Emerging bioeconomy

Assessing the implications of advanced bioenergy and biochemicals with bottom-up and top-down modelling approaches

Ioannis Tsiropoulos

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Ph.D. dissertation

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Emerging bioeconomy

Assessing the implications of advanced bioenergy and biochemicals with bottom-up and top-down modelling approaches

Opkomende bio-economie

Verkenning van effecten van geavanceerde productiepaden voor bio-energie en bio-chemicaliën met behulp van bottom-up en top-down modelbenaderingen

(met een samenvatting in het Nederlands)

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ABBREVIATIONS AND UNITS

ALCA	Attributional Life Cycle Assessment
AP	Acidification/Acidification Potential
ASTM	American Society for Testing and Materials
BAT	Best Available Technology
BDO	1,4-Butanediol
BECCS	Bioenergy and Carbon Capture and Storage
BIT	Biomass Intermodal Transportation
BOD	Biological Oxygen Demand
CARG	Compound Annual Growth Rate
CCS	Carbon Capture and Storage
CD	Cobb-Douglas
CDE	Constant Difference of Elasticity
CED	Cumulative Energy Demand
CES	Constant Elasticity of Substitution
CFED	Cumulative Fossil Energy Demand
CGE	Computable General Equilibrium
CHP	Combined Heat and Power
CLCA	Consequential Life Cycle Assessment
COD	Chemical Oxygen Demand
COP	Conference of Parties
DAP	Diammonium Phoshpate
db	dry basis
DEG	Diethylene Glycol
dLUC	Direct Land Use Change
EA	Economic Allocation
EA-C	Economic Allocation-Conservative
EA-O	Economic Allocation-Optimistic
EB	Ethylbenzene
EBP	Ethanol Blending Programme
ECN	Energy research Centre of the Netherlands
EG	Ethylene Glycol
EO	Ethylene Oxide
EP	Eutrophication/Eutrophication Potential
EQ	Ecosystem Quality
etOH	Ethanol
EU	European Union
FCC	Fluid Catalytic Cracking
FDCA	Furandicarboxylic Acid
FIT	Feed In Tariff
Fos	Fossil
FT	Fischer-Tropsch
GDP	Gross Domestic Product
GHG	Greenhouse Gas
Glob	Global
GTAP	Global Trade Analysis Project
GWP	Global Warming Potential
IGL	India Glycols
iLUC	Indirect Land Use Change
INDC	Intended Nationally Determined Contributions
ISBL	Inside Battery Limits
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LDPE	Low Density Polyethylene
LHV	Lower Heating Value
LLDPE	Linear Low Density Polyethylene
LowTech	Low Technology Development
LPG	Liquefied Petroleum Gas
LUC	Land Use Change
MAGNET	Modular Applied GeNeral Equilibrium Tool
MARKAL	MARket ALlocation
MEG	Monoethylene Glycol
MSWI	Municipal Solid Waste Incineration
NL	Netherlands
NREU	Non-Renewable Energy Use
NUTS	Nomenclature of territorial Units for STatistics
OSBL	Outside Battery Limits
pchem	Petrochemical
PDO	1,3-Propanediol
PE	Polyethylene
PEF	Polyethylene Furanoate
PEM	Partial Equilibrium model
PET	Polyethylene Terephthalate
PHA	Polyhydroxyalkanoates
PLA	Polylactic Acid
PM	Particulate Matter
PO	Propylene Oxide
PP	Polypropylene
PTA	Purified Terephthalic Acid
PTT	Polytrimethylene Terepthalate
PYR	Pyrolysis
R&D	Research and Development
RD&D	Research and Development and Demonstration
RED	Renewable Energy Directive
Ref	Reference
Reg	Regional
RES	Renewable Energy Sources
RJF	Renewable Jet Fuels
SA	Succinic Acid
SE-C	System Expansion-Conservative
SE-O	System Expansion-Optimistic
SNG	Synthetic Natural Gas
SSP	Single SuperPhosphate

HDPE	High Density Polyethylene
HEFA	Hydroprocessed Esters and Fatty Acids
HH	Human Health
HHV	Higher Heating Value
HighTech	High Technology Development
HRD	Hydrotreated Renewable Diesel
HTL	Hydrothermal Liquefaction
HTL	Hydrothermal Liquefaction
IEA	International Energy Agency
IEE	Intelligent Energy Europe
TEG	Triethylene Glycol
TFC	Total Final Consumption
TOPS	Torrefied Pellets
TPES	Total Primary Energy Supply
TSP	Triple SuperPhosphate
UCO	Used Cooking Oil
UP	Uttar Pradesh
WEO	World Energy Outlook
WSI	Water Stress Index

Measurements are reported in the International System of Units (SI) and their accepted derivatives. Units of life cycle impact assessment measurements are reported in line with the impact assessment method.

1

Introduction

1.1 THE SOCIETAL CHALLENGES AHEAD

It is historically evident that the release of anthropogenic Greenhouse Gas (GHG) emissions has been increasing rapidly since industrialisation. Over the course of the last four decades, GHG emissions, mainly comprised of CO_2, CH_4 and N_2O, almost doubled, from 27 $GtCO_{2eq}$/yr in 1970 to 49 $GtCO_{2eq}$/yr in 2010 (IPCC, 2014a). With great certainty, their unprecedented concentration in the atmosphere is the main cause of global warming and other climatic changes (IPCC, 2014b). Climate change is recognised as a potentially irreversible threat to humanity and the planet calling for urgent action and global cooperation (UNFCCC, 2015a). At the Conference of Parties in Paris in 2015 (COP 21), it was agreed to limit temperature increase to well below 2 ^oC, while aiming at 1.5 ^oC, above pre-industrial levels by 2100 (UNFCCC, 2015a). However, the aggregate of mid-term global GHG emission reduction addressed in Intended Nationally Determined Contributions (INDC) falls short in realising this goal by 10-16 $GtCO_{2eq}$/yr in 2030, compared to what is needed to reach the 2 ^oC target (Rogelj et al., 2016).

Fossil resources, namely coal, lignite, crude oil and natural gas, dominate the energy system as they contribute about 81% to the global Total Primary Energy Supply (TPES). They are used to meet the demand for electricity, heat, transport fuels and materials such as bitumen, coke, chemicals and derivatives. The provision of these products and energy services based on fossil resources is deemed unsustainable, as the use of fossil resources contributed 78% to the GHG emission increase since 1970, followed by emissions from forestry and other land use (IPCC, 2014a). Apart from being the largest source of anthropogenic GHG emissions, fossil fuels are deeply embedded in the world's economy. Infrastructure, deployed fossil fuel conversion (e.g. power plants, refineries) or end-use demand technologies (e.g. internal combustion engines) have been developed around fossil fuels. Furthermore, the installed capacities, such as oil refineries, are rather inflexible to supply-demand responses. Fossil fuel use raises concerns also on socio-economic impacts as their increasing consumption induces economic and political dependencies between energy supply and demand regions. Related issues include costs of supply, the strong correlation of fossil fuel consumption with economic growth, energy access and energy security. Major progress is required to ensure universal access to affordable, reliable, sustainable modern energy, in line with Sustainable Development goals (UN, 2016; World Bank, 2015). Decoupling economic growth from fossil fuel consumption is a complex task as the current economic system is largely dependent on these non-renewable energy resources and deep structural economic changes are required to transition to more sustainable socio-economic systems. These aspects are particularly relevant for individual countries (e.g. Germany, the Netherlands) that have large capital stock of oil refineries and other petrochemical conversion technologies (e.g. steam crackers) making their energy and economic system highly dependent on fossil fuels.

Similar to fossil resources, world food supply is not equally distributed across regions, food and feed production suffers from inefficiencies, for example in agricultural productivity, and is a major source of GHG emissions. Furthermore, food prices are vulnerable to disturbance, as it has been observed in recent subsequent periods (2006-2008, 2010-2011). A change in climate is only expected to exert more pressure on food production systems leaving the poorer parts of the world more vulnerable to meet basic food needs (IPCC, 2014a). Moreover, extinction of species is 10^2-10^3 greater than what can be attributed to natural reasons and has exceeded the safe operating space of the planet (Rockström et al., 2009). In this respect, changes in land resources are the key cause of increased loss of biodiversity and associated ecosystem services. Pressure on land is expected to increase as population grows and climate change intensifies.

These challenges frame the direction in which global coordinated action is required to reduce threats on the sustainability of the planet that the future of human society depends on. Global scenario assessment studies show that the mitigation of GHG emissions requires fundamental changes in the energy supply and demand system by means of large substitution of fossil energy resources through increased supply of renewable energy, deployment of CO_2 mitigation technologies and increase in energy efficiency (IPCC, 2014a). Most studies demonstrate that biomass, as a renewable energy resource, next to renewable energy supply from wind, solar, hydropower and geothermal, mitigation technologies and other measures will have an important role in realising long-term GHG emission reduction targets (IPCC, 2014a).

1.2 ASSESSING BIOECONOMY DEVELOPMENTS IN THE TRANSITION TO A SUSTAINABLE ENERGY SYSTEM

In the need to address climate change and other societal challenges, the concept of transitioning from a fossil-based economy to a bioeconomy is receiving increasing attention (see e.g. EC (2012a), OECD (2009), TKI-BBE (2015a), White House (2012)). In general, a bioeconomy covers all economic activities that are generated by renewable biological resources (McCormick and Kautto, 2013). This description illustrates the broad coverage of activities that bioeconomy is comprised of, from primary production to end-use, and implies the complex relationships and interdependencies that characterise it. The term bio-based economy is also frequently used, which describes all economic activities based on biomass, with the exception of human food and feed (Kwant et al., 2015). In this dissertation bioeconomy is assessed as the means to transition to a sustainable energy system, thereby leaning closer to the definition of Kwant et al. (2015).

1.2.1 Overview of current biomass use for energy and materials

Biomass has been used throughout the history of mankind for food, feed, traditional energy uses and materials. The main types of biomass include agricultural crops (sugar, starch, oil), forestry, their residues and grasses. Currently, the total demand for biomass utilised for all human purposes ranges between 215-270 EJ/yr (assuming 18 GJ/t average heating value; 12-15 billion tonnes dry matter (bnt_{dm}/yr) (Piotrowski et al., 2015; van den Born et al., 2014)). This amount is primarily consumed for food or feed (Figure 1.1). Bioenergy (i.e. electricity, heat and transport fuels from biomass) is the largest supplier of renewable energy (about 60 EJ in 2015 or about 10% of global TPES in 2015) and has been growing at around 2% per year since 2010 (REN21, 2016). In 2014, bioenergy supplied about 14% of global Total Final energy Consumption (TFC). Traditional bioenergy (mainly fuelwood and charcoal for cooking, lighting and space heating, primarily in developing countries) represented about two thirds of the global TFC (REN21, 2016). Such uses are characterised by low energy efficiency, labour intensity and cause negative impacts on health and living conditions (Chum et al., 2011). The remainder was modern bioenergy supply (i.e. efficient biomass conversion to bioelectricity, biomass heat and biofuels). Similar to bioenergy, materials from biomass are primarily conventional as they include wood products, construction materials, paper and pulp, and minor fractions are renewable organic inputs for the chemical industry. A characteristic example is the production of ethanol-based bulk chemicals (i.e. ethylene and ethylene derivatives) that account for about 50% of the global total bioplastics production capacity (IfBB, 2015).

1.2.2 Long-term developments of biomass use for energy and materials

The premise that bioenergy may become a significant supplier of modern renewable energy and other applications lies in that: (a) several primary feedstocks, which are widely distrib-

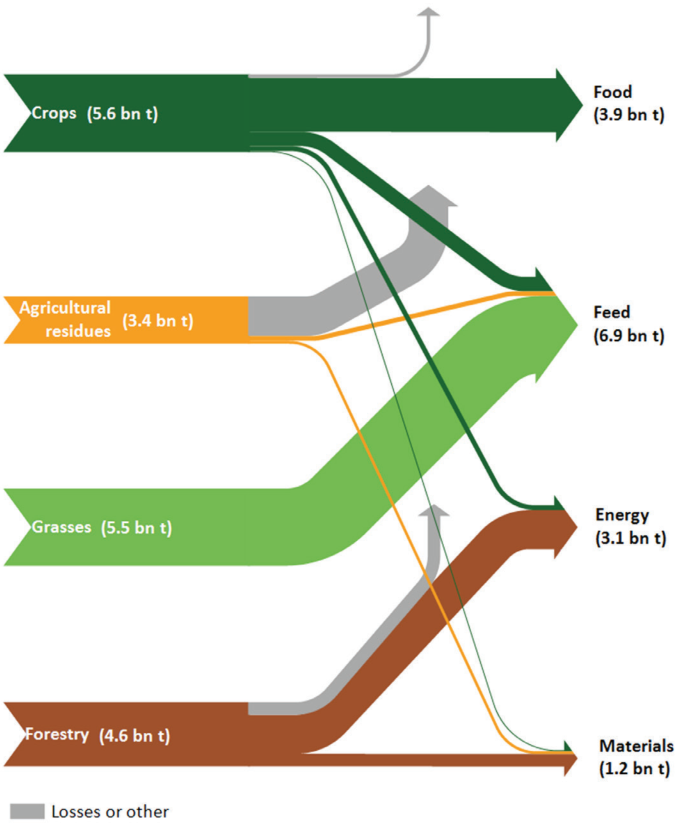

Figure 1.1 Global biomass flows from agriculture, grasslands, grazing lands and forestry in 2010. Secondary flows and conversion losses are not included in the illustration (based on data from van den Born et al. (2014))

uted across regions, can be used to supply energy, (b) most feedstocks and processed bio-based products can be stored for relatively long periods, transported and traded, (c) existing and prospective technologies enable the conversion of feedstocks for uses across the energy matrix, namely to supply electricity, heat and fuels, (e) recent advances in conversion technologies enable bioenergy supply to sectors that have no other renewable alternative options, namely jet fuel and shipping fuel but also non-energy, (f) biomass-derived energy carriers can be more easily integrated with the existing infrastructure of the energy system when compared to renewable alternatives, (g) if bioenergy is combined with carbon capture and storage (BECCS) it can result in negative emissions. Furthermore, studies indicate that levelised production costs of several modern biomass production systems are already cost-competitive to other renewable sources or the conventional fossil-based energy and products, while they can still benefit from cost reduction over time due to learning (Gerssen-Gondelach et al., 2014).

Modern bioenergy use is expected to increase given the significant deployment of renewable energy required to meet climate targets. Future outlooks on biomass use in the global energy system indicate that the demand for bioenergy is likely to be between 120-155 EJ/yr, or even up to 300 EJ/yr by 2050, which is higher than the current global biomass consumption for all purposes (Chum et al., 2011). Daioglou (2016) points out that the required biomass for bioenergy and biochemicals may lie between 129-163 EJ/yr to supply 13-15% of global TFC by 2050 (by 2100 it increases to 232-268 EJ/yr or 21-35% of TFC). Recent literature indicates that global biomass potential may range between 75-226 EJ/yr in 2050 (Daioglou, 2016; Saygin et al., 2014), which falls within the range of IPCC's technical potential with high-to-medium agreement for 2050 (i.e. 100-200 EJ; Smith et al. (2014)).

Production of biochemicals may become increasingly important over time due to the non-renewable energy resource substitution potential and the lower GHG emission profiles that they generally demonstrate when compared to fossil alternatives (i.e. 2-246 $MtCO_{2eq}$/yr per biochemical assuming complete replacement of their fossil counterpart (de Jong et al., 2012a), or up to around 6 tCO_{2eq}/t product in 2030 (Gerssen-Gondelach et al., 2014)). Today only a small share of the total global feedstock demand for chemicals originates from biomass (i.e. 0.6 EJ/yr; Saygin et al. (2014)). There is a very wide range of technical options and alternative pathways to produce biochemicals, with bioplastics representing a key product group with high growth rates, albeit lower than anticipated some years ago (EuBP, 2016; Shen et al., 2010). Similar to biofuels produced from lignocellulosic feedstocks (2nd generation biofuels), biochemicals have been facing barriers to rapid expansion over the last years. By 2050, global biomass demand for non-energy uses (i.e. the feedstock used as raw material that is not used for fuel purposes or transformed to fuels) may range from about 18 EJ (Daioglou et al., 2014), leading to a reduction of about 1.2 $GtCO_2$/yr, to 45 EJ (1.1. to 2.4 bnt_{dm}/yr, assuming 18 GJ/t average heating value) (Piotrowski et al., 2015). Dornburg et al. (2008) show a wide production range of biochemicals in the EU in the order of 4.8-113 Mt/yr by 2050, depending on fossil fuel price developments, technology maturity, biomass feedstock costs, and overall demand.

Global assessment scenarios also demonstrate that meeting long-term climate targets depends on the deployment of carbon mitigation technologies in fossil and biomass-based production systems. Carbon Capture and Storage (CCS) or its combination with bioenergy (i.e. BECCS) is a technology that captures CO_2 emissions from flue gases of stationary sources such as power plants or industrial processes and stores them in geological formations. CCS and BECCS have been proposed as technical solutions that may reduce total CO_2 mitigation costs compared to other options (Fuss et al., 2014). While there are major uncertainties on physical constraints on BECCS, on the response of geological formations to sequestered emissions, on the costs and financing of an untested technology and socio-institutional barriers (Fuss et al., 2014), the synergies of bioenergy with CCS, may be a critical pathway to achieve GHG emission reduction targets (Rose et al., 2014).

To summarise, in order to meet climate change mitigation targets, bioenergy demand for modern uses and emerging sectors (e.g. biochemicals) in combination with carbon mitigation technologies may increase substantially in the future. There are opportunities, which, if valorised, can position bioeconomy developments as a key strategy in meeting climate change and other sustainable development goals. In order to mitigate risks and valorise opportunities, short-term and mid-term assessments are required that address the relevance and urgency of bioeconomy to embark on pathways that tackle societal challenges.

1.2.3 The complexity of bioeconomy assessments

There is considerable complexity associated with assessments on the current and future role of bioenergy and biochemicals. Technically, any energy carrier and organic chemical can be produced from biomass. At each step of the supply chain, multiple options are available (e.g. different feedstocks, pretreatment technologies, biological, thermal, catalytic processes), often in varying combinations, with increasing complexity parallel to technological progress. Supply chains of established bioenergy pathways or advanced bioenergy and biochemical pathways have varying environmental and cost performance depending on the characteristics of the production system. Among others these include the type of feedstock used, its location, the agricultural management practices, the induced Land Use Change (LUC) and associated Direct (dLUC) and Indirect (iLUC) emissions, and the performance of the conversion technologies. For instance, combinations of land systems and conversion technologies are found to influence the GHG emission savings of biofuels (Hoefnagels et al., 2010), due to LUC emissions that may even be higher than the avoided emissions due to fossil fuel substitution (Fargione et al., 2008; Searchinger et al., 2008; Wicke et al., 2012). Furthermore, deployment levels of bioenergy and biochemicals are highly uncertain. In the mid term, supply chains may divert from current patterns. Supply potentials and their regional distribution will depend on developments in biomass producing sectors (e.g. agriculture, forestry). Demand regions may invest further in their domestic biomass conversion capacity if technologies become more competitive, thereby increasing the demand for biomass. Feedstock supply to the demand regions will depend on the progress of international biomass trade. Such developments might be accelerated or delayed depending on the level of policy ambitions to mitigate GHG emissions and future fossil fuel prices.

However, to assess the influence of these characteristics is not as straightforward because biomass deployment in the economy also depends on several dynamic factors based on interaction between biomass supply and the technologies in the energy system (including the chemical industry). If bioeconomy sectors grow, potential conflicts may arise from end-use sectors when in competition for the same feedstock. At the same time, cross-sectoral synergies may surface if multi-output conversion technologies (e.g. biorefineries) are deployed. Furthermore, other renewable energy conversion and carbon mitigation technologies (CCS, BECCS) compete with biomass in the supply of renewable energy and as a result in their potential contribution to emission reduction. However, there are uncertainties on the readiness and speed of technical change of biomass conversion, other renewable energy and carbon mitigation technologies that ultimately determine their economic competitiveness against the fossil energy system.

The complexity of bioeconomy developments is addressed by a variety of tools and methods that have fundamental differences in the activity they evaluate, their geographical detail, their temporal scope, the impact categories and their interactions with the human and natural system (van Leeuwen et al., 2015). Wicke et al. (2015) classify the bioeconomy assessment tools and methods into the following categories: (a) bottom-up models and analyses, (b) Partial Equilibrium Models (PEMs), (c) Computable General Equilibrium (CGE) models and (d) integrated assessment models (Box 1.1). Each tool and method is designed to answer specific types of questions and so far none can assess bioeconomy in its entirety. This entails that the available toolkit demonstrates strengths in addressing specific issues but limitations in representing the complexity of bioeconomy and its potential impacts.

1.3 KNOWLEDGE GAPS

The environmental and economic impacts, the potential size and the production costs of bioenergy and biochemicals are context-specific. Nonetheless, emerging bioeconomy sectors and supply chains are underrepresented in methods and tools, and as a result the underlying factors that can impede, safeguard or stimulate bioeconomy developments are not addressed with the required context-specific detail. Therefore, the implications of biomass production and conversion in different locations, point in time or deployment within economies that have varying structural characteristics need improved understanding. The most relevant knowledge gaps, though not limited to, are:

On the environmental impacts of emerging bioeconomy products across established supply chains. Among several other bottom-up tools and analyses, Life Cycle Assessment (LCA) is perhaps the most widely applied to assess the environmental performance of products and services (Jones and Azapagic, 2016). Several studies have assessed bioenergy, biofuels, biochemicals or other uses of biomass and compared them to their petrochemical counterparts (Chen and Patel, 2012; Cherubini and Strømman, 2011; Hoefnagels et al., 2010; Weiss et al., 2012). Nonetheless, results of such studies do not always agree. Reasons for such differences are closely related with methodological intricacies and with the characteristics of biomass production systems (Cherubini et al., 2009; Reap et al., 2008a, 2008b). One of the most fundamental and unresolved issues in LCA is related with the multi-functionality (or allocation) problem that occurs in multi-output processes. Biomass supply chains are faced with the multi-functionality problem from production to conversion and this often leads to divergent results even for the same production system (Cherubini et al., 2009; Hoefnagels et al., 2010). The multi-functionality problem in LCA is associated with the type of questions that the study aims to address.

Typically, LCA studies follow two approaches. The first approach, referred to as Attributional LCA (ALCA), is a static assessment, as it analyses the environmental impacts of a product or services based on the current (or recent) average state of the technology and infrastructure while ignoring dynamic economic interactions (Pawelzik et al., 2013). In ALCAs, typically, energy, mass, or economic allocation procedures are used to account for co-products. In contrast, the second approach, known as Consequential LCA (CLCA), assesses marginal impacts by assuming increase in production and is more dynamic as it aims to capture market-mediated effects. CLCAs typically imply system expansion when dealing with multi-output systems and rely on assumptions regarding the changes in the marginal system (e.g. land, energy) that are induced by increase in production. CLCA results are deemed more appropriate for policy makers (Pawelzik et al., 2013; Plevin et al., 2014), however, require careful interpretation. The results and the insights based on these two approaches vary significantly and may lead to different conclusions even when applied on the same product-system. Several other issues are also found to influence outcomes namely differences in data (year, quality), the geographic location, non-GHG emissions of bioenergy systems, assumptions on carbon storage in bio-based products and so forth (Cherubini et al., 2009; Reap et al., 2008a, 2008b). Furthermore, GHG emissions is the most-assessed impact category with LCA (Jones and Azapagic, 2016). Nonetheless, potential environmental impacts of bioeconomy products go beyond these as they extend to environmental impairment, human health and water scarcity (see e.g. Weiss et al. (2012)) and there is need to comprehensively explore the solution space with respect to trade-offs. The above are not well understood in established biochemical supply chains, namely those of bioplastics production. The focus of most environmental assessments, including LCA has been on products and services produced and used in the industrialised world, while there is need to adequately cover also existing and emerging systems in spite of the additional complication related to data and region-specific context. Therefore, there is need to comprehensively assess current bioplastics supply chains by taking the characteristics of their production systems within their region-specific context into account and provide insights based on the different methodological LCA applications.

On the potential role and contribution of biomass within a national context in the transition towards a more sustainable energy system. The representation of bioeconomy in energy systems models, whether with a long-term or mid-term temporal scope or with a global or region-specific geographic focus has shortcomings. Firstly, it does not explicitly describe emerging sectors such as biochemicals (that have a high growth potential) and thus does not account for all competing biomass uses. Secondly, technological details are typically aggregated at a high level, and thus bioeconomy assessments are not able to represent important aspects relevant for emerging pathways. Biorefineries are expected to be a key technology concept as they can process different biomass constituents to useful outputs for end-use sectors (Cherubini, 2010; de Jong et al., 2012a). Therefore the multi-functionality problem as it arises in product-specific studies (e.g. LCA) is less relevant. On the other

Box 1.1 Methods and tools for bioeconomy assessment

Bottom-up models and analyses are typically based on highly detailed representation of technologies, processes, agents or resources. Prominent examples are process-based technical models, environmental life cycle assessment (e.g. Wang et al., 2012), life cycle costing and social life cycle assessment (e.g. Hellweg & Milà i Canals, 2014), process-based biophysical models (e.g. Fischer et al., 2010; de Wit et al., 2014), land-use allocation models and spatial analysis (e.g. van der Hilst et al. (2012)), bioenergy supply and demand mapping (e.g. Masera et al. (2006)), statistical scenario analysis of biomass resource availability (e.g. Smeets et al. (2007)), cost-benefit analysis, multi-criteria analysis and prospective studies (e.g. van den Wall Bake et al. (2009)). By incorporating high level of detail, they can demonstrate with high precision potential impacts of the bio-based system they analyse. This, however, entails that they have narrowly defined regional, temporal and sectoral scope, which limits their capacity to capture interactions with systems outside their boundary. Furthermore, many bottom-up models tend to focus on a specific set of impacts or dynamics such as the effect of land use change on greenhouse gas emissions but fail to capture impacts on other areas of concern. Moreover, due to the highly complex dynamics that can be induced by bioeconomy developments, such methods are not suitable to assess indirect effects such as indirect land use change emissions, price responses and substitution effects (Wicke et al., 2015). Consequential life cycle assessment enabled by model collaboration, is a prominent exception as it is appropriate to analyse policy-induced changes in emissions; however, its application has focused mainly on evaluating indirect land use change emissions, while they are several other policy-induced emission changes (Creutzig et al., 2012).

Partial equilibrium models focus on detailed representation of specific economic sectors such as food markets or energy systems. They operate under neo-classical assumptions by clearing supply and demand of markets they describe. Their sectoral and geographical scope differs depending on the questions they are designed to address and may vary from simple supply-demand relationships between a single product and a region to multi-commodity markets and grid-cell level regional representation. They are highly flexible in incorporating large amount of bottom-up detail, for example on agricultural management practices or technologies. This makes them suitable to assess sectoral dynamics, and thus address first-order effects of policy intervention such as a carbon tax or a support of technologies. Nonetheless, as partial equilibrium models do not interact with all economic sectors they cannot represent impacts outside their boundaries, which due to the interconnected structure of bioeconomy with the rest of the economy can prove to be critical. Furthermore, in the absence of macro-economic closure they may introduce bias when sectors have a big role in an economy (Wicke et al., 2015).

Computable general equilibrium models have been established for long to offer policy advice in several areas such as taxation and trade, human diseases, international labour migration and climate change adaptation. Their strength lies in that they cover comprehensively key relationships between all economic sectors and regions. They are deep structural macro-economic models that explicitly solve the maximisation problem of consumers and producers. They allow to account for production factor and market price adjustments, associated changes in trade and market balance, to provide indicators on the full costs and benefits of policies, their distribution within and among regions and agents, and finally, their consequences on income, growth and job markets. Recent efforts led to the improved representation of biomass in computable general equilibrium models. The focal areas include endogenised modelling of land markets (Wicke et al., 2012), improvement of biofuel production chains (Banse et al., 2011; Laborde, 2011) and prospective biomass feedstocks for advanced biofuel production (Taheripour and Tyner, 2013; van Meijl et al., 2012). However, their level of aggregation is often too high and is unable to demonstrate the variation in underlying constituents elements (Wicke et al., 2015).

Integrated assessment models have been designed to assess interactions between human and natural systems such as land, energy, economy and climate. Typically, integrated assessment models have spatially explicit representation of land and biophysical processes while the human-economic system if often simplified. In terms of impacts they assess, integrated assessment models can be very useful for long-term holistic bioeconomy assessment. Due to their lack of detail in representing smaller scale dynamics, for instance, those enabled by technologies, they are not suitable for shorter-term analyses (Wicke et al., 2015).

hand, high-value biomass applications (Bozell and Petersen, 2010), cascading uses of biomass (Keegan et al., 2013), cross-sectoral synergies, and co-production are few of the issues that need to be included in modeling frameworks. As a result, the implications of bioeconomy are often masked and therefore possible shorter-term actions required for the transition towards a sustainable energy system are not stipulated explicitly. All these aspects need to be addressed in an environment where other renewable energy technologies and fossil technologies also develop. Key variables, such as biomass potentials and their effects on cost of supply, uncertainty in technology development, effects of fossil fuel prices can shed light on the threats and opportunities in the transition to a bioeconomy that leads to cost-efficient realisation of long-term climate targets. Focusing on the structure of the energy and industry system of a national economy and for the medium term is required to in order to embed key factors pertinent to bioeconomy developments in energy systems models.

On the macro-economic impacts of emerging bioeconomy developments. Analyses of macro-economic impacts of bioeconomy developments can provide insights into indicators such as economic growth, employment and regional development. Typical tools used for macro-economic analyses include CGEs, PEMs. CGE models are able to demonstrate economy-wide impacts as they can capture linkages and interactions between different sectors to demonstrate direct and indirect effects of biomass use. CGE models have been used to address implications of biofuel policies on agricultural markets, LUC and leakage effects (Banse et al., 2011; Laborde, 2011). Significant efforts took place to improve the representation of biomass flows among which endogenised modelling of land markets and enriched biomass supply modules to account for prospective technologies for biofuel production. On the demand side, however, most models focus on biofuels and less on other bioeconomy sectors. Nonetheless, significant investments are expected to take place in bioeconomy sectors apart from biofuels in order to support the transition to the world onto a 2 °C trajectory. The International Energy Agency (IEA) estimates that about 1.8 trillion € are required on bioenergy supply between 2014 and 2035 (IEA, 2014). Such investments will generate direct and indirect macro-economic impacts. Within modern bioenergy sectors, 2.4 million people are employed globally, almost half of which in modern bioenergy sectors (biofuels) (IRENA, 2013). According to the REmap 2030 scenario of IRENA (2013), global direct and indirect employment induced by renewable energy sectors can reach 16.7 million, of which bioenergy accounts for almost 60%. With the expected growth of emerging bioeconomy sectors, it is important to include them in a comprehensive framework to quantify the overall impact of bioeconomy on macro-economic trajectories. Existing CGE models are too aggregate to provide such information. By using bottom-up technology details existing frameworks can be improved and help close this gap. Model collaboration can take place as alignment and harmonisation of input data, detailed model comparison and model linkage (Wicke et al., 2015). Similar to energy systems models, focusing on a national economy is required to provide insights in the underlying elements that need to be accounted for in the structure of CGE models in order to assess advanced bioeconomy sectors and reveal implications on the macro-economy, taking global dynamics into account.

1.4 AIM AND OUTLINE OF THE DISSERTATION

In light of the above, this dissertation aims to address some of the knowledge gaps related with current and mid-term developments of emerging bioeconomy products and sectors. It provides different insights in the performance of bioenergy and biochemical production systems and the synergies and the conflicts between bioeconomy sectors by using bottom-up and top-down modelling approaches developed for and applied in region-specific supply chains and in a national context. These insights are needed for an effective design of bioeconomy in the medium term and provide evidence about the potential impacts of novel applications today or in the near future. To achieve this aim, it assesses the environmental performance of established biofuel and biochemical supply chains with core process steps occurring in developing and emerging economies (India and Brazil) by addressing methodological implications of LCA. Furthermore, it improves the representation of bioenergy and biochemical production chains in an energy systems and a macro-economic CGE model, paying particular attention to biomass supply and advanced conversion technologies. To obtain insights in the structure and key parameters that are critical for the models' extension, it places its focus on the energy system of the Netherlands. The Netherlands has been selected on the premise that it can support large-scale bioeconomy developments. The country has one of Europe's most efficient and advanced agricultural sectors, which is nevertheless limited by the domestic supply of biomass and land availability; therefore, it relies heavily on trade. Moreover, the Netherlands has developed a competitive logistics infrastructure over the years and it holds a strong position in the production of fuels and chemicals. Beyond 2030, the gradual depletion of natural gas reserves will change the country from a net gas exporter to a net gas importer; therefore, the transition to a more resource-secure and sustainable energy system is required. By improving the understanding of environmental and macro-economic impacts that emerging bioeconomy products and sectors may have, this dissertation seeks to demonstrate what the role and implications of large-scale bioeconomy deployment might be in the medium term as required in order to transition to a sustainable system that meets long-term societal challenges.

To this end, the following research questions are addressed:

I. What are current and potential mid-term environmental implications of biofuels and biochemical applications?
II. What is the potential size and contribution of bioeconomy in a national energy system in pursuing mid-term climate change mitigation targets?
III. What are the potential mid-term economic impacts of bioeconomy developments at a national level?
IV. How can tools and methods for assessing current and mid-term bioeconomy developments be improved?

