measures have only recently been implemented and vary in terms of the underlying policy goals: most policies aim at reducing waste and littering and not at reducing greenhouse gas emissions. In addition, the variation is also large when it comes to the material properties which are targeted: some measures target the *source* of the materials (supporting bio-based materials) and others target the *end-of-life* stage of the materials (supporting biodegradable materials). Our case study on the implementation of a carbon dioxide tax shows that the tax would only influence the market position of ethylene, whose market volume is so large that such a tax would indeed affect the market considerably, as well as lead to global greenhouse gas savings. We have also identified significant barriers to the market penetration of bio-based materials which arise from existing policy measures: many policies supporting the use of bioenergy explicitly restrict any (financial) benefits to the use of biomass for energy purposes. As energy and materials compete for the same biomass or agricultural land, this puts bio-based materials at a disadvantage. In order to increase the market penetration of biomaterials, it is therefore important that policy barriers are removed to create a level playing field for the use of biomass, whether for energy or material purposes. Additionally, the policy measures for biomaterials should be harmonised, and more measures introduced, one of which could

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be a carbon tax or other carbon pricing scheme. Finally, policy measures for new bio-materials should move away from niche applications such as plastic bags, expand into targeting more comprehensive applications such as general packaging, and receive more attention within green/sustainable procurement efforts.

6.1 Introduction

When discussing mitigation of and adaptation to climate change, a consensus has been reached that action is required at different levels of governance, from international to local levels. Within the field of mitigation, it is also clear that to effectively reduce the emissions of greenhouse gases, changes have to be implemented in various industrial sectors. The sector which has received much attention so far is the energy supply sector, including electricity and transportation fuels (REN21, 2009). The transition of this sector from using traditional, non-renewable energy sources such as oil, coal and natural gas towards using more and more renewable energy sources such as wind, solar energy and biomass has been supported by numerous policies on regional as well as national levels: biofuels for transportation have been supported regionally by the EU Biofuels Directive (EC, 2003a) and will continue to be supported by the 2009 Renewable Energy Directive (EC, 2009e), and nationally by the US Energy Policy Act (2005) and for example by the French eco-tax (2005). Bioenergy in terms of electricity and heat is stimulated on the regional level for example by the 2009 Renewable Energy Directive (EC, 2009f), and on national levels e.g. by the German Renewable Energy Law (EEG; BMU, 2004). These policies have boosted the production and development of electricity and fuels from biomass (Thornley and Cooper, 2008; REN21, 2009), leading to increases in production and consumption of biomass for energy purposes.

However, biomass can also be used to produce materials such as conventional renewable materials (e.g. paper and wood products) and bio-based chemicals, plastics, fibres, lubricants or composite materials (novel 'bio-based materials'). Several authors have shown that bio-based materials offer higher greenhouse gas savings than bioenergy per kg biomass used or per hectare (e.g. Dornburg et al., 2003; Anex and Ogletree, 2006; Vink et al., 2007, Chapter 3). Biomaterials have received some attention from policy-makers (e.g. the Biopreferred Program in the US), but analyses of either existing or potential policy measures are missing in this field. It is therefore the aim of this chapter to analyse how current and future policy measures affect market penetration of novel bio-based materials. To this end, we firstly provide an overview of current world-wide policies that are relevant for bio-based materials (section 6.2). We then compare these policies with those on transportation biofuels and other bioenergy (section 6.3) as well as current policy barriers (section 6.4). Finally, we present a case study of an ex ante policy analysis, modelling the effect of a carbon tax on (bio)materials (section 6.5).

6.2 Inventory of existing policy measures

In the following, we provide an overview of the existing laws, regulations, and agreements that directly or indirectly influence the political and financial status quo of biomaterials. The term ‘biomaterials’ in this article comprises bio-based materials (i.e. derived from biomass) as well as biodegradable materials. Many bio-based materials are biodegradable, and many biodegradable ones are bio-based, but the two groups are not identical (see Table 1.1 for a categorisation of materials).

6.2.1 Method and scope

The information on which this analysis is based stems from an extensive review of scientific as well as non-scientific sources. Scientific policy analyses are scarce, and we thus had to rely not only on legal texts of laws & directives, but also on secondary literature (publications about laws & directives). The information provided here is considered up-to-date at the time of writing (May 2010), however, the policy field can change rapidly and information therefore may have become outdated since then.

In the analysis, we distinguish between normative mechanisms (i.e. obligations and standards), economic incentives (i.e. subsidies and taxes) and communication measures (i.e. labelling and information campaigns). In several cases, policy measures are a combination of two types, for example in the case of a cap-and-trade system, with the cap being the normative part of the measure, and the trading of certificates being the economic part of the measure. Cases where categorisation of the policy measures is difficult have been indicated.

The analysis focusses on the situation in Europe, the USA and Japan, but includes some policy measures in other countries as well. We discuss only those mechanisms here that have a direct impact on the market penetration of bio-based materials, so research & development subsidies are excluded. Additionally, a number of other measures may appear relevant because they draw attention to bio-based materials, supply a framework and aim at unifying and focussing policy efforts in this area, but they do not contain any concrete policy measures and are therefore not discussed here either; examples are the Lead Market Initiative of the EU (EC, 2003d) and National Action Plans on bio-based materials (e.g. BMELV, 2009).

6.2.2 Normative mechanisms

Normative mechanisms have been used to support bio-based materials in two areas: public procurement and plastic (carrier) bags. The USA and many European countries mandate bio-based/green public procurement, where preference is given to bio-based/environmentally friendly products over traditional ones when purchasing for public administrations.

In the USA, the ‘Federal Biobased Products Preferred Purchasing Program (FB4P)’ started in 2002 for federal expenditures of at least US\$ 10,000 per year, stipulating that materials on the programme’s list are to be preferred over unlisted materials when federal purchases are made. Materials are considered if they are available, meet performance standards, are cost-competitive and are at least partially bio-based. Preference has to be given if they have been approved (‘designated’) by the US Department of Agriculture (USDA) in one of several rounds. To be approved, the materials have to consist fully or by significant parts of bio-based resources, the level of which is determined by ASTM D6866, which determines the share of bio-based carbon in total organic carbon of the materials. However, the exact requirements which these materials have to fulfill — in terms of bio-based content but also e.g. in terms of CO₂ savings — are not uniformly set but instead are determined by the US Department of Agriculture (USDA) for each round (USDA, 2005). The first round of designated items included diesel fuel additives, roof coatings, bedding and hydraulic fluids, the second round added, among others, composite panels and plastic insulating foam, the third round carpets and disposable cutlery, the fourth round bathroom and spa cleaners, metalworking fluids and wood & concrete sealers, and the fifth round lubricants, cleaners and corrosion preventatives (USDA, 2010). To our knowledge, no evaluation is yet available regarding the effectiveness of this policy.

In Japan, the national government’s ‘Biotechnology Strategy Guidelines’ and the ‘Biomass Nippon Strategy’ mandate that 20% of all plastics used in the country must be bio-derived by 2020 (Schut, 2009; Inomata, 2009). This has contributed to increased innovation and production of bio-based plastics for this market, and for Asia in general.

The European Commission has encouraged the drawing up of National Action Plans on Green Public Procurement (GPP) since 2003 (EC, 2003c) and emphasised its importance by another communication on GPP in 2008 (EC, 2008b). This has led to numerous initiatives on GPP in Europe: the UK published a National Action Plan for public procurement (Defra, 2007), aiming at a leading role within the European Union by 2009. Many other European countries such as Finland, Belgium, Slovakia also have green public procurement rules in place (EC, 2009c), specifying percentages of public procurement that should be carried out through GPP. France, Denmark and Austria also specify quotas for GPP, but these vary according to product groups. Italy specifies that 30% of goods purchased through public procurement have to fulfill ‘ecological criteria’, which could potentially be used as an argument for bio-based materials.

To discuss a specific example, in the Netherlands, sustainable public procurement is endorsed through a general national action plan on sustainable development (2003). The policy’s goal is to stimulate the market for sustainable products and to lead by example. ‘Sustainable’ encompasses social as well as environmental aspects, and the focus is on sustainable products, not necessarily bio-based ones. Clear targets for sustainable procurement have been set at different levels for the year 2010: 100% sustainable procurement at national level, 75% at local level and 50% at provincial level (Senter Novem, 2009b). Results of monitoring indicate that in 2008, only 25%, 27% and 26% of procurement was carried out following these guidelines at municipal, provincial and national level respectively (VROM, 2009), which suggests that the targets may not be reached. This may partly be due to the lack of established criteria for numerous product groups at the time

(2008). In July 2009, a new proposal for criteria was accepted by the Dutch government, detailing the criteria for 52 product groups. Examples of bio-based products considered for public procurement are hydraulic fluids and curtains from bio-based viscose (Senter Novem, 2009a).

The most common (and best-known) normative measure in the context of biomaterials is the plastic carrier bag ban. Numerous UK local communities, Macedonia, China, India, Bangladesh, Bhutan, South Australia, parts of Africa (South Africa, Eritrea, Rwanda, Somalia, Tanzania, Kenya and Uganda) and several US-American cities (such as San Francisco, Washington DC and Los Angeles) already have plastic carrier bag bans in place, other countries are discussing them (MCA, 2008; Packaging Europe, 2009; Kampagnen Kollektiv, 2007). Some of these bans exempt plastic bags that are made from renewable resources and/or are biodegradable (e.g. San Francisco). Many bans were established not necessarily to support the biomaterials industry but rather to combat an environmental problem, such as the clogging of waterways by plastic bags during heavy monsoon rainfalls in India and Bangladesh (Roach, 2008). The Spanish National Integrated Waste Plan (Plan Nacional Integrado de Residuos; Spain, 2009) stipulates that the use of single-use plastic bags must be reduced by 50% (by weight) by 2010 and that at least 70% of the remaining single-use plastic bags should be made from renewable resources by 2015. Similarly, the United Arab Emirates are phasing out plastic shopping bags by 2013 (Abdullah, 2010).

The plastic carrier bag bans target a relatively small market, but one which is tangible and easily understandable for consumers. Reliable statistics on plastic bag use and disposal are scarce, but a rough calculation based on an annual per capita consumption of 250–300 bags in developed countries (Dunne, 2004; Allan, 2005; Tough, 2007; Convery et al., 2007, higher in the US) and a bag weight of 6–18 g¹ results in a per capita consumption of 1.5–5.4 kg². Annual plastic packaging consumption in Europe is at 60 kg per capita (derived from PlasticsEurope, 2008, total plastics consumption per capita is 160 kg), so plastic carrier bags make up roughly 3–9% of that consumption by weight. Most bags are currently made from polyethylene, which can be substituted by bio-based plastics while reducing emissions to the environment from the plastics sector (see Chapter 3 and Murphy et al., 2008).

In Italy, legislation was passed at the beginning of 2008 which mandates the use of biodegradable bags or reusable containers for the collection of the degradable organic fraction of household waste (Parlamento Italiano, 2008). Using a biodegradable bin liner means that the liner can be collected and undergo waste treatment together with the waste, thus simplifying it. This law has boosted the demand for bio-based bin-liners.

Recently, a covenant was signed by the French Ministry of Ecology and Sustainable Development, the plastics, bioplastics and packaging industry associations, the French mayors and the federation of commerce. This framework agreement aims at promoting

¹Supermarket carrier bags were stated to be 5–7 g on average by Allan (2005) and 7–8 g on average by Ayalon et al. (2009), but Allan (2002) gave a range of 6, 18 and 35 g (the latter for reusable bags) and we therefore calculate with a range of values for the average weight.

²Worrell (1994) calculated 1.3 kg per capita.

the use of bio-based and biodegradable plastic waste bags in France. It stipulates that such bags should contain 40% renewable material in 2010 and 70% by 2018 (Eyre, 2009).

6.2.3 Economic mechanisms

6.2.3.1 Cap-and-trade systems

The European Emissions Trading Scheme (ETS) was not originally designed to affect biomaterials but to reduce CO₂ emissions from the energy sector and the manufacturing industry. The ETS has only included petrochemicals from the second (2008–2012) trading period onwards, and has since impacted the production of petrochemicals³. From 2013 onwards, the scope of considered emissions from the petrochemicals sector will broaden: during the second period it was only propylene and/or ethylene production (i.e. steam cracking) above 50 kt/year. Starting in 2013, the ETS will also cover N₂O emissions from nitric, adipic and glyoxylic acid and glyoxal production as well as CO₂ emissions from all petrochemical production processes if they are produced in large installations with a capacity of more than 100 t/day (EC, 2009b). This will more directly affect the competition between petrochemicals and bio-based materials, favouring the latter because more petrochemical installations will fall under the ETS than before⁴. However, the (bulk) chemicals market is a global one, and the ETS price does not necessarily lead to a correspondingly higher product price of petrochemicals because of global competition, and the costs derived from the carbon costs will therefore not necessarily be reflected in the products' market prices. The directive calls for an analysis of and proposal on how to deal with exposed industries such as the global petrochemical industry by early 2010, which may include possible adjustments to the share of free allowances (EC, 2009b). We model the effect of carbon prices in a carbon tax case study in section 6.5.

Another important aspect of the ETS is the exclusion of carbon emissions from hazardous as well as municipal waste incinerators (EC, 2009b, Annex I) throughout all trading periods. In relation to materials, this means that carbon which is stored in (petrochemical) materials during use but which will be released as CO₂ when they are incinerated, is not part of the ETS and is therefore also not priced. As a consequence, there is no incentive from the ETS to reduce material carbon emissions and limited support for the bio-based material industry.

³During the second trading period most allowances were allocated free of charge. The directive for the third trading period in its current form provides for 80% free allowances to be given initially and for reducing these evenly to 30% by 2020 aiming at zero by 2027 for the industrial sector.

⁴For the case study on the carbon tax (see section 6.5), we assume that bio-based plants fall under the ETS but in reality this is a complex issue that depends among others on the individual plant's capacity, the plant's combustion capacity as well as if such plants are 'opted in' nationally, and some plants may therefore be not fully covered by the ETS or not covered at all.

6.2.3.2 Subsidies

Germany subsidised insulating materials from renewable resources from 2003 until 2007 and biogenic lubricants and hydraulic fluids from 2001 until 2008 (BMELV, 2009). Both were market introduction schemes and as such designed to have a limited duration. The subsidy level was 25-30 € per m³ of insulating material, for quantities above a threshold value of 5 m³. The use of the subsidised bio-based insulating materials increased by a factor of 7 for hemp (14,000 m³ in 2003 to 100,000 m³ in 2007, equivalent to 500–4,000 tonnes and a market share of 0.4%), coupled with a decrease in market prices (Carus et al., 2008). Bio-based lubricants and hydraulic fluids both managed to increase their market shares as well: lubricants went from a share of 2.5% in 1999 to 7.6% in 2005 and hydraulic fluids from 5.7% to 7.1% in the same time-frame (Theissen, 2006).

6.2.3.3 Material taxation

This section deals with taxes imposed directly on materials, whether they are bio-based or petrochemical. Taxes on CO₂ emissions from materials and (reduced) value-added tax have not been applied in the context of biomaterials. There are three groups of materials that are commonly taxed: packaging materials, single-use materials and plastic bags.

A deposit system and packaging tax are in place in Germany and The Netherlands respectively, with both countries differentiating between bio-based and conventional materials. The German packaging law exempts plastic packaging from the return system until December 2012 if they are produced from biodegradable materials and all components are certified biodegradable. Additionally, as of April 2009 and also valid until December 2012, the law exempts all single-use beverage containers from the deposit (0.25 € per container) if they contain at least 75% bio-based materials (BMU, 2008b).

The Dutch packaging tax was introduced in 2008 and differentiates between petrochemical plastics, bioplastics, paper and a number of other materials. Originally, the bioplastics tax was set at 50% of that for petrochemical plastics, but in 2009 it was reduced further to the level of paper and cardboard – in detail: 0.07 €/kg vs. 0.43 €/kg for petrochemical plastics (Belastingdienst Nederland, 2009). For 2010, the taxes have been slightly increased for bioplastics and paper & cardboard to 0.08 €/kg and to 0.47 €/kg for petrochemical plastics.

Regarding taxes on single-use materials, Belgium introduced a ‘picnic tax’ in 2007 (Belgium, 2007b), after debating the introduction of a more general packaging tax. This tax is intended to make single-use products more expensive, targeting plastic bags, wrapping and aluminium foil as well as single-use plates, cups and cutlery (hence the name ‘picnic tax’). Certified biodegradable plastic bags are explicitly excluded from the tax (Belgium, 2007a). Tax levels are based on environmental profiles of materials, charging disposable plastic bags 3.0 €/kg, disposable cutlery 3.6 €/kg and aluminium foil 4.5 €/kg, among others.

Taxes on plastic carrier bags were introduced in many countries (e.g. Ireland, Canada and Taiwan), primarily as a means to reduce plastic waste and littering. The plastic bag

tax in Ireland ('PlasTax') was one of the earliest material taxation measures world-wide (2003) and was originally implemented to reduce littering. The tax applies to all plastic bags, irrelevant of the material used, and its level was increased from 0.15 € to 0.22 € per bag in 2007 (roughly 8–36 € per kg material for an average bag weight of 6–18 g); discussions are ongoing to increase the tax further in order to keep plastic bag consumption low. It is one of the few measures that was evaluated: Convery et al. (2007) quantified revenues from the tax at more than 30 times the costs of implementation (12–14 million € in annual revenues, ca. 3% of which are spent on collection and administration costs).

Some countries exempt bio-based plastics completely from such taxes, e.g. Romania (Packaging Europe, 2008). Other countries differentiate between biodegradable and non-degradable bags: Malta originally had tax levels of 0.01 € per bag for degradable bags, exempted plastic bags made from 100% organic material, and taxed conventional bags at 0.06 € per bag (Debono, 2006), the latter roughly corresponding to 10 €/kg material. The tax was changed in February 2009 and now amounts to 0.15 € on all plastic bags, irrespective of degradability (Debono, 2009). Savings in plastic bags as a result of material taxation are considerable: Malta reportedly saved 25 million plastic bags (i.e. more than 50%, corresponding to roughly 150 tonnes of plastic) in the first two years after introducing the tax, Ireland reports savings of more than 90% in plastic bags (roughly equal to savings of 7,000 tonnes of plastic). Hong Kong also introduced a tax of 50 cents (0.04 €) on plastic carrier bags in July 2008 (Hong Kong, 2009), it does not distinguish between bio-based or biodegradable and conventional materials. In November 2002, an awareness campaign began in Tokyo's district of Suginami (Edahiro, 2008): customers who refused a plastic bag in any of the participating local stores received a small sticker, 25 of these could be exchanged for a voucher of 100 Yen (equals 0.85 €/25 bags). Where all plastic carrier bags are taxed equally, this will lower the consumption of all plastic bags, conventional as well as bio-based/biodegradable ones.

6.2.3.4 Summary of the economic instruments discussed

We compare the economic instruments discussed so far on the basis of the incentive they provide to reduce CO₂ emissions. However, it should be noted that this is a simplification, because there are other environmental effects which these policy measures aim at. A comparison of the main part of Table 6.1 with the last row, which presents the calculations for the European Emission Trading Scheme, illustrates that the range of implicit CO₂ prices for biomaterials are much higher than the current market price for CO₂, differing by orders of magnitude. The table also shows that economic incentives vary considerably when they are translated into €/t CO₂. Most attention has gone out to measures mitigating the consumption of plastic carrier bags, most likely due to public perception of plastic bags as a major source of (visible) environmental pollution and their short use time, and the resulting support for such policy measures.

6.2.4 Communication mechanisms

6.2.4.1 Labelling

Labelling is one aspect of the US-American FB4P (see section 6.2.2): products that qualify for the FB4P programme are allowed to carry the label 'USDA certified biobased product' (USDA, 2009a). Products must fulfill the requirements of a US Department of Agriculture (USDA) 'designated item category' if the type of product has already been considered during one of the FB4P rounds, or else be at least 51% bio-based, with lower percentages possible upon application (USDA, 2009b). This measure is still under development, first labels are expected to appear on products in early 2010 (USDA, 2009a). Likewise, the European Union's Eco-Labeling scheme has stipulated requirements for renewable carbon contents for different classes of lubricants since 2005: $\geq 50\%$ for hydraulic and two-stroke oils, $\geq 45\%$ for greases and $\geq 70\%$ for chain-saw oils (EC, 2005), and revisions are under way which may include requirements for detergents and plastics as well as other bio-based products.

Carbon labelling has been on the rise in the UK, France, USA, Canada, and Switzerland and many other countries are following suit or considering to do so (Stancich, 2008). Although carbon labels are not directly aimed at strengthening biomaterials, most carbon labels require continuous reduction of the carbon footprint (i.e. greenhouse gas emissions) for products to retain the label (see e.g. Carbon Trust, 2010) and may therefore strengthen biomaterials indirectly.

6.2.4.2 Information Campaigns

In the UK, several measures have been implemented to increase public awareness of the bio-based economy, including the biomaterials sector; in November 2003, the National Non-Food Crops Centre (NNFCC) started working as a source of information on products and technologies using renewable resources.

6.2.5 Conclusions on existing policy measures

In general, most existing policy measures have only recently been implemented and are not based on a comprehensive policy strategy regarding biomaterials. Of the policy measures that have been implemented so far, the underlying policy goals vary: the most important environmental aspects for the implementation of policy measures in this field appear to be 1) the reduction of waste and littering, 2) the strengthening of an immature but promising market and 3) the reduction of greenhouse gas emissions. The materials targeted by existing policy measures vary as well, with some measures directed at ensuring that the *source* of the materials is renewable (bio-based materials) and other measures targeting the *end-of-life* stage of the materials to ensure that they are biodegradable. In addition, many measures target very specific product groups, most notably plastic bags. In Table 6.2 we provide an overview of the incentives from existing policy measures described

Table 6.1: Overview of implicit CO₂ prices of economic instruments; all CO₂ prices related to plastic carrier bags are estimates based on an average bag weight of 6–18 g; CO₂ savings calculated on the assumption that biomaterials save roughly 30% compared to petrochemical.

Material	distinguish	country	year	level of taxation €/unit	€/t CO ₂
Insulation	bio-based	Germany	2003–2007	25–35/m ³	310–440
Lubricants	bio-based	Germany	2001–2008	1.2/kg	380
Single-use plastic	biodegradable	Belgium	since 2007	3.00/kg	2,000
Beverage containers	bio-based	Germany	2009–2012	0.25/container	n/a
Plastic packaging	bio-based	Netherlands	since 2010	0.47/kg	260
Plastic bags	bio-based	Malta	until 2009	0.06/bag	200–600
Plastic bags	degradable	Malta	until 2009	0.01/bag	160–490
Plastics bags	biodegradable	Romania	2003–2007	0.05/bag	160–470
All plastic bags	no	Ireland	since 2007	0.15/bag	n/a
All plastic bags	no	Ireland	since 2007	0.22/bag	n/a
All plastic bags	no	Hong Kong	since 2008	0.04/bag	n/a
All plastic bags	no	Malta	since 2009	0.15/bag	n/a
All plastic bags	no	Suginami, Japan	since 2002	0.03/bag	n/a

For comparison:
PP, PE

EU (ETS)

since 2008

15

above, attempting to quantify their respective impacts to date. This quantification is difficult because most policies have only recently been implemented and only few policy measures have been evaluated. In general, market penetration of biomaterials remains low and few policies seem to have had a significant effect.

6.3 Comparison of biomaterial policies with bioenergy policies

This section provides examples of policies supporting bioenergy and biofuels and we carry out a simplified quantification of the incentives that were implemented through them, to compare these with the current incentives for biomaterials.

Biofuels have been supported in the EU by the 2003 Biofuels Directive and will be supported further by the Renewable Energy Directive (EC, 2009e). Nationally, the French eco-tax (2005), the German fossil diesel tax (1999), the Spanish biofuel plant tax exemption until 2012, the Dutch biofuels obligation starting in 2007 (van Thuijl and Deurwaarder, 2006) as well as the US Renewable Fuel Standard (2005, part of the Energy Policy Act) and the Energy Independence and Security Act (2007) all support biofuels as well, in many cases mandating a minimum share of renewable fuels in a given year.

In the US, there is also a federal production tax credit for biomass power valid through 2013 (REN21, 2009) as well as income tax credits and loan guarantees for small ethanol producers (USDA, 2008). The US Renewable Fuels Standard provides excise tax credits to ethanol and biodiesel producers amounting to US\$ 0.51 and US\$ 1.00 per gallon respectively, with an additional US\$ 0.10 per gallon for the first 15 million gallons of ethanol for small producers, where the latter are defined as having an annual capacity of less than 60 million gallons (USDA, 2008). On a sub-national level, all US states have an ethanol promotion programme in place (USDA, 2008) and several mandate a minimum share of bioethanol and/or biodiesel through a Renewable Portfolio Standard, such as E10 in Iowa and B5 in New Mexico (REN21, 2009). Other examples of state policies are: exempting transportation biofuels from state excise taxes, assigning tax credits for the use or sale of transportation biofuels, and requiring the state fleet to purchase transportation biofuels (USDA, 2008).

The European Union has import duties in place amounting to € 19.2 and 10.2 per hectolitre for un-denatured and denatured ethanol respectively and a rate of 6.5% for biodiesel. However, most imported ethanol entered the EU duty-free in the period 2002–2004 under trade schemes such as ‘Everything but Arms’, and crude palm oil (for non-food use) and crude soybean oil can be imported at zero and 3.2% respectively (Swinbank, 2009), supporting the use of biofuels. Biomass use to produce electricity and heat has been stimulated on the EU level for example by the 2008 climate & energy package, and on national levels e.g. by the German Renewable Energy Law (EEG; BMU, 2004), the US Energy Policy Act (US EPA, 2005), national targets in at least 73 countries and policies for renewable electricity in 64 (REN21, 2009), mostly through feed-in tariffs. These show

significant variation, with France paying 45 €/MWh, small-scale producers in Germany receiving up to 120 €/MWh and many other countries within that range. For larger-scale feed-in in Germany (≥ 5 MW), the tariff of 78 €/MWh translates into an incremental subsidy of 9 €/MWh (calculated based on BMU, 2010).

Table 6.3 presents an overview of economic incentives for transportation biofuels based on existing policy measures. Calculations for transportation biofuels are based on the rough assumption of 35% savings based on energy content of the fuels, which is the minimum greenhouse gas saving required according to the sustainability criteria which will be implemented starting in 2010 (EC, 2009e).

Bioenergy policies have often explicitly aimed at reducing greenhouse gas emissions, increasing energy security and supporting rural development. This is in contrast with biomaterials, for which the same goals could be formulated but which have instead been centred mostly around waste reduction. When putting Tables 6.1 and 6.3 side by side, it becomes clear that the range of economic incentives for biomaterials is even wider than for bioenergy, with the highest levels of implicit CO₂ prices for policy measures aiming at reducing waste and littering problems (Belgian ‘picnic tax’ and Irish ‘PlasTax’). From the above and Table 6.2 also it becomes clear that, in general, many policies with economic incentives have been implemented to support transportation biofuels but only a limited amount have been put into place for biomaterials.

In terms of total government expenditure, there are fundamental differences between bioenergy and biomaterial policy measures: for example, the German government spent 3 million € annually (2003–2007) on bio-based insulation material subsidies, and 4 million € annually (2001–2008) on bio-based lubricant subsidies (FNR, 2009). For comparison: US subsidies for bioelectricity were US\$ 246 million in 2007 (EIA, 2008), with renewable electricity a total of US\$ 1,000 million. The differential (incremental) payments for the feed-in scheme in Germany were at the level of 790 million € in 2009 alone (BMU, 2010), solid biomass for electricity use constituting roughly 100 million €. The comparison with transportation biofuels is based on average subsidy levels derived from Table 6.3 and total volumes of production according to REN21 (2009). The results indicate that spending on biodiesel subsidies in Europe is in the order of 2,400 million € and 1,000 million € for bioethanol annually⁵. Although these numbers on spending are estimates, this shows that subsidies on bioenergy are at a different order of magnitude compared to financial incentives for biomaterials.

⁵Volumes of production are much higher for biodiesel than for bioethanol in Europe: 8 billion litres vs. 2.8 billion according to REN21 (2009).

Table 6.2: Summary of existing policy measures with a direct effect on biomaterials; impact on biomaterials can only be assessed qualitatively at this point (0: no effect, +: slightly positive effect)

Programme	targeted material	policy goal	country	year	impact to date on biomaterials
FB4P	bio-based materials	support bio-based	USA	since 2002	+
GPP	sustainable products		NL	since 2003	0 ^a
Bans	all plastic bags	reduce waste & littering	several		0 ^a
Mandate	biodegradable bin liners	support biomaterials	Italy	since 2008	+
Mandate	bio-based plastic bags	reduce waste & littering	Spain	starts 2015	0 ^b
ETS	CO ₂ emissions	tackling climate change	EU	2008–2012	0 ^a
Subsidies	bio-based insulation	increase market penetration	Germany	2003–2007	+
Subsidies	bio-based lubricants	increase market penetration	Germany	2001–2008	+
Tax	packaging materials	reduce packaging	NL	2008	0 ^b
Tax	non-degradable, "single-use materials	reduce waste & littering	Belgium	since 2007	0 ^b
Tax	all plastic bags	reduce waste & littering	several		0 ^a
FB4P labelling	bio-based materials	increase awareness	USA	starts 2010	0 ^c
Eco-labelling	bio-based materials	increase awareness	EU	since 2005	0 ^c
Carbon labelling	all materials	reducing environmental footprint	several		0 ^c
Campaigns	bio-based economy	increase awareness	UK	since 2003	0 ^c

^aNot specifically targeting biomaterials.^bOnly recently implemented.^cIncentive is not strong enough.

Table 6.3: Summary of existing economic incentives from policy measures for transportation bio-fuels, where incentives are quantified per secondary energy content (GJ) and per unit of carbon dioxide saved. Data is based on van Thuijl and Deurwaarder (2006); Wiesenthal et al. (2009); Matsumoto et al. (2009); Swinbank (2009); EREC (2009); Tyner (2007).

country	Biodiesel		Bioethanol	
	[€/GJ _e]	[€/t CO ₂]	[€/GJ]	[€/t CO ₂]
Austria	9	340		
Czech Republic	10	360		
France	10	370	19	650
Germany	14	520	33	1100
Japan			23	760
Spain			17	580
Sweden	12	440	27	880
UK	9	330	15	510
USA	8	290	7–8	230–280
USA ^a			14	450

^a for fuels derived from cellulose

6.4 Policy barriers

Upon close examination, several existing policy measures in their current form put bio-based materials at a disadvantage. Many of these occur at the level of the feedstocks used for bio-based material production, either through competition for the same biomass such as biomass which can also be used for bioenergy, or through unequal tax systems such as on petrochemical vs. bio-based feedstocks for the production of chemicals. In the following, we discuss several examples of such measures, however, the list is not exhaustive.

At the level of material feedstocks, a discrepancy exists between petrochemical and bio-based materials: the imports of crude oil and petrochemical feedstocks are duty free and unrestricted in terms of quantities (IEA, 2008), whereas the imports of agricultural crops which are currently used as feedstocks for the production of bio-based materials are subject to an import duty. The level of the import duty for agricultural crops depends on the country of origin as well as the levels of import and the set EU intervention prices. This leads to a disadvantage for bio-based materials, also when comparing to imports of transportation biofuels (see above). Additionally, crude oil and petrochemicals production are exempt from excise tax in the US and Europe (IEA, 2008), providing an economic advantage for petrochemicals production.

The European sugar policy was revised in 2006 with a view of liberalising the market completely by 2014 (EC, 2006a). The reformed, lower minimum guaranteed price of 400 €/t only applies to quota sugar, most of which is used in the food industry. The industrial use of sugar (only 2% of total sugar consumption in the EU-15 (EC, 2006b)) falls under out-of-quota sugar, for which market prices spiked in 2007/2008 (MCCOAM, 2010). This means that before the sugar reform as well as during the first years since

then, European sugar feedstock costs have been higher than world market prices, and this has put the European bio-based material industry at a disadvantage if their starting point for production is fermentable sugar and not agricultural biomass (crops)⁶. The EU is therefore in the process of preparing a regulation which would allow ‘the duty-free import of 400 kt of sugar from the world market to be possibly used by the chemical industry in 2010/11 to ensure the long-term raw material supply planning for this sector’ (EC, 2010). In addition, importing raw sugar from Brazil, Australia and Cuba is to become easier for the year 2009/10. However, these measures are one-time implementations and a more structural approach is needed to support the availability of production feedstock for the existing biomaterials industry. This is especially true if the biomaterials industry is supposed to grow in the future, in which case 400 kt of sugar is very little⁷.