This dissertation addresses the research questions I to IV in Chapter 2 through Chapter 6 (Table 1.1). Each chapter uses a different methodological approach and varies in its regional and temporal scope, and the system it analyses in order to assess sustainability aspects of bioeconomy (Table 1.1, Figure 1.2). Chapter 2 examines the environmental impacts of ethanol production from sugarcane in India and Brazil and trade-offs therein. Particular focus is placed on the multi-functionality problem that occurs in multi-output processes. Chapter 3 uses the findings of the analysis on sugarcane ethanol production and extends it in a methodologically consistent manner to the production of bioplastics. It addresses methodological issues related with multi-functionality of product-systems, it examines trade-offs across environmental impact categories and compares the findings to European supply of petrochemical plastics. It also applies scenarios that focus on technology improvement and other interventions across the supply chain to reduce the environmental impact in the near term. Chapter 4 extends a cost-minimisation linear programming energy systems model of the Netherlands to include emerging bioeconomy sectors. It pays particular attention to biorefineries in order to demonstrate cross-sectoral synergies. Furthermore, it examines how different technology development rates may affect mid-term bioeconomy developments. Chapter 5 uses scenario analysis to assess energy transition pathways for the Netherlands in efforts to comply with the 2 °C target trajectories in the medium term. It focuses on how climate policy, biomass cost-supply, and fossil fuel price variation can influence bioeconomy developments. Chapter 6 demonstrates how emerging bioeconomy sectors can be incorporated in a macro-economic model by using bottom-up information and deploys it to gain insights in macro-economic impacts of mid-term bioeconomy developments in the Netherlands.

Following this introduction, the remainder of this dissertation is structured as follows. To address research question I, **Chapter 2** applies LCA to existing supply chains of ethanol production from sugarcane in Brazil and India. Sugarcane ethanol is selected because it contributes significantly to global biofuel supply and is currently a key feedstock used in the production of bioplastics. The selected regions are the world's largest sugarcane producers. In this chapter particular attention is paid to detailed bottom-up data of the production chain from feedstock to conversion to enable the assessment of a wide range of environmental impacts. Furthermore, the production systems are analysed in detail to address in a methodologically consistent manner the multi-functionality problem, where it arises, by applying three different approaches (two system expansion variants and economic allocation). These insights are also used to address research question IV.

Research question I is investigated further in **Chapter 3**. This chapter uses the key findings of the analysis on sugarcane ethanol chains and extends it in a methodologically consistent manner on the production of the plastics Polyethylene (PE) and Polyethylene Terephthalate (PET) from biomass and their supply to Europe. It addresses how the methodical choices and production systems can affect the environmental profiles of bioplastics and it reveals trade-offs across environmental impact categories that may occur (GHG emissions, non-renewable energy use, ecosystem quality, human health). Therefore, it also provides answers to research question IV. Furthermore, it investigates the influence of technology improvements and other measures across the supply chain on the environmental profile of bioplastics in the near term. By comparing the existing supply chains of bioplastics with the production of petrochemical counterparts in Europe, it draws conclusions as to the gains and concerns that can be expected by replacement of fossil resources by biomass.

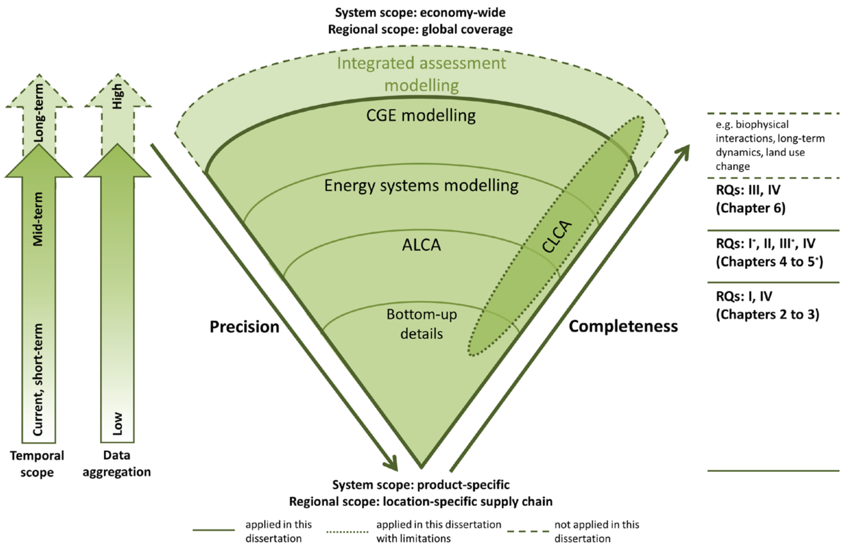

Figure 1.2 Methodological framework used to assess the implications of bioeconomy developments in this dissertation. ALCA stands for Attributional LCA, CLCA stands for Consequential LCA, CGE stands for Computable General Equilibrium modelling (adapted from Creutzig et al. (2012))

Table 1.1 Overview of the topics of this dissertation in relation to the research question they address

Chapter	Topic	Method	Regional scope	System scope	Temporal scope	Research question			
						I	II	III	IV
2	Life cycle assessment of sugarcane ethanol production in India in comparison to Brazil	LCA	India, Brazil	Product-specific	Current	X			X
3	Life cycle impact assessment of bio-based plastics from sugarcane ethanol	LCA	India, Brazil, Europe	Product-specific	Current, short-term	X			X
4	Emerging bioeconomy sectors in energy systems modelling – Integrated systems analysis of electricity, heat, road transport, aviation and chemicals: a case study for the Netherlands	Cost-optimisation	The Netherlands	Energy system	Mid-term		X		X
5	The role of bioenergy and biochemicals in CO_2 mitigation through the energy system – A scenario analysis for the Netherlands	Cost-optimisation	The Netherlands	Energy system	Mid-term	X	X	X	X
6	On the macro-economic impact of bioenergy and biochemicals – Introducing advanced bioeconomy sectors into an economic modelling framework with a case study for the Netherlands	CGE modelling	Global, focus on the Netherlands	Total economy	Mid-term			X	X

Chapter 4 combines existing and emerging biomass conversion technologies, CCS, BECCS, and the aviation and chemical sector in a cost-minimisation linear programming model of the energy system of the Netherlands. Firstly, it improves the representation of biomass cost-supply curves by incorporating several different biomass feedstocks and supply regions within the EU. Secondly, it extends its technology portfolio by including detailed cost-structures of 1st and 2nd generation biofuel technologies for road transport and aviation, biomass and fossil fuel conversion technologies to chemicals and other bioenergy options such as industrial biomass heat and biogas. Finally, it focuses on technological uncertainty by developing and applying two technology development scenario variants. This chapter addresses research questions II and IV in relation to energy systems modelling.

Chapter 5 assesses the required mid-term pathways for the Netherlands to embark on cost-efficient trajectories that pursue climate targets. To that end, it uses the model presented in Chapter 4. It applies scenario analysis as a method to assess cost-efficient pathways for the Dutch energy system for high and low technology development rates. Scenarios are assessed under the uncertain environment shaped by climate policy, biomass availability and cost of supply, and fossil fuel prices. Furthermore, it addresses the impact of sector-specific assumptions such as closure of coal-based capacity in the Netherlands and decrease in demand for chemicals. This chapter addresses research questions I through IV as it provides insights in the necessary preconditions to embark on cost-efficient CO_2 emission mitigation trajectories by estimating the renewable energy share, biomass consumption and CO_2 emissions under different scenario conditions until 2030.

Chapter 6 addresses the limitation of existing CGE models to account for emerging bioeconomy sectors. It demonstrates how detailed bottom-up information can be used to improve the sectoral representation of bioeconomy in top-down modelling, thereby addressing research question IV. By harmonising input data, soft-linking the bottom-up model developed in Chapter 4 and comparing model outcomes in detail it provides insights in the macro-economic impacts of bioeconomy developments in the medium term. The CGE model is global thereby taking into account all interactions of bioeconomy with the rest of the economy and indirect effects; however, results are presented for the Netherlands, which are used to provide answer on research question III.

Chapter 7 summarises and reflects on the findings of Chapter 2 to Chapter 6 in relation to the research questions. It provides recommendations that can be useful to policy makers and other stakeholders of bioeconomy and concludes by indicating directions that deserve further research.

2

Life cycle assessment of sugarcane ethanol production in India in comparison to Brazil

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ABSTRACT

Purpose: India's biofuel programme relies on ethanol production from sugarcane molasses. However, there is limited insight on environmental impacts across the Indian ethanol production chain. This study closes this gap by assessing the environmental impacts of ethanol production from sugarcane molasses in Uttar Pradesh, India. A comparative analysis with South-Central Brazilian sugarcane ethanol is also presented to compare the performance of sugarcane molasses-based ethanol with sugarcane juice-based ethanol.

Methods: The production process is assessed by a cradle-to-gate Life Cycle Assessment. The multi-functionality problem is solved by applying two variants of system expansion and economic allocation. Environmental impacts are assessed with Impact 2002+ and results are presented at the midpoint level for greenhouse gas emissions, non-renewable energy use, freshwater eutrophication and water use. Furthermore, results include impacts on human health and ecosystem quality at the damage level. Sensitivity analysis is also performed on key contributing parameters such as pesticides, stillage treatment, irrigation water.

Results and discussion: It is found that, compared to Brazilian ethanol, Indian ethanol causes lower or comparable greenhouse gas emissions (India: 0.09-0.64 $kgCO_{2eq}/kg_{etOH}$, Brazil: 0.46-0.63 $kgCO_{2eq}/kg_{etOH}$), non-renewable energy use (India: -0.3-6.3 MJ/kg_{etOH}, Brazil: 1-4 MJ/kg_{etOH}), human health impacts (India: 3.6·10-6 $DALY/kg_{etOH}$, Brazil: 4·10-6 $DALY/kg_{etOH}$) and ecosystem impairment (India: 2.5 $PDF{\cdot}m^2{\cdot}yr/kg_{etOH}$, Brazil: 3.3 $PDF{\cdot}m^2{\cdot}yr/kg_{etOH}$). One reason is that Indian ethanol is exclusively produced from molasses, a co-product of sugar production, resulting in allocation of the environmental burden. Additionally, Indian sugar mills and distilleries produce surplus electricity for which they receive credits for displacing grid electricity of relatively high CO_2 emission intensity. When economic allocation is applied, the greenhouse gas emissions for Indian and Brazilian ethanol are comparable. Non-renewable energy use is higher for Indian ethanol, primarily due to energy requirements for irrigation. For water use and related impacts, Indian ethanol scores worse due to groundwater irrigation, despite the dampening effect of allocation. The variation on greenhouse gas emissions and non-renewable energy use of Indian mills is much larger for high and low performance than the respective systems in Brazil.

Conclusion: Important measures can be taken across the production chain to improve the environmental performance of Indian ethanol production (e.g. avoiding the use of specific pesticides, avoiding the disposal of untreated stillage, transition to water efficient crops). However, to meet the targets of the Indian ethanol blending programme, displacement effects are likely to occur in countries, which export ethanol. To assess such effects a consequential study needs to be prepared.

2.1 INTRODUCTION

In 2008, India imported more than 70% of its oil requirements and more than 15% of its demand for high quality coal. Among the three end-use sectors, transport accounted for around one third of the total final energy consumption (OCED/IEA, 2011). While the country's domestic energy production is expected to increase, the combination of other drivers (e.g. population growth and economic growth) is expected to push the energy demand for transport even further, thus creating a stronger dependency on foreign oil. Recognising these dynamics, the Indian government launched a major programme for the production of biofuels in 2001, in order to introduce an alternative to petroleum-based fuels. Also, the programme intended to contribute to global greenhouse gas mitigation, reduce oil imports and generate employment (GoI, 2003).

Due to the prominent position of the sugar industry in India and the established ethanol production by distilleries for potable liquor and industrial use (Table 2.1), the Indian government mandated in 2009 a 10% blending target across 20 states and 4 Union Territories and proposed an indicative blending target of 20% for ethanol and biodiesel by 2017 (GoI, 2009). However, through the course of the Ethanol Blending Programme (EBP) this target has been partially met primarily due to fluctuating supply of sugarcane molasses and its impact on ethanol production costs. As a consequence, the Indian government revised the mandatory blending target to 5% (USDA, 2013). Biodiesel production for transport is not commercialised, as the potential for cultivation and utilisation of jatropha plantations remains untapped (USDA, 2013). Therefore the use of biofuels in transport relies on the success of the Indian EBP.

Table 2.1 Bio-ethanol production and consumption in India, million litres (USDA, 2013)

Year	Production	Total supply[a]	Consumption			
			Industrial use	Potable use	Transport	Total
2006	1,898	2,410	619	745	200	1,564
2007	2,398	3,160	650	800	200	1,650
2008	2,150	3,616	700	850	280	1,830
2009	1,073	3,025	700	880	100	1,680
2010	1,522	2,855	720	900	50	1,670
2011	1,681	2,766	700	850	365	1,915
2012	2,170	2,901	720	880	400	2,000
2013	2,239	2,955	740	910	450	2,100

[a]Includes beginning stocks and imports. Between 2006-2013, these account for 20%-55% and 1%-9% of total supply, respectively.

With the focus of the government and the industry on implementing the EBP, it becomes also important to address environmental concerns that characterise the ethanol production chain. For instance, Indian agricultural practices are characterised by excessive use of agrochemicals, especially nitrogen-based fertilisers (MoEF, 2009a). Also, the Indian Central

Pollution Control Board has classified the sugar and distillery industry among 17 industries with high polluting potential (CPCB, 2009). In addition, with sugarcane being a water-intensive crop, regional water stress is a significant resource constraint. Some studies report carbon emissions in the production of Indian ethanol and its use in the transport sector (e.g. Prakash et al. (2005)). Other types of emissions (e.g. Chemical Oxygen Demand; COD) have also been studied for specific steps of ethanol production (Tewari et al., 2007). However, in contrast to other countries that produce ethanol from sugarcane (e.g. Brazil, Australia, Thailand, Cuba, Mexico, Nepal), so far no comprehensive assessment has been carried out for molasses-based ethanol in India.

As molasses-based ethanol systems are characterised by multi-functionality due to co-production of sugar, molasses and bagasse or electricity in the sugar mill, the choice of allocation can be key in determining the results. Hoefnagels et al. (2010) demonstrated the latter for greenhouse gas and energy performance of biofuel production from thirteen feedstock types, including Brazilian sugarcane ethanol. However, sugarcane molasses-based ethanol was not included where different allocation options are also possible. Nguyen and Hermansen (2012), in line with ISO (2006a), show that system expansion is most appropriate to account for the multi-functionality problem of a sugar mill. In their study, molasses were assumed to displace feed, while in India molasses have traditionally been used for ethanol production. On the contrary, Renouf et al. (2011) note that system expansion is more valid for the determining product (sugar) while results for all products can be generated more consistently using allocation. However, in their study stillage was not digested anaerobically to cover energy demand of distilleries, thereby increasing the bagasse availability for additional power output as it is the case for most Indian distilleries. The multi-functionality problem of sugar mills and distilleries in India calls for an assessment of different approaches to account for impacts of molasses-based ethanol. Literature has extensively addressed the influence of allocation in GHG emissions of biofuels (e.g. Hoefnagels et al. (2010)). The authors analysed the energy and GHG performance of biofuel production from thirteen feedstock types and concluded that allocation is a key parameter in determining the results. It is the purpose of this study to provide an environmental assessment of Indian ethanol production, taking into account the system's intrinsic characteristics and their effects on allocation and to highlight potentials for improvement using life cycle assessment. To do so we assess sugarcane ethanol production in Uttar Pradesh (UP), India.

Furthermore it is unclear how the production of Indian sugarcane ethanol, which is exclusively based on sugarcane molasses, scores in environmental terms compared to production directly from sugarcane juice. To make this comparison, we also assess Brazilian sugarcane ethanol. Brazil has a long experience in sugarcane ethanol production, and therefore represents a good benchmark. For Brazilian sugarcane ethanol we extend analysis of previous studies to include additional impact categories (Seabra et al., 2011) and we divert from other studies on inventory data and method to account for impacts of pesticides (e.g. Cavalett et al. (2013), Ometto et al. (2009)).

For both systems we assess two extreme cases by assuming high and low conversion efficiencies. In parallel, we address the influence of allocation by applying three different approaches. The results include greenhouse gas emissions, non-renewable energy use, eutrophication, human health and ecosystem quality. In the following, we describe ethanol production in India and Brazil, highlighting their differences. We then present the methodology and data used to compare the two products. Finally, we present our main findings and discuss the most influential parameters for the various environmental impact categories and the sources of uncertainty.

2.2 SYSTEM DESCRIPTION

2.2.1 Ethanol production in India

India is the world's second largest sugarcane producer after Brazil. In 2009, Indian sugarcane production was 292 Mt (17% of the total global production of 1,700 Mt). Currently, sugarcane is being used to produce sugar, making India the world's second largest sugar producer and the world's largest sugar consumer (OECD/FAO, 2011; USDA, 2011). In 2009, India produced 21 Mt of centrifugal sugar, which represents 13% of total global production (USDA, 2011). Not all sugarcane is crushed in conventional mills to produce crystalline sugar; a significant share is used for the production of unrefined, mixed with molasses, non-centrifuged sugars (Gur and Khandsari). In this study, only sugarcane processed by conventional mills is taken into account (75% of total production).

In India, sugarcane grows in three distinct climatic-geographical regions: the subtropical Northern region, the Central-West subtropical peninsular region and the South-East tropical region. Across these regions there are differences in production practices, yield, sugar content and production cycle (Gopinathan and Sudhakaran, 2009). In most areas a one-year crop is followed by one ratoon crop (i.e. crop grown from the stubble of the harvested crop) (MoEF, 2010). Cultivation practices are almost exclusively manual, with the exception of ploughing, which is mechanised in some states (Gopinathan and Sudhakaran, 2009). This limits fossil inputs in sugarcane cultivation to agrochemicals and energy use for irrigation. Groundwater use is also significant since sugarcane is a water-intensive crop, especially when considering regional water scarcity in India. Additional inputs include manual labour and animal use. Unlike many other sugarcane producing countries, pre-harvest or post-harvest burning is not practiced in all regions. Sugarcane green tops are removed in the field and used as animal feed. Sugarcane is then transported to sugar mills by means of rickshaws, bullock carts and trucks (Kumar, 2013).

In the sugar mill, sugarcane is washed and shredded, and then the juice is separated from the fibrous bagasse (MoEF, 2010). Bagasse is predominantly utilised in co-generation facilities to cover energy requirements of the mill and to provide surplus electricity to the grid. Surplus bagasse is stored and used off-season to provide surplus electricity, is sold as solid biofuel, for paper production or animal feed (ISMA, 2011a). Crystalline sugar is produced by water evaporation after the juice has been heated, sulphitated, clarified and filtrated. The filtrate (called filtercake or mud, mixed with boiler ashes) is typically offered to sugarcane producers for free, who apply it back to the fields. After crystallisation of the clarified juice, residual sugars which cannot be recovered are separated by a centrifuge (MoEF, 2010). This co-product, known as molasses, is collected and used by distilleries to produce ethanol

(ISMA, 2011a). Approximately 95% of total molasses is directed to ethanol production. The remaining portion is mainly used as cattle feed (ISMA, 2011a). Distilleries are either adjacent to sugar mills or are stand-alone facilities (MoEF, 2009b). In the former case molasses are directly supplied to the facility and the energy requirements are met by the co-generation system of the sugar mill. Otherwise, molasses need to be transported and the distilleries cover their energy demand through other sources (e.g. bagasse, biogas) (Prakash et al., 2005). After dilution and fermentation of molasses the resulting broth contains 6-8% (v/v) ethanol and is passed through an analyser column for distillation to approximately 40% ethanol (v/v). The ethanol vapours are passed through a rectification column to produce hydrous ethanol of approximately 95% (v/v) concentration (rectified spirit). For fuel grade (anhydrous) ethanol, a dehydration step is required bringing the concentration to 99.5% (v/v). The effluent that exits the analyser column, known as stillage, has a very high chemical and Biological Oxygen Demand (BOD) and needs to be treated (MoEF, 2009b). Over 90% of Indian distilleries apply anaerobic treatment and recover biogas, which they use to cover own energy requirements (Tewari et al., 2007).

2.2.2 Ethanol production in Brazil

Brazil is the largest sugarcane producer of the world. In 2009 it produced 690 Mt, which represents 41% of global production (OECD/FAO, 2011). Sugar and ethanol are the two main products of sugarcane processing. In 2009, total production exceeded 31 Mt of sugar and 27.5 bn litres of ethanol (506 PJ; Lamers et al. (2011)), making Brazil the world's second largest ethanol producer and consumer after US. Brazil is also the world's leading ethanol exporter with exports peaking to 108 PJ in 2008 (Lamers et al., 2011). Most of the production is concentrated in South-Central Brazil. In the harvesting seasons 2004-2009, more than 85% of sugarcane, 90% of ethanol and 85% of sugar output in Brazil was produced in this region (UNICA, 2011). Brazil has long regulatory and technological experience in ethanol production. In 1975, large-scale development of ethanol plants was promoted under the ProAlcool programme. Since then ethanol plays an important role in the country's energy supply mix in the transport sector, accounting for 22% in road transport fuels in 2010 (IEA, 2012).

Sugarcane cultivation in Brazil offers high yields. It is harvested once per year in a 6-year cycle, during which five harvests (four of which are ratoon cultivations) and one field reforming cycle are performed (Macedo et al., 2008). However, there is some variation depending on local climate and cultivation practices. In the South-Centre 48% of the sugarcane is harvested with machinery (Seabra et al., 2011), while 52% is harvested manually. Until recently, sugarcane pre-burning was the dominating harvesting practice applied even on mechanically harvested areas. Based on state laws (No 11.241/02) and the industry association's protocol of intention, mechanisation is expected to increase and sugarcane trash pre-burning practices are expected to phase out (by 2031 based on State decree, or by 2017 based on the industry's protocol). Main inputs in Brazilian sugarcane production are agrochemicals, returned residues from ethanol production (filtercake, stillage, boiler ashes) and diesel used for land preparation, harvesting and ferti-irrigation. Contrary to production in UP, India, sugarcane crops in South-Central Brazil are not irrigated as the production is based on rainwater.

In Brazil, ethanol is produced in stand-alone or adjacent distilleries to sugar mills. The most important difference compared to the Indian system is that sugarcane juice is directly used for ethanol production, next to 10% of the Brazilian ethanol output, which originates from molasses (own calculations based on MME (2011) and UNICA (2011)). After harvested sugarcane has been transported to the mills by trucks, it is washed –if harvested by burning practices– and shredded so that juice can be extracted from bagasse. Apart from washing off the impurities, water is used to ensure higher sugar recovery. For physical treatment the juice passes a series of screens before entering the fermentation tanks. The filtercake is collected and applied as fertiliser on sugarcane fields. After fermentation, the resulting broth enters the analyser and the rectification column to produce 95% (v/v) hydrous ethanol. The stillage generated during ethanol production is sprayed on sugarcane fields as fertiliser. The majority of the bagasse is used in co-generation systems to cover all process energy requirements and to provide surplus electricity to the grid. Surplus bagasse is sold as solid biofuel. Brazil's car fleet includes 100% alcohol fuelled and flexible-fuel vehicles which use hydrous ethanol.

2.3 METHOD

2.3.1 System boundaries and functional unit

The main processes steps included in the system boundaries are: sugarcane production in UP (Northern India) and South-Central Brazil, sugarcane processing to sugar in UP, molasses processing to ethanol in UP and sugarcane processing to ethanol in South-Central Brazil (Figure 2.1, Figure 2.2). The system boundaries extend from cradle to gate, i.e. extraction of fuels and raw materials, production of material inputs and intermediate transport is included. The impact of infrastructure is excluded.

The functional unit is 1 kilogramme of hydrous ethanol (92.6-93.8% ethanol on a mass basis, the remainder is practically water) at the distillery gate. We exclude the use phase and consequently the comparison with conventional gasoline, firstly because we do not specify the transport and distribution of ethanol to gas stations; secondly, we do not study differences in fuel efficiency for ethanol-blends and gasoline; and thirdly, we intend to account for ethanol applications also in the chemical industry, where hydrous ethanol serves as suitable feedstock.

2.3.2 System range

In addition to assessing regional production in South-Central Brazil and UP, India (averge systemm performance), we present two extreme cases[1] (for greenhouse gas emissions and non-renewable energy use; extreme cases for impacts on human health and ecosystem quality are assessed by means of sensitivity analysis).

[1] The system's energy efficiency can be estimated either by accounting for the energy conversion efficiency (i.e. primary to secondary energy) and the process energy requirements (e.g. heat demand per unit of output) or by accounting for products (and co-products) per unit of total energy input, which is covered by bagasse. We assess a range based on conversion efficiency. However, efficiency improvements at the agricultural production phase (sugarcane productivity) are also important to assess the system's environmental and cost performance (van den Wall Bake et al., 2009).

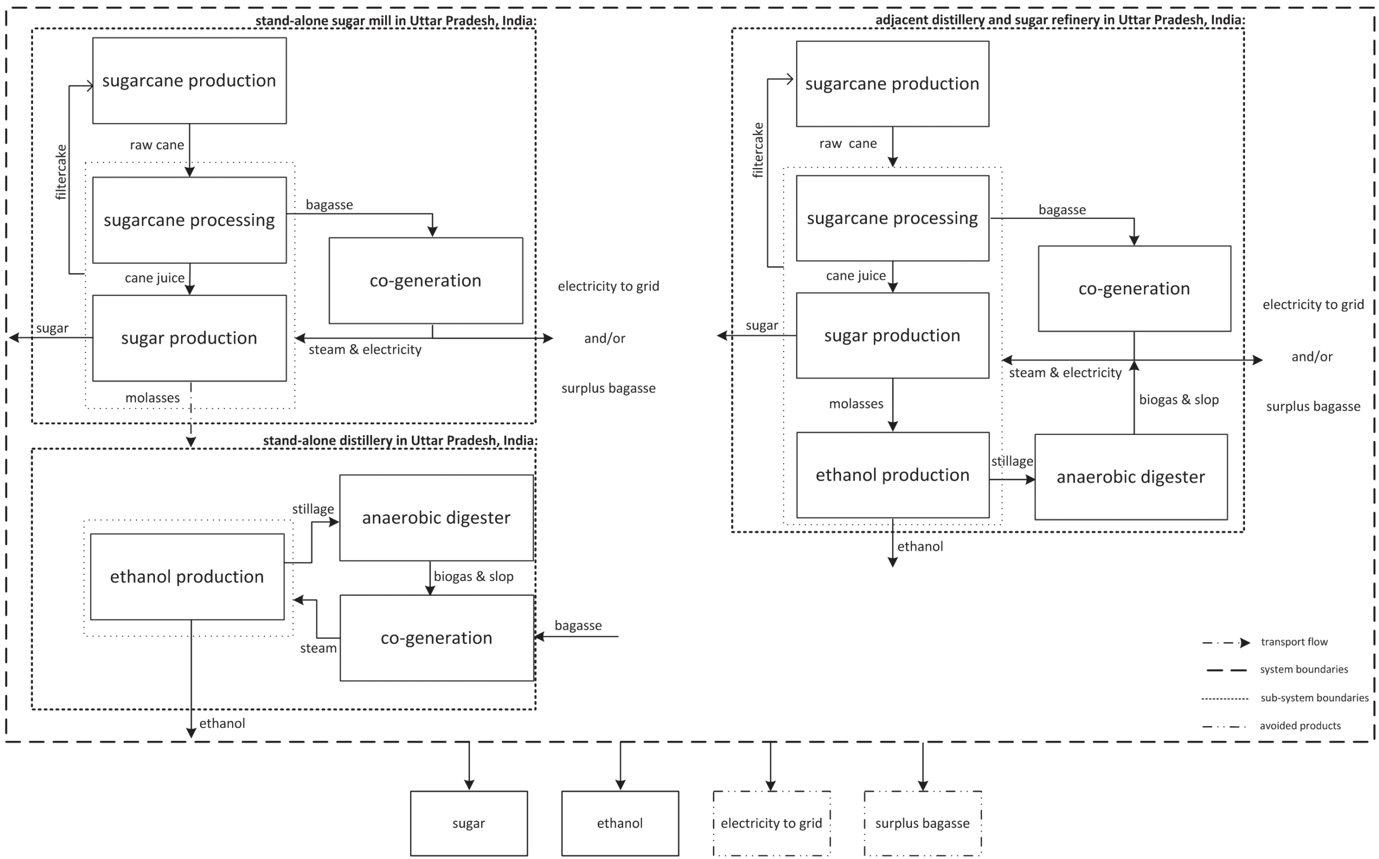

Figure 2.1 Foreground process-chain of ethanol production in India. Left: production at a stand-alone distillery, right: production at a distillery attached to a sugar mill. In this study a virtual average case is assessed on the basis of the products that exit the sector's boundaries within Uttar Pradesh, India

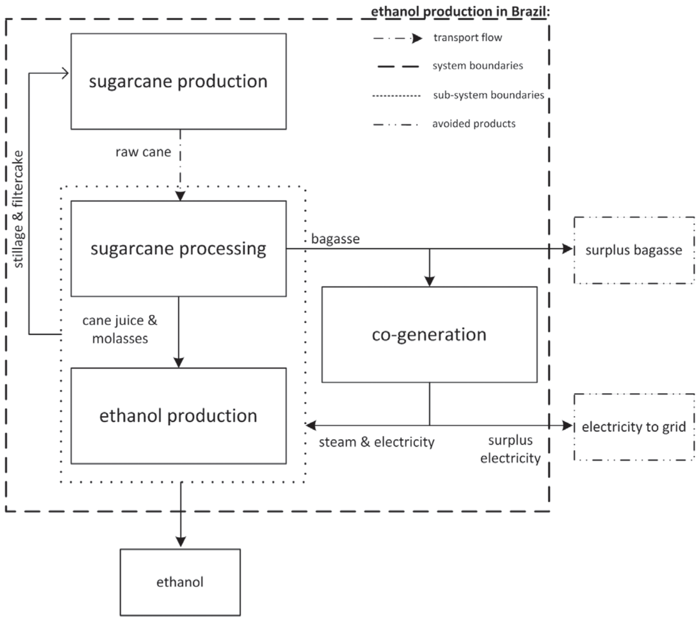

Figure 2.2 Foreground process-chain of ethanol production in South-Central Brazil

- *High system performance*: For India, only attached mills and distilleries that produce surplus electricity are accounted. The sugarcane input is assigned based on the mills' crushing capacity, which ranges from 2 to 11 kt_{cane}/day (DFPD, 2013; ISMA, 2011b). It is assumed that excess electricity is produced from all the bagasse available at the mills and from biogas that was recovered after anaerobic treatment of stillage. For Brazil, we assume higher process and co-generation efficiencies which lead to lower demand for process heat and higher surplus electricity per tonne of sugarcane processed (Seabra and Macedo, 2011).
- *Low system performance*: For India, we assume that no excess electricity is generated by mills. Mills use bagasse only to cover process heat requirement and sell any surplus biomass. Process electricity is supplied from the grid. Also, it is assumed that distilleries do not treat stillage anaerobically and cover heat demand by purchased bagasse and electricity demand from the grid. For Brazil, we assume all bagasse is consumed for own energy requirements, i.e. neither surplus electricity nor bagasse is provided by the mills.

The data used in this study are presented in Table 2.2 - Table 2.5.

2.3.3 System expansion and allocation

In our study we are confronted with multi-functional systems in several instances. In the Indian ethanol product-system, sugarcane processing produces sugar, molasses, surplus electricity and surplus bagasse. In the Brazilian product-system, sugarcane processing produces ethanol, surplus electricity and surplus bagasse. Other outputs produced within the system boundaries but have no market price (e.g. sugarcane trash, boiler ashes, filtercake) are assumed to be consumed within the system boundaries. The International Standardization Organization (ISO) recommends to solve multi-functionality problems by substitution (system expansion) and thus avoiding system partitioning (ISO, 2006a). Literature addressed the allocation problem of molasses-based ethanol production by applying different approaches to account for multi-functionality of sugar mills. Nguyen and Hermansen (2012) analyse stand-alone sugar mills and stand-alone distilleries and recommend system expansion to avoid allocation. In their consequential assessment, molasses are diverted from their use as feed and are assigned impacts of wheat production accounting for displacement effects. Renouf et al. (2011) applied system expansion to account for molasses-based ethanol for the system in situ, where sugar production is credited by avoided production of sorghum, which is used as feed. Consequently, sugar-mill co-products used for ethanol production have no impacts from sugarcane production. However, since in India molasses are used for ethanol production, system expansion cannot be applied following these approaches. In addition, applying substitution on surplus electricity output of Indian sugar mills may not be justified. The Indian electricity system is constrained since supply does not cover demand (CEA, 2011). Therefore it can be argued that in the short-term surplus electricity of sugar mills and distilleries stimulates additional consumption by marginal electricity consumers, while in the long term it may contribute to reducing the demand of additional capacity, which is primarily fossil-based (CEA, 2011).