There are potential *indirect* economic effects from the implementation of policy measures to support bioenergy, given the size of that market and its fast growth in recent years. Several studies have indicated that land prices have increased as a result of such policy measures (e.g. Carus et al., 2010; Banse et al., 2008), but it is difficult to translate them into crop price increases, because the prices of agricultural products and plant residues depend on numerous factors such as increased demand from the food and feed sector and the yield and quality of harvests. This suggests that more detailed modelling will be necessary, including the interaction of policies on fuels, electricity and materials from biomass.

An example of a specific policy measure leading to a disadvantage for bio-based materials is the EU Waste Framework Directive (EC, 2008e): it provides definitions of waste, stipulates a waste hierarchy for Europe and introduces extended producer responsibility (according to the polluter-pays principle). Straw and similar materials are exempted from its coverage, if this is used for farming, forestry or energy production. However, if the same material is used for material production, it will be covered by the directive. This implies that for straw for material use, for example, records would have to be kept on quantity, origin and type of treatment of this ‘waste’. In parallel, biomaterials (both bio-based and biodegradable ones) are not included in the directive’s definition of bio-waste, suggesting that these materials will be excluded from efforts to divert waste from landfill & incineration to composting & digestion plants.

Another example of a specific measure which only works in favour of bioenergy is the European Union’s Common Agricultural Policy (CAP, EC, 2003b): multi-annual crops for use in material applications (such as short rotation crops or miscanthus) only received the premium from the ‘first pillar’ (i.e. direct farm payment) if they were grown on set-aside land. Energy crops were allowed to use non-set-aside land as well, and received an energy premium of 45 € per hectare up to a limit of two million hectares within the EU (2004–2009) for crops used in transportation biofuels, electric or thermal energy production (EC, 2003b). Both aspects have led to an economic disadvantage of using biomass resources for material purposes. As a result of the 2008 ‘Health Check’ of

⁶Recently, however, the EU reference price was lower than the world market price for refined sugar (London No.5) due to exceptionally good sugar harvests (more than 50% above quota) in Germany (EC, 2010).

⁷The production of 100 kt bio-based ethylene for example would consume roughly 400 kt of sugar (Chapter 2). Existing ethylene production capacity in Western Europe is 22,200 kt.

the CAP, the obligation to set aside 10% of the land was abolished and the energy crop premium terminated (EC, 2009a), opening up new opportunities for using biomass crops for biomaterial production. Additionally, the EU Budget Review, which is still ongoing, may lead to a re-definition of European objectives with the aim to achieve a fair balance between continuity and response to new challenges, including climate change, energy and migration (Landgrebe et al., 2009). It is thus also expected to have a significant impact on policy regarding agricultural production, bioenergy and, potentially, bio-based materials.

In terms of country-specific measures, Germany has regulations in place on bio-waste and fertilisers, which impact bio-based materials. The bio-waste regulation (Bioabfall-Verordnung, 2007) defines which substances may enter bio-waste treatment, how they are to be treated, what requirements the resulting products (e.g. compost) have to fulfill and how & where treated bio-waste may be applied to (agricultural) soils. The fertiliser law (Düngemittel-Verordnung, 2008) specifies the requirements for synthetic as well as organic fertilisers, to which compost derived from biodegradable materials belongs. The bio-waste regulation in its current form requires bio-based materials to consist of 100% renewable materials in order to be considered bio-waste at the end of their life cycle. Most materials, however, contain some petrochemical additives for technical reasons and thus do not qualify. The fertiliser regulation was updated in 2008 and no longer requires 100% renewable materials, but instead requires certification of compostability. A new version of the bio-waste law has been in preparation since March 2008, which would change the requirements for bio-based materials: in order to be considered bio-waste, they would have to be certified compostable or biodegradable and to predominantly consist of renewable materials. This modified version would eliminate the 100% rule and enable bio-based materials to be collected separately as bio-waste. However, this would lead to a new conflict: the current (2004) Renewable Energy Law (Erneuerbare-Energien Gesetz, EEG) stipulates that a renewable energy producer permanently loses the economic premium when producing energy from bio-waste, because anything that falls under the German bio-waste regulation was explicitly excluded. Once the new bio-waste rule is in place and compostable/biodegradable biomaterials fall under that regulation, it will prevent cascade use of biomass (first for material, then for energy) even though biomass cascading has been shown to improve CO₂ savings and mitigation costs compared to direct conversion into energy (Dornburg and Faaij, 2005).

An entirely different measure is the obligation to register and evaluate chemicals used and/or discharged according to the EU Regulation on the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH; EC, 2008c). The regulation holds for petrochemical as well as bio-based products, but ca. 30 bio-based materials are currently exempted (e.g. starch, sugars, cellulose pulp). New bio-based materials may be at a disadvantage compared with petrochemicals because of the lack of required information, for example regarding their toxicity (de Klerk and van Es, 2008) and because the costs of implementation are distributed over a smaller total volume of production as plant capacities are generally much smaller than for petrochemicals. This implies that, comparatively, more efforts will have to be undertaken by the producers of such materials to fulfill the REACH requirements and thus to keep or gain access to the European market. The same report also notes that most companies interviewed did not consider REACH an incentive

to look at substituting petrochemical materials by bio-based ones. This suggests that bio-based chemicals will not profit from the regulation unless supportive policy measures are implemented.

Overall, many policies supporting the use of bioenergy explicitly restrict any (financial) benefits to the use of biomass for energy purposes. As energy and materials compete for the same biomass or agricultural land, this puts bio-based materials at a disadvantage. Altogether, these existing policies act as significant barriers to the market penetration of bio-based materials. This may be because they date from a time when bio-based materials were not on the political agenda at all, or may be the result of simply forgetting this area when drafting the policy. Either way, these policies unintentionally but irrefutably penalise the use of biomass for material production. Irrespective of the cause of such disadvantages, it would be crucial to adapt such policies to also encompass the material use of non-food and non-feed biomass in order to create a level playing field.

It is difficult to describe how that level playing field should be reached. The most consistent method would be a global CO₂-tax, but this is hardly realistic. Without it, various different policy measures would be implemented in different countries, most likely with differing aims. In doing so, it will be important that biofuels and biomaterials will be supported in the same way. Considering the currently existing support for the former, an important contribution to a level playing field could be including biomaterials in these measures. This could be done as a first step towards unlocking policy support for biomaterials. Other measures, which may be more specifically geared towards biomaterials only, could then build upon these policies. With any biomaterials measure, care should be taken with the differentiation between 'new' (such as PLA and bio-PE) and 'old' (such as paper and wood) biomaterials and whether these should be supported in the same way. On the one hand, the new ones have to be supported more in order to help them establish themselves on the market, on the other hand, however, this has to remain within reasonable bounds: for example, it should be avoided that window frames are made from wood-based bioplastics instead of wood due to policy support. These delimitations to policy support for biomaterials will be specific to the geography and type of support measure chosen.

6.5 Case study of a potential policy measure: Carbon tax on materials

As yet, there are hardly any *ex ante* analyses nor *ex post* evaluations of the desired and/or undesired effects of existing or potential policy measures on biomaterials. We therefore present an *ex ante* assessment of the effect that a tax on fossil carbon dioxide emissions could have on the economics and market penetration of selected bio-based materials as one example of a potential policy measure. We base our calculations on the methodology and results of previous work, namely an economic and environmental analysis that systematically modelled and evaluated the present and future production processes of bio-based bulk chemicals (Chapters 2 and 3). In this article we consider eight bio-based bulk

chemicals and their petrochemical equivalents for our calculations: adipic acid, acetic acid, butanol, ethyl lactate, ethylene, polylactic acid (PLA), polytrimethylene terephthalate (PTT; partially bio-based) and succinic acid.

6.5.1 Economic data and calculations

Our previous analysis (Chapter 2) was based on a low crude oil price (US\$ 25/bbl). However, in the light of current oil prices we have updated the economic data so that the cost calculations of the bio-based and the reference fossil based bulk chemicals now represent an oil price of US\$ 70 per barrel. Plant sizes can differ significantly between bio-based and petrochemical production sites: we assume for all bio-based chemicals and polymers that the plants in which they are produced have a capacity of 100 kt per year, corresponding to large-scale bio-based production. For the petrochemicals production, however, we worked with real plant sizes, which can be much larger than 100 kt (up to 500 kt). We then calculated the production costs plus profits (PCPP, also known as profited production cost) based on production costs, investment costs (depreciation, including interest, and profits) and operation & maintenance costs. The PCPP is a proxy for the market price that we use because market prices do not strictly follow increases or decreases in oil price and thus production costs. The capital charge is the total of depreciation and profits (including interest) and is set at 30%; for a detailed description of method and input data see Chapter 2. We also carried out a sensitivity analysis using a lower capital charge of 15% for petrochemical installations based on the fact that many of these installations can redeem their value over a longer time span (see section 6.5.4).

We then applied a CO₂ tax to all materials, where only fossil or non-renewable carbon is taxed, whereas carbon from renewable resources such as biomass is not. The carbon price is currently around 15 € per tonne CO₂ (May 2010) and projected to be in the order of only 20 € per tonne throughout the third trading period (Graus et al., 2009) as a result of the economic crisis, but could increase above this level if the European targets on greenhouse gas emission reduction are increased, e.g. from 20% to 30%. We also calculated an extreme scenario with a price of 100 € per tonne, which is equivalent to the fine for non-compliance under the ETS scheme (see section 6.2.3.1).

The price of biomass used as an input to the production processes for bio-based chemicals influences the economics of such products to a large extent (see Chapter 2). Prices of fermentable sugars have seen significant variations in the past and are likely to see significant variations in the near and longer-term future and to continue to differ among world regions. We therefore carried out calculations for four price levels of fermentable sugar between 70 €/t and 400 €/t⁸. We calculated results for tax levels between 0 and 100 € per tonne of CO₂.

⁸We intentionally present results for a range of sugar prices because of the geographical and seasonal variability of these prices. The price of 70 €/t represents low, local sugar prices in Brazil, whereas 400 €/t represents a high, 6 month average of world raw sugar prices (contract 11, November 2009–April 2010, US\$ 490/t) as traded at the ICE Futures US (formerly New York Board of Trade; ICE, 2010).

6.5.2 Environmental data and calculations

For the bio-based chemicals, CO₂ savings depend on the source of biomass/fermentable sugar as well as the energy used during the chemical production process. We differentiate between maize, sugar cane and lignocellulosics as biomass sources and the eight bio-based chemicals mentioned above, using the data from Chapter 3. Similar to a carbon tax on energy, we only taxed emissions of fossil, non-renewable carbon. Methane emissions are also considered and converted to CO₂-equivalents but are very small. Putting a carbon tax on materials presents a methodological challenge that is not relevant in the case of energy: when a material leaves the production site (cradle-to-factory gate), there is still carbon stored in the material. The question therefore arises whether that carbon should be included in or excluded from the tax. Excluding the carbon stored in the material would imply that the carbon tax only deals with the production processes of a product, and any sort of waste treatment during which that carbon may be released is ignored; this case is what is currently considered in the ETS. On the other hand, including emissions from that carbon would imply that there is a carbon tax on all aspects of society (full carbon tax), also including waste treatment. We carried out calculations for both cases⁹. For the latter we assume that the material is burnt without energy recovery, i.e. that all the carbon is released from the material but no credits are given for heat and electricity production¹⁰.

6.5.3 Results

The economic advantages or disadvantages change by very little for most bio-based bulk chemicals when a carbon tax is implemented. When considering current technology, the PCPPs of bio-based chemicals are lower than their petrochemical counterparts even without a carbon tax for ethyl lactate, for PLA (when compared to PET), for partially bio-based PTT (compared to either Nylon-6 or petrochemical PTT) and for succinic acid. This means that the respective bio-based products are economically advantageous compared to their petrochemical equivalents even without a carbon dioxide tax and remain advantageous independent of its level. However, the PCPPs of bio-based chemicals are significantly higher than their petrochemical counterparts for acetic and adipic acid, for

⁹From the point of view of carbon accounting and life cycle assessments, all carbon should be included and this would mean considering any carbon (also bio-based C) stored in the material. (This is what we do in Chapters 3–5.) In the context of the European ETS, any emission of embodied carbon from fully or partially petrochemical materials is excluded, and so is the sequestration of bio-based carbon. We follow the latter approach here to be consistent with the ETS.

¹⁰This is not a representative case for solid waste incineration plants in Europe (EU-25), where most incineration plants recover energy, with the net average export of power representing approximately 11% of the total gross calorific value (higher heating value) of the waste fed to the plant and the respective percentage for the net export of heat approximately 22% (Reimann, 2006). Nevertheless, the results are not affected by whether we consider energy recovery in incineration: most materials considered here have identical petrochemical equivalents, so the heat and electricity recovered from their waste will be identical too and thus will not influence the difference between PCPPs. In those cases where materials are compared based on functional equivalence (e.g. PLA vs. PET & PE), the energy content of the bio-based materials is generally lower than that of petrochemical materials, and therefore also credits for heat and electricity production (e.g. the HHV for PLA is roughly 19 MJ/kg, for PET 23 MJ/kg and PE 50 MJ/kg); see section 6.5.4 for a sensitivity analysis.

PLA when compared to PS, and somewhat lower for butanol (current technology). The levels of the carbon tax studied here (0–100 €/t CO₂) do not make current technology acetic and adipic acid economically viable. PLA (compared to PS) and butanol only become economically advantageous at extremely high levels of carbon dioxide prices as compared to current prices, for a sugar price of 200 €/t this is 400 and 600 €/t CO₂ and for a sugar price of 400 €/t it would be roughly 600 and 1000 €/t CO₂ respectively. For future technology, the comparative advantages and disadvantages hold, irrespective of the carbon tax, for acetic acid, ethyl lactate, PLA, PTT and succinic acid.

Two cases where the PCPPs of petrochemical and bio-based chemicals are comparable and where the carbon tax therefore would have an effect are current technology ethylene and future technology adipic acid, starting at a tax level of ca. 10–40 € per tonne CO₂. Figure 6.1 shows this comparison of petrochemical vs. bio-based materials for the system boundaries cradle-to-factory gate and cradle-to-grave (incineration without energy recovery) and a sugar price of 200 €/t¹¹. Note that the PCPPs of bio-based products using sugar cane as a feedstock decrease with increasing prices of CO₂; this is due to net greenhouse gas savings, i.e. credits for by-products such as electricity being larger than all emissions from processes and their material or energy inputs (cradle-to-factory gate). Note also that for bio-based materials the same values apply to cradle-to-factory gate (CF) and cradle-to-grave (CG) because incineration releases bio-based carbon which would not be taxed¹². From a carbon footprinting methodology point-of-view, the data for bio-based materials should by default be compared with the CG data for petrochemicals, but from the point-of-view of how the European ETS currently works, the comparison should be with CF data for petrochemicals.

With a production capacity of 22,200 kt/year in Western Europe alone, the ethylene market is very large; so if the carbon tax supports the introduction of bio-based ethylene it will have a very large effect on the CO₂ savings. The global potential greenhouse gas savings from bio-based chemicals at different levels of carbon dioxide prices are presented in Figure 6.2.

6.5.4 Sensitivity of results

Here we show the sensitivity of the results of this case study to important assumptions. The results of this case study have all been calculated based on a price of crude oil of US\$ 70/bbl. The results are sensitive to any changes in the crude oil price; see Chapter 2 for the effect of oil price on products' PCPPs.

By default, we used CO₂ data for materials considering they would be incinerated without energy recovery at the end of their life cycle. Figure 6.1 shows that whether the carbon tax is based on cradle-to-factory gate (CF), thus ignoring the carbon stored

¹¹At the higher and lower sugar prices that we considered (70, 135 and 400 €/t), the differences in PCPPs are so large that the carbon tax would not have an effect.

¹²For the system boundary CF, renewable carbon is stored (sequestered) in the product, but we assume likewise that no tax benefits are associated with it, see footnote 9.

6.5 Case study of a potential policy measure: Carbon tax on materials

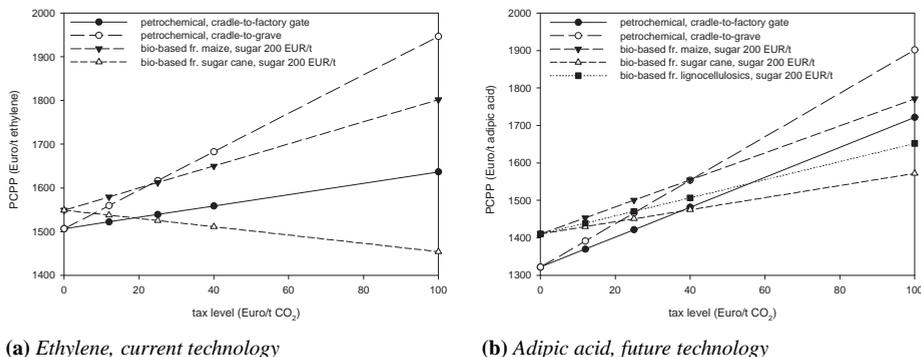


Figure 6.1: Effect of a carbon tax on the production cost plus profits (PCPP) of petrochemical and bio-based chemicals for three sugar sources: maize, sugar cane and lignocellulosics; system boundaries are cradle-to-factory gate and cradle-to-grave (incineration without energy recovery)

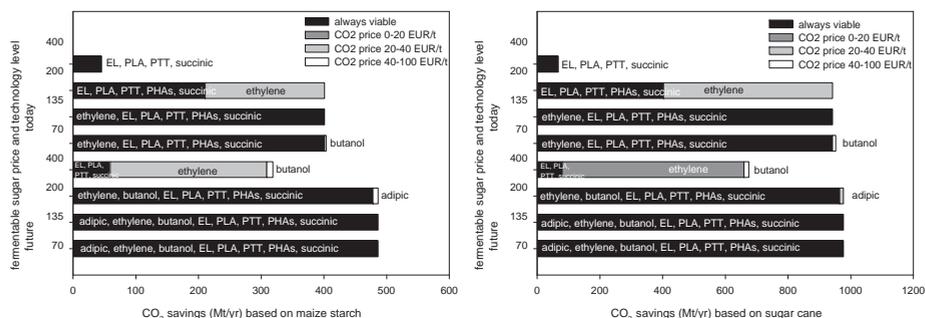


Figure 6.2: Effect of a carbon tax on global CO₂ savings from substituting petrochemicals by economically viable bio-based chemicals produced from maize starch (left) and sugar cane (right) for three carbon dioxide tax levels (0–20, 20–40 and 40–100 €/t), four sugar price levels (70, 135, 200 and 400 €/t) and two technology levels (today and future). Chemicals considered are adipic acid, acetic acid, butanol, ethyl lactate, ethylene, poly(lactic acid) (PLA), poly(trimethylene terephthalate) (PTT; partially bio-based) and succinic acid. Data on global installed capacity taken from Table 2.1, CO₂ savings are calculated including waste incineration of the material

in the material, or based on cradle-to-grave (CG) has an effect on the economic viability of ethylene and adipic acid above a CO₂ price of roughly 10–40 €/tonne. This methodological choice between basing a carbon tax on CF or CG data is significant for petrochemicals, but in most cases, the differences in economics of production which we calculated are so large that the carbon tax is ineffective independent of whether it is based on petrochemical CF or CG data.

However, for bio-based ethylene from sugar cane, the break-even points are roughly

5 and 18 €/t CO₂ depending on the system boundary (CF or CG). The current (May 2010) CO₂ price falls into this range at 15 € per tonne. The projected carbon price is approximately 20 € until 2020 (see section 6.5.1), or more than that if the European renewables targets increase (see section 6.5.1). Although the break-even points calculated here are very sensitive to changes in assumption, their relative values (roughly factor three) are comparatively robust. This difference means that whether or not carbon stored (CF vs. CG) is considered in the taxation or trading scheme is very likely significant. For an optimum support of bio-based materials and to minimise total greenhouse gas emissions, it would be desirable to consider the emissions during waste treatment as well, i.e. to set the system boundary at cradle-to-grave.

We also carried out a sensitivity analysis on the level of the capital charge: reducing it by 50% for petrochemical products (i.e. from 30% to 15% of the investment costs) only reduces the PCPP by 8% on average for a crude oil price of US\$ 70/bbl¹³. This difference is only significant for a few chemicals, where the PCPPs of the bio-based and petrochemical product are similar.

6.5.5 Conclusions regarding the carbon tax case study

Overall, this means that a carbon tax will only influence the market position of ethylene and its derivatives, but not that of most other bio-based materials considered here. However, ethylene has such a large market volume that the tax's potential effect on the market and on global greenhouse gas savings is significant. In Brazil, bio-based polyethylene (derived from sugar cane) is expected to start industrial scale production in 2010 (Shen et al., 2010), without the help of a carbon tax. Sugar cane-based bioethylene (current technology) competes with petrochemical ethylene at a carbon tax of 5–18 €/t CO₂. This means that, nominally at least, the existing policy measures discussed in section 6.2.3 are high enough to support increased market penetration. The reason why so far the existing stimuli have not yet led to wide-spread market penetration of biomaterials is probably that they are very specific measures, many of them for niche applications only (such as plastic carrier bags).

6.6 Conclusions

The inventory of existing policy measures shows that, so far, neither on the national level nor the supra-national level has there been a set of consistent and encompassing policy measures to support the introduction of renewable resources for material use. The inventory also demonstrates that most existing policy measures have only recently been implemented and vary in terms of the underlying policy goals: most policies aim at reducing waste and littering but not the reduction of greenhouse gas emissions. In addition, the

¹³The share of investment costs and thus also the sensitivity to the value of the capital charge will be higher for lower oil prices.

variation is also large when it comes to the material properties which are targeted: some measures target the *source* of the materials (supporting bio-based materials) and others target the *end-of-life* stage of the materials (supporting biodegradable materials). This is a significant difference and requires the materials' compliance with distinct standards.

The comparison of policies for biomaterials and bioenergy shows that policies for bioenergy are much more widespread and more consistent in geographic coverage of countries, and the range of economic incentives for biomaterials is even wider than for bioenergy, with very high implicit carbon prices for some plastic bag measures geared at low-weight, single-use bags. The case study on the effects of a carbon tax shows that a CO₂ price of 0–100 € would only impact ethylene, but not influence the market position of most other bio-based materials considered here; however, the market volume of ethylene is so large that the tax's potential effect would be significant, opening the door to potential savings in the order of 200 Mt CO₂/year (maize starch, current technology).

We have also identified significant barriers to the market penetration of bio-based materials which arise from existing policy measures, predominantly from laws and regulations on non-food and non-feed uses of biomass which restrict the policy support to biomass use for energy production. This may be the result of policies that date from a time when bio-based materials were not on the political agenda at all, or the result of simply forgetting this area when drafting the policy, but it unintentionally penalises the use of biomass for material production. Irrespective of the cause of such disadvantages, it would be crucial to adapt such policies so that they also encompass material use of biomass.

In conclusion, in order to increase the market penetration of biomaterials, it is important that policy barriers are removed to create a level playing field for the use of biomass, whether for energy or material purposes. Additionally, policy measures for biomaterials should move away from niche applications such as plastic bags and expand into more comprehensive measures such as general packaging and should receive more focussed attention in green/sustainable procurement efforts.

Summary

The central research aim of this thesis is *to investigate the environmental benefits and economic opportunities for bio-based materials and how these are affected by policy measures*. This is done by evaluating the entire life cycle of bio-based materials, from the production of bio-based chemicals, via the application of bio-based polymers in packaging and to the waste treatment of bio-based, biodegradable materials and by analysing how current and potential policy measures have influenced and can influence the opportunities for bio-based materials. For the economic analysis, the focus is on production costs and for the environmental analyses, it is on carbon and energy footprints, land and water use.

A discussion of the research context as well as important terminological and methodological clarifications are given in *Chapter 1*: with the world population growing from 6.5 billion in 2005 to approximately 9.1 billion by 2050 and economic growth expected to continue as well, the demand for materials is very likely to continue growing. Currently, more than 99% of plastics are produced from fossil, i.e. non-renewable, resources. When fossil feedstocks are used, their consumption is linked to a number of serious national, regional and global problems, such as the depletion of non-renewable resources, the dependence on imports of oil and natural gas or their derivatives, and the emission of greenhouse gases with their detrimental effects on the global climate. If the target of limiting global warming to not more than 2°C global warming compared to pre-industrial times, as laid down in the Copenhagen Accord, is to be reached, then a considerable reduction of global greenhouse gas emissions is necessary. This requires a major effort in terms of mitigating emissions from all sectors, including the materials sector.

Biomass can be used as a feedstock for the production of bio-based materials, an application which has so far received comparatively little attention from policy-makers despite the fact that the production of these materials may offer higher greenhouse gas savings per hectare of land than the production of bioenergy. As a renewable resource, it offers the

potential to counter-act the depletion of non-renewable resources, the import dependence and insecurity of supply, and climate change. This thesis therefore aims at contributing to a better understanding of the environmental and economic benefits across the entire life cycle of replacing conventional petrochemical materials by materials produced from biomass feedstocks, i.e. bio-based materials. Regarding the production of these materials, this thesis considers several biopolymers, such as polylactic acid, cellulose and paper, and focusses on the biochemical conversion of biomass to bulk organic materials using so-called white biotechnology (WB, which is the use of fermentation and/or enzymatic processes for the production of industrial products and fuels, potentially making use of breeding and genetic modifications of the micro-organisms used in fermentation). Feedstocks to produce the bio-based materials which are analysed in this thesis are readily fermentable sugars such as sucrose, dextrose and hydrolysed starches, representing current technology, and lignocellulosics such as miscanthus and maize stover, representing future (or ‘second generation’) technology.

Little information is yet available on the economic viability of the production of bio-based bulk chemicals and intermediates using white biotechnology. *Chapter 2* therefore determines to what extent bio-based bulk chemicals and intermediates offer economic opportunities compared with current petrochemicals, today and in the future. To this end, the chapter presents a techno-economic analysis of bio-based bulk chemicals production using white biotechnology. We detail a methodology to systematically evaluate the techno-economic prospects of present and future production routes of bio-based bulk chemicals produced using white biotechnology, which is based on available data and consistent assumptions regarding future (bio)technology developments. Current and future technology routes are evaluated for 15 products assuming prices of fermentable sugar between 70 €/t and 400 €/t and crude oil prices of US\$ 25/barrel and US\$ 50/barrel.

In general, a large number of WB chemicals are economically viable compared to their petrochemical equivalents, but this economic competitiveness depends to a large extent on the prices of crude oil and fermentable sugar. In more detail, the results show that WB-based ethanol, 1,3-propanediol, polytrimethylene terephthalate and succinic acid already offer economic opportunities for current state-of-the-art WB processes and a crude oil price of US\$ 25 per barrel. Ethylene and PLA are economically attractive at a higher crude oil price (US\$ 50/barrel).

Since we first conducted this study, the crude oil price has risen further and we therefore briefly discuss the results for a price of US\$ 70/barrel here (see Hermann et al., 2010, for more details). At this price, the following products are economically viable for current technology: 1,3-propanediol, polytrimethylene terephthalate, polylactic acid, succinic acid, ethyl lactate, ethylene and ethanol. For future technology, all studied products except for acetic acid are economically viable at fermentable sugar prices of 70–135 €/t. All analyses show that the level of technology has a large effect on the results: production cost plus profits of products directly obtained from fermentation are 20%–40% lower when assuming future instead of current technology; for products which require a chemical conversion step after fermentation, the production cost plus profits improve

by 15%–20% when considering future instead of current technology (for US\$ 70/barrel and across all sugar prices; similar values are true for lower oil prices). Technological progress in white biotechnology can thus contribute significantly to improved economic viability of bio-based products. If a comparison with other biomass-to-product conversion routes (e.g. thermochemical conversion) favours the production of WB-based bulk chemicals and if they also offer environmental benefits, a large-scale introduction would be desirable.

In principle, the use of renewable resources such as biomass for the production of materials can contribute to solving two important environmental problems: climate change and the depletion of fossil resources. However, there is a lack of reliable and consistent data-sets to quantify and compare savings by different materials. Based on the same principal approaches and data as detailed in chapter 2, *Chapter 3* quantifies the *extent* to which several bio-based bulk chemicals and intermediates offer environmental benefits compared with current petrochemicals. To do this, it presents the carbon and energy footprints as well as direct land-use for the production of these bio-based bulk chemicals. This chapter contains an analysis of current and future technology routes using white biotechnology to convert fermentable sugars derived from maize starch, sugar cane, or maize stover (lignocellulosics) into 15 bulk chemicals.

With the current state of the art in biotechnology, all products except for adipic acid and acetic acid allow significant savings in non-renewable energy and greenhouse gas emissions compared to petrochemical products. On average, savings are in the order of 30% and 65% using maize and sugar cane feedstocks respectively and current technology, and 50%–85% for future technology. With sugar cane as a feedstock, savings of more than 100% in non-renewable energy use and greenhouse gas emissions are already possible today, when the energy credits from co-combustion of waste biomass or from side-streams of agricultural or biochemical production are larger than the emissions and energy use associated with process inputs for the WB process chain. Substantial further savings (25%–35% higher in terms of greenhouse gas emissions than for current technology) are possible for the future by improved fermentation and downstream processing. To maximise savings in greenhouse gas emissions and non-renewable energy use by bio-based bulk chemicals, sugar cane is favoured over lignocellulosics, which in turn are preferable from an environmental point of view to maize starch as source of fermentable sugar.

The products with the highest savings (in % compared to their petrochemical counterparts) are acrylic acid, butanol (from the ABE process), ethanol, ethylene, PDO, and PHA. Worldwide greenhouse gas savings in the range of 0.5–1.0 Gt CO₂ per year are technically possible using future technology, considering only the 15 bulk chemicals selected for this study. This is a significant share of up to one third of the total annual CO₂ emissions by the chemical and petrochemical industry (3 Gt CO₂, estimate based on IPCC, 1996; IEA, 2009a,b). Together with the economic opportunities shown in Chapter 2, this builds a strong case for the production of bio-based bulk chemicals using white biotechnology. Employing white biotechnology to use biomass as a feedstock for chemicals production

hence offers excellent opportunities for mitigating greenhouse gas emissions and decreasing dependence on fossil resources and therefore could become an important strategy for the chemical industry's contribution to limit global warming.

The maximum *technical* potential (solely based on material properties) of bio-based polymers to substitute their petrochemical counterparts has been estimated to reach 90%, indicating that the material properties can enable a large-scale substitution of feedstocks for the production of bulk chemicals and plastics, if there are environmental and economic opportunities that can be seized. The question then becomes if there is enough land to produce these materials, and produce them sustainably. Other authors have extensively discussed the sustainability of feedstock production, or the lack of it, from the perspective of bioenergy production. This discussion applies equally to materials. The land requirements for bio-based materials were calculated by Dornburg et al. (2008) based on this thesis and the potential surplus agricultural land (Smeets et al., 2007). If we ignore the land requirements for bioenergy, the biomass potentials are easily large enough to enable a complete substitution of petrochemical materials by bio-based materials as discussed in this thesis.

Chapter 4 presents a full life cycle assessment for the use of bio-based plastics in packaging, an industry that uses nearly 40% of all polymers and makes use of very specific material functions (e.g. barrier properties). Our aim is to determine whether food packaging which is partly or wholly made from bio-based materials offers environmental advantages compared with the currently used petrochemical material. We do this through a case study on inner and outer packs for snack foods, where inner packs are in direct contact with food and therefore have to provide certain barrier properties, whereas outer packs are not. A comparison is made between bio-based materials such as paper, bio-based polyethylene, polylactic acid and another bio-based polyester on the one hand and conventional materials such as polypropylene and polyethylene on the other. The impact assessment is presented for non-renewable energy use, total energy use, global warming potential, depletion of abiotic resources, photo-oxidant formation, acidification, eutrophication, water use, and land use.

The comparison with the current, non-renewable materials (polypropylene) shows that, generally, films and laminates (i.e. multiple films laminated together) which (partly) consist of bio-based polymers offer opportunities for reducing environmental impacts of food packaging: 1. For inner packs, their environmental performance is in some cases equally good as that of the current material if the material is landfilled after use, and slightly better if incinerated with energy recovery. 2. For outer packs, many films and laminates offer environmental benefits compared to the current material, with a particularly low environmental impact for bio-based polyethylene film; bio-based polyester laminates also offer significant savings. 3. For inner as well as outer packs, laminates including polylactic acid offer environmental advantages when assuming a future technology level for polymer or film production; the environmental footprint of cellulose also improves significantly when increased technology maturity is assumed for the film production stage. Large variations in land-use are found across all packs, depending on the

type of bio-based material used.