Given the uncertainty that this system entails we distinguish the following approaches[2]:

- As 'reference' approach we assume that surplus electricity substitutes electricity of *low* CO_2 emission intensity. In UP this is justified due to the regional proximity to the Uttaranchal grid and in Brazil due to the average national electricity mix. Both grids consist of high hydropower capacity (CSO, 2012; OCED/IEA, 2011). For other co-products (sugar, molasses, bagasse) we apply economic allocation. We refer to this approach as 'SE-C', standing for "System Expansion-Conservative".
- As second approach we assume that surplus electricity substitutes grid electricity of *high* CO_2 emission intensity. In India, this is the average national electricity mix, which is primarily based on coal (OCED/IEA, 2011). Similarly, in Brazil, it reduces fuel use

2 System expansion is associated with consequential modelling. However, there are situations where it is applied to solve multi-functionality of foreground systems modelled by an attributional approach. These situations are encountered in product-related decision support studies that assess the life cycle of existing supply chains (EC 2010), similar to this study.

in the operational margin, which is predominantly natural gas (Seabra et al., 2011).[3] For surplus bagasse, we expand the system to include direct heat production from bagasse with 79% efficiency assuming that it displaces primary energy in fossil fuel-fired boilers. The fuels displaced depend on the fuels used for industrial heat production in each country. For India we assume displacement of coal-based heat, supplied with 80% efficiency.[4] For Brazil we assume oil-based boiler efficiency of 92%.[5] We apply economic allocation among the remaining co-products (sugar and molasses, applicable only for India). We refer to this approach as 'SE-O', standing for "System Expansion-Optimistic".

- In the third approach, we apply Economic Allocation (referred to as 'EA') across all products. This approach is justified, firstly because it accounts for the competitiveness of the sector's outputs based on economic criteria and secondly because it is consistently applied to all products, whereas in the other two approaches, a combination of system expansion and economic allocation is used. As electricity input we use national average grid.

In Nguyen and Hermansen (2012) and Renouf et al. (2011) feedstock energy is clearly separated among the subsystems of sugar and ethanol production. However, the system of our study includes both attached and stand-alone facilities, and biogas produced from stillage treatment of the ethanol subsystem also contributes to reducing net primary energy from other sources. This also calls for a different approach than found in literature. Therefore, at an intermediate level the system is broken down to the subsystems sugar and ethanol production and molasses are assumed as an intermediate product (Figure 2.1 - Figure 2.2). Surplus electricity that is produced from bagasse is assigned to the sugar product-system and surplus, which is produced by biogas to the ethanol product-system, based on primary energy. Depending on the allocation approach, surplus electricity is a critical parameter in determining the greenhouse gas emissions of ethanol production in UP. Therefore results are also presented for a range based on the electricity surplus of high and low system performance (section 2.3.2). In addition, it can be argued that since molasses do not exit the system boundaries, allocation approaches should consider only final products, namely sugar, ethanol, electricity and bagasse (Figure 2.1). This entails that the system is treated as 'black

3 In comparative life cycle assessment the two systems should have aligned regional scope. This entails for India and Brazil that each SE approach should consider displacement of national average electricity (based on JRC (2010)) or marginal electricity production. Instead, in this study we compare the systems on the basis of the credits assigned. This choice is made because the national average electricity fuel mix of the two countries differs significantly (fossil fuel-based and hydropower in India and Brazil, respectively). Considering the geographical context of the two systems and the JRC (2010) guidelines, the SE-O approach for India should be compared to the SE-C approach for Brazil.

4 Typically, coal boilers have higher efficiencies than biomass boilers. This study assumes similar efficiencies, which is likely for new biomass boilers that displace vintage coal-based boilers.

5 Alternatively, for surplus bagasse in Brazil we: (a) assume that it is combusted to increase power output with 25% efficiency (1.1 $kWh/kg_{bagasse,db}$), (b) assess pellet production, export to Europe and use in co-firing power plants where it displaces coal (section 2.5.1). Background information is included in the section 2.7.

box'. We solve the multi-functionality of the 'black box' based on: (a) price of sugar, ethanol and bagasse with credits assigned for surplus electricity with *low* CO_2 emission intensity (SE-C), (b) price of sugar and ethanol with credits assigned for surplus electricity with *high* CO_2 emission intensity and substitution of coal-based heat generation from surplus bagasse (SE-O) and (c) price of all final products.

Allocation factors, prices and system credits of each approach are presented in Table 2.6.

2.3.4 Impact assessment method

The impact assessment method used is Impact 2002+ v2.10 (Jolliet et al., 2003). We present results for the midpoint indicators greenhouse gas (GHG) emissions, Non-Renewable Energy Use (NREU), freshwater eutrophication, and water use. Furthermore, we present results on human health (HH) and ecosystem quality (EQ) at the damage level. For sugarcane production we also present the contribution of midpoint impact assessment results to the endpoints HH and EQ. GHG emissions are calculated for global warming potential of 100 years (IPCC, 2006). The characterisation factor for fossil and biogenic methane emissions is adapted to 27.75 $kgCO_{2eq}/kgCH_4$ and 25 $kgCO_{2eq}/kgCH_4$, respectively (Muñoz et al., 2013). We also provide an estimate on the influence of land use change emissions on the GHG profiles of ethanol, based on emission factors found in literature (section 2.5.1). We include the ozone depleting potential of nitrous oxides (N_2O), i.e. 0.017 $kgCFC\text{-}11_{eq}/kgN_2O$ (Ravishankara et al., 2009) because it is not considered in default characterisation factors of Impact 2002+. Lastly, impacts of aquatic acidification and freshwater eutrophication are linked with EQ, based on 8.82E-3 $PDF{\cdot}m^2{\cdot}y/kgSO_{2eq}$ and 1.4 $PDF{\cdot}m^2{\cdot}y/\ kgPO_{4\ eq}^{3}$, respectively (Humbert et al., 2012).

2.4 INVENTORY DATA

We have selected the study regions based on data availability and representativeness. For India, the state of UP is selected, being the largest sugarcane producing state accounting for 40% of the country's total production in 2009-2010. Moreover, approximately 32% of the country's total sugar production capacity and 34% of the country's total ethanol production is located in UP (ISMA, 2011a). The assessment of Brazilian ethanol production reflects practices of the south-central region, which is by far the most important cultivation area, i.e. 86% of the total planted area (UNICA, 2011).

For sugarcane production in UP, we use average sugarcane production yields from ISMA (2011c) and Kumar (2013), which are comparable with the average yields between 2001-2010. For agricultural inputs we use data from Kumar (2013), who compiled inventories for UP. Compared to agricultural statistical information on fertiliser consumption (GoI, 2013), data of Kumar (2013) indicate higher consumption for N and K_2O fertilisers (by 31% and 58%, respectively) and lower for P_2O_5 fertilisers (by 28%) per hectare. This difference is expected since the statistical information is not crop-specific while data in this chapter reflect sugarcane production. We include energy and groundwater requirement for irrigation based on MoEF (2010), Shah (2009) and Srivastava et al. (2009). We rely on survey-based irrigation data for the region (60 L/kg_{cane}), which also specify means of irrigation. Water footprint studies indicate higher consumption (140 L/kg_{cane}; Mekonnen and Hoekstra (2010)) but do not specify means of irrigation. As sensitivity analysis we assess impacts of low and high water use. Apart from seasonal variation, spatial variation in yields, inputs of agrochemicals, irrigation water consumption and practices is anticipated. However, there is limited information to support a further estimate on the range. For sugarcane production in South-Central Brazil we use industry-based data of the sugarcane technology centre, reported in Seabra et al. (2011). Parameters such as sugarcane productivity, unburned and mechanised area are representative for a large number of mills (up to 168), while the sample is smaller for diesel consumption, transport distances, and agrochemicals. When compared to aggregated regional data differences are expected. For example, in 2008, based on FAO statistics, Brazilian sugarcane yield was 79 t_{cane}/ha (FAOSTAT, 2011), while based on data in this study the yield in the South-Central region was approximately 10% higher (Table 2.2). We opt to use data from the sugarcane technology centre due to their traceability and reliability. Table 2.2 presents the inventory inputs of sugarcane production in India and Brazil. For sugarcane processing in India, we use sector-wide data (Table 2.3) on production volumes (ISMA, 2011a), sugarcane crushing capacity (DFPD, 2013) and ethanol production capacity (ISMA, 2011c).

Energy requirements of mills and distilleries that do not provide surplus power are estimated based on literature data on energy demand for sugar production (Jungbluth et al., 2007) and for distilleries (Prakash et al., 2005). Based on ISMA (2011a) and personal communication with the All India Distillers' Association energy requirements of stand-alone distilleries are met by biogas and biomass, which we assume to be bagasse (Gopinathan and Sudhakaran, 2009; Khatiwada et al., 2012). In this manner we estimate the net bagasse surplus assigned to sugar production as a co-product. Since in India bagasse flows are not monitored results include a range for different net output assuming high and low system performance. Material inputs for sugar and ethanol production are from ISMA (2011c) and Kumar (2013). For South-Central Brazil we use industry-based data from Seabra et al. (2011). Compared to earlier studies (e.g. Macedo et al. (2008)), these are the latest inventory data on Brazilian ethanol production. The inputs of ethanol production of the two product-systems are presented in Table 2.5. Background data used in this chapter originate from ecoinvent (2010) v2.2. For Indian average grid electricity production we use data from the International Energy Agency (IEA; IEA (2011), OCED/IEA, (2011); see section 2.7). Multi-functionality allocation factors, co-products and credits of the different approaches are presented in Table 2.6.

Table 2.2 Inputs for 1 tonne of sugarcane production in Uttar Pradesh, India (2009) and South-Central Brazil (2008)

Input	Unit	Uttar Pradesh, India	South-Central Brazil
Land occupation[a]	m^2a	169	147
Freshwater irrigation	m^3	59.5[b]	0
N-fertilisers	kgN	2.69	0.78
Ammonium sulphate		0.42	-
Ammonia		-	0.11
Ammonium nitrate		0.42	0.29
Diammonium phosphate (DAP)		0.37	-
Urea		1.23	0.37
Potassium nitrate		0.25	-
Monoammonium phosphate		-	0.08
P_2O_5-fertilisers	kgP_2O_5	1.31	0.25[c]
Diammonium phosphate (DAP)		0.62	-
Single superphosphate (SSP)		0.4	0.146
Triple superphosphate (TSP)		0.2	0.081
Phosphate rock		0.07	0.003
Monoammonium phosphate		-	0.022
K_2O-fertilisers	kgK_2O	0.82	0.98[d]
Potassium chloride		0.80	0.96
Potassium nitrate		$8.0 \cdot 10^{-3}$	0.01
Potassium sulphate		$8.0 \cdot 10^{-3}$	0.01
Pesticides	kg		
Herbicides, unspecified[e]		0.056	0.031
Triazine compounds[e]		0.011	0.006
Phenoxy compoundse		0.003	0.001
Glyphosate[e]		0.004	0.002
Diuron[e]		0.009	0.005
Insecticides, unspecified		0.050	0.003
Fungicides, unspecified		0.003	$1.0 \cdot 10^{-5}$
Other inputs	kg		
Sugarcane, as seed[f]		100	-
Lime		-	5.18[g]
Ash[h]		2	2
Gypsum		-	2.30

Table 2.2 (continued)

Input	Unit	Uttar Pradesh, India	South-Central Brazil
Other inputs	kg		
Stillage		-	570
Filtercake		40[i]	31
Energy			
Diesel, transport of inputs	l	$4.1\cdot10^{-3}$	$5.6\cdot10^{-3}$
Diesel, field operations	l	-	3.62[j]
Diesel, irrigation[k]	l	0.54	-
Electricity, irrigation[k]	kWh	12	-

[a]India: Based on 59.2 t_{cane}/ha ISMA (2011a). Brazil: Based on 86.7 $t_{cane}/ha_{harvested\ area}$ which represents 83% of total area. The remaining area was not harvested due to reforming cycle or bad weather conditions. [b]Based on 14.6 kg_{cane}/m^3_{water} (Srivastava et al., 2009) for 92.3% of area irrigated (MAC, 2009). [c]Assuming 91% of total P_2O_5 fertilisers (Seabra et al., 2011) are DAP, SSP, TSP as in Jungbluth et al. (2007). [d]Assuming that total K_2O fertilisers are applied in the same ratio of potassium chloride, sulphate and nitrate as in Jungbluth et al. (2007). [e]Brazil: types of pesticides calculated based on shares of specified and unspecified pesticides as in Jungbluth et al. (2007) based on total pesticides (herbicides, acaricides and other defensives) reported in Seabra et al. (2011). India: Same approach was applied. For Brazil, the active ingredients differ from other studies (e.g. Cavalett et al. (2013)). However, given the uncertainties in pesticide use in both Brazil and India, the ecoinvent inventory is preferred, to provide a more a conservative estimate. [f]India: includes losses or other non-productive uses (ISMA, 2011a). Brazil: The inventory inputs of Seabra et al. (2011) account for seed requirement of sugarcane. Macedo et al. (2008) report seed efficiency of 6.9 ha_{cane}/ha_{seed}. [g]Energy use in production of lime based on UNICA (2009). [h]In this study no impacts are associated to ash. Same quantity assumed for India and Brazil based on Seabra et al. (2011). [i]Based on 4.5% of sugarcane crushed in mills (ISMA, 2012). [j]Includes land preparation (reforming, tillage, ploughing), seeding, agro-chemicals application, harvesting, ferti-irrigation. In Seabra et al. (2011) diesel consumption is given for the total area including transport of sugarcane to the mills and agrochemicals to the fields (274 L/ha). This value is adjusted per t_{cane} taking into account the productivity and total area and by subtracting the diesel requirement for transport of sugarcane to mills and agrochemicals to fields for an average truck efficiency of 55 tkm/L and distance of 42 km (round trip).[k]India: Groundwater pumping for UP (MoEF, 2010). Average water depth is 36.7 m (Srivastava et al., 2009) and 20% diesel and 80% electric pumps (Shah, 2009). Diesel requirement for groundwater pumping based on Kägi and Nemecek (2007), which is 0.059 L diesel/L water·depth (in m) and diesel density of 0.832 kg/L. Brazil: sugarcane production is rainwater-dependent. Energy requirement for ferti-irrigation is accounted under diesel, field operations.

Table 2.3 Sugarcane processing products and sector analysis in Uttar Pradesh, India for the 2009–2010 season. Calculated based on ISMA (2011a, 2011b, 2011c)

	Unit	Quantity
Processing products		
Sugarcane processed	Mt	56.7
Sugar	Mt	5.2
Molasses	Mt	2.9
Ethanol	Mt	0.51
Surplus electricity[a]	TWh	3.11
Bagasse (total)	Mt	18.2
Biogas (total)[b]	TJ	2,785
Sector analysis		
Crushing capacity[c]	kt/day	565
Crushing capacity of facilities which provide surplus electricity[c]	kt/day	263
Ethanol capacity[d]	Mt/yr	0.9

[a]84.5% of total surplus was produced during the sugarcane crushing season (103 days in 2009–2010) and the remaining was produced outside the sugarcane crushing season (ISMA, 2011b). On-season energy output calculated based on on-season surplus capacity, multiplied by the number of days of the on-season period and 24 hours in a day. Similarly for off-season. [b]Calculated based on 0.35 nm^3/kg_{COD} removed, with 72% COD removal efficiency (typical for mesophilic treatment technologies), 100,000 $mg_{COD}/L_{stillage}$ and biogas energy content of 16.6 MJ/kg (Tewari et al., 2007). 90% of distilleries apply anaerobic treatment (Satyawali and Balakrishnan, 2008) and 12.5 $L_{stillage}/kg_{etOH}$ is generated. [c]Based on DFPD (2013) and ISMA (2011a). 77.5% of the sugarcane crushing capacity is installed in stand-alone mills and the remaining 22.5% is installed in mills attached to distilleries (see section 2.7). [d]50% of the total ethanol capacity is in stand-alone distilleries and 50% is attached to sugar mills (NFCSF, 2012). Based on ISMA (2011a) the capacity of the distilleries that provide surplus to the grid represents 26.5% of the total. Since Indian stand-alone distilleries do not process sugarcane, only attached distilleries are associated with production of surplus electricity.

Table 2.4 Inputs and (co-) products for 1 tonne sugarcane processing in Uttar Pradesh, India

	Unit	Quantity
Products	kg	
Sugar		91.4
Molasses		50.3
Bagasse[a]		6
Electricity[b]	kWh	54.2
Inputs	kg	
Sugarcane		1,000
Sulphur dioxide		1.5
Limestone		1.9
Sodium hydroxide		0.5
Superphosphate		0.1
Soda		0.03
Organic chemicals		0.01
Lubricating oil		0.6
Phosphoric acid		0.01
Water		30
Transport, sugarcane	tkm	12
Transport, inputs	tkm	0.6

[a]By subtracting surplus bagasse from total available amount we estimate the quantity of bagasse used within the system boundaries to provide process energy requirements and excess electricity. Surplus bagasse is estimated based on total availability in sugar mills that do not produce excess electricity but supply bagasse to stand-alone distilleries to supplement their primary energy requirements. Process energy requirements are assumed to be met exclusively from bagasse and are 313 kWh/t_{sugar} and 16.9 GJ/t_{sugar} (Jungbluth et al., 2007) and 237 kWh/t_{etOH} and 11.5 GJ/t_{etOH} (Prakash et al., 2005). Bagasse requirement for steam and electricity generation based on Prakash et al. (2005). For distilleries, net primary energy requirement based on Prakash et al. (2005). Heating value of bagasse is 16 MJ/kg_{db}. [b]Since power is produced both on- and off- season, we assume that all the available bagasse is consumed in the mills that produce surplus power and that no additional bagasse or other biomass source is supplied from other mills. Surplus electricity allocated between the two product-systems on a primary energy basis; i.e. 98% of the total surplus was produced by bagasse and 2% by biogas. By dividing total surplus electricity by total sugarcane processed in UP sugar mills we calculate that 54.8 kWh/t_{cane} are produced (Table 2.5, Figure 2.7). Biogas recovery estimated based on Tewari et al. (2007).

Table 2.5 Inputs and co-products for 1,000 kg hydrous ethanol production in Uttar Pradesh, India and South-Central Brazil

	Unit	Uttar Pradesh, India	South-Central Brazil
Products			
Ethanol	kg	1,000	1,000
Bagasse	kg	-	130
Electricity	kWh	60	160
Inputs	kg		
Molasses		5,060	-
Sugarcane		-	14,960[a]
Lubricating oil		-	0.15
Lime		-	13.1
Sulphuric acid		0.41	9.3
Biocides		-	0.1
Organic chemicals		-	0.86
Magnesium sulphate		0.11	-
Urea		1.3	-
Phosphoric acid		0.14	-
Chlorine		0.38	-
Soda		0.06	-
Chromium oxide		0.1	-
Sodium hydroxide		0.6	-
Zinc		0.12	-
Formaldehyde		0.02	-
Water	m^3	11.4[b]	24.7
Transport, sugarcane (Brazil)	tkm	-	8.32[c]
Transport, molasses (India)	tkm	380[d]	197

[a]Based on hydrous ethanol yield of 84.7 L/t_{cane} and ethanol density of 0.789 kg/L. [b]Net water consumption based on Tewari et al. (2007), taking into account gross water requirement and freshwater returned to nature. [c]Based on total diesel consumption presented in Seabra et al. (2011) by subtracting diesel requirement of sugarcane harvesting operations [d]Considering that approximately half of the total ethanol capacity is attached to sugar mills (NFCSF, 2012). Only part of the molasses needs to be transported. The value includes transport of chemical inputs (0.58 $kgkm/kg_{etOH}$). Not including transport of bagasse to stand-alone distilleries.

Table 2.6 Products and multi-functionality allocation factors and credits based on the three different approaches

Products	Sugarcane processing [per t_{cane}]	Price[b]	Multi-functionality allocation factors[c] and credits: Default, SE-C	Default, SE-O	Default, EA	Black box, SE	Black box, EA
Uttar Pradesh, India							
Sugar	91.4 kg	0.546 $/kg	91.5(92,89)[a] %	92(92,92)%	85(79,89)%	86.5%	80.8%
Molasses	50.3 kg	0.087 $/kg	8(8,8)%	8 (8,8)%	7.5(7,8)%	-	-
Bagasse	6 (0,36) kg	0.043 $/kg	0.5(0,3)%	0.38(0,2) MJ/kg_{etOH}	0.4(0,3)%	0.5%	0.4%
Electricity (bagasse)	54.2 (117,0) kWh	0.076 $/kWh	0.44(0.95,0) kWh/kg_{etOH}	0.44(0.95,0) kWh/kg_{etOH}	7.1(14,0)%	54.8kWh/t_{cane}	6.8%
Electricity (biogas)	0.60 (2.5,0) kWh	0.076 $/kWh	0.06(0.25,0) kWh/kg_{etOH}	0.06(0.25,0) kWh/kg_{etOH}	0.6(2.5,0)%		
Ethanol	9.9 kg	0.747 $/kg	100(100,100)%	100(100,100)%	99.4(97.5,100)%	13%	12%
South-central Brazil							
Ethanol	66.8 kg	0.64 $/kg	99.5(99.5,100)%	100(100,100)%	97.5(93.5,100)%		
Electricity	10.7 (32,0) kWh	0.085 $/kWh	0.16(0.48,0) kWh/kg_{etOH}	0.16(0.48,0) kWh/kg_{etOH}	2(6,0)%		
Bagasse	8.6 kg	0.023 $/kg	0.5(0.5,0)%	0.9(0.9,0) MJ/kg_{etOH}	0.5(0.5,0)%		

[a]Values in parentheses indicate parameters for the high and low performance cases (left and right value, respectively). [b]Indian prices from Kumar (2013), ISMA (2011c) and personal communication with ISMA considering exchange rate US$1=INRs45.8 (2010). Brazilian prices from Cavalett et al. (2011), where US$1=R$1.76 (2010). Price for hydrous ethanol obtained by converting the price of anhydrous ethanol based on average difference as estimated by UNICA (2011). [c]Economic allocation factors. Price of sugar, molasses and ethanol in India largely depends on availability of sugarcane, which varies on an annual basis. However, the price ratio of co-products, used to calculate the economic allocation factors, is not expected to vary significantly.

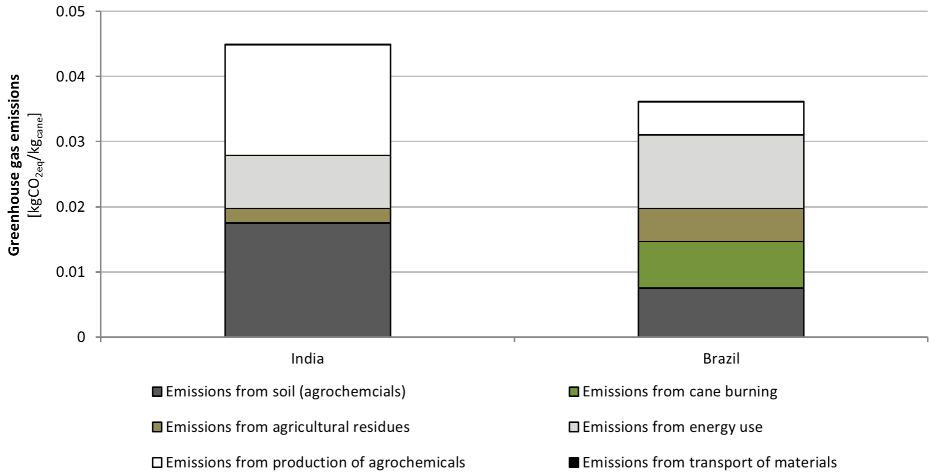

Figure 2.3 Cradle-to-gate emissions of sugarcane production in India and Brazil, excluding animal use and human labour

2.5 RESULTS AND DISCUSSION

2.5.1 Greenhouse gas emissions and non-renewable energy use

GHG emissions of Indian sugarcane production are higher than those of Brazilian sugarcane production (India: 0.045 $kgCO_{2eq}/kg_{cane,}$ Brazil: 0.036 $kgCO_{2eq}/kg_{cane}$ in SE-C; Figure 2.3). The difference is due to high N_2O emissions from oxidation of nitrogen in N-fertilisers and CO_2 release from decomposition of urea[6] ('Emissions from soil'; Figure 2.3) and CO_2 emissions from energy intensive production of N-fertilisers ('Emissions from production of agrochemicals'; Figure 2.3). On the other hand, pre-harvesting burning and energy use related to mechanisation in Brazil reduce the difference of sugarcane emissions between the two countries. Note that the source of grid electricity in electric irrigation pumps in India is aligned with the source of electricity that credits are given for surplus electricity in the sugar product-system. Therefore, emissions of Indian sugarcane production under SE-O and EA are 0.057 $kgCO_{2eq}/kg_{cane}$. The difference with SE-C (0.012 $kgCO_{2eq}/kg_{cane}$) is due emissions from electricity production for irrigation. The emissions of Brazilian sugarcane are the same under all approaches. Note that the assessment of Indian sugarcane excludes the impact of animal use and labour, while the assessment of Brazilian sugarcane includes the impact of machinery use.

For ethanol, we present results for GHG emissions and NREU (Figure 2.4). For each allocation approach the gross results for average system performance are broken down to contribution of sugarcane production, energy use for irrigation in India and agricultural

6 1.325% of nitrogen in N-fertilisers and 1.225% of nitrogen in unburned trash is converted to N in N_2O (Macedo et al., 2008).

operations in Brazil (subsumed as 'energy in agriculture'), molasses production (India), ethanol production and transport. The net impact of ethanol production after deducting the credits is represented by symbols, thereby distinguishing between high, low and average system performance (section 2.3.2).

Net GHG emissions of Indian ethanol are lower when compared to Brazilian ethanol across the results for the system expansion approaches and average system performance (excluding high and low cases). This difference is due to the credits of the Indian system, which are by a factor 8 higher in SE-C (India: 0.27 $kgCO_{2eq}/kg_{etOH}$, Brazil: 0.035 $kgCO_{2eq}/kg_{etOH}$) and by a factor 3 higher in SE-O (India: 0.61 $kgCO_{2eq}/kg_{etOH}$, Brazil 0.18 $kgCO_{2eq}/kg_{etOH}$) when compared to Brazil.[7] Credits are given based on the electricity output per tonne of sugarcane allocated to ethanol production, which as Table 2.6 shows, is higher for Indian ethanol (India: 0.5 kWh/kg_{etOH}, Brazil: 0.16 kWh/kg_{etOH}) and the CO_2 emission intensity of the electricity that is displaced under each approach.[8] In addition, Indian ethanol is associated with a fraction of the impacts from the agricultural phase due to allocation between sugar and molasses[9] but also because impacts of animal use and labour are not included. Khatiwada and Silveira (2011) estimated that human labour contributes 3.5% to GHG emissions of ethanol production in Nepal. Similar contribution in India would increase emissions to 0.31, 0.09 and 0.67 $kgCO_{2eq}/kg_{etOH}$ under SE-C, SE-O and EA, respectively. Contribution to Brazilian ethanol would be lower because of high mechanisation and hence lower human labour in agriculture, which is accounted in results for Brazilian ethanol (Figure 2.4). Assuming that surplus bagasse in Brazil is used to produce additional power then GHG emissions increase slightly to 0.5 $kgCO_{2eq}/kg_{etOH}$ (SE-O). This is due to lower conversion efficiency of biomass to electricity but also due to lower emission factor of natural gas when compared to oil. If surplus bagasse were used for pellet production, displacing coal in European co-firing power plants, the emissions would remain unchanged (0.45 $kgCO_{2eq}/kg_{etOH}$

7 Distilleries in Uttar Pradesh also produce fuel grade (anhydrous) ethanol. Due to aggregated reporting of hydrous and anhydrous ethanol by available statistics the approach of this study might lead to an underestimation of co-products associated with hydrous ethanol. To assess the influence of our assumption (i.e. all production reported for Uttar Pradesh is hydrous ethanol), we correct the avoided energy requirement related to the conversion of hydrous to anhydrous ethanol based on the values in Prakash et al. (2005). The underestimation would be in the range of 1% for surplus electricity and 3% for surplus bagasse, which would only slightly affect the results of SE-O, i.e. GHG emissions and NREU would be lower by 0.007 $kgCO_{2eq}/kg_{etOH}$ and 0.05 MJ/kg_{etOH}, respectively.

8 GHG emissions from Brazilian dams are controversial. Fearnside and Pueyo (2012) estimate higher emissions than those published by the national Brazilian electricity authority. The latter are also used in the ecoinvent inventories (Dos Santos et al., 2006) and have been used in this study. Upward correction of these values in our analysis would entail that the CO_2 intensity of the national average Brazilian electricity mix was higher. By analogy, higher credits would be assigned to surplus electricity provided by the sugar mills, therefore reducing the relative difference between Indian and Brazilian ethanol.

9 Due to allocation (Table 2.6), Indian ethanol is associated with the impacts of 8 kg_{cane}/kg_{etOH} (based on), while Brazilian ethanol is associated with the impacts of 15 kg_{cane}/kg_{etOH}. The difference in GHG emissions associated with the agricultural phase between the two product-systems is 45% in SE-C and 10% in SE-O between Indian and Brazilian ethanol.

instead of 0.46 $kgCO_{2eq}/kg_{etOH}$ in SE-O). High credits for coal displacement are reduced due to high energy demand for drying wet bagasse and lower surplus electricity supplied to the grid due to electricity requirement of milling, pressing and handling. When EA is applied results for GHG emissions are comparable, since both systems do not receive any credits. The range between the high and low performance cases is wider in India than in Brazil because in the low performance case we assume that Indian mills use grid electricity but not the Brazilian mills. Similarly, in the high performance case it is assumed that Brazilian mills produce lower electricity surplus than Indian mills per kilogramme of ethanol. The comparable results of the EA and the range between the high and low performance particularly for Indian ethanol indicate the importance of surplus power output and the effect that different methodological choices have. The 'black box' approach yields higher impacts in SE-C and EA as shown in Figure 2.4. For SE-C this is explained because the system is not subdivided and the allocated credits of ethanol do not outweigh the allocated burdens. Following SE-O, the overall credits allocated to ethanol are higher in the 'black box' approach, which is illustrated by the slightly lower impacts when compared to the reference approach. The effect of the 'black box' approach is significant under EA because ethanol has higher price compared to the other co-products thus is assigned with higher burden share. Additional results are provided in section 2.7.

Based on the emission factor of Cavalett et al. (2013), direct land use change reduces the emissions of Brazilian ethanol by approximately 2% (-0.01 $kgCO_{2eq}/kg_{etOH}$). Accounting for indirect land use change based on emission factors of the California Air Resource Board (CARB, 2010) increases the emissions of Brazilian ethanol by 1.2 $kgCO_{2eq}/kg_{etOH}$. Other studies propose much lower emission factors (see e.g. review by Wicke et al. (2012)). For example, assuming the average land use change emission factor of Tipper et al. (2009) we estimate that emissions of Brazilian ethanol increase by 0.08 $kgCO_{2eq}/kg_{etOH}$. Given the wide range and the absence of methodological consensus in accounting for additional emissions due to indirect land use change these results should be interpreted with caution.

For India, to our knowledge land use change emission factors for molasses-based ethanol do not exist. Due to the increasing ethanol demand (Table 2.1) and the high EBP targets in India, it can be anticipated that molasses used as an animal feed ingredient to be directed to ethanol production. This will increase the demand for feed crops, especially grain. Impacts of molasses diverted to ethanol production chain will be equivalent to impacts of feed crops in India or elsewhere. As Nguyen and Hermansen (2012) show, accounting for impacts of molasses diverted from feed with system expansion shows higher emissions when compared to economic allocation. Similarly, if ethanol is diverted from the potable liquor or chemical sector to transport, additional ethanol would have to be imported or produced domestically. Ethanol production in India is constrained by sugar demand, therefore, displacement of crops is likely to occur in ethanol exporting countries like US, South Africa, Thailand, Brazil (Gopinathan and Sudhakaran, 2009; USDA, 2013). Marginal increase in ethanol demand for fuel in India will be associated with emissions of ethanol production in those countries. In view of the ambitious EBP targets such impacts should be taken into account by a consequential approach to assess GHG emissions associated with the increase in demand.

2.5.2 Human health and ecosystem quality

Impacts of sugarcane ethanol production on HH and EQ are presented in Figure 2.5. A range is included based on critical assumptions on pesticides, which are explained further on. The range is discussed only for SE-C since results are similar across all approaches, with the exception of results based on the 'black box' approach, which are 60% higher. Indian sugarcane has higher impacts on HH than Brazilian sugarcane due to high use of pesticides and fertilisers (Figure 2.6). Indian ethanol has comparable impacts with Brazilian ethanol, with only 10% lower HH and 25% lower EQ. This is partly associated with the allocation in the subsystem of sugar production (footnote 9). Contrary to GHG emissions and NREU, the credits do not influence the results.

Impacts on HH are to a large extent related with pesticide application on soil and more specifically with the arsenic-containing daconate. This input is included in the inventory because a large fraction of pesticides is unspecified (Table 2.9). While detailed data on consumption of chemicals in sugarcane production were not available, its usage is plausible since it is not banned according to the list of Persistent Organic Pollutants of Stockholm Convention (UNEP, 2013). In Brazil, daconate is listed under the agrochemicals produced and applied in sugarcane production (MAPA, 2012). According to Figure 2.6, in India the highest contribution to HH is due to carcinogenic and non-carcinogenic effects, which are related to daconate (99%). The remaining impact is associated with respiratory inorganics from NH_3 and N_2O emissions related to fertilisers. By excluding daconate from the inventory then impacts on HH remain lowest for India ('Best' in Figure 2.5). If, on the other hand, all unspecified pesticides are assumed as daconate there is a threefold increase on HH impacts ('Worst' in Figure 2.5). For Brazil, carcinogenic and non-carcinogenic emissions are also associated with daconate and they contribute 80% to HH. The remaining contribution originates from particulate matter emissions from pre-harvesting burning. Phasing out daconate reduces the impact on HH by 80-90%. Eliminating pre-harvesting practices also reduces greenhouse gas emissions by 10%. However, this estimation does not account the trade-off with increased mechanisation. Last, for Brazilian ethanol, approximately 5% of the impact on HH is associated with bagasse use in co-generation facilities. This effect is lower for Indian ethanol due to the effect of allocation. Note that other studies on Brazilian sugarcane (Cavalett et al., 2013), report lower quantities for pesticide application by a factor 5 (10 g/t_{cane}). This factor difference, in combination with different active ingredients assumed in this chapter (mainly arsenic and atrazine; see section 2.7) lead to significantly higher impacts for ethanol production due to pesticides. Even when the 'Best' scenario (no daconate application) is assessed, the impacts of pesticide application in sugarcane production are a factor 30 higher (on HH) and 15% higher (on EQ) compared to Cavalett et al. (2013).

The method applied to estimate the fate of pesticides in different environmental compartments influences greatly the impact assessment results related with human and environmental toxicity. Our approach, in line with ecoinvent, assumes that the agricultural field is part of the ecosphere, thus the full dose of pesticides is emitted to soil. For Australia, Renouf et al. (2010) account for a fraction of pesticides that runs off to other environmental compartments (1.5% of the active ingredients). Similar run-off factors for Brazilian or In-

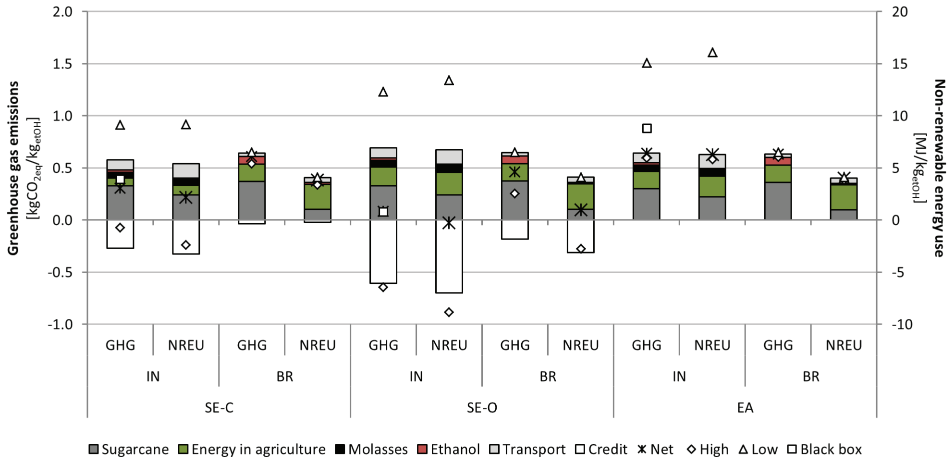

Figure 2.4 Cradle-to-gate greenhouse gas emissions and non-renewable energy use of ethanol production in India and Brazil

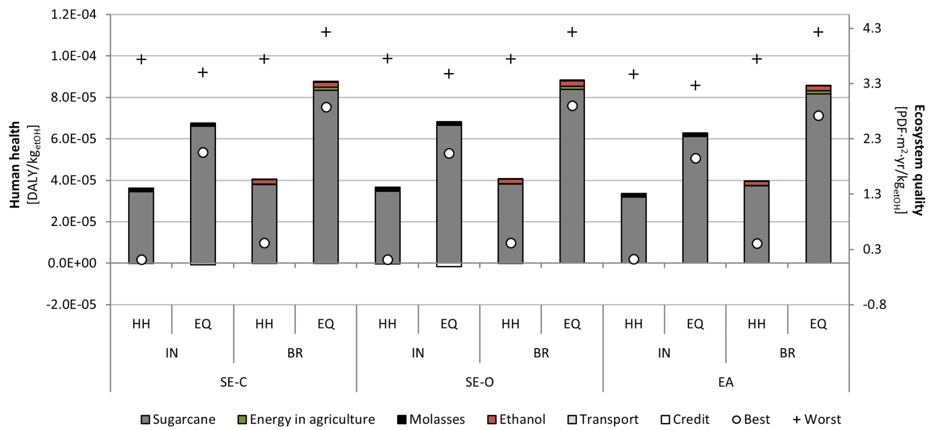

Figure 2.5 Potential impacts of sugarcane ethanol production in India and Brazil on human health and ecosystem quality

dian conditions are not available. If similar run-off factors were applied, the results on HH and EQ would be influenced, primarily due to the low run-off percentage. Differences in the contribution analysis are expected (Figure 2.6). Other methods assume the agricultural field as part of the technosphere (e.g. PestLCI) and argue that only a fraction of the applied dose is emitted to the environmental compartments (Dijkman et al., 2012). Such methods could lead to lower impacts on ecotoxicity and human toxicity by two orders of magnitude (Ometto et al., 2009).

In sugarcane production, impact categories that contribute significantly to ecosystem damage are land occupation, terrestrial ecotoxicity, acidification and nutrification (Figure 2.6). Per kilogramme of sugarcane, land occupation in Brazil is lower by 15% compared to India whereas per kilogramme of ethanol, land occupation contributes more in Brazil due to the direct use of sugarcane juice for ethanol production (Brazil: 2.2 m^2org.arable/kg_{etOH}, India: 1.6 m^2org.arable/kg_{etOH}). Due to data availability, for India we accounted only for harvested land for productive use and seed while for Brazil we accounted also for non-harvested land, which is typically 17% of the total area.[10] The impacts of terrestrial ecotoxicity are associated with daconate. Figure 2.5 ('Best') shows that if daconate is eliminated then the impact on EQ is reduced by 20% in India and by 14% in Brazil. Remaining impacts on terrestrial ecotoxicity are associated with heavy metals (e.g. copper, zinc). Impacts on terrestrial acidification and nutrification are associated with NO_x emissions from bagasse use in co-generation facilities and pre-harvesting burning practices.

On EQ, due to the high contribution of the impact categories mentioned above, freshwater eutrophication does not appear to be significant. Nevertheless, this impact category is particularly important for freshwater quality. We calculate emissions of approximately 0.5 gPO_4 and 0.2 gPO_4 for Indian and Brazilian ethanol, respectively. If instead we assume a 10% P surface run-off factor, which is considered characteristic for Brazilian soils (Ometto et al., 2009), the impact of eutrophication increases by approximately a factor 3 in both countries (India: 1.5 $gPO_4/kg_{etOH,}$ Brazil: 0.6 gPO_4/kg_{etOH}). Nevertheless this hardly increases the impact on EQ (only by 1% for Indian ethanol). On the other hand, stillage treatment is significant for the Indian production system (MoEF, 2009b; Satyawali and Balakrishnan, 2008). This study assumes that stillage is treated anaerobically followed by a secondary treatment (Tewari et al., 2007). However, if distilleries do not apply treatment methods and dispose the effluents directly on soils or water streams, then eutrophication increases by approximately 2 orders of magnitude, primarily due to high phosphorus (soil disposal) and phosphorus and COD content (water disposal). The impact on EQ increases by 25% and 35% for disposal on soils and water, respectively. If anaerobic conditions prevail methane releases increase the GHG emissions of Indian ethanol. Based on 5.52 $kgCO_{2eq}/kgBOD_{stillage}$ (Nguyen et al., 2010) and 36,500 $mgBOD/L_{stillage}$ (Satyawali and Balakrishnan, 2008) the net GHG emissions of Indian ethanol range from 2.6 to 3.1 $kgCO_{2eq}/kg_{etOH}$ depending on the approach (compared to 0.09 to 0.64 $kgCO_{2eq}/kg_{etOH}$; Figure 2.4). CH_4 emissions from stillage disposal account for 80 to 95% of the total emissions.

[10] In 2008, non-harvested land was as high as 28% primarily due to bad weather conditions. In that year, compared to Indian ethanol, Brazilian ethanol would show higher EQ by 45%, instead of 30% as shown in Figure 2.6.

2.5.3 Net water consumption and contribution to human health and ecosystem quality

Inventory results of net water consumption in ethanol production (Table 2.7) are calculated on the basis of freshwater extracted for irrigation and process water consumed for ethanol production (excluding the release back to the environment). Water consumption in Indian ethanol production is significantly higher than in Brazil. This is primarily due to groundwater extraction for irrigation, which in India is as high as 68 L/kg_{cane} (Srivastava et al., 2009), while some studies report higher consumption (100 L/kg_{cane}; IISR (2011)). In South-Central Brazil, sugarcane production is based on rainwater (UNICA 2007). Irrigation also has an effect on GHG emissions of Indian ethanol due to energy requirements for groundwater pumping. The lower estimate indicates net reduction of 0.07 (-8%), 0.06 (-73%) and 0.06 (-9%) $kgCO_{2eq}/kg_{etOH}$ for SE-C, SE-O and EA, respectively. The net increase for the highest estimate is 0.02 (8%), 0.2 (240%) and 0.19 (29%) $kgCO_{2eq}/kg_{etOH}$ for SE-C, SE-O and EA, respectively.

Table 2.7 Net water consumption of ethanol production in Uttar Pradesh, India and South-Central Brazil, in L/kg_{etOH}

	Uttar Pradesh, India	South-Central Brazil	Source
Reference case	543	28	Srivastava et al. (2009), UNICA (2007)
Lower estimate	361[a]	19.5[b]	IISR (2011), UNICA (2007)
Higher estimate	1,150[c]	-	Mekonnen and Hoekstra (2010)

[a]Water use efficiency in India seldom exceeds 35-45%. Demonstrated water saving techniques (skip furrow irrigation, critical growth stage irrigation, trash mulching and ring-pit planting) can enhance water use efficiency by 1.5 to 2.5 times. This value corresponds to the ring-pit planting method, taking into account increase in yield (IISR, 2011). [b]Calculated from 1.23 m^3/t_{cane}. Excluding mills with highest specific water consumption (8% of the sample).

Large difference between Brazil and India is illustrated when considering impacts of water use on HH and EQ. Using the characterisation factors presented in Pfister et al. (2009) we estimate an increase of Indian ethanol on HH by 4% and on EQ by 11%, compared to results in Figure 2.6 (see section 2.7). The lower estimate for irrigation water increases the impacts on HH and EQ by 3% and 7% while the higher estimate increases the impact by 9% and 23%, respectively. For South-Central Brazil, no increase on HH and EQ is estimated since it is not a water-stressed region.

Groundwater use for irrigation in India poses a serious constraint especially when considering expansion for meeting sugar and ethanol demand. It is imperative to promote practices which improve irrigation water use and efficiency. However, even if improvements take place, expansion of a water-intensive crop such as sugarcane only partly alleviates pressure on groundwater. As Table 2.7 shows, lower estimates on water use based on efficient water practices in UP are a factor 20 higher than Brazil. A transition to more water-efficient, drought-resistant crops could be a viable strategy to increase ethanol production without compromising scarce water resources (e.g. sweet sorghum, perennial grasses produced on marginal or degraded lands, other feedstocks for 2[nd] generation biofuels).

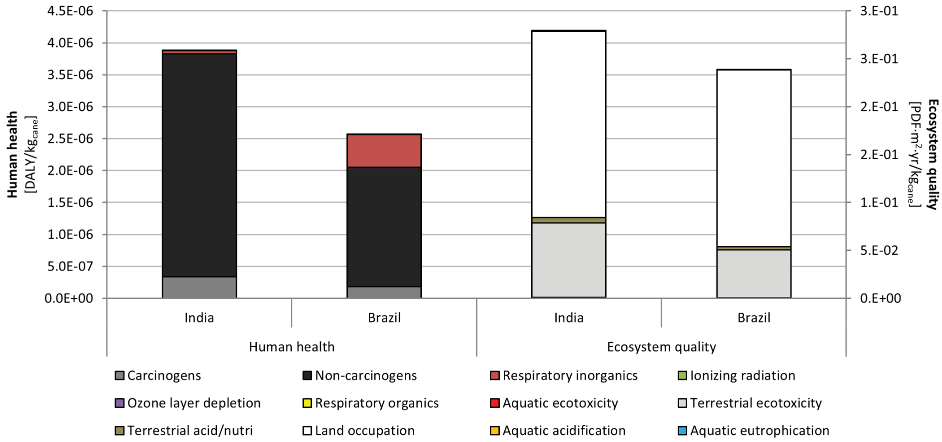

Figure 2.6 Potential impacts of sugarcane production in India and Brazil on human health and ecosystem quality

2.6 CONCLUSIONS AND RECOMMENDATIONS

The environmental performance of the Indian product-system relies on the sector's capacity to provide surplus electricity to the grid. The electricity demand covered by the sugarcane-processing sector reduces the demand for electricity generation by the power sector and the ethanol system is credited depending on whether it displaces localy or nationally produced electricity. However, since not all Indian mills and distilleries produce surplus (46% of the Indian sugarcane processing capacity is associated with surplus electricity) the environmental profiles of ethanol in individual facilities differs significantly. This demonstrates the importance for the sector to modernise and increase its co-generation capacity in order to cover its own electricity requirements by utilising renewable resources (bagasse) and to export electricity to the grid. Mills and distilleries, which rely on grid electricity have significantly higher emissions and non-renewable energy use when compared to the sector average (Figure 2.4). Unless individual distilleries treat stillage in a manner that does not induce anaerobic conditions (e.g. storage ponds) the GHG emission performance is heavily affected. From an environmental perspective it is preferable to capture CH_4 and produce biogas for the system to benefit from reduced primary energy from other sources. When the system does not receive credits (i.e. applying EA) the environmental profile of Indian ethanol is similar to Brazilian ethanol. Although GHG emissions and NREU of Indian sugarcane production are higher than in Brazil, the impact of ethanol is comparable due to the characteristic of the Indian sector, which produces ethanol exclusively by molasses. Therefore only part of the environmental impacts associated with sugarcane production and processing is allocated to ethanol. Our study confirms the findings of Hoefnagels et al. (2010), that allocation is key in determining results, and extends it to ethanol production from sugarcane in different world regions. We show that different choices in system expansion (SE-C, SE-O) also impact the results. The influence is greater for Indian compared to

Brazilian ethanol. Economic allocation yields higher GHG emissions compared to system expansion approaches. This conclusion is in line with results of Renouf et al. (2011). However, this finding contradicts results of Nguyen and Hermansen (2012), who estimate that system expansion leads to higher emissions than economic allocation. The main difference is that in the attributional study of Renouf et al. (2011), ethanol production is assigned only impacts of conversion of molasses to ethanol, while in the consequential approach of Nguyen and Hermansen (2012), ethanol additionally carries the impact of displaced feed production. In our study molasses do not displace feed because there is a long tradition of using molasses for ethanol in India; in applying allocation, molasses received impacts from sugarcane production but also part of the credits for surplus electricity based on economic allocation. For attributional modelling, we find that economic allocation provides most consistent results, since it is uniformly applied across the system co-products. Nevertheless, we recommend presenting results for all allocation approaches. For consequential modelling, increased ethanol demand in India stimulated by the EBP holds the risk of displacing molasses use for feed and diverting ethanol from the potable liquor and chemical sectors. In this event impacts of Indian ethanol will be associated with impacts of crop production for feed, ethanol production in exporting countries or domestic Indian ethanol production from other feedstocks. A consequential study should be performed to account for marginal increase in ethanol demand for transport in order to assess the environmental performance of different marginal suppliers, including domestic 1st and 2nd generation ethanol production.

Production of N-fertilisers and oxidation of nitrogen increase GHG emissions and the high application of P-fertilisers and stillage disposal to soil or water bodies increase freshwater eutrophication. It is recommended to focus efforts on reducing fertiliser inputs of Indian sugarcane cultivation (e.g. to levels similar to Brazil), while maintaining or increasing sugarcane productivity. With regard to HH and EQ it is important to monitor the amount and types of chemicals used. This is also relevant for Brazilian sugarcane production where chemicals such as daconate –if applied– and burning practices lead to high impacts. This can be supported by establishing chains of custody, which focus on agrochemicals. Due to data and methodological uncertainties absolute results using other types of pesticides differ. Given that impacts of Indian sugarcane production are allocated to estimate the environmental profiles of Indian ethanol, the relative difference with Brazilian ethanol is small across all impact categories. Furthermore, as increase in biomass production may lead to a wide array of land-use related impacts such as habitat degradation and loss of ecosystem services, efforts should focus on developing impact assessment methods that quantify and characterise impacts at a higher disaggregation level than the EQ indicator used in this study.

This study did not address efficiency improvements in the agricultural phase. Literature showed that increase of yields in Brazilian sugarcane production was key behind cost reduction that Brazilian ethanol production met over the last three decades (van den Wall Bake et al., 2009). Therefore, efficiency improvements on agricultural production and their effects on the environmental performance of ethanol are likely to demonstrate new improvement potentials for both the Indian and Brazilian system.

Groundwater irrigation was shown to determine water use, GHG emissions (associated with energy for pumping) and EQ of Indian ethanol. In India groundwater use poses a serious resource constraint and it is important to decouple expansion of ethanol production from water-intensive crops such as sugarcane. Ethanol production from water-efficient crops such as sweet sorghum or agricultural residues used for 2^{nd} generation ethanol is a step required to alleviate the pressure from the depleting groundwater resources. For current production of sugarcane water efficiency measures are needed.

Although data for Brazilian ethanol call for higher quality in specific parameters (e.g. types of pesticides) they are characterised by completeness and robustness when compared to Indian ethanol production data. It is recommended to improve the quality and coverage of the latter to levels similar to Brazilian ethanol. For sugarcane production, statistics for resource input (fertilisers, pesticides) should become available at high spatially explicit levels to assess regional variability. Data for animal and human labour are also important for a complete assessment. Improved datasets need to include the sector's bagasse flows. Reporting should include bagasse consumption and surplus per mill and attached distillery, inter-sectoral flows, intra-sectoral flows (e.g. paper industry) and losses. In addition, similar to reporting of fossil-fuel use, material flows should also be monitored if other biomass sources are used. Until bottom-up data become available, we recommend subdividing the sector to account for energy recovery from distilleries in order to assign appropriate credits to ethanol. Due to regional variability of important parameters such as sugarcane productivity, mechanisation and irrigation it is recommended that future studies assess the environmental performance of Indian ethanol at the national level, including effects of direct and indirect land use change.

Overall it can be concluded that the Indian government's plan on introducing biofuels to the market does not cause higher environmental impacts than those of other sugarcane ethanol production chains. However, this conclusion is limited to the production in UP. Also, the low yields, their dependency on groundwater irrigation, the constrained ethanol production based on sugarcane molasses and potential displacement effects, the high input of agrochemicals and the current electricity of sugar mills indicate unexploited opportunities for global players, governments and other stakeholders to support implementation of better practices and improve the GHG emission, NREU, HH and EQ performance of Indian ethanol production.

2.7 APPENDIX

2.7.1 Average Indian power generation

At the time of this study, inventories for average Indian power generation were not available. In order to calculate the CO_2 emissions, energy use and other impacts of average Indian power production delivered at the consumer we used data from the International Energy Agency (IEA) (IEA, 2011, 2010). For electricity generation IEA provides the primary energy use per fuel (e.g. coal and peat, oil products, gas), gross power generated per fuel (i.e. including use by power plants and delivered to consumers), own use of electricity by power plants as well as transmission and distribution losses. By subtracting own electricity use and transmission and distribution losses from the gross power output we estimated the net power delivered at the consumer. Dividing the primary energy consumption per fuel by the net power output per fuel we estimated the net electricity efficiency per fuel type. We then applied the calculated efficiencies in related processes of electricity generation by fuel type in Simapro v7.3 and we generated the final inventory by considering the energy mix of electricity output as provided by IEA (2011). With this approach we estimate that the emissions of average Indian electricity delivered at the consumer is 1.5 $kgCO_{2eq}/kWh$. If we use emission factors of IPCC (2006) we calculate 1.4 $kgCO_{2eq}/kWh$. In comparison, ecoinvent v3 for the process 'market for electricity, medium voltage' provides an estimate of 1.4 $kgCO_{2eq}/kWh$.

2.7.2 Emissions of sugarcane and ethanol production

Table 2.8 Emissions from bagasse combustion in sugarcane processing

Output	Unit	South-Central Brazil [1 t_{etOH}]	Uttar Pradesh, India [1 t_{cane} processing]
Air emissions[a]			
Carbon dioxide, biogenic	t	3.45	0.28
Dinitrogen monoxide	kg	0.12	0.01
Methane, biogenic	kg	0.92	0.08
Nitrogen oxides	kg	2.21	0.18
Particulates, 10 µm	kg	2.51	0.21
Particulates, 2.5 µm	kg	1.25	0.10
Sulfur dioxide	kg	0.12	0.01
Carbon monoxide, biogenic	kg	2.22	0.18
Volatile organic compounds	kg	0.16	0.01
Other emissions [b]	-	-	-

[a]Calculated based on the emission factors from the GREET model for small industrial boiler for bagasse input of 3.82 t bagasse/t_{etOH} (0.25 $t_{bagasse}/t_{cane}$) in Brazil and 0.31 $t_{bagasse}/t_{cane}$ in Uttar Pradesh, India. These emissions could be an overestimation for larger boilers used by distilleries and sugar mills. [b]Included in the model by scaling the emissions from co-generation in ecoinvent (Jungbluth et al., 2007). Note that emissions for ethanol production in Uttar Pradesh, India also include emissions from biogas combustion based on Stucki et al. (2011). According to Cavalett et al. (2013) the emissions inventory should also include ethanol emissions from the distillation process. Based on their computational model these are $2.02 \cdot 10^{-3}$ kg/kg_{etOH}. Including these emissions to the inventory would not have an effect on the total impact on HH or EQ. It would, however, increase the impact on respiratory organics by 25%.

Table 2.9 Emissions (in kg) of production of 1 t_{cane} South-Central Brazil and Uttar Pradesh, India. Background emissions from production or use of inputs presented are not included in this table

Output	South-Central Brazil	Uttar Pradesh, India
Air emissions		
Ammonia[a]	0.106	0.315
Nitrogen oxides[b]	0.212	0.013
Carbon monoxide[c]	7.53	-
Particulate matter, 10 µm[c]	0.639	-
Particulate matter, 2.5 µm[c]	0.319	-
Volatile organic compounds[c]	0.573	-
Sulfur oxides[c]	0.033	-
Carbon dioxide, fossil[d]	2.74	0.903
Methane, biogenic[c]	0.221	-
Dinitrogen monoxide[c,e]	0.041	0.013
Water emissions		
Phosphorus to river[f]	$1.0 \cdot 10^{-3}$	$3.0 \cdot 10^{-3}$
Phosphorus to groundwater[g]	$0.8 \cdot 10^{-3}$	$1.2 \cdot 10^{-3}$
Nitrate to groundwaterh[h]	0.044	0.067

Output	South-Central Brazil	Uttar Pradesh, India
Soil emissions		
Atrazine[i]	$5.7 \cdot 10^{-3}$	$1.1 \cdot 10^{-2}$
2,4-D[i]	$1.4 \cdot 10^{-3}$	$2.6 \cdot 10^{-3}$
Glyphosate[i]	$2.0 \cdot 10^{-3}$	$3.8 \cdot 10^{-3}$
Linuron[i]	$4.8 \cdot 10^{-3}$	$8.9 \cdot 10^{-3}$
Arsenic[i]	$1.6 \cdot 10^{-3}$	$2.9 \cdot 10^{-3}$
Finopril[j]	$4.8 \cdot 10^{-5}$	$8.0 \cdot 10^{-3}$
Endosulfan[j]	$1.6 \cdot 10^{-3}$	$2.7 \cdot 10^{-2}$
Carbofuran[j]	$5.4 \cdot 10^{-4}$	$9.1 \cdot 10^{-3}$
Terbufos[j]	$6.0 \cdot 10^{-4}$	$10.0 \cdot 10^{-3}$
Acephate[j]	$1.8 \cdot 10^{-4}$	$3.0 \cdot 10^{-3}$
Cadmium[k]	$2.8 \cdot 10^{-5}$	$2.8 \cdot 10^{-5}$
Chromium[k]	$2.1 \cdot 10^{-4}$	$2.1 \cdot 10^{-4}$
Copper[k]	$1.3 \cdot 10^{-4}$	$1.3 \cdot 10^{-4}$
Nickel[k]	$1.1 \cdot 10^{-4}$	$1.1 \cdot 10^{-4}$
Lead[k]	$2.2 \cdot 10^{-4}$	$2.2 \cdot 10^{-4}$
Tin[k]	$2.2 \cdot 10^{-3}$	$2.2 \cdot 10^{-3}$

[a] NH_3-N emissions of N-fertilisers are calculated from the share of N emitted in the form of NH_3 as in Jungbluth et al. (2007) (i.e. 8% ammonium sulphate, 15% urea, 2% ammonium nitrate, 4% MAP, DAP) and from NH_3 losses from stillage based on the model for liquid slurry described in Nemecek and Kägi (2007). The N-content in stillage is considered as NH_4 due to the low pH of stillage (typically lower than 5). [b] $NO_x = 0.21 \cdot N_2O$ as in Nemecek and Kägi (2007) and from trash burning (see next footnote) [c] Related with sugarcane pre-harvesting burning. Calculated based on emission factors from GREET model for burning of dry straw (GREET, 2010). [d] Carbon oxidation to CO_2 from urea and lime (0.2 kgC/kg urea and 0.13 kgC/kg dolomite) based on IPCC (2006). Carbonates from lime are considered as dolomite. N oxidation to N_2O from N-fertilisers and unburned trash left on the field, based on Macedo et al. (2008) and IPCC (2006). Emission factors 1.325% of nitrogen in N-fertilisers and 1.225% of nitrogen in unburned trash is converted to N in N_2O [f] Assuming the same ratio of P emissions to surface waters from P_2O_5 fertilisers as in Jungbluth et al. (2007)(i.e. 0.41% P-emissions from total P_2O_5-fertilisers). In the article sensitivity is presented for 10% factor. [g] Calculated from ecoinvent v2.2 guidelines for phosphorus emissions to groundwater (Nemecek and Kägi, 2007). [h] 2.5% of the N content in N-inputs to soil (Nemecek and Kägi, 2007). [i] Brazil: triazine compounds emitted as atrazine, phenoxy compounds emitted as 2,4-D, glyphosate emitted as glyphosate, diuron emitted as linuron. These soil emissions are accounted from the same elementary flows as in Jungbluth et al. (2007). Arsenic is contained in the herbicide daconate. To estimate this flow we calculate the share of arsenic emissions over the total unspecified pesticides as in Jungbluth et al. (2007) and we apply it to the total unspecified pesticides as reported in Seabra et al. (2011). India: we apply the same share of the elementary flows over the total pesticides applied in Brazil to the quantity of total pesticides applied in India. For arsenic, we deduct the elementary flows that were already attributed to pesticides from the total pesticides and we apply to the remaining quantity the same factor as in Brazil. [j] Calculated according to the active ingredients in Brazil based on Hassuani et al. (2005) adapted for quantities of insecticides for Brazil and India respectively. [k] Assumed to be the same both for Brazil and India as in Jungbluth et al. (2007).

2.7.3 Sector analysis

In Figure 2.7 the sugar mill and distillery sector of Uttar Pradesh, India is broken down based on the numbers of mills that provide surplus electricity and on the distilleries that are attached to mills. We present this analysis because it forms the basis of the top-down approach we took to assess the ethanol sector in Uttar Pradesh in the following instances: (a) to assess the surplus bagasse available, (b) to assess the high system performance case, where only mills and distilleries that provide surplus electricity are taken into account, (c) to assess the impact of the assumption that all ethanol produced is hydrous ethanol (rectified spirit).

2.7.4 Indian molasses-ethanol system modelling

In Uttar Pradesh, India the sugarcane processing sector produces sugar, ethanol, surplus electricity and bagasse as final outputs. The sector analysis shows (Figure 2.7) that based on capacities, approximately two-thirds of the sugar mills are stand-alone (not attached to distilleries) therefore they produce molasses as a co-product, which are then sold to distilleries. Therefore, for a significant part of the sector, molasses are an intermediate co-product, which are the main feedstock for ethanol production; hence they need to be associated with impacts of sugarcane processing. In our systems we try to capture this by subdividing the system in order to account for the impacts of this intermediate co-product, which are then carried forward to ethanol production. To model the sugarcane processing sector we distinguish two mutually exclusive cases (Figure 2.1): (a) sugar mills and attached distilleries that *produce surplus electricity*, (b) stand-alone sugar mills, stand-alone distilleries, and attached distilleries that *do not produce surplus electricity*. This is required because bagasse flows in these different settings are not monitored by the sugar mill association in India.

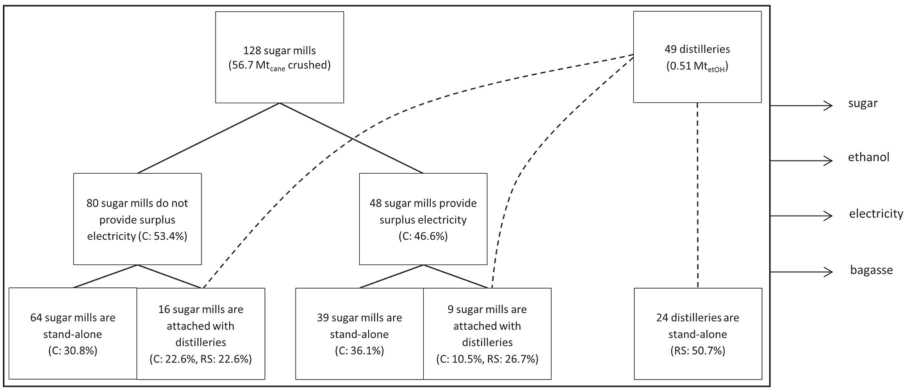

Figure 2.7 Schematic representation of sugar mill and distillery sector in Uttar Pradesh, India. Values in parenthesis represent the sugarcane crushing capacity (C) and rectified spirit (RS, i.e. hydrous ethanol) production capacity. Note that 15 distilleries have the capacity to produce anhydrous ethanol, 7 of which are attached to sugar mills and produce surplus electricity. For the remaining 8, which do not produce surplus electricity, it cannot be concluded if they are stand-alone distilleries or attached to sugar mills.

- For sugar mills and distilleries that *produce surplus electricity*, we assume that all bagasse extracted from sugarcane at the milling phase is consumed to cover energy requirements of sugar and ethanol production and also provide on- and off-surplus power to the grid. Moreover, since surplus electricity is partly produced by attached sugar mills and distilleries they make use of biogas available on site from anaerobic treatment of stillage. Based on primary energy ratio of bagasse and biogas the surplus electricity is partitioned in order to capture that molasses (and hence ethanol) contribute to meeting energy requirements of the mills and attached distilleries. The effect of the latter choice is assessed by the 'black box' approach, which does not assume this partitioning.

  In this system 1 t_{cane} processed yields: S_1 t_{sugar}, M_1 $t_{molasses}$ (intermediate co-product), E_{1a} kWh surplus electricity partitioned to bagasse, Et_1 t_{etOH} and E_{1b} kWh surplus electricity partitioned to biogas.

- For sugar mills and distilleries that *do not produce surplus electricity* we calculate the bagasse requirement to meet total energy requirements of sugar and ethanol production. This also leaves a bagasse surplus which can be supplied for other purposes (e.g. feedstock in pulp production, solid biomass fuel). We assume that stand-alone facilities (mills and distilleries) consume part or the surplus bagasse and their onsite biogas production to meet energy requirements (stand-alone distilleries only). For sugar-mills this amounts to approximately 3.3 $t_{bagasse}/t_{sugar}$. For ethanol production this amounts approximately to 0.4 $t_{bagasse}/t_{etOH}$. We then subtract the total bagasse use of facilities that do not produce surplus electricity from the total bagasse available to estimate the 'net' surplus bagasse produced by the sugarcane-processing sector.

  In this system 1 t_{cane} processed yields: S_2 t_{sugar}, M_2 $t_{molasses}$, B_2 $t_{bagasse}$ and Et_2 t_{etOH}.

The total sugarcane processed (C) in Uttar Pradesh, India gives $(S_1+S_2)\cdot C$ Mt_{sugar}, $(M_1+M_2)\cdot C$ $Mt_{molasses}$, $B\cdot C$ $Mt_{bagasse}$, $(E_{1a} + E_{2a})\cdot C$ TWh electricity and $(Et_1+Et_2)\cdot C$ Mt_{etOH} (Table 2.3). The system of this study represents a 'virtual' case which is composed by two subsystems: the sugarcane processing subsystem which produces sugar, molasses (as an intermediate), 'net' surplus bagasse and electricity partitioned to bagasse (Table 2.5) and the ethanol subsystem which uses molasses as an input and also produces the assigned electricity surplus partitioned to biogas (Table 2.5). The multi-functionality of the subsystems sugarcane processing and ethanol production is then treated with three different approaches (SE-C, SE-O, EA; Table 2.6). Separately, we assess the total sector as a 'black box' taking into account only the final outputs, i.e. without performing intermediate allocation on molasses and partitioning surplus electricity to bagasse and biogas. In the 'black box' approach we treat multi-functionality following SE-C, SE-O, EA.