Already today, the options with the lowest global warming potential are partly or fully bio-based. However, lack of experience and investment in converting bio-based polymers into final products and comparatively unfavourable material properties result in lower environmental advantages for some novel bio-based materials than one might expect on the basis of a comparison per kg raw material. Methodologically, this means that comparisons of bio-based materials with conventional materials using a functional unit of 1 kg do not sufficiently account for differences in material properties which become important for a specific application, in this case the materials' barrier properties. This also means that the savings calculated in Chapter 3 can become smaller when the comparison is not made per unit weight of the material but accounting for its functionality.

Overall, outer packs are environmentally more promising than inner packs. Starting with the outer packs when introducing bio-based wrappings to replace the current petrochemical material would allow investment and learning, from which inner packs could benefit. When considering that bio-based materials are likely to benefit more from technological progress than conventional materials because their production processes have been optimised less, this makes certain bio-based laminates potentially highly attractive options for the future.

Many life cycle assessments of bio-based and biodegradable materials neglect the post-consumer waste treatment phase because of a lack of consistent data, even though this stage of the life cycle may strongly influence the conclusions. The aim of *Chapter 5* was to approximate carbon and energy footprints of the waste treatment phase and to find out what the best waste treatment option for biodegradable materials is. We do this by modelling home and industrial composting, anaerobic digestion and incineration. We have compiled data-sets for the following biodegradable materials: paper, cellulose, starch, polylactic acid (PLA), starch/polycaprolactone (MaterBi), polybutyrate-adipate-terephthalate (PBAT, Ecoflex) and polyhydroxyalkanoates (PHA). We have done this on the basis of an extensive literature search, experiments and analogies with materials for which significant experience has been made.

It is a particularity of the biological waste treatment options that the materials are metabolised during the waste treatment process, so a part of their embodied carbon is emitted into air and the remainder is stored as compost or digestate. The compost or digestate can replace soil conditioners supporting humus formation, which is a benefit that cannot be achieved artificially. Experimental data on biodegradable materials shows a certain range across the amount of carbon stored of these materials, and more trials will be required in the future to reduce these uncertainties. Experimental data has also shown that home and industrial composting differ in their emissions of nitrous oxide and methane, but it should be noted that data availability on home composting is rather limited.

Of the waste treatment methods considered, anaerobic digestion has the lowest footprint for the current level of technology, but incineration may become better in the future if

energy efficiency in waste incineration plants improves significantly. Home composting is roughly equal to incineration with energy recovery in terms of carbon and energy footprint when carbon credits are considered. The same applies to industrial composting if carbon credits are assigned for compost which replaces straw. Carbon credits can therefore considerably affect the results, but there are significant uncertainties in how they are calculated. More research is therefore needed on quantifying the long-term carbon benefits to soils from adding soil conditioners such as compost. Additionally, the lack of knowledge regarding long-term soil carbon dynamics implies that there is also uncertainty around the carbon footprints of agricultural and silvicultural production of feedstocks for bio-based materials. Despite these uncertainties, however, biological waste treatment options should be chosen to safeguard soil carbon when that is a limiting factor, despite incineration with energy recovery possibly performing better in the future due to efficiency improvements.

The use of renewable resources for the production of materials offers higher greenhouse gas savings than bioenergy per kg feedstock as well as per hectare and has received some attention from policy-makers, but analyses of either existing or potential policy measures are almost entirely missing in this field. Chapter 6 investigates how current and potential policy measures affect market penetration of bio-based materials. In general, market penetration of biomaterials remains low and few policies seem to have had a significant effect. Our inventory of existing measures reveals a lack of consistent and comprehensive policy measures to support the introduction of renewable resources for material use. Furthermore, most measures have only recently been implemented and vary in terms of the underlying policy goals: most policies aim at reducing waste and littering and not at reducing greenhouse gas emissions. In addition, the variation is also large when it comes to the material's life cycle stage which is targeted: some measures target the *source* of the materials (supporting bio-based materials) and others target the *end-of-life* stage of the materials (supporting biodegradable materials).

Our case study on the implementation of a carbon tax shows that the tax would only influence the market position of ethylene. However, its market volume is so large that such a tax would indeed have a considerable effect and lead to global greenhouse gas savings. We have also identified significant barriers to the market penetration of bio-based materials which arise from existing policy measures: many policies supporting the use of bioenergy explicitly restrict any (financial) benefits to the use of biomass for energy purposes. As energy and materials compete for the same biomass or agricultural land, this puts bio-based materials at a disadvantage. In order to increase the market penetration of biomaterials, it is therefore important that policy barriers be removed to create a level playing field for the use of biomass, whether for energy or material purposes. Additionally, the policy measures for biomaterials should be harmonised, and more measures introduced, one of which could be a carbon tax or another carbon pricing scheme. Finally, policy measures for new biomaterials should move away from niche applications such as plastic bags, expand into targeting more comprehensive applications such as general packaging, and receive more attention within green/sustainable procurement efforts.

Overall, little was known at the start of these studies regarding the environmental

impacts of bulk chemicals production from biomass and whether they could be produced economically. We have therefore analysed the entire life cycle of biomaterials: the production of bio-based chemicals, the application of bio-based polymers in packaging and finally the waste treatment of biodegradable materials. Numerous bio-based chemicals offer economic opportunities, the extent of which depends on the prices of the petrochemical and bio-based feedstocks and can be further improved by technological progress in the future (chapter 2). Almost all bio-based chemicals have lower carbon and energy footprints than their petrochemical counterparts, and savings can be substantially increased in the future (chapter 3). Bio-based materials also offer savings when used for a specific food packaging application, but these savings can become smaller when the comparison is made not per kg of material but considering the functionality and material properties (chapter 4). Biodegradable materials are advantageous in that they are suitable for biological waste treatment options whose carbon and energy footprints are currently at least equally good as incineration, but have the additional benefit of producing a soil conditioner which can improve soil carbon content (chapter 5). Finally, consistent policy measures supporting bio-based and/or biodegradable materials are necessary to ensure this market's success (chapter 6). The progress made in terms of quantifying the benefits, the start of major research & development programmes and the start-up of production facilities have increased the likelihood for many bio-based chemicals and plastics to enter the market. So there are clear opportunities for biomaterials, but policy measures are needed to ensure that they can start competing on a larger scale.

Samenvatting

De centrale doelstelling van dit proefschrift is om *te onderzoeken waar de milieubaten en economische kansen voor biomaterialen liggen en hoe deze beïnvloed worden door beleidsmaatregelen*. Dit wordt gedaan door enerzijds de gehele levenscyclus van biomaterialen te evalueren, van de productie van biobased chemicaliën, via de toepassing van biobased polymeren in verpakkingen, tot de afvalverwerking van biologisch afbreekbare materialen. Anderzijds gebeurt dit door te analyseren hoe de huidige en potentiële beleidsmaatregelen de kansen voor biomaterialen hebben beïnvloed en zullen beïnvloeden. De economische analyse is gefocust op de productiekosten en de milieuanalyse op de broeikasgasemissies en het energie-, land- en watergebruik.

In hoofdstuk 1 wordt de onderzoekscontext bediscussieerd en de belangrijkste terminologie en methodieken uitgelegd. De wereldbevolking groeit van 6,5 miljard in 2005 tot ca. 9,1 miljard in 2050 en daarnaast wordt verwacht dat de wereldeconomie verder groeit. Dit betekent dat de vraag naar materialen hoogstwaarschijnlijk ook verder toeneemt. Op dit moment wordt meer dan 99% van al het plastic uit fossiele, niet-hernieuwbare hulpbronnen geproduceerd. Het gebruik van deze hulpbronnen wordt gerelateerd aan een aantal belangrijke nationale, regionale en wereldwijde problemen, zoals de uitputting van niet-hernieuwbare grondstoffen, de afhankelijkheid van geïmporteerde olie en gas of hun derivaten, en de emissie van broeikasgassen met zijn negatieve effect op het wereldklimaat. In het verdrag van Kopenhagen is de doelstelling opgenomen om de opwarming van de aarde niet hoger te laten worden dan 2°C boven het pre-industriële tijdperk. Als men deze doelstelling wil behalen dan is een duidelijke vermindering van de uitstoot van broeikasgassen nodig. Dit vraagt om een grote inspanning van alle sectoren om de emissies te reduceren, inclusief de materialensector.

Biomassa kan gebruikt worden als grondstof voor de productie van biomaterialen, een toepassing die tot nu toe relatief weinig aandacht van beleidsmakers heeft gekregen. Dit terwijl de productie van deze materialen hogere besparingen van broeikasgasemissies

per hectare land oplevert dan de productie van bio-energie. Omdat het een hernieuwbare grondstof is, biedt het de kans om de uitputting van grondstoffen alsmede de afhankelijkheid van import, de risico's rond voorzieningszekerheid en de klimaatverandering tegen te werken. Dit proefschrift wil daarom een bijdrage leveren aan een beter inzicht in de milieu- en economische winsten, die over de gehele levenscyclus te behalen zijn als conventionele, petrochemische materialen vervangen worden door materialen die uit biomassa geproduceerd worden, de zogenaamde biobased materialen.

Dit proefschrift analyseert de productie van meerdere van dit soort materialen, onder andere polymelkzuur (PLA), cellulose en papier. Het richt zich op de biochemische omzetting van biomassa in bulk organische materialen met behulp van de zogenaamde witte biotechnologie. Deze omvat het gebruik van fermentatie en/of enzymatische processen voor de productie van industriële producten en brandstoffen, mogelijk gebruikmakend van kweken en genetisch modificeren van de gebruikte micro-organismen. Grondstoffen voor de productie van biobased materialen die in dit proefschrift geanalyseerd worden zijn gemakkelijk fermenteerbare suikers als sacharose, dextrose en gehydrolyseerd zetmeel, die de huidige technologie representeren alsmede lignocellulosics zoals maïsstro en miscanthus die de toekomstige (of 'tweede generatie') technologie representeren.

Tot nu toe was weinig informatie beschikbaar over hoe economisch rendabel de productie van biobased bulkchemicaliën en grondstoffen met witte biotechnologie (WB) is. Hoofdstuk 2 beschrijft daarom in hoeverre biobased bulkchemicaliën en tussenproducten economische kansen bieden in vergelijking met de huidige petrochemische stoffen, vandaag en in de toekomst. Dat hoofdstuk laat daarom een techno-economische analyse van de productie van biobased bulkchemicaliën met witte biotechnologie zien. Een methode om systematisch de techno-economische vooruitzichten te kwantificeren van de huidige en toekomstige productieroutes van biobased bulkchemicaliën geproduceerd met WB wordt in dit hoofdstuk beschreven. Deze methode is gebaseerd op de huidige beschikbare gegevens alsmede consistente aannames ten aanzien van de toekomstige (bio)technologische ontwikkelingen. Routes van de huidige en toekomstige technologieën worden geëvalueerd voor 15 producten, ervan uitgaande dat de prijzen van fermenteerbare suiker tussen 70 €/t en 400 €/t liggen en de aardolie prijs tussen US\$ 25 en US\$ 50 per vat.

Over het algemeen zijn een groot aantal witte biotechnologie chemicaliën economisch rendabel in vergelijking met hun petrochemische equivalenten. De onderlinge concurrentie positie is sterk afhankelijk van de prijzen van ruwe olie en fermenteerbare suiker. In detail tonen de resultaten aan dat de huidige state-of-the-art processen met witte biotechnologie ethanol, 1,3-propaandiol (PDO), polytrimethyleen tereftalaat (PTT) en barnsteenzuur, bij een ruwe olieprijs van US\$ 25 per vat, vandaag al economische kansen bieden. Ethyleen en polymelkzuur zijn economisch aantrekkelijk bij een hogere aardolieprijs (US\$ 50 per vat).

Sinds de start van dit onderzoek is de prijs van ruwe olie verder gestegen. Daarom worden hier kort de resultaten besproken voor een prijs van US\$ 70 per vat (voor meer

details, zie Hermann et al., 2010). Bij deze prijs zijn de volgende producten met de huidige technologie economisch rendabel: 1,3-propaandiol, polytrimethyleen tereftalaat, polymelkzuur, barnsteen zuur, ethyl lactaat, ethyleen en ethanol. Met de toekomstige technologie zijn alle onderzochte producten met uitzondering van azijnzuur economisch rendabel voor fermenteerbare suikerprijzen van 70–135 €/t. Alle analyses laten zien dat het niveau van de technologie een grote invloed heeft op de resultaten. De productiekosten plus winst (PCPP) van producten die direct bij de fermentatie ontstaan zijn 20%–40% lager bij de toekomstige in plaats van de huidige technologie. Voor producten waarvoor een chemische omzetting nodig is na de fermentatie, verbetert de PCPP met 15%–20% (voor US\$ 70 per vat en alle suikerprijzen; vergelijkbare waarden voor lagere olieprijsen). Technologische vooruitgang in de witte biotechnologie kan dus aanzienlijk bijdragen aan de verbetering van de economische rentabiliteit van de biobased producten. Als een vergelijking met andere conversie routes naar biomassa-producten (bijv. thermochemische conversie) in het voordeel uitvalt van de productie van WB-gebaseerde bulkchemicaliën en als deze daarnaast voordelen voor het milieu bieden, zou een grootschalige introductie wenselijk zijn.

In principe kan het gebruik van hernieuwbare hulpbronnen zoals biomassa voor de productie van materialen bijdragen aan de oplossing van twee belangrijke milieuproblemen: klimaatverandering en de uitputting van fossiele hulpbronnen. Er is echter een gebrek aan betrouwbare en consistente gegevens om de besparingen door de verschillende materialen te kwantificeren en te vergelijken. Gebaseerd op dezelfde benaderingen en gegevens zoals beschreven in hoofdstuk 2 wordt in hoofdstuk 3 gekwantificeerd in hoeverre verschillende biobased bulkchemicaliën en tussenproducten voordelen voor het milieu bieden in vergelijking met de huidige petrochemische producten. Om dit te doen, worden de koolstof- en energiebalansen evenals direct landgebruik voor de productie van deze biobased bulkchemicaliën weergegeven. Hoofdstuk 3 bevat een analyse van huidige en toekomstige technologieën om met behulp van witte biotechnologie, fermenteerbare suikers, gemaakt van maïszetmeel, suikerriet of maïsstro (lignocellulosics), om te zetten naar 15 bulkchemicaliën.

Met de huidige state-of-the-art in de biotechnologie kunnen alle producten, met uitzondering van adipinezuur en azijnzuur, aanzienlijke besparingen bieden ten opzichte van petrochemische producten wat betreft niet-hernieuwbare energie en broeikasgasemissies. Gemiddeld zijn de besparingen, met de huidige technologieën, in de orde van 30% bij het gebruik van maïs en 65% bij suikerriet als grondstof en 50% en 85% voor de technologie van de toekomst. Bij het gebruik van suikerriet als grondstof zijn besparingen van meer dan 100% niet-hernieuwbare energiegebruik en broeikasgasemissies vandaag reeds mogelijk, indien de energie van verbranding van biomassa-afval, nevenstromen van de landbouwproductie of de biochemische credits groter zijn dan de emissies en het energiegebruik voor de WB procesketen. Aanzienlijke besparingen (25%–35% hoger wat betreft broeikasgassen dan voor huidige technologie) zijn verder mogelijk in de toekomst door betere fermentatie en productscheiding. De hoogste besparingen in broeikasgasemissies en niet-hernieuwbare energiegebruik door biobased bulkchemicaliën worden met suikerriet behaald, gevolgd door lignocellulosics, die op hun beurt beter zijn vanuit het

milieustandpunt dan maïszetmeel als bron van fermenteerbare suiker.

De producten met de hoogste besparingen (in % ten opzichte van hun petrochemische equivalenten) zijn acrylzuur, butanol (van het ABE-proces), ethanol, ethyleen, PDO en PHA. Mondiale broeikasgasreducties van 0,5–1,0 Gt CO₂ per jaar zijn mogelijk met behulp van toekomstige technologie, op basis van alleen de 15 bulkchemicaliën in deze studie. Dit is bijna eenderde van de totale jaarlijkse CO₂ emissie van de chemische en petrochemische industrie (3 Gt CO₂, een schatting op basis van IPCC, 1996; IEA, 2009a,b). Samen met de economische kansen die in hoofdstuk 2 werden aangetoond, is dit een sterk argument voor de productie van biobased bulkchemicaliën met behulp van witte biotechnologie. Het gebruik van witte biotechnologie om biomassa als grondstof in te zetten voor de productie van chemicaliën biedt dus uitstekende mogelijkheden voor het beperken van de uitstoot van broeikasgassen en het verminderen van de afhankelijkheid van fossiele hulpbronnen. Daarom zou het een belangrijke strategie kunnen worden voor de bijdrage van de chemische industrie aan de beperking van klimaatverandering.