2.7.5 Alternative uses of surplus bagasse in Brazil

Under the system expansion-optimistic (SE-O) approach, we assumed that surplus bagasse (0.065 $kg_{bagasse,dry}/kg_{etOH}$) are used in industrial boilers, hence displacing fuel oil. This bagasse application is used as reference. We also assess two alternative cases in which (a) surplus bagasse is used to produce additional electricity output and (b) surplus bagasse is used to produce pellets which are transported to Europe and used in co-firing plants, displacing coal.

The credit due to the reference use of surplus bagasse is: -0.08 $kgCO_{2eq}/kg_{etOH}$.

- higher surplus electricity output:
  - efficiency: 1.1 $kWh/kg_{bagasse,dry}$;
  - fuel input: 0.065 $kg_{bagasse,dry}/kg_{etOH}$;
  - additional surplus electricity: 0.0715 kWh/kg_{etOH};
  - credit due to natural gas displacement: 0.647 $kgCO_{2eq}/kWh$ (natural gas-based electricity generation);
  - credit of the ethanol product-system: -0.05 $kgCO_{2eq}/kg_{etOH}$.

When considering the above use of bagasse, the credit that the ethanol system receives is lower by 0.03 $kgCO_{2eq}/kg_{etOH}$, which increases the overall greenhouse gas emission profile of ethanol to 0.5 $kgCO_{2eq}/kg_{etOH}$ under the SE-O approach.

- Bagasse pelletisation and use in co-firing plants in Europe:
  - feedstock requirement for pellet production: We assume that bagasse pellets need to dry from 50% to 6% water content in drum dryers, similar to wood pellets in order to be combusted in co-firing plants (Sikkema et al., 2010). Therefore the feedstock requirement is 1.88 $kg_{bagasse,50\%wet}$ $/kg_{pellet}$, where 0.88 kg_{H2O} need to be evaporated. Including 3% transport losses for transport and handling the total feedstock requirement is 1.94 $kg_{bagasse,50\%wet}$ $/kg_{pellet}$;
  - biomass requirement for drying: Energy requirement for drying is 3.96 MJ/ $kg_{H2O,evap.}$ (Uasuf, 2010). Assuming a 79% bagasse boiler efficiency (Seabra et al., 2011), bagasse requirement for drying is 0.63 $kg_{bagasse}/kg_{H2O,evap.}$, or 0.55 $kg_{bagasse,50\%wet}/kg_{pellet}$;
  - the total feedstock requirement for pellet production and drying is: 2.5 $kg_{bagasse,50\%wet}/kg_{pellet}$;
  - emissions for bagasse combustion (CH_4, N_2O) for bagasse drying are from GREET (2010) and amount to 8.5 gCO_2/kg_{pellet};
  - electricity requirement for milling, pressing and cooling, handling and storage is 0.161 kWh/kg_{pellet} (Sikkema et al., 2010). We assume that the electricity requirement for milling, pressing and cooling, handling and storage is supplied by the mills and the distilleries, thus their net surplus is reduced;
  - emissions for transport are from ecoinvent v2.2 (ecoinvent, 2010), which are 8.8 gCO_2/t-km. The transport distance assumed is 11,000 km (Rio Grande, Brazil – Rotterdam, Netherlands). Land transport by train is excluded. Emissions for bagasse pellet transport to Europe are 97 gCO_{2eq}/kg_{pellet};
  - we assume an efficiency of the pellet-based plant to be 40%. This entails a pellet requirement of 0.5 kg_{pellet}/kWh for an LHV of 16 MJ/kg_{pellet}. The respective emissions (CH_4, N_2O) are 16 gCO_{2eq}/kWh;
  - the life cycle emissions of coal-based electricity generation in the Netherlands are 1.07 $kgCO_{2eq}/kWh$ ecoinvent v2.2 (ecoinvent, 2010).

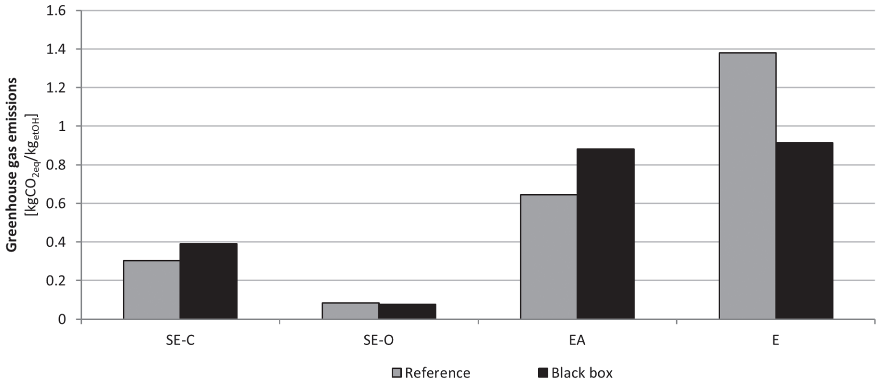

Figure 2.8 Comparison of net cradle-to-gate greenhouse gas emissions of ethanol production in Uttar Pradesh, India based on the reference approach and the 'black box' approach. (E) stands for allocation on an energy basis

Per kilogramme of ethanol surplus bagasse is 0.13 $kg_{bagasse,50\%wet}$, which corresponds to 0.05 kg_{pellet}/kg_{etOH} after deducting feedstock and biomass for energy requirement. The additional emissions are 6.8 gCO_{2eq}/kg_{etOH} and pellets are used to produce 0.1 kWh/kg_{etOH} and the credit due to bagasse-based surplus electricity is lower by 5 gCO_{2eq}/kg_{etOH}. Therefore the net avoided emissions are: -0.095 $kgCO_{2eq}/kg_{etOH}$, which are in a similar range with the net credit in the reference case use (-0.08 $kgCO_{2eq}/kg_{etOH}$).

2.7.6 Energy allocation and sensitivity of economic allocation on prices

As a variant of EA, we apply also allocation on an energy basis between the outputs of the system (Figure 2.8). Energy allocation shows the highest impact across all approaches. The 'black box' approach leads to comparable emissions with EA. However, subdividing the system and allocating the impacts based on the energy content of the products, increases the allocation factor of molasses (highest across all other approaches) and hence the impacts of ethanol are highest. For NREU the effect is similar with GHG emissions, while a factor 2 difference is noticed in HH and EQ.

Finally, we assess the influence of economic allocation factors based on different price ratios between the co-products of the Indian product-system. If the molasses price is decreased by 25% and the price of sugar and electricity are increased by 25% and 30%, respectively, then impacts of Indian ethanol decrease by 20% to 30% compared to results presented in Figure 2.4. This is because the economic allocation factor of molasses decreases, resulting in lower impacts for ethanol.

2.7.7 Impact of water stress on human health and ecosystem quality

The Water Stress Index (WSI) of the Uttar Pradesh region in India is equal to 1 (Pfister et al. 2009). Based on a consumptive water use of 543 kg_{H2O}/kg_{etOH} we calculate that the water deprivation is 543 kg_{H2O}/kg_{etOH}. Using the characterisation factors of Pfister et al. (2009) for Uttar Pradesh (i.e. $2.9 \cdot 10^{-6}$ DALY/m^3 H_2O and 5.02 PDF·m^2·yr/m^3 H_2O) we calculate that the impact of water use on human health and ecosystem quality is $1.6 \cdot 10^{-6}$ DALY/kg_{etOH} and 0.27 PDF·m^2·yr/kg_{etOH}. This corresponds to a 4% and 11% increase of the calculated impact on human health and ecosystem quality, respectively.

Note that the impact factors described in Pfister et al. (2009) were developed based on the Ecoindicator 99 impact assessment method, while the impact assessment results of this study were calculated based on the Impact 2002+ method. If we calculate the impact assessment results based on the Ecoindicator 99 method, then the increase is 17% and 12%, on human health and ecosystem quality, respectively.

3

Life cycle impact assessment of bio-based plastics from sugarcane ethanol

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ABSTRACT

The increasing production of bio-based plastics calls for thorough environmental assessments. Using Life Cycle Assessment, this study compares European supply of fully bio-based high-density polyethylene and partially bio-based polyethylene terephthalate from Brazilian and Indian sugarcane ethanol with production of their petrochemical counterparts in Europe. Bio-based polyethylene results in greenhouse gas emissions of around -0.75 $kgCO_{2eq}/kg_{PE}$, i.e. 140% lower than petrochemical polyethylene; savings on non-renewable energy use are approximately 65%. Greenhouse gas emissions of partially bio-based polyethylene terephthalate are similar to petrochemical production (±10%) and non-renewable energy use is lower by up to 10%, partly due to the low bio-based content of the polymer. Assuming that process energy is provided by combined heat and power reduces the greenhouse gas emissions from partially bio-based polyethylene terephthalate production to a range from -4% (higher) to 15% (lower) compared to petrochemical polyethylene terephthalate depending on the methodological choices made. Production from Brazilian ethanol leads to slightly higher impacts than production from Indian ethanol due to dampening effects of allocation on Indian ethanol produced from sugarcane molasses, different sugarcane pre-harvesting practices and inter-continental transport of Brazilian ethanol to India. Internal technical improvements such as fuel switch, new plants and best available technology offer savings up to 30% in greenhouse gas emissions compared to current production of petrochemical polyethylene terephthalate. The combination of some of these measures and the use of biomass for the supply of process steam can reduce the greenhouse gas emissions even further. In human health and ecosystem quality, the impact of the bio-based polymers is up to 2 orders of magnitude higher, primarily due to pesticide use, pre-harvesting burning practices in Brazil and land occupation. When improvements are assumed across the supply chain, such as pesticide control and elimination of burning practices, the impact of the bio-based polymers can be significantly reduced. Realising such improvements will minimise the greenhouse gas and other emissions and resource use associated with bio-based polyethylene terephthalate and will allow to alleviate further pressure on fragile ecosystems.

3.1 INTRODUCTION

Since 1980, petrochemical plastics production increased by an average compound annual growth rate of about 5%, resulting in a global production volume of 288 Mt in 2012 (PlasticsEurope, 2013). This production accounts for 5% of the global total primary energy supply (BP, 2013; PlasticsEurope, 2013).[11] In Europe, low-density, linear low-density and high-density polyethylene (LDPE, LLDPE and HDPE, respectively) and polyethylene terephthalate (PET) together represent 36% of plastics demand (PlasticsEurope, 2013).

It is known that the use of renewable resources for applications other than fuels, such as chemicals, oleochemicals, papers and textiles, generally offers higher value added (nova-Institut, 2010). Recently, the use of bio-based plastics for packaging has received a lot of attention due to emerging technological options (Shen et al., 2010). Polylactic acid, bio-based polyethylene (bio-PE), and partially bio-based PET (bio-PET) are notable examples. In 2011, bio-PE and bio-PET represented 56% of the global bioplastics' production capacity reaching 650 kt (EuBP, 2012). The capacity is expected to further increase since several producers have commissioned new production plants (JBF, 2012; TTS, 2011). Daioglou et al. (2014) estimate that the global feedstock energy demand for chemicals and refinery products is expected to increase from 30 EJ today to over 100 EJ by 2100. Biomass can supply over 40% of the total primary energy required for non-energy purposes and thus reduce greenhouse gas emissions by 20% in 2100 (Daioglou et al., 2014). Bio-based products and plastics could hence become an important strategy in the transition process towards a sustainable bioeconomy (CSES, 2011; EC, 2011a, 2009a). To ensure that adequate decisions are made, it is essential to assess the potential environmental impacts of the entire process chain taking into consideration local production practices and boundary conditions.

The purpose of this chapter is to assess the environmental impacts of bio-PE and bio-PET from sugarcane ethanol. The selected products represent a large share of current bioplastics production capacity and will continue to do so in the short and medium term (Shen et al., 2010). While numerous studies have been published on biofuel production from various feedstocks (e.g. Börjesson and Tufvesson (2011), von Blottnitz and Curran (2007)), to our knowledge, there is only one peer-reviewed article that assesses the environmental impacts of bio-LDPE (Liptow and Tillman, 2012). However, Brazilian sugarcane ethanol data and data on ethanol conversion to bioethylene need to be updated. Polymer producers also publish environmental profiles of their bio-based products without, however, disclosing vdetailed background information (Hunter et al., 2008). Other studies, in which ethylene is a precursor, do not report environmental impacts of bioethylene, but aggregated results for the final polymer (bio-PVC; Alvarenga et al. (2013)). Chen and Patel (2012) used literature data to prepare a rough estimate of non-renewable energy use and greenhouse gas emissions for bio-PET from sugarcane and maize. However, process data on ethanol dehy-

[11] Based on total global primary energy supply of 522 EJ (87% is fossil-based; BP (2013)). The contribution of petrochemical plastics (288 Mt in 2012) is calculated based on the weighted average specific energy consumption of plastics (76.7 GJ/t), of which approximately 46% is process energy requirements.

dration need to be revisited and, for a comprehensive analysis, it is important to assess additional environmental impacts on ecosystem quality, human health, water-use and land use. In the following, we describe the production of bio-PE and bio-PET from sugarcane ethanol. We then present the methodology used to assess their environmental performance, and compare the results with the production of their petrochemical counterparts in Europe.

3.2 PROCESS DESCRIPTION

Both bio-PE and bio-PET are currently produced from 1st generation ethanol, i.e. ethanol derived from food crops such as sugarcane. Ethanol is subsequently catalytically dehydrated to ethylene and (a) is polymerised to polyethylene or (b) is oxidised to ethylene oxide and then hydrolysed to bio-based mono-ethylene glycol (bio-MEG), the bio-based component of bio-PET. Regardless whether the feedstock is bio-based or petrochemical, further conversion of ethylene to these polymers remains the same. The comparability of bio-PE and bio-PET with their petrochemical counterparts is ensured since they are identical polymers. Although ethanol is produced from various food crops such as sugarcane, maize and wheat, we concentrate on production from sugarcane since it is currently the only feedstock used to produce bio-PE and bio-PET. Also, we focus on Brazilian and Indian production because they are the world's largest sugarcane and sugarcane ethanol producers and today's production of bio-PE and bio-PET is established in Brazil and India, respectively (de Jong et al., 2012b).

3.2.1 Sugarcane ethanol production in Brazil and India

The production chain of ethanol in Brazil and India is described in detail in Chapter 2. This section focuses on main differences between ethanol production in South-Central Brazil and Uttar Pradesh, India. Brazilian sugarcane cultivation offers high yields (around 85 t_{cane}/ha) and is highly mechanised; pre-harvesting burning practices are partly applied but they are gradually being phased out. In India, agricultural practices rely mainly on human and animal labour, yields are significantly lower (around 55 t_{cane}/ha) and irrigation is required. In Brazil, fresh sugarcane juice is directly fermented and distilled to ethanol whereas in India only sugarcane molasses are used.

In both countries, ethanol production yields co-products, which are used internally and reduce inputs (e.g. fertilisers), make the process less dependent on external energy sources and provide surplus electricity and biomass. During sugarcane juice extraction, juice is separated from the fibrous stalks and the obtained shredded bagasse is used in co-generation facilities to produce steam and electricity to meet process energy requirements. An increasing number of mills both in Brazil and in India generate surplus electricity, which they sell to the national grid. The remaining bagasse is typically sold as a solid biofuel or as feedstock for the paper industry (ISMA, 2011b, 2011c; Seabra et al., 2011). Residues of juice filtration, typically referred to as filtercake or mud, are mixed with ashes from boilers and are returned to sugarcane fields as fertilisers. The distillation generates a significant amount of wastewater (stillage). In Brazil, after cooling in open ponds, stillage is distributed onto the fields and valuable nutrients are recycled (Lisboa et al., 2011). In India, stillage is typically treated in anaerobic digesters to generate biogas; the biogas is used in co-generation facilities and contributes to on-site energy supply (Tewari et al., 2007). Depending on filtercake availability a number of distilleries use part of the stillage to produce bio-compost, which is either sold or offered to farmers for free (ISMA, 2012).

3.2.2 Bio-PE production

Historically, bioethylene was derived from ethanol dehydration. However, after the mid-1940s, with the rise of the petrochemical industry, steam cracking of petroleum liquids and heavier fractions of natural gas became the dominant processes for ethylene production (Kochar et al., 1981). As the industry has renewed its attention to produce ethylene from ethanol, the process has been further optimised and new, improved catalysts have been developed (Chematur, 2011).

Ethanol is catalytically dehydrated in a vapour phase reaction to remove one water molecule per molecule of ethanol, thereby yielding ethylene. The process is endothermic and based on the theoretical reaction enthalpy it requires 1.63 $MJ/kg_{ethylene}$. Diethyl ether is formed as intermediate product at temperatures between 150 °C and 300 °C, while ethylene formation is predominant between 300 °C and 500 °C (Morschbacker, 2009). Due to their high selectivity, productivity and resistance to deactivation, alumina or silica-alumina catalysts are used in fixed-bed or fluidised-bed reactors. Minor quantities of ethane, propylene, butylenes, acetaldehyde and negligible amounts of methane, carbon monoxide and dioxide, ethyl ethers and hydrogen are formed from side-reactions. The effluent stream consists mainly of water containing acetaldehyde, diethyl ether and non-reacted ethanol. It is treated by stripping, reaching a chemical oxygen demand level lower than 100 ppm. Commonly, light organic by-products are flared and heavy organics are collected for fuel use, which reduces the net energy requirement of the process (Chematur, 2011).

Depending on the desired application ethylene is converted to LDPE in high-pressure tubular or autoclave reactors (PlasticsEurope, 2008a). HDPE is mainly produced in low-pressure reactors via suspension or gas phase polymerisation (PlasticsEurope, 2008b). LLDPE is produced at relatively low pressures and temperatures by solution or gas phase polymerisation (PlasticsEurope, 2008c). Among these polymers, LDPE has the highest primary energy demand due to high electricity requirement. The primary energy demand of HDPE and LLDPE is lower despite high steam demand, because steam is typically produced with higher conversion efficiencies than electricity (JRC, 2007).[12]

3.2.3 Bio-PET production

Ethylene is mixed with oxygen, CO_2, argon and methane or nitrogen and the dilute gas mixture is fed to a tubular catalytic reactor. The reaction to ethylene oxide (EO) is highly exothermic; the reaction temperature is controlled by producing steam and by controlling the pressure in the steam drum. The EO is scrubbed with water. By-product CO_2 is removed

[12] Primary energy requirement is 6.4, 8.2, 4.8 MJ/kg HDPE, LDPE and LLDPE, respectively. Calculated from steam demand assuming 90% production efficiency (Saygin et al., 2011) and from electricity demand assuming 42% production efficiency (HP, 2010).

and returned to the reactor loop, and the EO is steam-stripped and recovered as concentrated aqueous solution. The EO stream is sent to the glycol reactor in which ethylene glycols are produced by reaction with water. After the reactor a multi-effect evaporator system is used for water removal. Glycols are dried, cooled and sent to a distillation train for separation and purification, where MEG is separated from the heavier Diethylene Glycol (DEG) and Triethylene Glycol (TEG) (HP, 2010).

Bio-MEG represents 27.7% of the inputs in mass terms required for bio-PET production.[13] The other monomer is Purified Terephthalic Acid (PTA), produced from paraxylene. So far no commercial bio-based route to paraxylene exists. Xylenes are produced mainly by solvent extraction and fractionated distillation of aromatic rich streams in refineries. Paraxylene and acetic acid are used to produce PTA. The monomer is polymerised with MEG to amorphous PET in a direct esterification or melt polymerisation process. For bottle applications a second polymerisation in solid state is required (PlasticsEurope, 2011).

3.3 METHODOLOGY

To evaluate the environmental impacts of products and services, Life Cycle Assessment (LCA) is used (ISO, 2006a, 2006b) which is widely applied in environmental assessments of bio-based materials (Álvarez-Chávez et al., 2012; Uihlein et al., 2008; Weiss et al., 2012), and can also be used for comparative assertions between products that deliver equivalent functions (ISO, 2006a).

3.3.1 System boundaries, functional units and data

The systems are assessed from cradle to gate. Both Brazilian and Indian sugarcane ethanol are used for bio-PET, while only Brazilian ethanol is used for bio-HDPE (henceforth referred to as bio-PE, unless specified otherwise). For bio-PE we consider bioethylene production and polymerisation to bio-PE in Brazil, and transport to Europe. For bio-PET we consider bio-MEG production in India. We analyse three cases, where bio-MEG is produced at India Glycols (IGL) from: (a) Indian sugarcane ethanol production in Uttar Pradesh (bio-MEG_{IN}), (b) ethanol of the distillery attached to the MEG facility in India Glycols (bio-MEG_{IGL}), (c) Brazilian sugarcane ethanol (bio-MEG_{BR}). These routes currently supply bio-MEG to the market. We include transport of bio-MEG to Europe, PTA production and polymerisation to bio-PET in Europe. Final transport is included because we compare with European production of fossil-based PE and PET. The functional units are 1 kilogramme of bio-based PE (bio-PE) and 1 kilogramme partially bio-based PET, produced in three product-systems: bio-PET_{IN}, bio-PET_{IGL}, and bio-PET_{BR} (Figure 3.1).

For production of sugarcane ethanol we use cradle-to-gate results from our previous study, which focused on the comparison of Brazilian and Indian ethanol (Chapter 2). For conversion of ethanol to bioethylene in Brazil, data from technology licensors are used. We

13 The share of ethylene glycol based on the carbon content of the reactants (ethylene glycol and terephthalic acid) is 20%. The share of the molecular weight of MEG relative to the repeating unit of PET is 31%.

assume that process electricity is sourced from the national grid or produced by natural gas depending on the allocation approach. For steam production we assume the use of natural gas and oil, on a 76:24 primary energy ratio, based on ecoinvent v2.2. Other heat requirements are supplemented by natural gas. Biomass is also suitable for industrial heat production, however, this option is not assessed in this study (Saygin et al., 2014). For ethylene polymerisation to bio-PE we estimate gate-to-gate impact assessment results, by deducting the cradle-to-gate data for the monomer from cradle-to-gate data for the polymer (ecoinvent v2.2 data; Hischier, 2007). These data are representative for European production sites; however, due to lack of more accurate information, we assume that they are representative also for Brazil. For bio-MEG production in India, we use data from the producer. We also present results based on data provided by technology licensors (section 3.4). Similar to bio-PE, process electricity is sourced from the national grid or from the local grid depending on the allocation. Steam is produced from coal (bio-PET$_{BR}$, bio-PET$_{IN}$) or from coal and biogas (bio-PET$_{IGL}$). Emissions for biogas combustion are based on Stucki et al. (2011). Other heat requirements are supplemented by natural gas. We also present results assuming that heat and electricity is supplied by an on-site coal-based combined heat and power (CHP) plant (section 3.4). For PTA production and polymerisation, we use cradle-to-gate results calculated from the latest environmental profiles published by the European Plastics association (Mersiowsky, 2011; PlasticsEurope, 2011) and ecoinvent v2.2 (section 3.6). We adapt the electricity mix of electricity-intensive processes, namely oxygen, nitrogen and sodium hydroxide production, to the regional electricity mix. In addition, for oxygen production we adapt process electricity requirements based on information provided by the bio-MEG producer. As background database for secondary inputs and processes we use ecoinvent v2.2 (ecoinvent, 2010). Table 3.1 gives an overview of the data used in this study. The impact assessment results we used for the process steps ethylene polymerisation to bio-PE, PTA production and polymerisation to bio-PET are presented in section 3.6.

We compare the results with cradle-to-gate environmental profiles of petrochemical PET (pchem-PET) and PE (pchem-PE) production in Europe. Ethylene polymerisation to PE, PTA production and polymerisation to bottle-grade PET are the same for the bio-based and the petrochemical route.[14] For computation of impact assessment results we use the software Simapro v 7.3 (Pré Consultants, 2011).

3.3.2 Multi-functionality

For multi-functional processes a suitable method must be applied to assign the environmental interventions to the multiple outputs. Based on the International Organization for Standardization, one should first subdivide the multi-functional system into distinct sub-processes and associate process-related impacts with individual products or co-products. This is not always possible due to non-separable processes or data availability. The

[14] PlasticsEurope released a new version of impact assessment results for petrochemical ethylene and MEG (PlasticsEurope, 2012). It is expected that results for pchem-HDPE will be updated accordingly. While preparing this study, these were not available. Therefore, the comparison is based on ecoinvent v2.2. We discuss findings of the latest PlasticsEurope report in section 3.4. For pchem-PET we use the latest available data published by PlasticsEurope (Table 3.1).

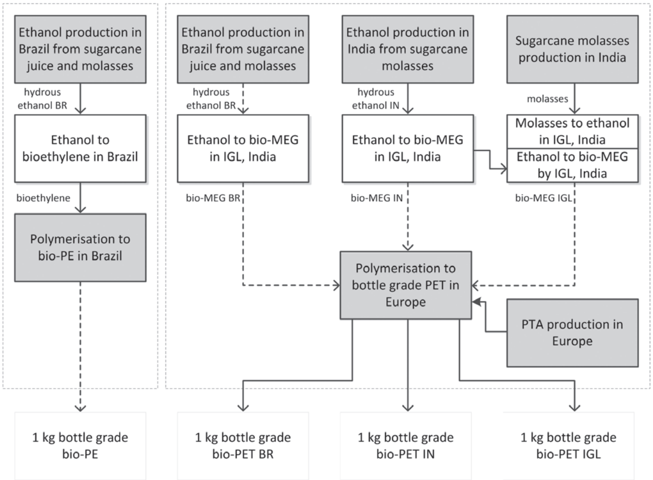

Figure 3.1 Main process steps in bio-PE (left) and bio-PET (right) production for further use in Europe. Data for processes marked with highlighted boxes are based on literature sources, databases or industry averages. Data for processes marked with clear boxes are based on primary producers or technology licensors. Dashed arrows indicate international transport.

Table 3.1 Representativeness, regional and temporal information on data used in this study

Process step	Region	Period	Representativeness	Source
Sugarcane ethanol	South-Central Brazil	2008-2009	Regional	Chapter 2
	Uttar Pradesh, India	2009-2010	Regional	
Ethanol to bioethylene	Brazil	2011	Technology licensor[a]	Personal communication
Bioethylene to bio-PE	Brazil	2006	European industry average	Hischier (2007)
Ethanol to bio-MEG	India	2011	Producer[a], technology licensor[a]	Personal communication
PTA	Europe	2009/2000[b]	European industry average	PlasticsEurope (2011)
Polymerisation to bio-PET	Europe	2009/2000[b]	European industry average	
Transoceanic transport[c]	International	2003	LCA database	Spielmann et al. (2007)
Petrochemical PET	Europe		European industry average	PlasticsEurope (2011)
Petrochemical PE	Europe		European industry average	Hischier (2007)

[a]Conversion of ethanol to bioethylene and bio-MEG is based on proprietary data. For this reason it is not possible to provide an extensive life cycle inventory. [b]For PTA production and for polymerisation we use cradle-to-gate impact assessment results for greenhouse gas emissions and non-renewable energy use calculated from the latest PlasticsEurope report (PlasticsEurope, 2011). These data are representative for 2009. For other impact categories we use data from ecoinvent v2.2 (Hischier, 2007), which are representative for 2000. [c]GHG emissions of sea transport are estimated at 8.8 gCO_{2eq}/tkm. They are in line with the emissions reported by the European Environment Agency (8 gCO_2/tkm) for bulk dry sea transport (EEA, 2009).

Table 3.2 Allocation approaches chosen for multi-functional processes in sugarcane processing in Brazil and India (Chapter 2)

Approach[a]	Products	Description	Credits/Allocation factors [per kg_{etOH}]
System expansion, conservative (SE-C)	Energy outputs: electricity (Brazil, India) Material outputs: ethanol and bagasse (Brazil), sugar, molasses and bagasse (India)	Displacement of *low* CO_2 emission intensity grid power Economic allocation between material outputs	Brazil: 0.16 kWh (0.22 $kgCO_{2eq}$/kWh) India: 0.5 kWh (0.55 $kgCO_{2eq}$/kWh) / Sugar 91.5%, Molasses 8%, Bagasse 0.5%
System expansion, optimistic (SE-O)	Energy outputs: electricity, heat from bagasse (Brazil, India) Material outputs: ethanol (Brazil), sugar, molasses and bagasse (India)	Brazil: Displacement of *high* CO_2 emission intensity power from natural gas and oil-based heat India: Displacement of *high* CO_2 emission intensity grid power and coal-based heat Economic allocation between material outputs	Brazil: 0.16 kWh (0.65 $kgCO_{2eq}$/kWh), 0.9 MJ_{heat} (0.09 $kgCO_{2eq}/MJ_{heat}$) India: 0.5 kWh (1.1 $kgCO_{2eq}$/kWh), 0.38 MJ_{heat} (0.13 $kgCO_{2eq}/MJ_{heat}$) / Sugar 92%, Molasses 8%
Economic allocation, conservative (EA-C) and optimistic (EA-O)	Material outputs: ethanol, electricity and bagasse (Brazil). Sugar, molasses, electricity and bagasse (India)	Economic allocation between material outputs	Brazil: Ethanol 97.5%, Electricity 2%, Bagasse 0.5% India: Sugar 85%, Molasses 7.5%, Bagasse 0.5%

[a]In the system expansion approaches process electricity input in the foreground system and energy intensive material inputs has the same emission intensity with surplus electricity. Chapter 2 present results for one economic allocation approach in which process electricity is supplied from the national grid. In this study we extend the analysis by assuming (a) process electricity for bioethylene from the national grid and for bio-MEG from the regional grid in Uttar Pradesh (EA-C) and (b) process electricity for bioethylene from natural gas and for bio-MEG from the national Indian grid (EA-O).

second approach, termed system expansion, enlarges the system boundaries to include the additional functions of the co-products.[15] The third and fourth options allocate environmental interventions between co-products based on physical or other relationships (e.g. economic value), respectively (ISO, 2006b).

In this study there are several process steps that lead to multiple outputs. The sugarcane ethanol product-system involves co-production of sugarcane trash, filtercake, surplus electricity and bagasse from sugar mills and distilleries. In the Indian ethanol product-system we are faced with the multi-functionality problem due to co-production of sugar and molasses. To account for these outputs we apply four different allocation approaches, which are explained in Chapter 2. An overview is presented in Table 3.2. Furthermore, in bio-MEG

15 Although not explicitly stated in ISO (2006a) this approach is typically considered equivalent to subtracting the additional functions of the system, i.e. by deducting credits (Heijungs, 2014). This “credit approach” is applied in this study.

production, DEG, TEG and heavier glycols are also produced. These are all valuable products with applications in the automotive and packaging industry (IGL, 2011). We allocate their impacts based on mass.

3.3.3 Impact assessment methodology

We present results for greenhouse gas (GHG) emissions over a 100 year timeframe (IPCC, 2006).[16] GHG emissions are the most commonly used metric to assess the sustainability impacts of products. However, as Laurent et al. (2012) indicate, GHG emissions are correlated with other environmental impacts only when these predominantly originate from fossil fuels. When toxicity to humans or ecosystems and land-use are of concern, then GHG emissions alone are a weak indicator. For bio-based products, the use of agrochemicals during biomass cultivation is expected to contribute to toxicity-related impacts, as opposed to GHG emissions and non-renewable energy use (NREU), for which bio-based products generally show savings when compared to petrochemical products (Weiss et al., 2012). Therefore, we extend the analysis to NREU, land-use, freshwater eutrophication, water-use at the midpoint level and to the endpoints human health (HH) and ecosystem quality (EQ). All categories are analysed using Impact 2002+ (Jolliet et al., 2003).[17] Water-use is reported at the inventory phase and at the endpoint using factors in Pfister et al. (2009). The impact of infrastructure is excluded from the analysis.

3.4 RESULTS AND DISCUSSION

3.4.1 Bio-based carbon

We include the bio-based carbon content of the final polymer as carbon storage (negative bar section in Figure 3.2 - Figure 3.3, amounting to 3.2 $kgCO_2/kg_{bio\text{-}PE}$ and 0.45 $kgCO_2/kg_{bio\text{-}PET}$). It is important to account for the bio-based carbon in cradle-to-gate systems as it may affect the final ranking of alternative options when compared to ranking based on cradle-to-grave assessments (Pawelzik et al., 2013). In this case, net emissions of the bio-based polymers (symbols in Figure 3.2 - Figure 3.3) should be compared with gross emissions of the petrochemical counterpart (shaded stacked bars, excluding end-of-life). Next to the base case ('Net') the graph shows high and low performance cases for mills and distilleries with high surplus electricity versus no surplus electricity and dependency on the grid based on Chapter 2. Figure 3.2 - Figure 3.3 also include results at the end-of-life, assuming incineration without energy recovery. In this case, gross results for the bio-based polymers (without carbon storage) should be compared with the petrochemical polymers at the end-of-life. Apart from process emissions from polymer production (shaded stacked bars in Figure 3.2 - Figure 3.3) also feedstock carbon emissions are accounted for (dotted bars in Figure 3.2 - Figure 3.3). Bio-based carbon emissions are neutral while fossil carbon

[16] Fossil and biogenic methane characterisation factors adapted to 27.75 $kgCO_{2eq}/kgCH_4$ and 25 $kgCO_{2eq}/kgCH_4$, respectively (Muñoz et al., 2013).

[17] Including the ozone depleting potential of nitrous oxides (N_2O), i.e. 0.017 $kgCFC\text{-}11_{eq}/kgN_2O$ (Ravishankara et al., 2009). Aquatic acidification and freshwater eutrophication are associated with ecosystem quality based on $8.82\cdot10^{-3}$ $PDF\cdot m^2\cdot y/kgSO_{2eq}$ and 1.4 $PDF\cdot m^2\cdot y/kgPO_{4eq}$, respectively (Humbert et al., 2012).

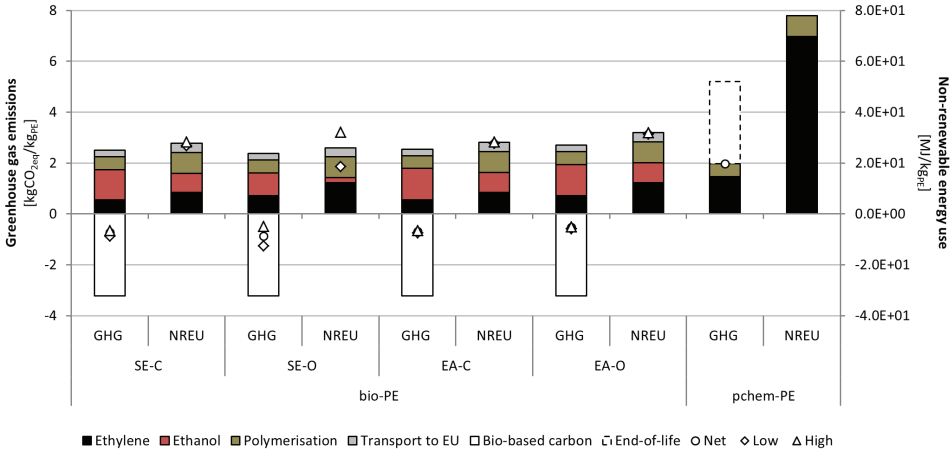

Figure 3.2 Greenhouse gas emissions and non-renewable energy use of bio-PE production from Brazilian sugarcane ethanol

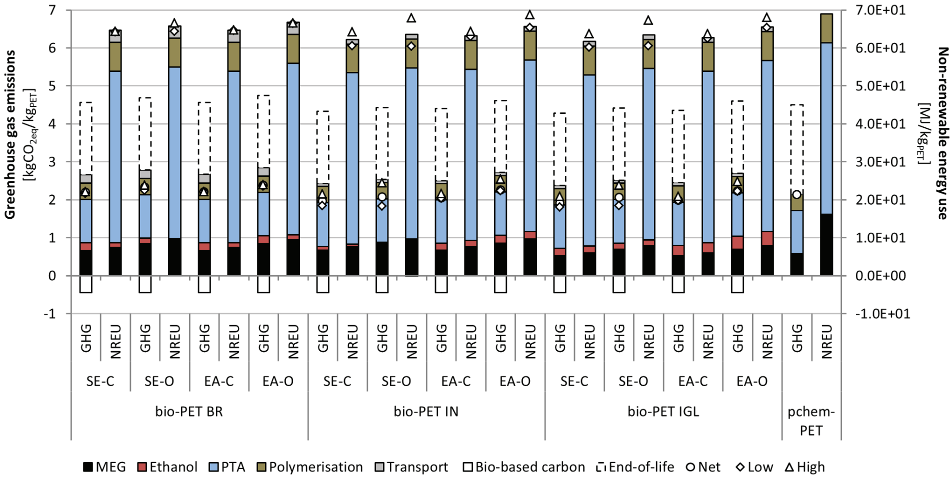

Figure 3.3 Greenhouse gas emissions and non-renewable energy use of bio-PET production from Brazilian and Indian sugarcane ethanol

emissions contribute to global warming. Bio-PE is fully bio-based hence there are no fossil end-of-life emissions. In bio-PET emissions from fossil carbon in PTA are included. For petrochemical polymers all carbon content is of fossil origin and is accounted for its contribution to global warming.

3.4.1.1 Greenhouse gas emissions and non-renewable energy use

GHG emissions are presented for bio-PE and bio-PET in Figure 3.2 and Figure 3.3, respectively. Bio-PE results are broken down to ethanol production, bioethylene production, polymerisation to bio-PE and final transport of the polymer from Brazil to Europe. Bio-PET results are broken down to ethanol production, bio-MEG production, transport of bio-MEG from India to Europe[18], PTA production and polymerisation to bio-PET in Europe.

Production of bio-PE results in average net CO_2 storage in the range of 0.75 $kgCO_{2eq}/kg_{bio\text{-}PE}$ (Figure 3.2). When compared to pchem-PE it leads to approximately 140% savings. The result is similar across the four allocation approaches (SE-C, SE-O, EA-C, EA-O) but the absolute contribution of each step differs slightly. In SE-O, ethanol production contributes less than in SE-C, EA-C and EA-O, while bioethylene production is somewhat higher. In SE-O, ethanol receives large credits for the co-products surplus electricity and surplus bagasse (Table 3.2), thus the cradle-to-gate impact of ethanol is lowest. However, due to the high CO_2 emission intensity of the electricity in ethanol dehydration the contribution of bioethylene step increases when compared to SE-C, EA-C and EA-O. While this trade-off does not influence the base case it has an impact when applied in the two extreme cases (high, low) under SE-O. The overall net GHG emissions for bio-PE range approximately ±50% in the high and low performance case.

Bio-PET production leads to comparable GHG emissions as pchem-PET, even after deducting the bio-based carbon of bio-MEG (Figure 3.3). These cases do not consider improvement potentials by integration of a CHP, process integration and other options which will be discussed in section 3.4.1. Depending on the source of ethanol (Brazil, India) and the allocation approach (SE-C, SE-O, EA-C, EA-O) the difference to pchem-PET (2.15 $kgCO_{2eq}/kg_{PET}$) ranges by approximately ±10%. Bio-PET_{BR} results in slightly higher GHG emissions than bio-$PET_{IN,\ IGL}$, partly due to the higher GHG emissions of Brazilian ethanol (Chapter 2) and partly due to the contribution of transport of Brazilian ethanol to India. PTA production, polymerisation and transport of bio-MEG to Europe account approximately for two-thirds of the GHG emissions. The former two steps are common in the bio-based and petrochemical route. The remainder of the contribution is due to bio-MEG production and originates primarily from process energy requirements, which are met by coal-based steam and grid electricity.

The system expansion approaches (SE) are associated with credits assigned to the product-systems of ethanol production in Brazil, and molasses and ethanol production in India. SE-C assumes lower credits than SE-O, which ultimately affect the ethanol cradle-to-gate results. More specifically, ethanol has higher impacts under SE-C when compared to SE-O.

[18] Transport results for bio-PET_{BR} also include transport of ethanol from Brazil to India.

It may be expected that results for bio-PET would follow a similar pattern, however, the reverse is noticed. Bio-PET has lowest cradle-to-gate emissions under SE-C, followed by EA-C, SE-O, and finally EA-O shows most conservative estimates. This is due to electricity consumption at the bio-MEG production step, the source of which (national, highly coal-based or local, highly hydro-based) depends on the allocation approach. Per kilogramme of bio-MEG the total electricity consumed by the system is larger than the surplus provided at the ethanol step. Therefore the influence of the credits is overcompensated by the electricity input for bio-MEG production, which is the final determinant for ranking the allocation approaches. Economic allocation (EA) is not related with system credits; however, it assumes regional grid (*low* CO_2 emission intensity in EA-C) and national grid electricity input for bio-MEG production (*high* CO_2 emission intensity in EA-O). The difference between the two EA approaches is approximately 10%. The different electricity source also explains the varying contribution of the bio-MEG step across the four allocation approaches.

For bio-PET_{IGL} the contribution of bio-MEG production is lowest, because steam is not only produced from coal but also from biogas available from the attached distillery. However, the latter has an impact on the contribution of ethanol production, which is higher than in bio-PET_{IN}. This is because, the integrated bio-MEG and ethanol production (bio-MEG_{IGL}) has the same steam source, while heat requirements of ethanol production in Uttar Pradesh, India are covered by bagasse and biogas. In other words, the benefit of using biogas is fully assigned to ethanol in bio-PET_{IN}, while it is shared among ethanol and MEG production in bio-PET_{IGL}.[19] The SE-O approach leads to high spread between the high and the low performance case for bio-$PET_{IN,GL}$, when compared to the base case (approximately ±15% from the average 2 $kgCO_{2eq}/kg_{bio\text{-}PET,IN}$). The smallest variation is noticed for bio-PET_{BR} under EA-C (approximately 1% from the average 2.3 $kgCO_{2eq}/kg_{bio\text{-}PET,BR}$).

Similar to GHG emissions, bio-PE shows significant savings in NREU when compared to pchem-PE (around 65%, Figure 3.2). Apart from fossil energy use to meet process energy requirements, pchem-PE is produced from petrochemical feedstocks, which is reflected in the large NREU.[20] For bio-PE, fossil energy requirements are primarily due to process energy requirements, production of material inputs, transport, and NREU due to ethanol production (diesel use in sugarcane production) while the feedstock is exclusively bio-based. The range across the allocation approaches does not show significant variation. When the higher and the lower performance cases are assumed, then the results range only for SE-O approximately by ±25%, compared to the base case result.

Bio-PET offers NREU savings from 3 to 11% compared to pchem-PET (Figure 3.3). While the NREU for PTA and polymerisation is identical for both routes, savings occur from producing bio-MEG from renewable feedstocks. However, the contribution of transport decreases the overall difference. Similar to GHG emissions, bio-PET_{BR} causes higher NREU

[19] In bio-PET_{IGL} the ethanol is partly from the integrated distillery and partly from ethanol procured from distilleries in Uttar Pradesh (31% and 69%, respectively; share estimated based on plant capacities of 80,000 kl_{etOH}/yr and 200,000 t_{MEG}/yr).

[20] The higher and lower heating value of polyethylene is 42.8 and 42.5 MJ/kg, respectively.

than bio-PET$_{IN, IGL}$, due to higher NREU of Brazilian ethanol and its transport to India. The higher and lower performance cases lead to the widest variation for SE-O, which however, does not influence significantly the relative savings.

3.4.1.2 Emissions from land use change

Due to the large influence that land use change (LUC) may have in the GHG emissions of bioplastics (Piemonte and Gironi, 2011) and the significance they receive in the policy agenda (EC, 2012b), we attempt to account for both direct and indirect LUC emissions as described in Chapter 2. For bio-PE, additional GHG emissions due to LUC, range from 0.16 to 2.38 kgCO_{2eq}/kg$_{bio\text{-}PE}$, for low (3 gCO_{2eq}/MJ$_{etOH}$) and high (46 gCO_{2eq}/MJ$_{etOH}$) LUC emission factors, respectively (Wicke et al., 2012). Adding this value to the results presented in Figure 3.2, the net GHG emissions range from -0.7 to 1.8 kgCO_{2eq}/kg$_{bio\text{-}PE}$, which are approximately 130% and 20% lower than the GHG emissions of today's pchem-PE, for low and high LUC emission factor, respectively. For bio-PET$_{BR}$ the LUC GHG emissions range from 0.03 to 0.4 kgCO_{2eq}/kg$_{bio\text{-}PET,BR}$. The highest LUC emission factor essentially cancels the bio-based carbon storage credit of bio-PET$_{BR}$. This entails that bio-PET$_{BR}$ production may even lead to an increase in CO_2 emissions compared to pchem-PET, by 4-13% and 22-30%, for low and high LUC emission factor, respectively. Due to the wide range in LUC emission factors, these results should be interpreted with caution.

Taking into account only direct LUC, emissions of bio-PE range from -0.55 to -0.88 kgCO_{2eq}/kg$_{bio\text{-}PE}$ depending on the allocation approach (2-3% lower compared with bio-PE GHG emissions without direct LUC, Figure 3.2). For bio-PET$_{BR}$ there is very limited influence of direct LUC emissions (less than 0.2% of emissions shown in Figure 3.3). For bio-PET production from Indian sugarcane molasses, we do not account for direct LUC and indirect LUC because to our knowledge, there is a lack of publicly available reliable data that could support calculations on the effects of land use change for Indian sugarcane ethanol production. However, if ethanol production in India increases displacement effects are likely to take place and LUC emissions should be accounted for (Chapter 2).

3.4.1.3 Stillage treatment

The assumption that stillage is treated via anaerobic digestion is critical for estimating CO_2 emissions. In particular, if stillage is disposed without prior treatment, and anaerobic conditions prevail, then Indian ethanol emissions may range from 2.6 to 3.1 kgCO_{2eq}/kg$_{etOH}$, as opposed to 0.09 to 0.64 kgCO_{2eq}/kg$_{etOH}$ which are the emissions of the reference system (Chapter 2). This increase is primarily due to the high global warming potential of methane but also due to the lower biogas recovery of the system –thus lower credits for surplus power provided to the grid. The emissions of bio-PET$_{IN}$ increase significantly to 2.8-3.1 kgCO_{2eq}/kg$_{bio\text{-}PET,IN}$, as opposed to 2-2.3 kgCO_{2eq}/kg$_{bio\text{-}PET,IN}$ in the base case.

3.4.1.4 CHP plant and other improvement options

In the production premises of the MEG producer's ethanol distillery a CHP plant is operated (DNV, 2009). In Figure 3.4 we present results for the GHG emissions of bio-PET$_{BR,IN}$ which assume that the CHP plant covers all steam demand of MEG produc-

Table 3.3 Approaches chosen for the multi-functionality problem at the CHP plant

	SE-C	SE-O	EA-C	EA-O
Electricity				
FIT	Displacement of power grid electricity:		Allocation factor:	
	Low CO_2 emission intensity (0.55 $kgCO_{2eq}$/kWh)	*High* CO_2 emission intensity (1.1 $kgCO_{2eq}$/kWh)	40%	40%
	Process uses electricity from local grid	Process uses electricity from national grid	Process uses electricity from local grid	Process uses electricity from national grid
No FIT	No displacement of power grid electricity:		Allocation factor:	
	CHP electricity used in process	CHP electricity used in process	40%	40%
	Remainder electricity supplied from local grid	Remainder electricity supplied from national grid	Remainder electricity supplied from local grid	Remainder electricity supplied from national grid
Steam				
FIT/no FIT	Used in process	Used in process	60%	60%
			Used in process	Used in process

tion.[21] For co-production of steam and electricity we apply the four allocation approaches as described in this chapter (Table 3.2 - Table 3.3). In one case, we assume that a feed-in tariff (FIT) for CHP electricity is in place, which makes it profitable for the producer to sell CHP electricity and purchase grid electricity for process requirements (CHP FIT). In this case, only CHP steam is used in the MEG process and electricity is supplied from the grid. To assess the effect of the FIT we also present results assuming that no FIT is in place (CHP no FIT). In this case both CHP steam and electricity are used in the MEG process, and only the remaining (net) process electricity requirement is supplied from the grid. Incorporating the CHP plant to the study reduces the GHG emissions of bio-PET by 3-10% when compared to the base case results of Figure 3.3 (section 3.4.1). The difference is smallest (i.e. showing a limited effect of CHP on the GHG emissions of bio-PET) under the SE-C approach due to the low credits assigned for displacement of grid electricity by the CHP. The difference is largest (i.e. showing a large effect of CHP on the GHG emissions of bio-PET) under economic allocation because impacts from CO_2 intensive coal-based steam generation are allocated due to co-generation of electricity. The emissions of bio-PET estimated with a FIT are by 3-8% lower compared to a no FIT case in SE-C, EA-C and EA-O. However, under SE-O the FIT assumption leads to higher emissions than assuming no FIT because emissions from process electricity from the grid (CHP FIT) are higher than the emissions from process electricity from the CHP (CHP no FIT).

[21] It depends on the size of the CHP plant and the needs of the entire site whether or not the CHP should be included for the conversion of Brazilian and Indian ethanol to MEG (as required for bio-$PET_{BR,IN}$). Such decisions are often difficult to take for a concrete industrial setting if the provided data are ambiguous.

In addition to CHP, we assume four further improvement scenarios. These include: (a) integration of the bio-MEG plant with a distillery that fully meets the ethanol demand for bio-MEG, (b) fuel switch from coal to natural gas as primary fuel for steam production, (c) advanced MEG production technology based on technology licensor instead of producer information and (d) implementation of best available technology for bio-MEG production. Other possible improvements, e.g. process heat integration or the fuel switch from coal or natural gas to biomass at the CHP plant, are not assessed in this paper. To ensure comparability with the base case bio-PET production, we assume that all improvements are applied in India.

- Integration: The integrated production of bio-MEG and ethanol makes it possible to utilise biogas for process steam production, thus reducing net coal input required to meet the steam demand for bio-MEG and ethanol production. In the case studied above (bio-PET$_{IGL}$), the distillery's capacity is not sufficient to meet the bio-MEG plant's maximum capacity and the ethanol demand is supplemented from other distilleries. This constrains the availability and use of biogas at the bio-MEG production site. Assuming that bio-MEG production is integrated with a distillery that fully meets its ethanol demand, the biogas availability per kilogramme of bio-MEG increases; hence the net coal input decreases. The latter has an impact on GHG emission reduction in the range of 15%, 5%, 12% and -4%, when compared to pchem-PET for SE-C, SE-O, EA-C and EA-O, respectively ('Integration', Figure 3.4). To place this in perspective, the GHG emission reduction of a non-integrated case (bio-PET$_{IN}$) is 10%, 4%, 7% and -5% for SE-C, SE-O, EA-C and EA-O, respectively ('Base case', Figure 3.4). This scenario is applied only in bio-PET$_{IGL}$.
- Fuel switch: India has abundant coal supplies, which are used as a primary fossil fuel for several energy applications. The high emission factor of coal combustion along with other environmental impacts compromises the environmental benefits of the systems studied. A fuel switch from coal to natural gas for steam generation in bio-MEG production would result in further reduction of approximately 10% of the GHG emissions of bio-PET production when compared to the base cases ('Fuel switch', Figure 3.4). This scenario is applied in bio-PET$_{BR}$, bio-PET$_{IN}$ and bio-PET$_{IGL}$.
- Licensor: For this scenario we use data provided by technology licensors. The results could apply to a newly commissioned bio-MEG plant ('Licensor', Figure 3.4). To compare this scenario with the base case we assume steam production from coal. Compared to the base case bio-PET, this would lead to additional GHG emission reduction in the range of 20%. This scenario is applied in bio-PET$_{BR}$ and bio-PET$_{IN}$.
- BAT: This scenario assumes for bio-PET similar process energy requirements for ethylene dehydration as assumed for bioethylene production (i.e. the technology assumed in bio-PE, which was found to have lowest energy requirements per kilogramme of ethylene). Furthermore, for the steps ethylene oxidation and hydrolysis-to-ethylene glycol we assume the same process energy requirements as in petrochemical MEG production. In addition, we assume that natural gas is used for steam generation. By aggregating process energy requirements of each step we conclude that significant

GHG emission reduction can be achieved ('BAT', Figure 3.4). These savings range from 10% to 20% and 15% to 30%, compared to today's pchem-PET, for bio-PET_{BR} and bio-PET_{IN}, respectively.

Figure 3.3, Figure 3.4 and Table 3.4 show that 'BAT' offers the highest savings on GHG emissions compared to the base case and pchem-PET ('BAT' is displayed on the upper side on the improvement options shown in Figure 3.4). Other options also offer improvements; in order of decreasing contribution these are 'Licensor', 'Fuel switch' and 'CHP'. The combination of some of these measures (e.g. 'Fuel switch' and 'CHP') and the use of biomass for the supply of process steam can potentially reduce the GHG emissions even further, depending on the implemented technological option and the chosen assessment methodology.

3.4.2 Human health and ecosystem quality

Potential impacts of bio-PE and bio-PET on HH and EQ are significantly higher than those of their petrochemical counterparts. The impact of bio-PE on HH is 50 times higher and on EQ is 2 orders of magnitude higher than pchem-PE (Figure 3.5). The results are similar across all four allocation approaches. Ethanol production dominates (98%) in both impact categories, with agriculture being the main contributing factor. The remaining impacts originate from ethanol dehydration, polymerisation and final transport to Europe. The contribution analysis of Brazilian sugarcane ethanol production on HH shows that carcinogenic and non-carcinogenic emissions due to pesticide application are responsible for around 80% of the impact. The remainder is primarily due to pre-harvesting burning practices but also to some extent due to bagasse combustion in boilers in Brazil (Chapter 2).

Due to uncertainties in active ingredients of pesticides applied in sugarcane cultivation we analyse two cases: (a) we exclude impacts of a highly toxic substance (daconate) from unspecified pesticides ('Pesticide control', Figure 3.5), (b) we assume that all unspecified pesticides are the pesticide daconate ('High', Figure 3.5). In addition, we consider a third case (c) where we assume that no pre-harvesting burning practices are applied ('Low', Figure 3.5). The total impact of bio-PE production on HH declines by 75% if daconate is eliminated and an additional 70% reduction is achieved if pre-harvesting burning practices are phased out. The remaining impact is four times higher than pchem-PE, of which one-fourth is due to international transport of bio-PE to Europe, polymerisation and ethanol dehydration.

For EQ, land occupation contributes approximately 80% to the total impact (4.3 m^2org. arable/$kg_{bio\text{-}PE}$). Excluding land occupation, 90% of the remainder impact is associated with terrestrial ecotoxicity, and 9% with terrestrial acidification and nitrification. Eliminating daconate reduces the EQ impact of bio-PE by approximately 50% (if land occupation is included reduction is 13%; 'Low', Figure 3.5). However, the comparison with pchem-PE still indicates a factor 40 higher impact of bio-PE on EQ. This is primarily related with heavy metals in sugarcane production. 10% of the impact is due to bioethylene production and transport to Europe. For terrestrial acidification and nitrification the two processes that contribute are pre-harvesting burning and bagasse burning in co-generation facilities.

Table 3.4 Net cradle-to-gate greenhouse gas emissions of bio-PET produced from Brazilian and Indian ethanol under different improvement options in $kgCO_{2eq}/kg_{PET}$

	bio-PET BR				bio-PET IN				bio-PET IGL				pchem-PET
	SE-C	SE-O	EA-C	EA-O	SE-C	SE-O	EA-C	EA-O	SE-C	SE-O	EA-C	EA-O	
Base case	2.21	2.34	2.22	2.39	1.98	2.08	2.06	2.27	1.93	2.07	2.00	2.25	
CHP FIT	2.15	2.22	2.01	2.17	1.92	1.97	1.85	2.05					
CHP no FIT	2.15	2.18	2.16	2.23	1.92	1.92	2.00	2.10					
Integration									1.83	2.05	1.90	2.23	2.15
Fuel switch	2.01	2.14	2.02	2.19	1.78	1.88	1.85	2.07	1.76	1.90	1.84	2.08	
Licensor	1.84	1.92	1.85	1.97	1.62	1.68	1.70	1.85					
BAT	1.68	1.85	1.70	1.91	1.45	1.60	1.53	1.78					

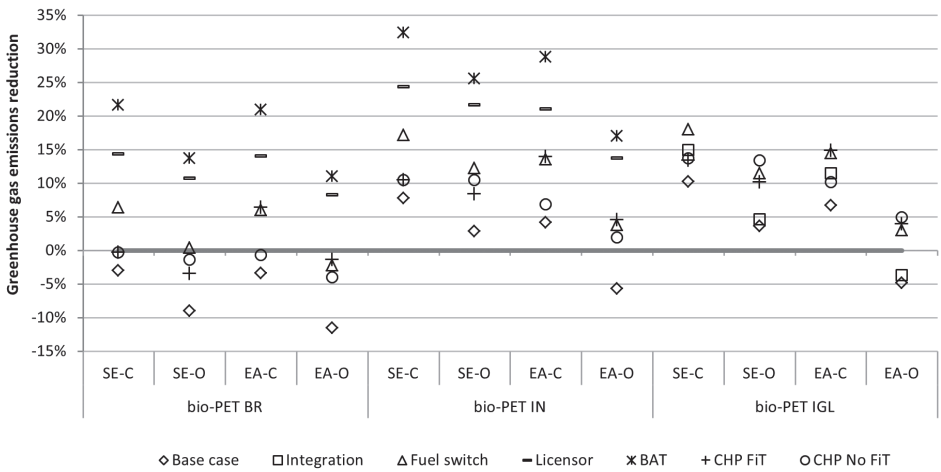

Figure 3.4 Cradle-to-gate greenhouse gas emission reduction potentials of bio-PET compared to today's petrochemical PET (0% line)

Eliminating the former reduces further the impact by 3%. On the contrary, if the unspecified fraction of pesticides is assumed to be daconate, the potential impact of bio-PE on HH and EQ increases by 140% and 25%, respectively ('High', Figure 3.5).

Next we normalise the results based on factors of Impact 2002+ (Jolliet et al., 2003). The impacts on HH appear to be significantly higher than those on EQ, climate change or resources. The impact categories that contribute most are non-carcinogens, carcinogens and respiratory inorganics. Eliminating daconate and pre-harvesting burning practices slightly changes the pattern and respiratory inorganics become the dominating impact category, while carcinogens and non-carcinogens are significantly reduced. Even though HH is still the most important area of protection, the relative difference to EQ, climate change and resources is reduced from more than 1 order of magnitude to a factor 2, 4 and 5, respectively (section 3.6). Despite benefits that bagasse co-generation exhibits (eliminates the need for external fuels and generates surplus electricity, assigned as credit to the ethanol product-system) it contributes to impacts of bio-PE production on HH through particulate matters emissions. These emissions need to be reduced by appropriate combustion technologies. It should be noted that spatial differentiation and local conditions are critical for appropriately assessing the fate of and exposure to these pollutants.

For bio-PET, impacts on HH and EQ are shown in Figure 3.6. Similar to bio-PE the allocation approach does not influence the final results. Both HH and EQ impacts related to bio-PET$_{IN, IGL, BR}$ are between a factor 14 and 19 higher than for pchem-PET. PTA production contributes approximately 20% to the HH impacts and approximately 6-8% to the EQ impacts of bio-PET. To estimate the impacts of PTA production and polymerisation on HH and EQ we use data from ecoinvent (Hischier, 2007), while the comparison with pchem-

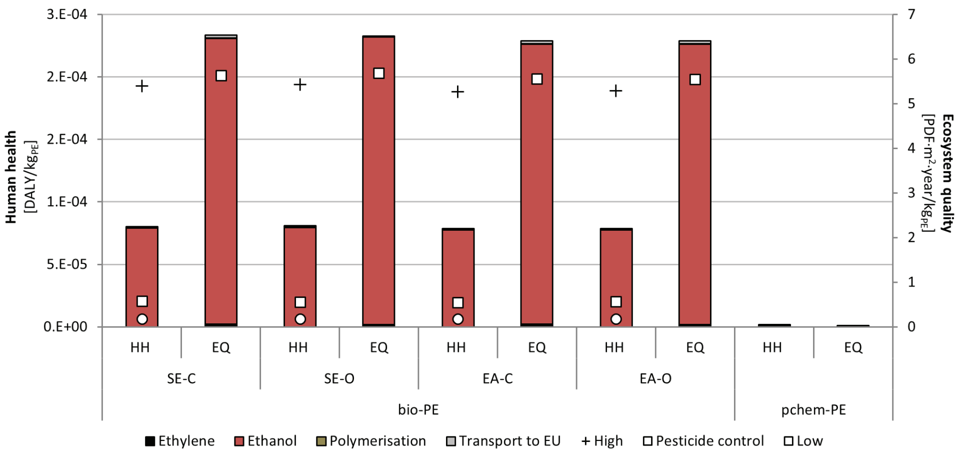

Figure 3.5 Potential impacts of bio-PE production from Brazilian sugarcane ethanol on human health and ecosystem quality

PET is based on new eco-profile data (PlasticsEurope, 2011). This entails that the comparison for HH and EQ is not fully aligned for PTA production and polymerisation. These steps should have the same absolute contribution to both bio-based and petrochemical PET. However, even when excluding the contribution of these steps from bio-PET impacts, it still has higher impact than pchem-PET, which is a plausible outcome (by approximately a factor of 15 for bio-PET_{BR} and 12 for bio-PET_{IN} on HH and EQ). Similar to bio-PE, most of the impacts originate from ethanol production. Ethanol contributes approximately 70% on HH and 80% on EQ, for bio-PET_{BR} and bio-PET_{IN}, respectively. In addition, 4% of the impact on HH comes from transport, and MEG production (each contribute from 1 to 3%). Bio-MEG production contributes 7 to 11% depending on the ethanol source and allocation, while transport contributes between 2-5% on the EQ impacts, for bio-PET_{BR} and bio-PET_{IN}. The absolute impact of bio-PET_{BR} is slightly higher because in this process chain all impacts of sugarcane production are allocated to ethanol. In the process chain of bio-$PET_{IN,IGL}$, impacts of sugarcane production are allocated among sugar and molasses. The (allocated) sugarcane required for bio-PET from Brazilian ethanol is 5 $kg_{cane}/kg_{bio\text{-}PET,BR}$, while it is only 2.6 $kg_{cane}/kg_{bio\text{-}PET,IN}$ from Indian ethanol. While the Brazilian sugarcane input is twice the Indian sugarcane input, the relative difference on HH and EQ impacts is significantly lower, by 10% and 20%, respectively. This is a consequence of the higher HH and EQ impacts of Indian sugarcane as compared to Brazilian sugarcane production (Chapter 2).

Eliminating the pesticide daconate reduces the impact on HH by approximately 55% in bio-PET_{BR} and 65% in bio-PET_{IN}. Phasing out pre-harvesting burning practices in Brazil reduces the impact on HH further by 30%. Thereafter bio-PET is a factor 6 higher than pchem-PET (pre-harvesting burning is not applied in Uttar Pradesh) ('Low' for bio-PET_{BR} and 'Pesticide control' for bio-PET_{IN}, Figure 3.6). Assuming all unspecified pesticides as daconate practically doubles the impact on HH for both process chains ('Pesticide control' and 'High', Figure 3.6). For EQ the results are similar, i.e. excluding daconate reduces the impact by approximately 10 to 15% ('Pesticide control', Figure 3.6). Assuming that all unspecified pesticides have the impact of daconate increases the EQ impact by 20% and 30% for bio-PET_{BR} and bio-PET_{IN}, respectively ('High', Figure 3.6). A possible fuel switch from natural gas further reduces the impacts of bio-PET on HH by 5% and on EQ by 5 and 10% for bio-PET_{BR} and bio-PET_{IN} ('Low', Figure 3.6). When excluding the contribution of land occupation on EQ (0.7 m^2org.arable/$kg_{bio\text{-}PET,BR}$, 0.5 m^2org.arable/$kg_{bio\text{-}PET,IN}$), the lowest estimate is 4 times higher for bio-PET_{BR} and approximately 3 times higher for bio-PET_{IN}, when compared to pchem-PET. PTA production, polymerisation and transport account for approximately 50% of the remaining impact. Terrestrial ecotoxicity and acidification/nutrification of ethanol production are contributing most to the remaining impacts, due to heavy metals, other pesticides, NH_3 and NO_x emissions due to fertilisers and bagasse burning.

Normalising the results of bio-PET_{IN} shows that impacts on HH are significantly higher than impacts on EQ, namely by a factor 30. However, when compared to climate change and resources they are larger only by a factor 10 and 6, respectively. This indicates that both impacts on HH and on climate change and resources have high importance. When daconate is excluded from Indian sugarcane production, then the impacts on HH are still dominating but the difference with climate change and resources is reduced to a factor 3 and 2, respectively (section 3.6). The normalised results of bio-PET$_{BR}$ show a similar pattern, with the exception of normalised respiratory inorganics, which are comparable with GHG emissions and NREU.

3.4.2.1 Net water consumption and its contribution to human health and ecosystem quality

Due to large use of water resources in agriculture and consequently in the production chain of bio-based products, we present results for bio-PE and bio-PET, and assess their impact on HH and EQ based on cause-effect relationships explained in Pfister et al. (2009). Net water consumption is calculated based on freshwater use for irrigation and water consumed for the processing steps that lead to the final polymers. Water outputs such as stillage recycling by ferti-irrigation in Brazilian sugarcane fields or other effluents returned to nature are deducted to calculate net water input. Results are shown in Figure 3.7.

In South-Central Brazil, sugarcane crops are practically not irrigated and only rainwater is used, which is not included in the above results. On the contrary, in Uttar Pradesh, India sugarcane plantations are irrigated by freshwater, which is included in the results. This explains why water consumption for bio-PET$_{BR}$ is by a factor 10 lower from bio-PET$_{IN}$. Assuming water efficiency improvements in sugarcane irrigation for existing crops in Uttar Pradesh (Chapter 2) would reduce water consumption to 120 L/kg$_{bio\text{-}PET,IN,IGL}$, which is still a factor 6 higher than bio-PET$_{BR.}$ Given the local water stress in the region, such a difference can be considered significant as additional demand for ethanol or Indian bio-based materials may put more pressure in the depleting groundwater resources. Taking into account regional water stress indices and damage factors (Pfister et al., 2009) the impact of bio-PET$_{IN}$ increases by 3% on HH and 9% on EQ, respectively (section 3.6).