Het maximale technische potentieel van bio-polymeren om hun petrochemische equivalenten te vervangen werd op 90% geschat, uitsluitend gebaseerd op materiaaleigenschappen. Dit geeft aan dat de eigenschappen van het materiaal een grootschalige vervanging van grondstoffen voor de productie van bulkchemicaliën en kunststof mogelijk maken, als de milieu- en economische kansen benut kunnen worden. De vraag is dan of er voldoende grond is om deze stoffen te produceren, en ze duurzaam te produceren. Andere auteurs hebben de duurzaamheid van de productie van grondstoffen, of het gebrek daaraan, uitvoerig onderzocht vanuit het perspectief van de productie van bio-energie. Deze discussie geldt ook voor materialen. De benodigde hoeveelheid land voor biobased materialen werd berekend door Dornburg et al. (2008) op basis van zowel dit proefschrift als de potentiële overschotten aan landbouwgrond (Smeets et al., 2007). Zonder de eisen aan hoeveelheden grond voor bio-energie mee te nemen is het biomassa potentieel gemakkelijk groot genoeg voor een volledige vervanging van petrochemische materialen door biomaterialen, zoals besproken in dit proefschrift.

Hoofdstuk 4 biedt een volledige levenscyclusanalyse voor het gebruik van biobased materialen voor kunststofverpakkingen, een industrie waarvoor bijna 40% van alle polymeren gebruikt wordt en waar zeer specifieke materiaalfuncties (bijv. barrière-eigenschappen) benut worden. Ons doel is om te bepalen of levensmiddelenverpakkingen die geheel of gedeeltelijk uit hoogwaardige biomaterialen bestaan voordelen voor het milieu bieden, in vergelijking met de momenteel gebruikte petrochemische stoffen. We doen dit door middel van een case-studie van binnen- en buitenverpakkingen voor snacks. Binnenverpakkingen staan in rechtstreeks contact met levensmiddelen en moeten daarom bepaalde barrière-eigenschappen hebben, terwijl dat voor buitenverpakkingen niet geldt. We maken een vergelijking tussen biomaterialen zoals papier, bio-polyethyleen, poly-melkzuur en een andere bio-polyester enerzijds, en de conventionele materialen zoals polypropyleen en polyethyleen anderzijds. De effecten worden beoordeeld en gepresenteerd voor het gebruik van niet-hernieuwbare energie, het volledige energiegebruik, de klimaatverandering, de uitputting van niet-hernieuwbare grondstoffen, foto-oxidant vor-

ming, verzuring, eutrofiëring en het water- en landgebruik.

De vergelijking met de huidige, niet-hernieuwbare materialen (polypropyleen) laat zien dat in het algemeen, films en laminaten (d.w.z. meerdere films samen gelaagd) die (gedeeltelijk) bestaan uit biopolymeren mogelijkheden bieden om de milieueffecten van voedselverpakking te verlagen. 1. Voor binnenverpakkingen is de prestatie op milieugebied in sommige gevallen even goed als het huidige materiaal in het geval het materiaal wordt gestort na gebruik, en iets beter dan het huidige materiaal als het verbrand wordt met energie terugwinning. 2. Bij de buitenverpakkingen bieden veel films en laminaten voordelen voor het milieu in vergelijking met het huidige materiaal, met bijzonder weinig milieueffecten voor bio-polyethyleenfilm. Ook bio-polyester laminaten bieden aanzienlijke besparingen. 3. Voor zowel binnen- als buitenverpakkingen bieden laminaten met polymelkzuur voordelen voor het milieu wanneer men uitgaat van een toekomstig technologieniveau voor de productie van het polymeer of de film. De milieubalans van cellulose verbetert ook aanzienlijk wanneer men veronderstelt dat de technologie voor filmproductie verbetert. Grote verschillen in landgebruik zijn te vinden tussen alle verpakkingen, afhankelijk van het type gebruikte biomateriaal.

Nu al zijn de opties met de laagste broeikasgasemissies geheel of gedeeltelijk bio-based. Echter een gebrek aan ervaring en investeringen in de omzetting van biobased polymeren naar eindproducten alsmede relatief ongunstige materiaaleigenschappen resulteren in lagere milieuvordelen voor sommige nieuwe biomaterialen dan men op basis van een vergelijking per kg grondstof zou verwachten. Methodologisch betekent dit dat vergelijkingen van biobased materialen met conventionele materialen op basis van een functionele eenheid van 1 kg de verschillen in materiaaleigenschappen niet voldoende in aanmerking nemen terwijl deze wel belangrijk zijn voor een specifieke toepassing, in dit geval de barrière eigenschappen. Dit betekent ook dat de besparingen berekend in hoofdstuk 3 kleiner kunnen worden wanneer de vergelijking niet per gewichtseenheid van het materiaal gemaakt worden maar rekening gehouden wordt met de functionaliteit.

In totaal zijn buitenverpakkingen veelbelovender dan binnenverpakkingen. Als men begint met het vervangen van de huidige petrochemische materialen in de buitenverpakkingen dan zou dit de mogelijkheid bieden om te investeren en te leren, waarvan de ontwikkelingen in de binnenverpakkingen zouden kunnen profiteren. Als men bedenkt dat biobased verpakkingen waarschijnlijk meer baat hebben bij technologische vooruitgang dan conventionele materialen, omdat de biomaterialen nog minder zijn geoptimaliseerd, dan maakt dit de biobased laminaten potentieel zeer aantrekkelijk voor de toekomst.

Vele levenscyclusanalyses van biobased en biologisch afbreekbare materialen verwaarlozen de afvalbehandeling van het product zelf vanwege een gebrek aan consistente gegevens. Dit onderdeel van de levenscyclus kan echter sterk de conclusies beïnvloeden. Het doel van hoofdstuk 5 is om de koolstof- en energiebalansen te ramen en om uit te zoeken wat de beste afvalbehandelingsoptie is voor biologisch afbreekbare materialen, door thuis en industriële compostering, anaërobe vergisting en verbranding te modelleren. We hebben gegevens verzameld voor de volgende biologisch afbreekbare

materialen: papier, cellulose, zetmeel, polymelkzuur (PLA), zetmeel/polycaprolacton (MaterBi), polybutyraat-adipaat-terephthalaat (PBAT, Ecoflex) en polyhydroxyalkanoaten (PHA). Dit is gedaan op basis van een uitgebreide literatuur studie, experimenten en vergelijkingen met stoffen waar men aanzienlijke ervaring mee heeft.

Het is een kenmerk van biologische afvalbehandeling dat de materialen worden gemetaboliseerd tijdens de behandeling van afval, zodat een deel van de opgeslagen koolstof wordt uitgestoten in de lucht en de rest wordt omgezet in compost of digestaat en opgeslagen. De compost of het digestaat kunnen worden gebruikt voor het vervangen van bodemverbeteraars ter ondersteuning van de vorming van humus. Dit voordeel kan niet kunstmatig bereikt worden. Experimentele gegevens over biologisch afbreekbare materialen tonen een bandbreedte wat betreft de hoeveelheid opgeslagen koolstof van deze materialen, en meer proeven zijn nodig om deze onzekerheden in de toekomst te verminderen. Uit experimentele gegevens blijkt ook dat thuis- en industriële compostering verschillen in hun emissies van lachgas en methaan, maar daarbij moet opgemerkt worden dat de beschikbaarheid van de gegevens betreffende thuiscompostering beperkt is.

Van de onderzochte methoden van afvalbehandeling heeft anaërobe vergisting de laagste koolstof- en energiebalans met de huidige technologie, maar verbranding kan in de toekomst aantrekkelijker zijn als de energie-efficiëntie in afvalverbrandingsinstallaties aanzienlijk verbetert. Thuiscomposteren is ongeveer gelijk aan verbranding met energierugwinning voor wat betreft de koolstofemissies en energiegebruik in het geval koolstof credits voor compost worden meegerekend. Hetzelfde geldt voor industriële compostering als koolstofcredits toegewezen worden voor compost die stro vervangt. Koolstofcredits kunnen de resultaten dus aanzienlijk beïnvloeden, maar er zijn veel onzekerheden over de manier waarop deze worden berekend. Daarom is meer onderzoek nodig om de lange-termijn vastlegging van koolstof in de bodem te kwantificeren als men grondverbeteraars zoals compost toevoegt. Bovendien impliceert het gebrek aan kennis over de lange-termijn dynamiek van bodemkoolstof dat er ook sprake is van onzekerheid rond de koolstofbalansen van grondstoffen voor biobased materialen die uit landbouw en bosbouw zijn verkregen. Ondanks deze onzekerheden geldt dat in het geval de bodemkoolstof een beperkende factor is, men het beste kan kiezen voor biologische afvalbehandeling ter bescherming van de bodemkoolstof ondanks dat verbranding met energierugwinning in de toekomst mogelijk beter presteert als gevolg van verbeterde efficiëntie.

Het gebruik van hernieuwbare grondstoffen voor de productie van materialen biedt hogere broeikasgasreducties dan het gebruik van bio-energie. Dit geldt per kg grondstof alsmede per hectare. Biobased materialen hebben enige aandacht gekregen van beleidsmakers, maar analyses van bestaande of potentiële beleidsmaatregelen ontbreken bijna volledig. Hoofdstuk 6 onderzoekt daarom hoe de huidige en potentiële beleidsmaatregelen de marktpenetratie van biomaterialen beïnvloeden. In het algemeen is de marktpenetratie van biomaterialen nog steeds laag en lijken er weinig beleidsmaatregelen te bestaan die een aanzienlijk effect hebben gehad. Onze inventarisatie van de bestaande maatregelen toont naast een gebrek aan samenhang ook een tekort aan brede, veelomvattende beleidsmaatregelen ter ondersteuning van de introductie van biobased materialen.

Bovendien zijn de meeste maatregelen pas onlangs geïmplementeerd en verschillen zij wat betreft de onderliggende beleidsdoelstellingen. De meeste beleidsmaatregelen zijn gericht op de vermindering van afval en zwerfvuil en niet op de vermindering van broeikasgasemissies. Daarnaast is de variatie ook groot als het gaat om de levenscyclusfase van het materiaal waarop men doelt. Sommige maatregelen richten zich op de bron van de materialen (ondersteunen biobased materialen) en anderen op het einde van de levensduur van de materialen (ondersteunen biologisch afbreekbare materialen).

Onze case-studie over de invoering van een koolstofheffing laat zien dat deze alleen invloed op de marktpositie van ethyleen zou hebben. Het marktvolume van ethyleen is echter zo groot dat deze belasting de markt aanzienlijk zou beïnvloeden, alsmede leiden tot een reductie van de wereldwijde broeikasgasemissies. We hebben ook een aantal aanzienlijke belemmeringen voor de marktintroductie van biobased materialen geïdentificeerd, die uit bestaande beleidsmaatregelen voortvloeien. Vele beleidsmaatregelen voor bio-energie beperken hun (financiële) ondersteuning expliciet tot het gebruik van biomassa voor energie doeleinden. Omdat energie en materialen met elkaar concurreren voor dezelfde biomassa of landbouwgrond, benadeelt dit de biomaterialen. Ter vergroting van het marktaandeel van biomaterialen is het daarom belangrijk dat beleidsbarrières worden verwijderd om op deze manier gelijke voorwaarden voor het gebruik van biomassa voor energie en materialen te creëren. Bovendien moeten de beleidsmaatregelen voor biomaterialen worden geharmoniseerd en aanvullende maatregelen worden ingevoerd. Eén daarvan zou een koolstofheffing of een andere koolstofprijstelling kunnen zijn. Tot slot moet de focus van beleidsmaatregelen voor nieuwe biomaterialen verschuiven van niche-toepassingen zoals plastic zakken, naar bredere toepassingen zoals algemene verpakkingen, en moeten biomaterialen meer aandacht krijgen binnen maatregelen voor milieuvriendelijke aankoop.

Aan het begin van dit onderzoek was er weinig bekend over de milieueffecten van de productie van bulkchemicaliën uit biomassa en of zij economisch rendabel geproduceerd kunnen worden. We hebben daarom de hele levenscyclus van biomaterialen onderzocht; de productie van biobased chemicaliën, de toepassing van de biobased polymeren in verpakkingen en ten slotte de afvalbehandeling van biologisch afbreekbare materialen. Vele biobased chemicaliën bieden economische kansen, maar de omvang daarvan hangt af van de prijzen van de petrochemische en biobased grondstoffen en kan verder worden verbeterd door technologische vooruitgang in de toekomst (hoofdstuk 2). Bijna alle biobased chemicaliën hebben lagere koolstof- en energiebalansen dan hun petrochemische tegenhangers en besparingen kunnen aanzienlijk worden verhoogd in de toekomst (hoofdstuk 3). Biomaterialen bieden ook besparingen als zij gebruikt worden voor een specifieke toepassing in voedselverpakkingen, maar deze besparingen kunnen afnemen als bij de vergelijking rekening gehouden wordt met de functionaliteit en materiaaleigenschappen in plaats van per kg materiaal (hoofdstuk 4). Biologisch afbreekbare materialen zijn voordelig omdat ze geschikt zijn voor biologische afvalverwerking waarvan de koolstof- en energiebalansen momenteel minstens even goed als verbranding zijn, maar die het extra voordeel hebben dat zij een bodemverbeteraar produceren die het koolstofgehalte van de bodem kan verbeteren (hoofdstuk 5). Tenslotte is een complex van samenhangende be-

leidsmaatregelen noodzakelijk om biobased en/of biologisch afbreekbare materialen succesvol te laten worden (hoofdstuk 6). De vooruitgang op het gebied van het kwantificeren van de baten, het opstarten van belangrijke onderzoeks- en ontwikkelingsprogramma's en het opzetten van productiefaciliteiten hebben de kansen voor vele biobased chemicaliën en kunststoffen verhoogd om op de markt te komen. Er bestaan dus goede kansen voor biomaterialen, maar er zullen beleidsmaatregelen nodig zijn om ervoor te zorgen dat biomaterialen grootschaliger kunnen concurreren.

Zusammenfassung

Das zentrale Forschungsziel dieser Dissertation ist es *zu untersuchen, welche ökologischen Vorteile und ökonomischen Chancen für biobasierte Materialien bestehen und wie diese durch politische Maßnahmen beeinflusst werden*. Dies wird erreicht, indem wir den gesamten Lebenszyklus von biobasierten Materialien evaluieren, von der Produktion biobasierter Chemikalien über die Verwendung von biobasierten Polymeren im Verpackungsbereich bis hin zur Entsorgung biobasierter und biologisch abbaubarer Materialien. Es wird auch analysiert in welchem Ausmaß aktuelle und potenzielle politische Maßnahmen die Chancen biobasierter Materialien bisher beeinflusst haben oder beeinflussen können. Bei der ökonomischen Analyse liegt der Fokus auf den Produktionskosten und bei der ökologischen Analyse liegt er auf Kohlenstoff- und Energiebilanzen, sowie auf Land- und Wasserbedarf.

Eine Diskussion des Forschungszusammenhangs sowie wichtige terminologische und methodologische Abgrenzungen werden in Kapitel 1 vorgenommen: Angesichts des Anstiegs der Weltbevölkerung von 6,5 Milliarden im Jahr 2005 auf geschätzte 9,1 Milliarden bis zum Jahr 2050 und des prognostizierten Wirtschaftswachstums wird mit großer Wahrscheinlichkeit auch die Nachfrage nach Materialien steigen. Derzeit werden mehr als 99% der Kunststoffe aus fossilen, d.h. nicht-erneuerbaren Rohstoffen hergestellt. Gebraucht man fossile Grundstoffe, so ist diese Nutzung mit einer Reihe ernster nationaler, supranationaler und globaler Probleme eng verknüpft, wie z.B. mit der Erschöpfung nicht-erneuerbarer Ressourcen, der Import-Abhängigkeit von Erdöl und Erdgas oder ihren Derivaten, sowie der Emission von Treibhausgasen samt ihren schädlichen Auswirkungen auf das Weltklima. Wenn, wie in der Vereinbarung von Kopenhagen festgelegt, das Ziel erreicht werden soll, die Erderwärmung auf höchstens 2°C im Vergleich zu vorindustriellen Zeiten zu begrenzen, dann ist eine Verminderung der globalen Treibhausgasemissionen nötig. Dies verlangt eine merkliche Anstrengung in allen Sektoren, hinsichtlich der Verringerung von Emissionen, auch im Materialsektor.