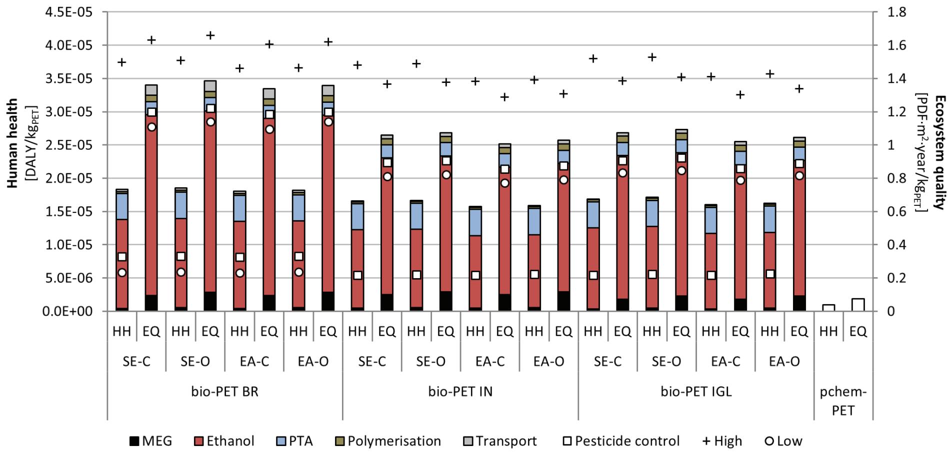

Figure 3.6 Potential impacts of bio-PET production from Brazilian and Indian sugarcane ethanol on human health and ecosystem quality

3.4.2.2 Eutrophication

Increasing nutrient loads in water bodies due mineral fertiliser use in agricultural production is a major concern for water quality. We estimate that freshwater eutrophication related to the production of the bio-based polymers is 0.45 $gPO_4/kg_{bio\text{-}PE}$ and bio-PET_{BR} while it is 0.55 $gPO_4/kg_{bio\text{-}PET,IN,IGL}$. If we assume higher P-surface runoff factor (10% P of P-fertilisers, representative for Brazilian soils; Ometto et al., (2009)) then the emissions increase to 1.3 $gPO_4/kg_{bio\text{-}PE}$, 0.6 $gPO_4/kg_{bio\text{-}PET,BR}$ and 1 $gPO_4/kg_{bio\text{-}PET,IN}$. We notice a factor 3 increase in bio-PE because high quantities of sugarcane are required for its production. For bio-PET the increase is 30% and 50% from Brazilian and Indian ethanol due to lower sugarcane requirement. For bio-PET_{IN} the increase is higher compared to bio-PET_{BR} due to significantly higher P-fertiliser input in sugarcane production (Chapter 2). The contribution of freshwater eutrophication to EQ is very low (<1%). When compared to petrochemical counterparts, freshwater eutrophication of bio-PE is approximately 60 times higher. Note that latest eco-profile data for petrochemical ethylene suggest that eutrophication is 1.1 $gPO_4/kg_{ethylene}$ (3 times higher when compared to petrochemical ethylene profiles used in this study (PlasticsEurope, 2012). Assuming a similar increase for pchem-PE then eutrophication of bio-PE is a factor 20 higher.[22] For bio-PET_{IN}, if stillage is not treated but instead is released to the soil, eutrophication increases by a factor 30 and the impact on EQ increases by 18%. The impact is even higher if stillage is disposed to water bodies (factor 50 increase in eutrophication by approximately and 28% on EQ).

3.4.3 Comparison with other studies

The GHG emissions of bio-PE in this study differ with those reported in literature. The results of our EA-C are higher than those of the attributional approach of Liptow and Tillman (2012) by roughly 1.2 $kgCO_{2eq}/kg_{PE}$. More than 50% difference is associated with sugarcane production, for which Liptow and Tillman (2012) in their attributional approach did not include mechanised harvest, fossil CO_2 emissions from lime and urea, N_2O and CH_4 emissions from unburned trash. The remaining difference is associated with ethanol production, dehydration and polymerisation. Ethanol production in Liptow and Tillman (2012) is not associated with GHG emissions. The small difference in ethanol dehydration to ethylene (0.15 $kgCO_{2eq}/kg_{PE}$) can be related with different process energy requirements based on different data sources. For polymerisation we assumed European production, while Liptow and Tillman (2012) assumed Brazilian production. The difference between our findings and those of Hunter et al. (2008) is in a similar range (1.3 $kgCO_{2eq}/kg_{PE}$). However, our study differs in scope and the product-systems studied. In section 3.6 we present a set of assump-

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22 PlasticsEurope indicates that the eutrophication potential of pchem-PET is 0.81 gPO_4/kg_{PET} (PlasticsEurope, 2011). It is calculated based on CML impact assessment, which includes characterisation factors for N-emissions, relevant for near shore eutrophication. In our analysis we calculate freshwater eutrophication potential limited to P-emissions and chemical oxygen demand. To compare freshwater eutrophication potential of pchem-PET with bio-PET we calculate the impacts of the former with Impact 2002+. The calculated emissions are by a factor 2 higher for bio-PET_{BR} and by a factor 3 higher for bio-PET_{IN} than pchem-PET. Similarly, accounting for the eutrophication potential of nitrogen emissions in bio-PE production results in a factor 10 higher estimate compared to current pchem-PE.

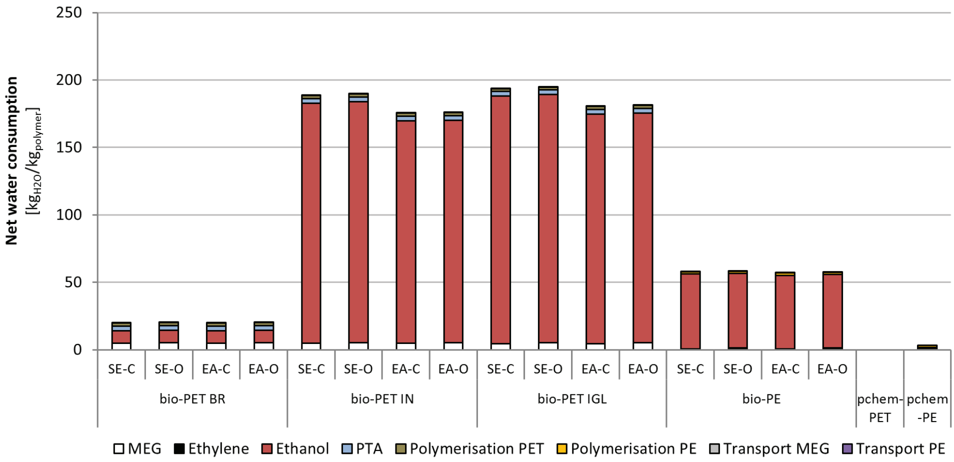

Figure 3.7 Net water consumption of bio-based and petrochemical PE and PET production

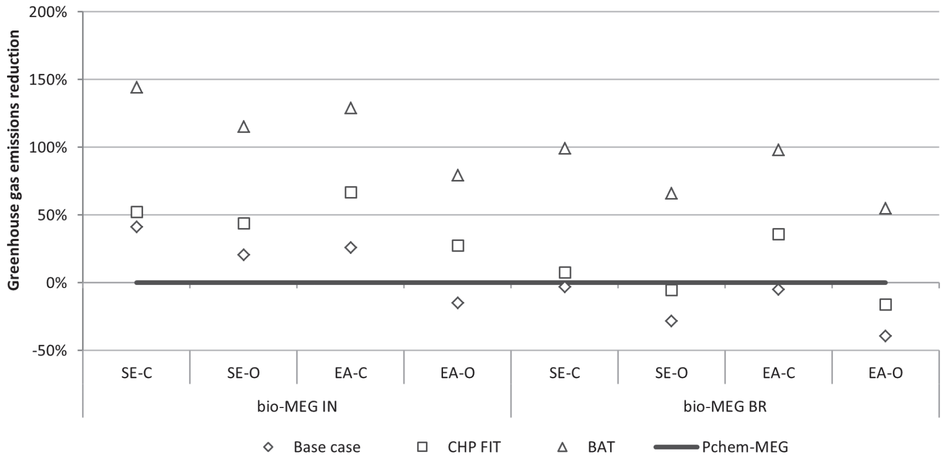

Figure 3.8 Comparison of petrochemical MEG production (0% line) with bio-MEG production from Indian and Brazilian ethanol

tions made to limit the difference between the two studies (e.g. excluding transport, process energy for dehydration met by bagasse), which reduce the difference on GHG emissions to 20%. High difference remains in eutrophication and acidification potential, which is associated with N-fertilisers and returned N-residues to soil and bagasse use in boilers.

The comparison of the base case results for bio-MEG production (excluding transport to Europe) with pchem-MEG (PlasticsEurope, 2012) reveals that bio-MEG_{IN} has better performance on GHG emissions than pchem-MEG by 25-50% under SE-C, SE-O and EA-C (difference of 0.33-0.65 $kgCO_{2eq}/kg_{MEG}$). Only under EA-O bio-MEG_{IN} results in 15% more GHG emissions than pchem-MEG (0.23 $kgCO_{2eq}/kg_{MEG}$). Bio-MEG_{BR} shows worse performance on GHG emissions by 3%-40% under the different allocation approaches. Chen and Patel (2012) estimate that sugarcane-based bio-PET emits 1.0 $kgCO_{2eq}/kg_{PET,BR}$, which is lower by 50% from the results of this study. Main differences are associated with ethanol requirement for ethylene where the authors use stoichiometric yields, energy requirement for dehydration which the authors based on the theoretical heat of formation, and cradle-to-gate emissions of ethanol production which are based on older Brazilian sugarcane ethanol data. In addition, the authors do not take into account that production of bio-MEG takes place in India in order to account coal-based process related emissions and do not include emissions from transoceanic transport.

The results become clearly more favourable when improvement potentials are considered (section 3.4.1): If a CHP plant (with FIT) is implemented then the emission reduction of bio-$MEG_{IN,BR}$ increases by 10%-40%. In this case emissions from bio-MEG_{BR} under the SE approaches are comparable to pchem-MEG. Under EA-C bio-MEG_{BR} shows lower emissions by 35% (0.57 $kgCO_{2eq}/kg_{MEG}$) and under EA-O it shows higher emissions by 15% (0.25 $kgCO_{2eq}/kg_{MEG}$). When assuming implementation of BAT bio-$MEG_{BR,IN}$ have lower emission profile by 50% to 150% across all allocation approaches. Other improvement options discussed in section 3.4.1 also bring benefits on GHG emissions, as well as their potential combinations (e.g. combination of 'Licensor' or 'BAT' with 'CHP'; Figure 3.8). It should be noted that bio-based carbon from bio-MEG is deducted (1.45 $kgCO_2/kg_{MEG}$). In section 3.6, comparison between other impact categories is presented.

3.5 CONCLUSIONS AND RECOMMENDATIONS

The purpose of this chapter was to assess the potential environmental impacts of bio-PE and partially bio-PET production across their production chain, to highlight where environmental pressures may be caused and indicate potentials for improvements. At the same time, results were compared with petrochemical counterparts of the bioplastics to demonstrate savings and trade-offs between impact categories. This study confirms the findings of a review study on bio-based materials which suggests that they may lead to savings of NREU and GHG emissions relative to conventional materials in NREU and GHG emissions but may on the other hand increase impacts associated with application of fertilisers (e.g. eutrophication) and pesticides during biomass cultivation (Weiss et al., 2012).

Compared to pchem-PE, the bio-based route offers significant savings in GHG emissions and NREU. These savings are reduced when iLUC emissions are considered, which does not change the general conclusion that bio-PE production is beneficial with regard to these impact categories.

Bio-PET production as assumed in the base case (without CHP) is in a similar range as pchem-PET production, with respect to GHG emissions and NREU (±10%). For the base case different allocation approaches were found to influence the results significantly. The difference between the highest and the lowest estimation on GHG emissions of different allocations is 9, 14 and 15 percent points in bio-$PET_{BR,IN}$ and bio-PET_{IGL}, respectively. When comparing with GHG emissions of pchem-PET, the difference due to allocation in bio-PET systems may lead to different conclusions. The comparison with other studies revealed that previous estimates on GHG emissions of bio-PET were more optimistic with regard to potential savings taking the base case production into account. In view of comparable profiles with pchem-PET for the base case, uncertainties in the supply chain, but also taking a conservative and precautionary approach by accounting for iLUC emissions, it becomes evident that the production of bio-PET needs further improvements with regard to its environmental performance.

If the on-site CHP plant is assumed within the system boundaries then bio-PET has up to +15% lower GHG emissions than pchem-PET; again, the additional emission reduction by CHP varies depending on the case (BR, IN, IGL) and the allocation method. Bio-PET offers limited savings of GHG emissions when compared with fully bio-based plastics (bio-PE) but this is expected since the biogenic carbon content of bio-PET is much lower (approximately 70% of the polymer's weight is petrochemical PTA, which is the same in bio-PET and pchem-PET; hence comparative savings are limited). A first step could be to switch the primary energy source of steam production in the bio-MEG plant from coal to natural gas. With regard to Indian ethanol production, it is important that the industry moves towards providing surplus electricity to the grid (for those 55% of mills that currently do not yet do so) and that stillage is treated prior to any disposal to soils or water bodies. Strong efforts should be made to implement the latest technology developed for the petrochemical MEG production, which would offer significant improvement potentials for bio-PET (which in some cases were shown to reach approximately 30% compared to pchem-PET). Such performance can be expected by new bio-MEG plants. Other potential improvement options, which however are not assessed in this study, include a combination of the above measures (e.g. 'BAT' with 'CHP') and replacement of coal or natural gas by biomass. Nevertheless, as explained above, the environmental profile is expected to improve in the future.

With respect to HH and EQ, bio-PE and bio-PET were found to have a higher impact by factors or even orders of magnitude compared to pchem-PE and pchem-PET. The largest contribution comes from sugarcane production. However, this result is subject to large uncertainty concerning specific pesticides used. It is therefore advised to establish a chain of custody, which ensures that certain substances such as daconate are not used. Also for bioplastics produced from Brazilian ethanol it is important to ensure supply from areas where no pre-harvesting burning practices are applied or to promote termination of burning practices by other means. Apart from carcinogenic and non-carcinogenic impacts on human health, the normalisation analysis showed that for both bio-PE and bio-PET from Brazilian ethanol, respiratory inorganics caused by bagasse use in co-generation facilities and pre-harvesting burning practices play an important role. It is therefore advised to monitor closely and on a local level the possible pathways of those pollutants and the impacts that they may have to the population and sugarcane field workers surrounding fields and distilleries. Also proper technologies may be installed to reduce particle emissions to the atmosphere. Further work is needed on inventories of sugarcane pesticides but also on impact assessment methodologies related with toxicity. Moreover, further analysis is required for estimating PTA profiles and land use change emissions of Indian sugarcane production. For bio-PET from Indian ethanol, water consumption is very high; given the water stress of the region, this not only reduces available water resources but it is also related with an increase of the impact on HH and EQ. Last, in India the eutrophication can increase significantly if higher emission factors are assumed due to the high P-fertiliser use. It becomes evident that there is untapped potential in the production chains of bio-PE and bio-PET, which if fully exploited can contribute to further improvements in environmental and human health impacts compared to petrochemical plastics.

3.6 APPENDIX

3.6.1 Environmental impacts of bioethylene polymerisation to bio-HDPE (bio-PE)

The environmental impacts (*EI*) of ethylene polymerisation to HDPE (EI_{pol}) are calculated based on the following equation:

$$EI_{pol} = EI_{pchem\text{-}HDPE} - EI_{pchem\text{-}ethylene} \bullet 1.027 \frac{kg\ ethylene}{kg\ HDPE}$$

Equation 3.1

where $EI_{pchem\text{-}ethylene}$ and $EI_{pchem\text{-}HDPE}$ refer to the environmental impacts of the ecoinvent processes “ethylene, average, at plant/RER U” and “Polyethylene, HDPE, granulate, at plant/RER U”, respectively. The impact assessment results were calculated using the impact assessment method Impact 2002+ and the software Simapro v7.3. Then the environmental impacts of polymerisation of bioethylene were added to the environmental impacts of bioethylene.

3.6.2 Environmental impacts of PTA production and polymerisation to bio-PET

In order to calculate the environmental impact of bio-PET we need to include the process steps of PTA production and polymerisation. There are two approaches that we could take. We could either consider inventories as published in ecoinvent v2.2 based on industry data of 2005 (Hischier, 2007) or we could estimate the environmental impacts based on the most recent PlasticsEurope report (PlasticsEurope, 2011). We opt to follow the second approach since the industry association provides the most up to date environmental profile of the pet-

Table 3.5 Environmental impacts of bioethylene polymerisation to HDPE

	Midpoint		Endpoint	
Impact category	**Value**	**Unit**	**Value**	**Unit**
Carcinogens	1.58E-01	kgC_2H_3Cl eq	4.43E-07	DALY
Non-carcinogens	3.58E-03	kgC_2H_3Cl eq	1.00E-08	DALY
Respiratory inorganics	3.08E-04	kg$PM_{2.5}$ eq	2.16E-07	DALY
Ionizing radiation	2.84E-02	BqC-14 eq	5.96E-12	DALY
Ozone layer depletion	2.84E-09	kgFC-11 eq	2.98E-12	DALY
Respiratory organics	1.31E-03	kgC_2H_4 eq	2.80E-09	DALY
Aquatic ecotoxicity	6.22E+00	kgTEG water	3.12E-04	PDF·m^2·yr
Terrestrial ecotoxicity	2.74E-02	kgTEG soil	2.17E-04	PDF·m^2·yr
Terrestrial acid/nutri	6.75E-03	kgSO_2 eq	7.02E-03	PDF·m^2·yr
Land occupation	1.11E-06	m^2org.arable	1.21E-06	PDF·m^2·yr
Aquatic acidification	2.45E-03	kgSO_2 eq	2.16E-05	PDF·m^2·yr
Aquatic eutrophication	3.19E-06	kgPO_4 P-lim	3.64E-05	PDF·m^2·yr
Mineral extraction	2.84E-06	MJ surplus		
Global warming 100 GWP	5.04E-01	kgCO_2 eq		
Non-renewable energy	8.10E+00	MJ primary		

rochemical route. Also these saving are attributed to improved PTA production.[23] Based on personal communication with PlasticsEurope the cradle-to-gate greenhouse gas emissions of PTA are 1.3-1.35 kgCO_{2eq}/kg_{PTA} and the non-renewable energy use is 50-55 MJ/kg_{PTA}. Considering that 0.86 kg_{PTA}/kg_{PET} and 0.32 kg_{EG}/kg_{PET} are required we can also estimate the GHG and NREU of the polymerisation step. This approach was also followed in Chen and Patel (2012). Note that such a decoupling was not possible for the other impact categories as per Impact 2002+. Thus for all other impact categories we use the PTA and polymerisation inventories as reported in ecoinvent v2.2.

3.6.3 Environmental impacts of transport

The transport distances that are assumed in this study are:

- transport of bio-PE from Brazil to Europe
  - polymerisation (plant) to Rio Grande (port): 1,407 km (assumption)
  - Rio Grande (port) to Rotterdam (port): 10,989 km (Searates, 2012);
- transport of bio-MEG from India to Europe
  - Bio-MEG production (plant) to Mumbai (port): 1,407 km

[23] When estimating the bio-PET environmental profile with recent PTA data the comparison should be made with the recent pchem-PET as reported by PlasticsEurope (this is the choice we made). If we would have opted to use older PTA data then the comparison should have been made with old pchem-PET.

  - Mumbai (port) to Rotterdam (port): 11,660 km (Searates, 2012);
- transport of ethanol from Brazil to India
  - Ethanol production (plant) to Rio Grande (port): 1,407 km (assumption)
  - Rio Grande (port) to Mumbai (port): 14,897 km (Searates, 2012)
  - Mumbai (port) to Bio-MEG production (plant): 1,407 km;
- transport of ethanol and/or molasses to bio-MEG plant in India
  - Ethanol production (plant) to bio-MEG (plant): 180 km (applicable to bio-MEG_{IN})
  - Sugar mills (plant) to bio-MEG (plant): 150 km (applicable to bio-MEG_{IGL}).

For international transport we use the process "Transport, transoceanic freight ship". For road transport we use the process "Transport, lorry>16t, fleet average/RER". For inland transport in India of Indian ethanol and molasses we use the process "Transport, lorry 7.5-16t, EURO3/RER". In India, for transport of materials to the field or to the plants we use the process "Transport, lorry 3.5-7.5t, EURO3/RER". All processes are from ecoinvent v2.2. Note that for international transport we assume losses of 1%. We calculate the environmental impacts of transport based on Impact 2002+ and the software Simapro v7.3.

3.6.4 Normalisation results for bio-PE and bio-PET

The damage factors reported in ecoinvent v2.2 are normalised by dividing the impact per unit of emission by the total impact of all substances of the specific category for which characterisation factors exist, per person per year (for Europe). The normalisation factors used in this study are:

Table 3.6 Normalisation factors used in this study (Jolliet et al., 2003)

Area of protection	Normalisation factor
Human health	1.41E+02
Ecosystem quality	7.30E-05
Climate change, CO_2	1.01E-04
Resources	6.58E-06

Note that Figure 3.9 to Figure 3.15 show the normalised results based on the SE-C approach. The normalised results for the SE-O and EA approach follow a similar pattern.

3.6.5 Impacts of water use on Human Health and Ecosystem Quality

Based on Pfister et al. (2009), the region of Uttar Pradesh, India has a water stress index (WSI) equal to 1. This entails that all the consumptive use of water is accounted as the mid-point indicator water deprivation.

Based on our analysis, process water consumption is equal to 0.2 m^3 H_2O/kg bio-PET_{IN}. By multiplying the process water consumption and the WSI we obtain that water deprivation is 0.2 m^3/kg bio-PET_{IN}. Based on Pfister et al. (2009), the regional impact factors for human health (HH) and ecosystem quality (EQ) per m^3 of water deprived are $2.898 \cdot 10^{-6}$ DALY/m^3 H_2O and 0.502 PDF/m^2·yr/m^3 H_2O.

By multiplying the water deprivation indicator with each of the impact factors we obtain that the impact of water consumption of bio-PET$_{IN}$ on HH and EQ is $0.57 \cdot 10^{-6}$ DALY and 0.1 PDF/m^2·yr per kilogramme of bio-PET$_{IN}$. In a similar manner we add the impacts of PTA production and polymerisation, assuming regional factors for the Netherlands. Then we aggregate the results with the results for HH and EQ that were calculated for bio-PET$_{IN}$ without impacts of water use (shown in Figure 3.5 - Figure 3.6) and we estimate that the final impact (including water use) is increased by 3% and 9%, for HH and EQ, respectively. If we calculate the impact assessment results based on the Ecoindicator 99 method, then the increase is 11% and 13% on human health and ecosystem quality, respectively.

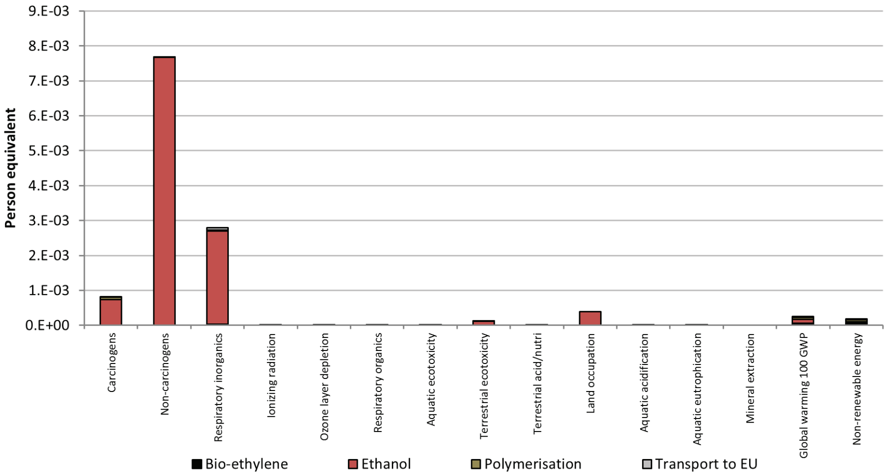

Figure 3.9 Normalised results for 1 kilogramme of bio-PE production. Results on GHG emissions do not subtract bio-based carbon

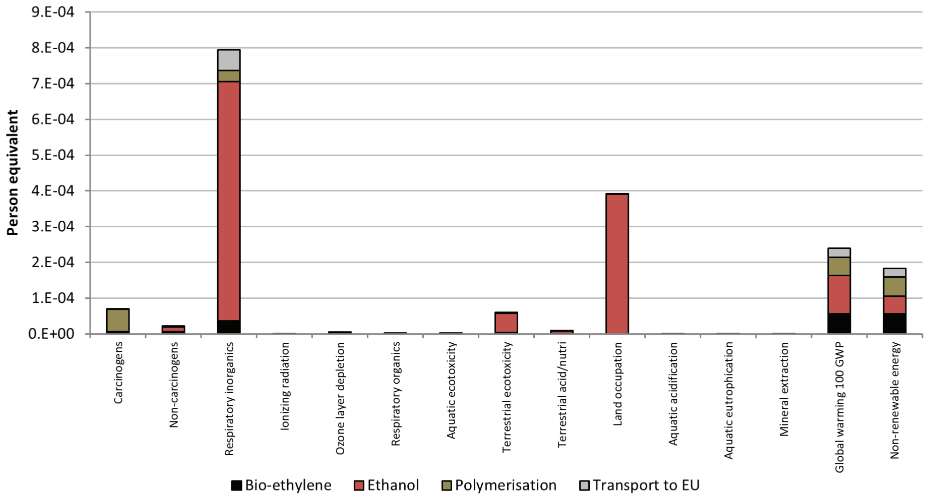

Figure 3.10 Normalised results for 1 kilogramme of bio-PE production excluding pre-harvesting burning practices and daconate application in sugarcane production in Brazil. Results on GHG emissions do not subtract bio-based carbon

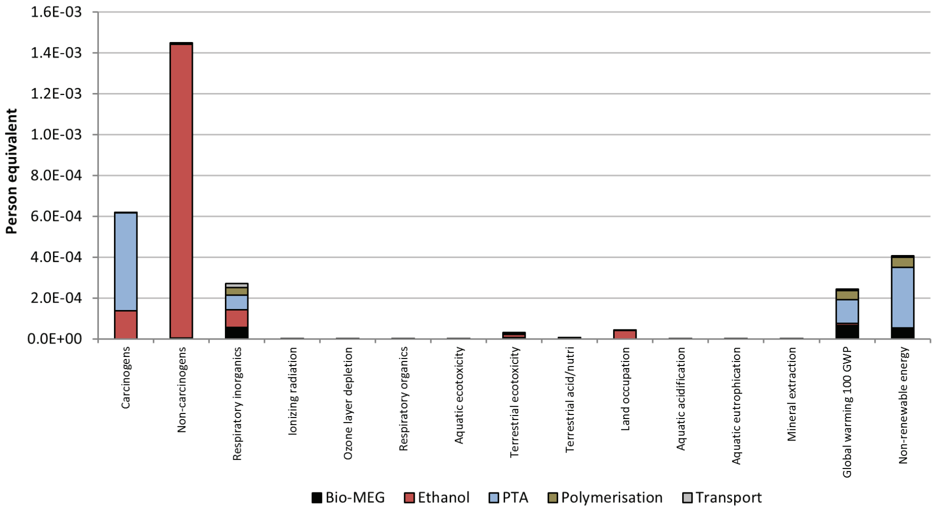

Figure 3.11 Normalised results for 1 kilogramme of bio-PET$_{IN}$ production. Results on GHG emissions do not subtract bio-based carbon

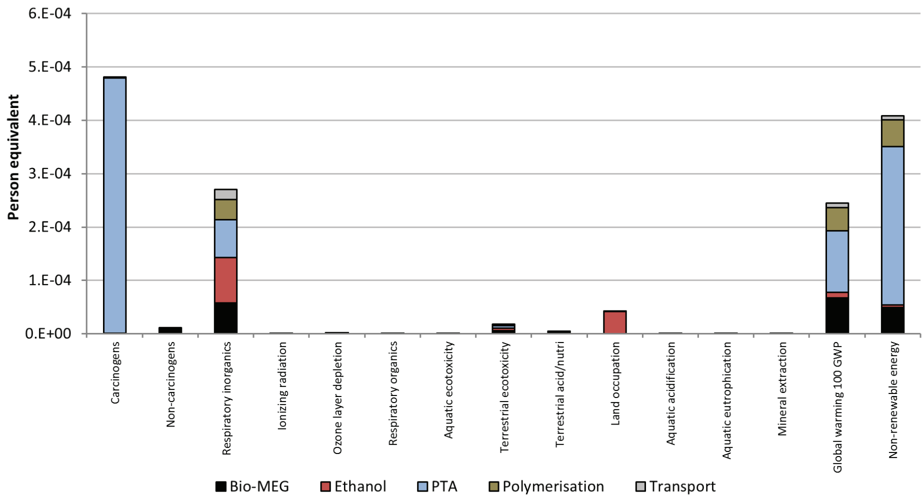

Figure 3.12 Normalised results for 1 kilogramme of bio-PET$_{IN}$ production excluding daconate application in Indian sugarcane production. Results on GHG emissions do not subtract bio-based carbon

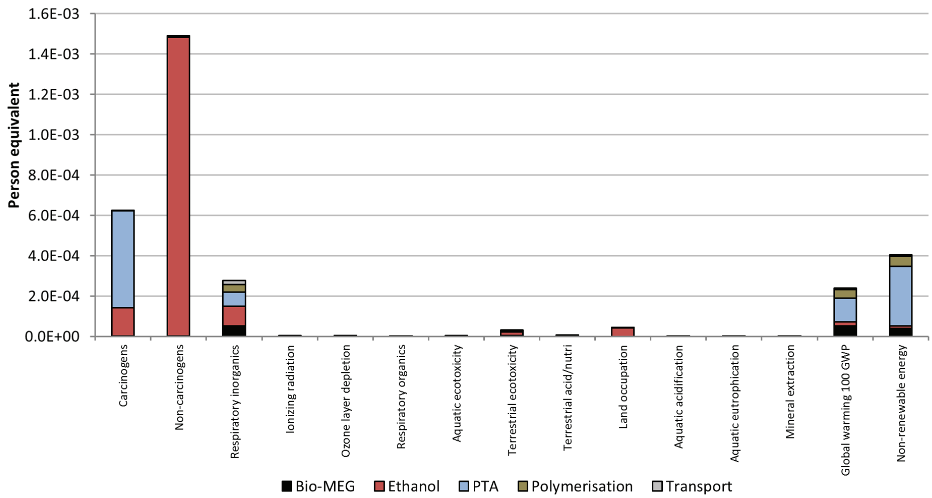

Figure 3.13 Normalised results for 1 kilogramme of bio-PET$_{IGL}$ production. Results on GHG emissions do not subtract bio-based carbon

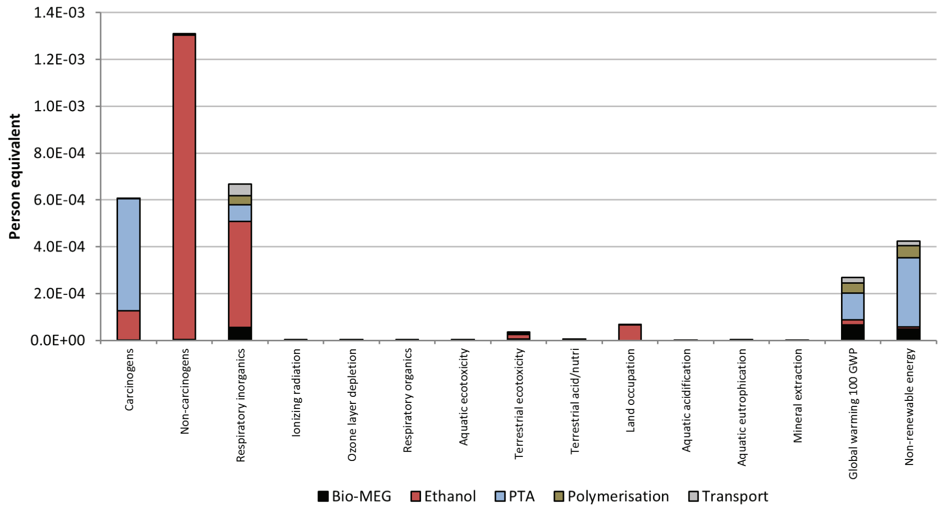

Figure 3.14 Normalised results for 1 kilogramme of bio-PET$_{BR}$ production. Results on GHG emissions do not subtract bio-based carbon

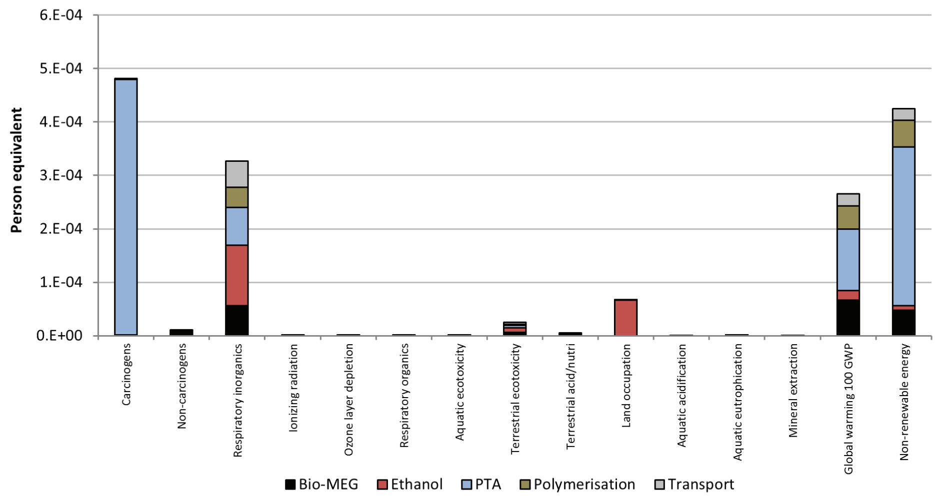

Figure 3.15 Normalised results for 1 kilogramme of bio-PET$_{BR}$ production excluding pre-harvesting burning practices and daconate application in sugarcane production in Brazil. Results on GHG emissions do not subtract bio-based carbon

3.6.6 Comparison with latest data of petrochemical MEG production in Europe

In 2011, PlasticsEurope released the latest environmental profiles (eco-profiles) for petrochemical PET production in Europe (PlasticsEurope, 2011). Later in 2012, they also released environmental profiles of MEG (PlasticsEurope, 2012). However, since data on PTA production and polymerisation to PET have not yet become available, the data on MEG and PET cannot be compared for consistency. For this reason, we compared the results of bio-PET with the latest results for petrochemical PET, thus not taking into account the latest report on petrochemical MEG production. We performed a separate analysis for GHG emissions of bio-MEG with the latest PlasticsEurope pchem-MEG in section 3.4.3. In order to provide a comparison based on the recent pchem-MEG eco-profiles, in the figures that follow (Figure 3.16 - Figure 3.18) we compare the GHG emissions, NREU and cumulative energy demand of petrochemical and bio-based MEG production from Indian and Brazilian ethanol for three cases: (a) base case without CHP plant, (b) base case production with CHP plant and a feed-in-tariff system and (c) for BAT technology. In Figure 3.19 we present results for other impact categories but only for the EA-O case (results are similar for all other allocation approaches). Note that for each impact category we used the impact assessment methods that petrochemical MEG was assessed in the PlasticsEurope report.