Biomasse kann als Grundstoff für die Produktion biobasierter Materialien genutzt werden, eine Anwendung, die bislang vergleichsweise wenig Aufmerksamkeit der politischen Entscheidungsträger bekommen hat — und dies trotz der Tatsache, dass die Produktion dieser Materialien pro Hektar Land mehr Einsparung von Treibhausgasen ermöglichen könnte als die Produktion von Bioenergie. Als erneuerbare Rohstoffquelle hat sie das Potenzial, der Erschöpfung nicht-erneuerbarer Ressourcen, der Importabhängigkeit und dem Risiko von Versorgungsengpässen, sowie dem Klimawandel entgegen zu wirken. Deshalb zielt diese Dissertation darauf ab zu einem besseren Verständnis — über den kompletten Lebenszyklus hinweg — der ökologischen und ökonomischen Vorteile beizutragen, die ein Ersetzen konventioneller petrochemischer Materialien durch aus Biomasse gewonnene Grundstoffe (d.h. biobasierte Materialien) mit sich bringt.

Hinsichtlich der Produktion dieser Materialien behandelt diese Dissertation mehrere Biopolymere, wie Polylaktide (PLA), Zellulose und Papier, und konzentriert sich auf die biochemische Konversion von Biomasse in Basis-Chemikalien unter Nutzung der sogenannten weißen Biotechnologie (WB); darunter versteht man die Nutzung von Fermentation und/oder enzymatischen Prozessen für die Herstellung von Industrieprodukten und Treibstoffen, wobei möglicherweise Züchtung und genetische Veränderung der bei der Fermentation verwendeten Mikroorganismen eingesetzt werden. Die Grundstoffe für die Herstellung der biobasierten Materialien, die in dieser Dissertation analysiert werden, sind leicht fermentierbare Zucker wie Saccharose, Dextrose und hydrolisierte Stärken (entsprechend heutiger Technologie), sowie Lignozellulosen wie Miscanthus und Maisstroh (entsprechend der Technologie von morgen, der “zweiten Generation”).

Noch liegen nur wenig Informationen bezüglich der Rentabilität der Produktion biobasierter Basis-Chemikalien und Zwischenprodukte mithilfe weißer Biotechnologie vor. Kapitel 2 untersucht daher, in welchem Ausmaß biobasierte Basis-Chemikalien und Zwischenprodukte ökonomische Chancen im Vergleich zu den momentan genutzten petrochemischen Produkten bieten — heute und zukünftig. Um dies zu quantifizieren, bietet das Kapitel eine technisch-ökonomische Analyse der Produktion biobasierter Basis-Chemikalien mit Hilfe weißer Biotechnologie. Um die technisch-ökonomischen Chancen gegenwärtiger und zukünftiger Produktionswege biobasierter WB-Basis-Chemikalien systematisch zu evaluieren, wird ein detailliertes methodisches Vorgehen entwickelt, das sich auf verfügbare Daten und schlüssige Annahmen bezüglich zukünftiger (bio)technischer Entwicklungen stützt. Gegenwärtige und zukünftige Produktionswege werden für 15 Produkte evaluiert, wobei wir von Preisen für fermentierbaren Zucker zwischen 70€/t und 400€/t und von Rohölpreisen von US\$ 25/Barrel und US\$ 50/Barrel ausgehen.

Im Allgemeinen ist eine große Zahl von WB-Chemikalien ökonomisch rentabel, wenn man sie mit ihren petrochemischen Äquivalenten vergleicht, diese wirtschaftliche Wettbewerbsfähigkeit hängt jedoch zu einem großen Teil von den Preisen für Rohöl und fermentierbaren Zucker ab. Stärker im Detail betrachtet zeigen die Ergebnisse, dass WB-basiertes Ethanol, 1,3-Propandiol (PDO), Polytrimethylen-Terephthalat (PTT) und Bernsteinsäure schon heute ökonomisch chancenreich sind — WB-Prozesse nach heutigem

Stand der Technik und einen Rohölpreis von von US\$ 25/Barrel vorausgesetzt. Ethylen und PLA sind ökonomisch attraktiv, sobald man von einem höheren Rohölpreis ausgeht (US\$ 50/Barrel).

Im Zeitraum seit Abschluss dieser Studien ist der Rohölpreis weiter gestiegen und deshalb diskutieren wir hier kurz die Ergebnisse für einen Preis von US\$ 70/Barrel (weitere Details in Hermann et al., 2010). Bei diesem Preis sind bei gegenwärtiger Technik die folgenden Produkte ökonomisch rentabel: 1,3-Propandiol, Polytrimethylen-Terephthalat, Polylaktide, Bernsteinsäure, Ethyllaktat, Ethylen und Ethanol. Bei zukünftiger Technologie sind außer Essigsäure alle untersuchten Produkte ökonomisch rentabel bei Preisen für fermentierbaren Zucker von 70 bis 135 €/t. Alle Analysen zeigen, dass das technische Niveau einen weitreichenden Einfluss auf die Ergebnisse hat: Die Produktionskosten plus Profit (PCPP) von Produkten, die direkt aus Fermentation gewonnen werden, sind 20%-40% niedriger, wenn man zukünftige anstelle heutiger Technologie zu Grunde legt. Für Produkte, die nach der Fermentation noch eine chemische Umsetzung benötigen, verbessern sich die Produktionskosten plus Profit um 15%–20%, wenn man von zukünftiger statt heutiger Technologie ausgeht (für US\$ 70/Barrel und quer durch alle Zuckerpreise; ähnliche Werte gelten bei niedrigeren Rohölpreisen). Der technische Fortschritt im Bereich weißer Biotechnologie kann demnach einen signifikanten Beitrag zur Verbesserung der ökonomischen Wettbewerbsfähigkeit biobasierter Produkte leisten. Wenn ein Vergleich mit anderen Herstellungswegen von Biomasse-zu-Endprodukt (z.B. thermochemische Umsetzung) zugunsten von WB-Basis-Chemikalien ausfällt, und wenn sie auch ökologische Vorteile bieten, wäre ihre Einführung in großem Ausmaß wünschenswert.

Prinzipiell kann der Gebrauch erneuerbarer Rohstoffquellen (wie z.B. Biomasse) zur Herstellung von Materialien zur Lösung zweier wichtiger Umweltprobleme beitragen: des Klimawandels und der Erschöpfung der fossilen Ressourcen. Allerdings fehlen zuverlässige und konsistente Datensätze um Einsparungen zu quantifizieren und entsprechend verschiedener Materialien miteinander zu vergleichen. Auf der Basis derselben prinzipiellen Vorgehensweisen und Daten, wie sie detailliert in Kapitel 2 dargestellt worden sind, quantifiziert Kapitel 3 das Ausmaß, bis zu dem mehrere biobasierte Basis-Chemikalien und Nebenprodukte ökologische Vorteile bieten im Vergleich zu heutigen petrochemischen Produkten. Zu diesem Zweck werden die Kohlenstoff- und Energie-Bilanzen wie auch die direkte Landnutzung für die Produktion dieser biobasierten Basis-Chemikalien dargestellt. Dieses Kapitel enthält auch eine Analyse gegenwärtiger und zukünftiger Technologien bei der Nutzung weißer Biotechnologie zur Umwandlung von aus Maisstärke, Zuckerrohr oder Maisstroh (Lignozellulose) gewonnenen Zuckern in 15 Basis-Chemikalien.

Nach dem heutigen Stand der Technik im Bereich der Biotechnologie bieten — außer Adipinsäure und Essigsäure — alle Produkte beachtliche Einsparungspotenziale bezüglich nicht-erneuerbarer Energie und Treibhausgasemissionen verglichen mit petrochemischen Produkten. Im Durchschnitt bewegen sich die Einsparungen in der Größenordnung von 30% bei der Verwendung von Mais als Ausgangsstoff bzw. bei 65% bei der Verwendung von Zuckerrohr, jeweils bei heutiger Technologie. Bei zukünftiger Technologie

liegen diese Werte bei 50% bis 85%. Mit Zuckerrohr als Ausgangsstoff sind Einsparungen von über 100% beim Verbrauch nicht-erneuerbarer Energie und bei Treibhausgasemissionen bereits heute möglich, wenn die Energiegutschriften aus der Verbrennung von Biomasseresten oder von landwirtschaftlichen oder biochemischen Nebenprodukten größer sind als die Emissionen und der Energieverbrauch, die mit den Ausgangsstoffen und dem WB-Prozess verbunden sind. In der Zukunft sind weitere substanzielle Einsparungen möglich (in puncto Treibhausgasemissionen um 25%-35% höher als mit heutiger Technologie) durch verbesserte Fermentation und Produkttrennung. Um mittels biobasierter Basis-Chemikalien größtmögliche Einsparungen bei Treibhausgasemissionen und beim Verbrauch nicht-erneuerbarer Energie zu erreichen, ist Zuckerrohr vorteilhafter als Lignozellulosen, welche andererseits — aus ökologischer Sicht — als Ausgangsstoff für fermentierbaren Zucker der Maisstärke vorzuziehen sind.

Die Produkte mit den größten Einsparungen (in % verglichen mit ihren petrochemischen Äquivalenten) sind Akrylsäure, Butanol (aus dem ABE-Prozess), Ethanol, Ethen, PDO und PHA. Weltweit sind Einsparungen von Treibhausgasen von 0,5 bis 1,0 Gt CO₂ pro Jahr bei Einsatz zukünftiger Technologie möglich, wenn man nur die 15 Basis-Chemikalien berücksichtigt, die für diese Studie ausgewählt wurden. Dies stellt einen beachtlichen Anteil von bis zu einem Drittel der jährlichen CO₂-Emissionen durch die chemische und petrochemische Industrie dar (3 Gt CO₂, Schätzung auf der Grundlage von IPCC, 1996; IEA, 2009a,b)). Zusammen mit den ökonomischen Chancen, die in Kapitel 2 vorgestellt worden sind, bildet dies ein überzeugendes Plädoyer zugunsten der Produktion von biobasierten Basis-Chemikalien mit Hilfe weißer Biotechnologie. Deren Einsatz bei der Nutzung von Biomasse als Ausgangsstoff für die Produktion von Chemikalien bietet demnach hervorragende Chancen, um Treibhausgasemissionen deutlich zu reduzieren und die Abhängigkeit von fossilen Ressourcen zu verringern, und könnte deshalb für die chemische Industrie zu einer wichtigen Strategie werden um dazu beizutragen, die Erderwärmung einzuschränken.

Man hat geschätzt, dass biobasierte Polymere allein auf der Basis von Materialeigenschaften das technische Potenzial besitzen, maximal 90% ihrer petrochemischen Äquivalente zu ersetzen. Dies legt nahe, dass sich auf Grund der Materialeigenschaften die Ausgangsstoffe für die Produktion von Basis-Chemikalien und Kunststoffen in großem Maßstab ersetzen lassen, sofern man dadurch ökonomische und ökologische Chancen nutzen kann. Das führt zu der Frage, ob genügend Land verfügbar ist um diese Materialien zu produzieren — auf nachhaltige Weise. Andere Autoren haben die Frage wie nachhaltig erneuerbare Rohstoffe produziert werden (können) in aller Ausführlichkeit diskutiert, wenn auch aus der Perspektive der Bioenergieproduktion. Diese Diskussion lässt sich jedoch 1:1 auf Materialien übertragen. Der Landbedarf für biobasierte Materialien wurde von Dornburg et al. (2008) auf der Grundlage dieser Dissertation sowie dem möglicherweise überschüssigen Ackerland (Smeets et al., 2007) berechnet. Wenn wir den Landbedarf für Bioenergie ausklammern, ist das Biomassepotenzial leicht groß genug, um eine komplette Substitution petrochemischer durch biobasierte Materialien, wie in dieser Dissertation diskutiert, zu ermöglichen.

Kapitel 4 bietet eine vollständige Lebenszyklusanalyse für die Verwendung von Biokunststoffen im Verpackungsbereich. Dieser Industriezweig verbraucht fast 40% aller Polymere und nutzt sehr spezifische Materialfunktionen wie z.B. Barriereigenschaften. Unser Ziel ist es festzustellen, ob Lebensmittelverpackungen, die teilweise oder vollständig aus biobasierten Materialien bestehen, ökologische Vorteile gegenüber dem derzeit verwendeten petrochemischen Material bieten. Zu diesem Zweck führen wir eine Fallstudie für innere und äußere Snackverpackungen durch; dabei stehen innere Verpackungen in direktem Kontakt mit Lebensmitteln und müssen daher bestimmte Schutzfunktionen erfüllen, wogegen dies für äußere Verpackungen nicht gilt. Man vergleicht biobasierte Materialien wie Papier, biobasiertes Polyethylen, Polymilchsäure und einen weiteren biobasierten Polyester mit konventionellen wie Polypropylen und Polyethylen. Die Ökobilanz quantifiziert den Verbrauch nicht-erneuerbarer Energie, Gesamtprimärenergieverbrauch, Treibhausgasemissionen, Erschöpfung abiotischer Ressourcen, Photo-Oxidanzienbildung, Versauerung, Eutrophierung, Wasser- und Flächenbedarf.

Der Vergleich mit den derzeit verwendeten, nicht-erneuerbaren Materialien (Polypropylen) zeigt, dass im Allgemeinen Folien und Lamine (d.h. mehrschichtige Folien), die (teilweise) aus biobasierten Polymeren bestehen, Chancen eröffnen die ökologischen Effekte von Lebensmittelverpackungen zu reduzieren: 1. Im Bereich der inneren Verpackungen ist in manchen Fällen die Ökobilanz mit dem aktuellen Material gleichwertig, wenn die Materialien nach Gebrauch deponiert werden, und etwas besser wenn es mit Energierückgewinnung verbrannt wird. 2. Im Bereich der äußeren Verpackungen bieten viele Folien und Lamine ökologische Vorteile gegenüber dem aktuellen Material, etwa die besonders niedrige Umweltbelastung für biobasierten Polyethylenfolien; biobasierte Polyesterlamine bieten ebenfalls erhebliche Einsparungen. 3. Im Bereich innerer und äußerer Verpackungen bieten Lamine mit Polylaktiden ökologische Vorteile, wenn man ein zukünftiges Technologieniveau bei der Herstellung von Polymeren oder Folien zugrunde legt; auch die Ökobilanz von Zellulose verbessert sich erheblich, wenn man für das Stadium der Folienherstellung von erhöhter technologischer Reife ausgeht. Große Unterschiede im Landbedarf sind quer durch alle Verpackungen zu verzeichnen, je nach Typ des verwendeten biobasierten Materials.

Bereits heute sind die Optionen mit den niedrigsten Treibhausgasemissionen teilweise oder vollständig biobasiert. Ein Mangel an Erfahrung mit und Investitionen in die Konversion biobasierter Polymere in Endprodukte, sowie vergleichsweise ungünstige Materialeigenschaften führen allerdings bei einigen neuartigen biobasierten Materialien zu geringeren ökologischen Vorteilen als man auf der Basis eines Vergleichs pro kg Rohstoff erwarten würde. Aus methodischer Sicht bedeutet dies, dass Vergleiche biobasierter mit konventionellen Materialien unter Verwendung einer funktionellen Einheit von 1 kg die Unterschiede in den Materialeigenschaften nicht ausreichend berücksichtigen, welche für eine spezifische Anwendung wichtig werden, in diesem Fall die Barriereigenschaften des Materials. Dies bedeutet auch, dass die in Kapitel 3 berechneten Einsparungen geringer ausfallen können, wenn der Vergleich nicht pro Gewichtseinheit des Materials sondern in Hinblick auf seine Funktionalität erfolgt.

Insgesamt sind äußere Verpackungen ökologisch vielversprechender als innere Verpackungen. Wenn man bei der Einführung biobasierter Verpackungsmaterialien mit dem Ersetzen der aktuellen, petrochemischen äußeren Verpackungen anfängt, würde dies die Tür öffnen für Investitionen und Lernprozesse, was den inneren Verpackungen zu Gute käme. Berücksichtigt man, dass biobasierte Materialien mit großer Wahrscheinlichkeit stärker als konventionelle Materialien vom technischen Fortschritt profitieren werden (weil ihre Produktionsprozesse bisher weniger optimiert worden sind), lässt dies bestimmte biobasierte Lamine zu potenziell hochattraktiven Zukunftsoptionen werden.

Viele Lebenszyklusanalysen biobasierter und biologisch abbaubarer Materialien vernachlässigen die Phase der Abfallbehandlung aus Mangel an konsistenten Daten, obwohl dieses Stadium des Lebenszyklus die Schlussfolgerungen stark beeinflussen könnte. Das Ziel von Kapitel 5 war es, Näherungswerte für die Kohlenstoff- und Energiebilanz der Abfallbehandlung zu etablieren und für Haus- und Industriekompostierung, anaerobe Vergärung und Verbrennung herauszufinden, welches die beste Abfallbehandlungsmethode für biologisch abbaubare Materialien ist. Wir haben für die folgenden biologisch abbaubaren Materialien Datensätze zusammengetragen: Papier, Zellulose, Stärke, Polylaktide, Stärke/Polycaprolacton (MaterBi), Polybutyrat-Adipat-Terephthalat (PBAT, Ecoflex) und Polyhydroxyalkanoate (PHA). Diese Daten beruhen auf intensiver Literaturrecherche, Experimenten und Vergleichen mit Materialien, für die umfangreiche Erfahrungen vorliegen.

Es ist eine Besonderheit der biologischen Abfallbehandlung, dass die Materialien während des Behandlungsprozesses metabolisiert werden, so dass ein Teil ihres gespeicherten Kohlenstoffs freigesetzt wird und der Rest im Kompost oder Digestat verbleibt. Der Kompost oder das Digestat können Bodenverbesserungsmittel ersetzen, welche die Humusbildung unterstützen — ein Vorzug, der sich nicht künstlich herstellen lässt. Experimentelle Daten über biologisch abbaubare Materialien zeigen eine gewisse Bandbreite bezüglich der Menge des in Kompost bzw. Digestat gespeicherten Kohlenstoffs, und weitere Untersuchungen werden erforderlich sein um diese Ungenauigkeiten zu reduzieren. Experimentelle Daten haben auch gezeigt, dass sich häusliche Eigen- und industrielle Kompostierung in ihren Emissionen von Lachgas und Methan unterscheiden, aber es sollte festgehalten werden, dass nur wenige Daten zur häuslichen Eigenkompostierung verfügbar sind.

Von den untersuchten Methoden der Abfallbehandlung hat anaerobe Vergärung die günstigste Ökobilanz bei derzeitigem Stand der Technik, aber die Verbrennung kann in Zukunft günstiger werden, sofern es zu erheblichen Verbesserungen bei der Energierückgewinnung in Müllverbrennungsanlagen kommt. Die häusliche Eigenkompostierung ist hinsichtlich Kohlenstoff- und Energiebilanz grob der Verbrennung mit Energierückgewinnung gleichwertig, wenn man Kohlenstoffgutschriften berücksichtigt. Dasselbe gilt für industrielle Kompostierung, wenn Kohlenstoffgutschriften vergeben werden für Kompost, der Stroh ersetzt. Diese Gutschriften können sich daher spürbar auf die Resultate auswirken, aber es gibt große Unsicherheiten dahingehend, wie sie berechnet werden sollen. Deshalb bedarf es weiterer Forschung um den langfristigen Nutzen in puncto Kohlenstoff in Böden zu quantifizieren, der aus dem Aufbringen von Bodenverbesserungsmitteln

keln wie z.B. Kompost resultiert. Die Wissenslücken hinsichtlich der Dynamik langfristiger Kohlenstoffgaben an Böden impliziert außerdem eine Unsicherheit auch im Bereich der Kohlenstoffbilanzen von land- und forstwirtschaftlicher Produktion von erneuerbaren Ressourcen für die Materialnutzung. Wo jedoch Kohlenstoff im Boden ein limitierender Faktor ist, sollte biologische Abfallbehandlung gewählt werden um ausreichend Kohlenstoff im Boden sicherzustellen, auch wenn die Verbrennung mit Energierückgewinnung wegen Wirkungsgradverbesserungen in der Zukunft möglicherweise bessere Ergebnisse liefern wird.

Obwohl die Verwendung erneuerbarer Ressourcen für die Produktion von Materialien pro kg Grundstoff wie auch pro Hektar ein höheres Einsparungspotenzial für Treibhausgase bietet als dies für Bioenergie gilt hat sie hat zwar einige Beachtung durch politische Entscheidungsträger gefunden, aber Analysen bestehender oder potenzieller politischer Maßnahmen fehlen auf diesem Gebiet fast vollständig. Kapitel 6 untersucht, wie aktuelle und potenzielle politische Maßnahmen die Marktdurchdringung biobasierter Materialien beeinflusst. Diese bleibt im Allgemeinen niedrig, und scheinbar hatten nur wenige politische Maßnahmen einen signifikanten Effekt. Unser Inventar bestehender Maßnahmen verdeutlicht einen Mangel an konsistenten und umfassenden politischen Maßnahmen, welche die Einführung erneuerbarer Ressourcen für die Materialnutzung unterstützen können. Außerdem sind die meisten Maßnahmen erst vor kurzem implementiert worden und unterscheiden sich zudem hinsichtlich der zugrundeliegenden politischen Ziele: Das Ziel der meisten ist Müllreduzierung und nicht die Verringerung von Treibhausgasemissionen. Darüber hinaus gibt es große Unterschiede, wenn es um das Stadium im Lebenszyklus des jeweiligen Materials geht, an dem die Maßnahmen ansetzen sollen: Einige zielen auf die Ausgangsstoffe der Materialien ab (Förderung biobasierter Materialien), andere auf die Verwertung dieser Materialien nach ihrer Nutzung (Förderung biologisch abbaubarer Materialien).

Unsere Fallstudie zur Implementierung einer CO₂-Steuer zeigt, dass diese Steuer nur die Marktposition von Ethylen beeinflussen würde; doch ist dessen Marktanteil so groß, dass eine solche Steuer sich in der Tat beachtlich auf den Markt auswirken und zudem zu globalen Treibhausgaseinsparungen führen würde. Wir haben auch signifikante Hemmnisse für die Marktdurchdringung durch biobasierte Materialien ermittelt, die aus bestehenden politischen Maßnahmen erwachsen: Viele von ihnen, welche die Verwendung von Bioenergie fördern, beschränken explizit jegliche (finanzielle) Unterstützungen auf die Nutzung von Biomasse für Energiezwecke. Da Energie und Materialien in Konkurrenz um dieselbe Biomasse oder dasselbe Ackerland stehen, werden biobasierte Materialien dadurch benachteiligt. Um die Marktdurchdringung durch biobasierte Materialien zu steigern, ist es deshalb wichtig, dass politische Hindernisse beseitigt werden, damit für den Gebrauch von Biomasse, egal ob für Energie- oder Materialnutzung, gleiche Rahmenbedingungen geschaffen werden. Darüber hinaus sollten die politischen Maßnahmen für Biomaterialien harmonisiert und weitere Maßnahmen ergriffen werden, wovon eine die Einführung einer CO₂-Steuer oder eines anderen Weges zu einem Kohlenstoffpreis sein könnte. Schließlich sollten politische Maßnahmen zugunsten neuer Biomaterialien sich von Nischenanwendungen wie Plastiktüten wegbewegen und sich umfassendere Anwen-

dungsbereiche wie Verpackung insgesamt zum Ziel setzen. Sie sollten außerdem stärker berücksichtigt werden im Rahmen der Bestrebungen, das öffentliche Beschaffungswesen umweltorientierter zu gestalten.

Insgesamt gab es zu Beginn dieser Untersuchungen Wissenslücken bezüglich der ökologischen Auswirkungen der Herstellung von Basis-Chemikalien aus Biomasse und der Frage, ob sie wirtschaftlich produziert werden können. Wir haben deshalb den gesamten Lebenszyklus von Biomaterialien analysiert: die Produktion biobasierter Chemikalien, die Verwendung biobasierter Polymere Verpackungsbereich und schließlich die Abfallbehandlung biologisch abbaubarer Materialien. Zahlreiche biobasierte Chemikalien bieten ökonomische Chancen, deren Ausmaß von den Preisen für die petrochemischen und biobasierten Ausgangsstoffe abhängt, und können durch den technischen Fortschritt in der Zukunft weiter verbessert werden (Kapitel 2). Fast alle biobasierten Chemikalien haben eine günstigere Kohlenstoff- und Energiebilanz als ihre petrochemischen Äquivalente, und Einsparungen können in der Zukunft substanziell gesteigert werden (Kapitel 3). Biobasierte Materialien bieten auch Einsparungen, wenn sie ganz spezifisch für Lebensmittelverpackungen eingesetzt werden, diese Einsparungen können jedoch geringer ausfallen, wenn der Vergleich nicht pro kg Material vorgenommen wird sondern hinsichtlich Funktionalität und Materialeigenschaften (Kapitel 4). Der Vorteil biologisch abbaubarer Materialien liegt darin, dass sie sich für die biologische Abfallbehandlung eignen, deren Kohlenstoff- und Energiebilanzen gegenwärtig mindestens gleich gut sind wie die für Verbrennung, und dass sie zusätzlich einen Bodenverbesserer herstellen, der dazu dient den Kohlenstoffanteil im Boden zu erhöhen (Kapitel 5). Abschließend ist zu sagen, dass einheitliche politische Maßnahmen zugunsten biobasierter oder biologisch abbaubarer Maßnahmen notwendig sind um den Erfolg dieses Markts zu sichern (Kapitel 6). Die Fortschritte, die hinsichtlich der Quantifizierung der Vorteile erzielt worden sind, der Start bedeutender Forschungs- und Entwicklungsprogramme, sowie die Inbetriebnahme von Produktionsanlagen haben die Wahrscheinlichkeit erhöht, dass viele biobasierte Chemikalien und Kunststoffe auf den Markt kommen. Es bestehen also gute Chancen für Biomaterialien, aber es bedarf politischer Maßnahmen um sicherzustellen, dass sie in großem Umfang in den Wettbewerb eintreten können.

Abbreviations & Glossary

ABE acetone-butanol-ethanol

ADP abiotic depletion potential

AP acidification potential

AlO_x aluminium oxide

bbf barrel

BBP bio-based polyester

Biodegradation is the decomposition of organic material by micro-organisms or their enzymes into smaller products such as carbon dioxide and water

CAP common agricultural policy (EU)

CENTREL electricity mix of Czech Republic, Hungary, Poland and Slovakia

CF cradle-to-factory gate

CG cradle-to-grave

E_{el} electric energy

EL ethyl lactate

EP eutrophication potential

ETS emissions trading scheme (EU)

EU European Union

EVA ethyl vinyl acetate

FB4P Federal Biobased Products Preferred Purchasing Program

fte full-time equivalent

- GHG** greenhouse gas
- GJ_e** electric energy (GJ)
- GJ_p** primary energy (GJ)
- GPP** green public procurement
- GWP** global warming potential
- H2O** water use (impact category), consisting of process, cooling and irrigation water
- HDPE** high density polyethylene
- HHV** high heating value, also called gross calorific value
- Humification** is the conversion of organic matter into humus.
- ISBL** Inside battery limits, deals with the core process equipment, piping, instrumentation etc.
- kt** kilotonne (all tonnes are metric tonnes)
- LU** Land use
- M €** million €
- Mt** Megatonne (metric)
- Mineralisation** (in soil science) is the decomposition of organic matter into forms that are accessible to plants.
- MOPP** metallised oriented PP
- MPET** metallised PET
- MPLA** metallised PLA
- NORDEL** electricity mix of Denmark, Finland, Norway and Sweden, excluding Iceland
- NREU** non-renewable energy use
- OPP** oriented polypropylene
- OSBL** Outside battery limits, deals with steam and power generation and supply, wastewater treatment, cooling towers, etc.
- PBAT** polybutyrate adipate terephthalate
- PBS** polybutyrate succinate
- PCL** polycaprolactone
- PCPP** production cost plus profits

- PDO** propanediol
- PE** polyethylene
- PET** polyethylene terephthalate
- PHAs** polyhydroxyalkanoates
- PHB** polyhydroxybutyrate
- PHBV** polyhydroxybutyrate-co-valerate
- PLA** polylactic acid
- PLA-NG** next generation (future) PLA
- POF** photochemical oxidant formation
- PP** polypropylene
- PTT** polytrimethylene terephthalate
- PUR** polyurethane
- PVdC** polyvinylidene chloride
- R&D** research and development
- REU** renewable energy use
- SiO_x** silicone oxide
- t** tonne (metric)
- TEU** total energy use (renewable and non-renewable)
- TPS** thermoplastic starch
- TFC** total fixed capital
- UCTE** electricity mix of Austria, Belgium, Bosnia-Herzegovina, Croatia, France, Germany, Greece, Italy, Luxembourg, Macedonia, Netherlands, Portugal, Serbia & Montenegro, Slovenia, Spain and Switzerland
- VFG** vegetable, fruit and green waste
- WB** white biotechnology

List of Figures

1.1	Market shares of bulk materials	2
2.1	Simplified flowsheet as used in the generic approach	18
2.2	Procedure to calculate production costs plus profits (PCPP)	24
2.3	Production cost plus profits (PCPP) for PDO production	28
2.4	Cost composition of PDO production for low and high sugar prices	29
2.5	Economic viability of today's WB products	31
2.6	Economic viability of future's WB products	31
2.7	Effect of high crude oil price on PCPPs	33
3.1	Subsystems and system boundaries	42
3.2	Greenhouse gas savings per tonne of WB chemical	48
3.3	Greenhouse gas (GHG) savings vs. land use for sugar from sugar cane	49
3.4	Greenhouse gas (GHG) savings vs. land use for sugar from maize stover	50
4.1	Total weight of films and laminates (inner and outer Packs)	66
4.2	Savings of inner packs relative to reference material	70
4.3	Cradle-to-grave: Global warming potential for inner packs	71
4.4	Agricultural and silvicultural land use of inner packs	72
4.5	Savings of outer packs relative to reference material	73
4.6	Agricultural and silvicultural land use of outer packs	74
4.7	Cradle-to-grave: Global warming potential for outer packs	75
4.8	Global warming potential of laminates assuming future technology	76
4.9	Effect of wind credits on film production and lamination	78
5.1	Biodegradable materials in four types of biological waste treatment	85
5.2	Humus formation and degradation for various soil conditioners	90
5.3	Comparison of four waste treatment types for biomaterials	101
6.1	Effect of a carbon tax on bio-based chemicals	127

6.2	Global CO ₂ savings resulting from a carbon tax	127
-----	----------------------------------------------------------------------	-----

List of Tables

1.1	Classification of materials	6
1.2	Examples of polymer production from bulk chemicals	7
2.1	Production capacities of selected petrochemicals in Western Europe	17
2.2	Key data on fermentation processes	20
2.3	Separation technologies for downstream processing	21
2.4	Specific energy use in fermentation and downstream processing	22
2.5	Process inputs and outputs for the production of 1 tonne of PDO	27
2.6	Production cost plus profits (€/t) of WB products	30
2.7	Viability of WB-chemicals for four price levels of fermentable sugar	32
3.1	Background data for 1 tonne of fermentable sugar	46
3.2	Potential worldwide GHG savings of 15 WB products	51
3.3	Non-renewable energy use (NREU) of WB products	55
3.4	Greenhouse gas emissions (GHG) of WB products	56
3.5	Land use of WB products	57
4.1	Effect of wind credits on primary energy use of PLA	64
4.2	Laminates included in this study	67
5.1	Key properties of biodegradable materials	84
5.2	Extent of biodegradation of materials during composting	86
5.3	Carbon sequestration properties of various soil conditioners	92
5.4	Overview of the methodological differences between perspectives	93
5.5	Weighted GWP fertiliser data for Europe	94
5.6	Literature data on industrial composting	96
5.7	Literature data on home composting	97
5.8	Carbon and nitrogen emissions for home composting of biomaterials	98
5.9	Carbon balance for anaerobic digestion	99
5.10	Global warming potential for incineration	100

List of Tables

5.11	Contribution of waste processes to overall GWP	102
6.1	Implicit CO ₂ prices of economic instruments	116
6.2	Existing policy measure which affect biomaterials	119
6.3	Existing economic incentives for transportation biofuels	120

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Curriculum Vitae



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