3.6.7 Comparison with other studies on bioplastics from sugarcane ethanol

The comparison of the results on GHG emissions for bio-HDPE (i.e. bio-PE in this chapter) with those reported for bio-LDPE (Liptow and Tillman, 2012) reveals differences.[24] The results of this study are approximately 90% higher. Approximately 50% of the difference is attributed to different emissions in sugarcane cultivation (Figure 3.18). Part of the difference on these emissions can be explained due to the absence of mechanised harvesting in the attributional approach of Liptow and Tillman (2012). Despite that the authors use older data on Brazilian sugarcane ethanol production, which can partly explain the difference it appears that fossil CO_2 emissions from urea and lime application, but also N_2O and CH_4 emissions from unburned trash have not been accounted for. With respect to the other process steps, there are differences noticed in the range of 15 to 20% (Figure 3.18). To some extent, these differences can be explained due to different data sources used, different methods and so forth.

When comparing the results of this study with the one prepared by Hunter et al. (2008), it should be kept in mind that the two studies differ significantly in their scope. For example, Hunter et al. (2008) assess an integrated cane-to-PE process, the energy requirements of which are exclusively met by bagasse (Hunter et al., 2012). In addition, they assess a best-in-class ethanol production process. Therefore apart from methodological differences, the different scope is critical for an aligned comparison of the results. Although there is lack of

[24] Note that the comparison is made on the attributional approach of Liptow and Tillman (2012) excluding land use change emissions with the results for EA without deducting the bio-based carbon from bio-PE of this study. Among all the approaches taken in the two studies, the selected comparison is regarded as the one with the least methodological differences and the most aligned scope.

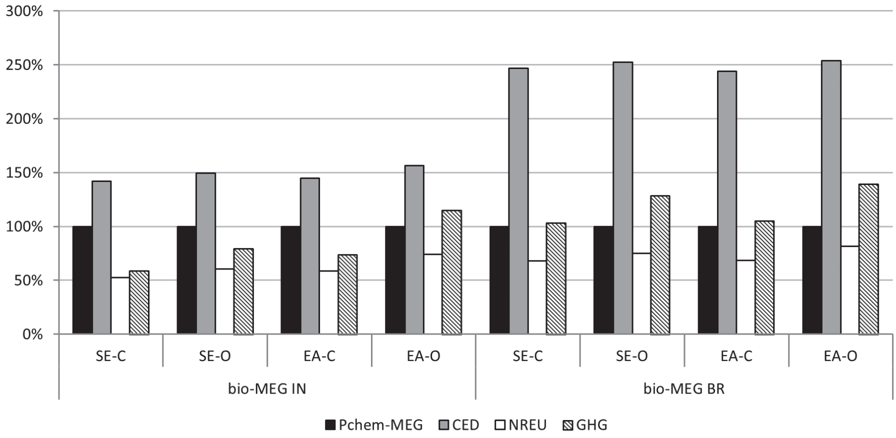

Figure 3.16 Cradle-to-gate results for cumulative energy demand (CED), non-renewable energy use (NREU) and greenhouse gas emissions (GHG) of base case bio-MEG production from Indian (bio-MEG_{IN}) and Brazilian (bio-MEG_{BR}) ethanol in comparison with petrochemical MEG production in Europe. The bio-based carbon content is deducted from the GHG results. The gate of bio-MEG production is in India

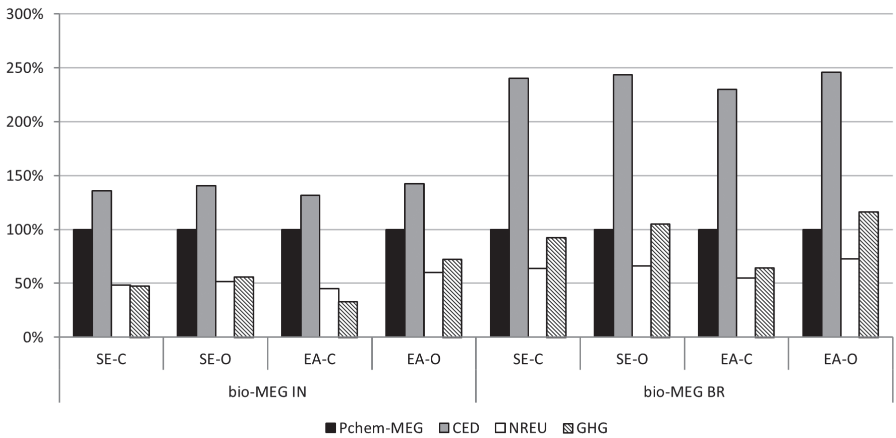

Figure 3.17 Cradle-to-gate results for cumulative energy demand (CED), non-renewable energy use (NREU) and greenhouse gas emissions (GHG) of bio-MEG production from Indian (bio-MEG_{IN}) and Brazilian (bio-MEG_{BR}) ethanol assuming a CHP plant and a feed in tariff system, in comparison with petrochemical MEG production in Europe. The bio-based carbon content is deducted from the GHG results. The gate of bio-MEG production is in India

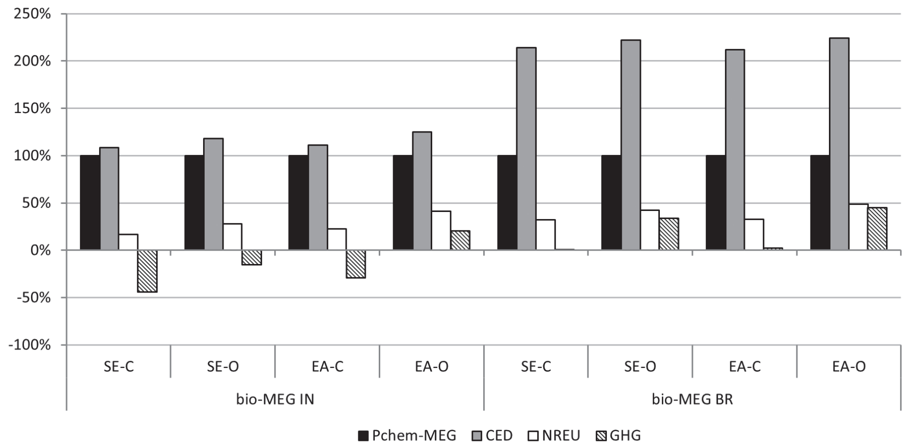

Figure 3.18 Cradle-to-gate results for cumulative energy demand (CED), non-renewable energy use (NREU) and greenhouse gas emissions (GHG) of bio-MEG production from Indian (bio-MEG_{IN}) and Brazilian (bio-MEG_{BR}) ethanol assuming BAT technology, in comparison with petrochemical MEG production in Europe. The bio-based carbon content is deducted from the GHG results. The gate of bio-MEG production is in India

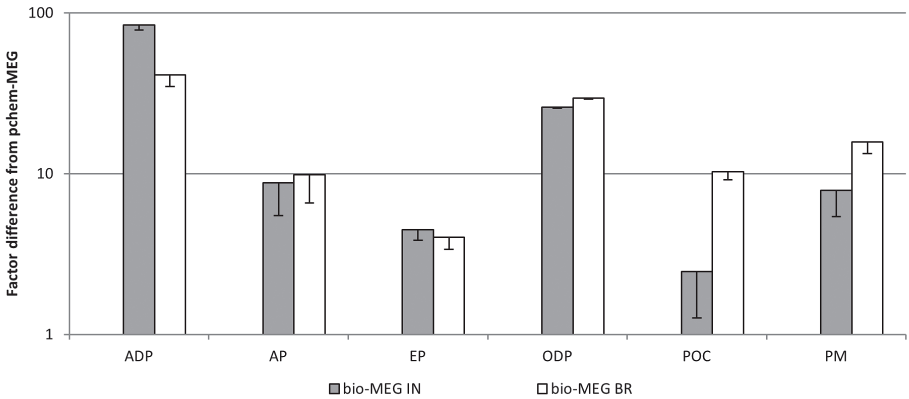

Figure 3.19 Cradle-to-gate results for abiotic depletion (ADP), acidification (AP), eutrophication (EP), ozone depletion (ODP), photochemical ozone creation (POC) and dust and particulate matter (PM) of bio-MEG production from Indian (bio-MEG_{IN}) and Brazilian (bio-MEG_{BR}) ethanol in comparison with petrochemical MEG production in Europe. The gate for bio-MEG production is in India. A lower estimate is also presented assuming the adoption of best available technology (BAT)

background information on the Hunter et al. (2008) study, we attempt a comparison by a set of assumptions that tries to limit the difference in scope and product-systems. Therefore the system of this study is adapted based on the following:

- transoceanic and intermediate transport of ethanol to the ethylene plant is excluded;
- process energy requirements are exclusively met by bagasse, assuming that sufficient bagasse is available;
- no pre-harvesting burning practices are applied;
- impact assessment method is CML 2001.

We then plot in Figure 3.19, the relative difference of bio-LLDPE and pchem-LLDPE as presented in Hunter et al. (2008) and the relative difference of bio-HDPE and pchem-HDPE as calculated in this study based on the adaptations described above (Figure 3.20).

It can be concluded that the results for Cumulative Energy Demand (CED), Cumulative Fossil Energy Demand (CFED) and Global Warming Potential (GWP) are similar for both studies. Any differences noticed could be due to process yields, process energy requirements for ethanol-to-ethylene, lower field-to-ethanol plant distances, allocation procedures, background data, process energy requirements of polymerisation that are also met by bagasse, etc. The results on Acidification Potential (AP) reveal the reverse conclusion and the relative difference on Eutrophication Potential (EP) of this study is a factor 3 higher from the relative difference of the Hunter et al. (2008) study. The inventory inputs that contribute most to these impact categories are N-fertilisers and returned N-residues to soil, and bagasse use in boilers, which are likely to explain the difference.

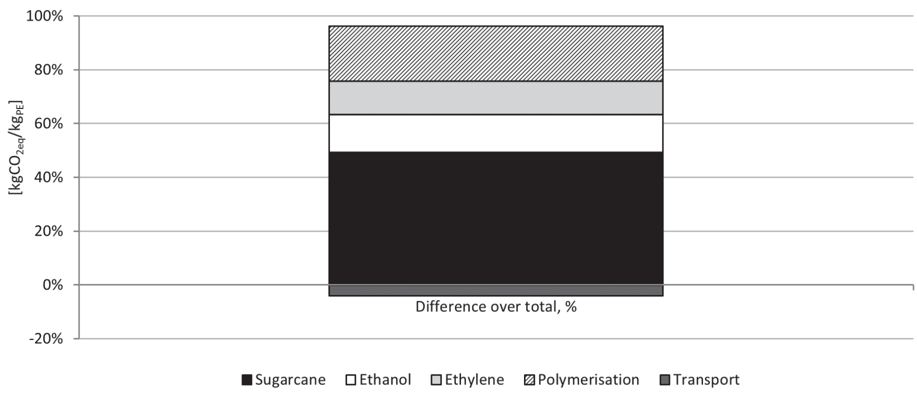

Figure 3.20 Percent differences per process step, between this study and the results of Liptow and Tillman (2012)

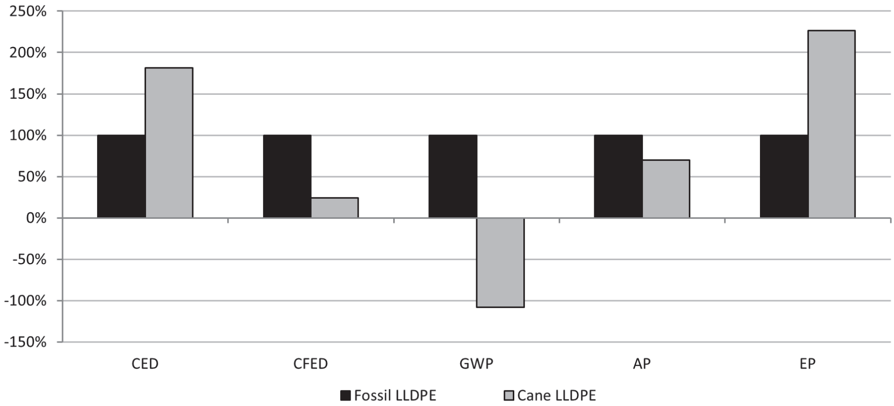

Figure 3.21 Relative difference of fossil and bio-LLDPE based on ecoinvent v2.2 results and Hunter et al. (2008), respectively. Results are presented for CED (Cumulative Energy Demand), CFED (Cumulative Fossil Energy Demand), GWP (Global Warming Potential), AP (Acidification Potential) and EP (Eutrophication Potential)

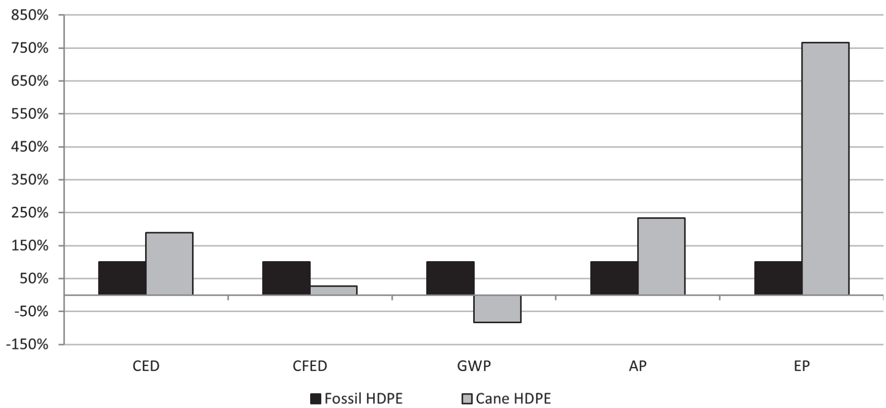

Figure 3.22 Relative difference of fossil and bio-HDPE based on ecoinvent v2.2 results and this study (adapted), respectively. Results are presented for CED (Cumulative Energy Demand), CFED (Cumulative Fossil Energy Demand), GWP (Global Warming Potential), AP (Acidification Potential) and EP (Eutrophication Potential)

4

Emerging bioeconomy sectors in energy systems modelling – Integrated systems analysis of electricity, heat, road transport, aviation and chemicals: a case study for the Netherlands

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Submitted

ABSTRACT

Many studies have addressed the future role of biomass in reducing fossil energy use and greenhouse gas emissions. So far, the focus has been on electricity, heat and road transport, although sectors that have few alternatives to biomass (such as aviation and the chemical industry) are expected to become increasingly important. We have extended an existing bottom-up energy system model with fossil-based chemicals and biochemicals as well as with renewable jet fuels to assess the deployment of biomass conversion technologies in the Netherlands until 2030. The model comprises detailed cost-structures and mid-term developments for the fossil energy system and mitigation options. It includes detailed cost-supply curves for biomass, renewable energy conversion technologies and carbon capture and storage. The framework incorporates multi-output processes, such as biorefineries, to address cross-sectoral synergies. To capture the uncertainty in technical progress, different technology development scenarios are used to assess cost-optimal biomass utilisation pathways over time. Slow technical progress leads to biomass applications for heating, 1st generation biofuels from hydrotreated oils and biochemicals based on 1st generation fermentation systems. Enhanced technology development allows the production of 2nd generation biofuels from solid biomass. Furthermore, large volumes of diverse biochemicals are produced (1.9 Mt) as well as 13 PJ renewable jet fuel. The required biomass will range from 230 to 300 PJ in 2030, supplied primarily from non-domestic resources. Under existing policies, CO_2 emissions will only gradually be reduced to reach 1990 levels (140-145 MtCO_2). Further scenario analysis is recommended to assess model sensitivity and the necessary preconditions for future biomass conversion pathways and robust directions towards the required greenhouse gas mitigation pathways.

4.1 INTRODUCTION

The role of biomass in the global energy mix is frequently highlighted for its potential to mitigate greenhouse gas (GHG) emissions and secure energy supply (Chum et al., 2011; Deng et al., 2012). In contrast to other renewable sources, biomass can substitute fossil fuels in sectors for which there are no or few other alternatives (e.g. shipping, aviation, feedstock for chemicals). In the European Union (EU), biomass is the largest source of renewable energy and it is expected to remain so, increasing from 4 EJ in 2010 to 5.8 EJ in 2020, providing almost 55% of the EU renewable energy target (AEBIOM, 2014; EC, 2009b; Stralen et al., 2013). A similar growth can also be expected post-2020, based on the European Commission's proposal to reduce GHG emissions in the EU by 40% in 2030 compared to 1990, and the Commission's anticipation that in 2030 27% of final energy consumption will be covered by renewable energy.

The role and contribution of biomass is ambiguous in view of the EU's intention to strategically expand its bioeconomy sectors (EC, 2014a, 2011a).[25] The development of bioeconomy sectors, especially those that deliver raw materials (biomass feedstocks), energy and industrial products, is seen as key to meeting societal challenges (EC, 2012a). However, there is little understanding of how biomass will be distributed across the bioeconomy sectors or where it will be used most cost-effectively. This impedes the design of improved biomass policies that promote its optimal deployment. One of the most notable omissions has been the use of biomass for biochemicals. The potential contribution of selected biochemicals to future GHG emission reduction ranges from 2 to 246 $MtCO_2$/yr per biochemical, assuming the complete replacement of the fossil counterpart (de Jong et al., 2012a). However, despite their high added value, bio chemicals have so far been excluded from policy frameworks and their diffusion in the market is limited (Carus et al., 2014; Daioglou et al., 2014; Dornburg et al., 2008; Posen et al., 2014). Furthermore, the EU has the ambition to consume 2 Mt of renewable jet fuel (RJF) by 2020 (EC, 2011b). However, their uptake has been negligible to date, primarily due to limited production capacity and high production costs (Aviation Transport Action Group, 2012; de Jong et al., 2015).

An integrated systems perspective is required to overcome the knowledge gap on synergies and trade-offs of fossil energy reduction and GHG mitigation options beyond 2020, both in energy sectors (electricity, heat, transport fuels) and in non-energy applications (e.g. feedstock for biochemicals). Several models have been used to provide such a perspective to support informed decision-making. A MARKAL (MARket ALlocation) model focusing on the power sector was used to assess the impact of international climate policies on carbon capture and storage (CCS) in the Netherlands (van den Broek et al., 2011). The same model was used to assess the potential deployment of hybrid vehicles and of synthetic fuels with electricity generation and CCS, thus expanding the model to the road transport sector (van Vliet et al., 2011). Other studies analysed biomass deployment in the electricity, heat

[25] According to EC (2012a), "the bioeconomy includes the sectors of agriculture, forestry, fisheries, food and pulp and paper production, as well as parts of chemical, biotechnological and energy industries". The present study assesses the energy, biotechnological and chemical sectors.

and transport sectors in the context of the EU's National Renewable Energy Action Plans (Stralen et al., 2013). However, none of these studies assessed biomass use for chemicals or RJF, even though these products had been analysed in dedicated studies (e.g. Eerhart et al. (2012), Cok et al. (2014), Chapter 3). One study used a MARKAL framework to assess competitive uses of biomass for energy and materials in Europe (Gielen et al., 2001). However, this study was conducted in 2001 and new insights are required that take new policies and technology developments into account. More recently, potential non-energy uses of biomass have been assessed from a global systems perspective (Daioglou et al., 2014). Nevertheless, interactions with the rest of the energy system have been ignored and a large temporal and geographical scope has been applied, which is not suitable to capture short-term developments. Other studies have used linear programming to model system interactions, but they included only a limited number of products (Meesters, 2006). Large-scale production of bioenergy and biochemicals in the Netherlands has been assessed; however, parallel developments in the deployment of fossil and renewable energy technologies were ignored (Hoefnagels et al., 2013). Moreover, a critical aspect has not been sufficiently assessed: the multi-output production from biorefineries that can supply different sectors. The above omissions indicate the need for a comprehensive framework that considers a timeframe to 2030 and that includes the required sector-specific and region-specific details to assess the optimal uses of biomass across all competing sectors.

The main goal of the present study is to design a modelling framework that accounts for competitive and synergetic uses of biomass for energy and non-energy applications, including the emerging sectors of the bioeconomy. The framework should be able to capture future uncertainty regarding the technical progress of advanced conversion technologies. This uncertainty influences production costs and thus the competition of alternative technology options. Furthermore, the study should address how the competitiveness of different biomass value chains is affected by the development of competitive renewable energy technologies. Biomass cost-supply curves can be a major determinant for optimisation. The emphasis of this article is on the method applied. To demonstrate the modelling framework, it is applied in a context with sufficient regional and temporal detail. The Netherlands has been selected on the premise that it can support large-scale bioeconomy development. The country has one of Europe's most efficient and advanced agricultural sectors, which is nevertheless limited by the domestic supply of biomass and land availability; therefore, it relies heavily on trade. Moreover, the Netherlands has developed a competitive logistics infrastructure over the years and it holds a strong global position in the production of chemicals. Between 2025 and 2030, the gradual depletion of natural gas reserves will change the country from a net gas exporter to a net gas importer; therefore, the transition to a more resource-secure and sustainable energy system is required (ECN, 2014).

4.2 METHODS

4.2.1 Model description

We employed MARKAL, a bottom-up, technology-rich and technology-explicit model, which uses linear optimisation techniques to calculate an intertemporal partial equilibrium on energy and non-energy markets. It generates a least-cost pathway for the total system by minimising the system's present value to deliver demand services: electricity (PJ_e), heat (PJ_{th}), vehicle-kilometres, jet fuel (PJ) and chemicals (Mt) (Loulou et al., 2004). Supply and demand conversion technologies are represented by detailed cost-structures and process efficiencies. They are deployed in 5-year intervals within scenario constraints such as feedstock supply at a specific cost-price, the maximum total capacity of a specific technology and the minimum supply of a specific fuel, e.g. biofuels (Figure 4.1).

This study builds on an existing model for the Netherlands (MARKAL-NL-UU; van den Broek et al. (2011), 2008, van Vliet et al. (2011)). To assess the energy and chemical industry markets, we extended the model to include jet fuel production and the Dutch chemical industry, thus taking into account energy use as feedstock (otherwise referred to as non-energy use). Furthermore, we expanded the model's existing technology portfolio of electricity, heat and fuel production technologies. The extended version presented in this chapter covers a substantial number of biomass conversion technologies. Table 4.1 through Table 4.3 give an overview of all the technologies included in the extended model. The model extension is described in section 4.2.2. An explicit characterisation of all technologies is included in section 4.7.

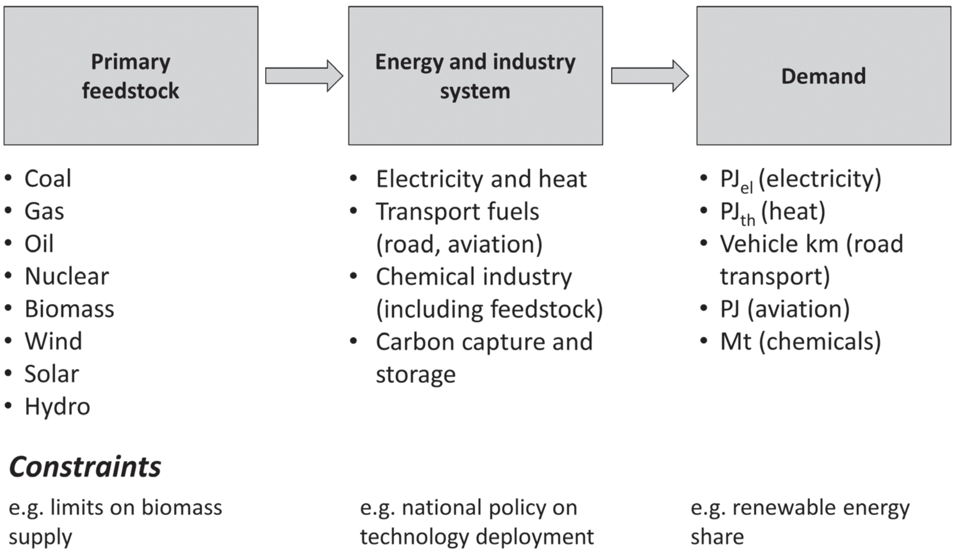

Figure 4.1 Illustration of the key modules of the MARKAL-NL-UU modelling framework: supply, conversion and demand

Table 4.1 Overview of electricity and heat technologies included in MARKAL-NL-UU[a]

Fossil conversion	Biomass conversion	Other renewable conversion
Electricity[a,b]		
Combined cycle power plant on natural gas	Steam cycle power plant on biomass	On-shore wind turbine
Gas turbine power plant on natural Gas	Pulverised coal power plant on biomass co-firing with coal	Off-shore wind turbine
Pulverised coal power plant	Integrated gasification combined cycle power plant on biomass co-gasification with coal	Solar photovoltaic panel
Integrated gasification combined cycle power plant on coal	Gas engines on biogas	Hydroelectric power plant
Nuclear power plant		
Electricity and co-generated heat[a,c]		
Small-scale gas engine combined heat and power (CHP) on natural/ landfill gas	Steam cycle CHP on biomass	
Combined cycle CHP on natural gas	CHP[d,e] on biogas	
Gas turbine CHP on natural/landfill Gas	Integrated gasification combined cycle CHP on biomass co-gasification	
Steam turbine CHP on natural gas	MSWI – organic waste fraction[f]	
MSWI – fossil waste fraction[f]		
Heat[d]		
Natural gas boiler[g]	Industrial biomass boiler	
	Wood stove (fuelwood for space heating)	
Electricity/heat other		
	Grid injection of green gas[c,h]	

[a]Production costs of electricity and co-generation technologies have been updated by Brouwer et al. (2015). [b]All large-scale electricity production technologies can be coupled with CCS. Exceptions are dedicated biomass steam cycle plants and municipal solid waste incinerator (MSWI). [c]Electricity and/or heat is also co-produced by CHP units of transport fuel or chemical conversion technologies. These are not included in this table. [d]Added in the present study. [e]Upgrade of biogas from anaerobic digestion of liquid manure to green gas is also included. Green gas is assumed to substitute natural gas only for heat applications. Synthetic natural gas from biomass gasification can be another direct natural gas substitute; however, it has been excluded from the analysis [f]MSWI (fossil, organic fraction) are aggregated to a single technology. In this table, the fossil and organic fractions of municipal solid waste are referred to separately for categorisation purposes. [g]Natural gas-based boilers are implicitly included (i.e. without incorporating detailed cost-structures) by assuming a process efficiency of 90%, which is representative for industrial heat generation in member-countries of the Organisation for Economic Co-operation and Development (Saygin et al., 2014). This is a simplification, as efficiencies may vary per sector (within industry or across other end-users such as households) or fuel type. Furthermore, input fuels may vary; however, for the Netherlands, natural gas is the main energy carrier for heat. [h]Green gas is injected into the natural gas grid and can substitute natural gas applications such as electricity, as listed in this table.

Table 4.2 Overview of road and jet fuel production technologies included in MARKAL-NL-UU

Fossil conversion[a]	**Biomass conversion**	**Other renewable conversion**
Road transport		
Refining of crude oil to petrol	Fermentation of sugar to 1st generation ethanol[b]	
Refining of crude oil to diesel	Pretreatment of biomass followed by extraction of lignocellulosic sugar and fermentation to 2nd generation ethanol[b]	
Reforming of natural gas to Hydrogen	Esterification of vegetable and/or used cooking oil to biodiesel[b]	
	Gasification of biomass to syngas followed by Fischer-Tropsch (FT) synthesis to FT-fuels (diesel, petrol and jet fuel)[b]	
	Gasification of biomass followed by methanol synthesis	Supply of renewable electricity (Table 4.1) to electric vehicles
	Gasification of biomass followed by dimethyl ether synthesis	
	Gasification of biomass followed by hydrogen synthesis	
	Supply of bioelectricity (Table 4.1) to electric vehicles	
	Pyrolysis of biomass to bio-oil followed by hydrotreatment to petrol[b]	
	Co-production of petrol from methanol-to-olefins synthesis[b]	
Aviation[b]		
Refining of crude oil to kerosene	Hydrotreatment of used cooking oil to renewable diesel (HRD)[c,d]	
	Hydroprocessing of used cooking oil to renewable diesel (Hydroprocessed Esters and Fatty Acids; HEFA)[c,d]	
	Hydrothermal liquefaction of biomass to renewable diesel[c]	
	Catalytic pyrolysis of biomass to diesel[c]	

[a]Coal-gasification and FT synthesis to FT-fuels (petrol, diesel, jet fuel) have been excluded from the present study. [b]Added in the present study. [c]Biomass conversion technologies for aviation also applicable to road transport. [d]Renewable diesel supplied to road transport may also use vegetable oil.

4.2.2 Model extension with chemicals, aviation fuels and other biomass technologies

4.2.2.1 Structure of the chemical industry and model representation

The chemical industry converts fossil *feedstocks* such as light oil naphtha (henceforth referred to as naphtha), liquefied petroleum gas (LPG), natural gas liquids and heavy gas oil to key organic *basic chemicals* such as olefins (ethylene, propylene, butadiene) and aromatics (benzene, toluene, xylene). Natural gas is used as a fossil feedstock to produce hydrogen, mainly for methanol and ammonia synthesis. In the Netherlands, steam cracking of naphtha is the most important production process in terms of production volume and energy use (Neelis et al., 2005; Saygin et al., 2011).[26] In 2012, industrial final energy use in the Netherlands was 1,214 PJ (energy and non-energy use) of which 672 PJ was feedstock (CBS, 2014). Fossil feedstock use in organic basic chemical and fertiliser production accounted for 80% of the Dutch non-energy use (Table 4.8, section 4.7). Since 2000, non-energy use has increased at a compound annual growth rate (CARG) of 1.3%. In 2012, the main feedstocks used for the production of basic chemicals were naphtha (330 PJ), LPG (36 PJ) and natural gas liquids (86 PJ). During the same year, natural gas consumption for nitrogen fertiliser production reached 64 PJ (CBS, 2014). Figure 4.2 shows how these sectors are captured in MARKAL-NL-UU. Basic chemicals are converted further to key commodity chemical *intermediates*, which are synthesised to a range of *final products* such as polymers, rubbers, adhesives, solvents and fertilisers.

The products and (interlinked) processes of the chemical industry are complex and numerous. In MARKAL-NL-UU we used a simplified representation based on the description of Petrochemicals Europe (Petrochemicals Europe, 2014), which describes basic chemicals as building blocks, intermediates as derivatives and final products as everyday products (Petrochemicals Europe, 2014). Such a distinction has also been used in other studies that apply a systems perspective (Meesters, 2006). Studies with a different scope include different products in these categories (e.g. Neelis et al. (2005) describes 22 different products as basic chemicals). We included biomass conversion technologies that produce a direct naphtha substitute as feedstock for the chemical industry and biochemicals that are identical or functionally equivalent to fossil-based chemicals. Thus, bio-based alternatives can be provided at the following four levels: feedstock, basic chemical, intermediate and final product (Figure 4.2; for an expanded flowchart see section 4.7 (Figure 4.19).

Selection of chemicals

The focus of this study is on chemicals from naphtha and natural gas due to their significant contribution to the Dutch non-energy use (section 4.2.2). We select downstream fossil-based chemicals from naphtha (i.e. basic and intermediate chemicals as well as final products) based on volume according to historic consumption of basic chemicals by

[26] Fluid catalytic cracking (FCC) and catalytic reforming are refinery processes that produce propylene from gas oil and aromatics from naphtha-based reformate, respectively. In this study, refinery processes are not modelled explicitly, however; FCC propylene and catalytic reforming aromatics are assumed as import commodities.

derivative in Western Europe (as defined in Petrochemicals Europe (2014)) and production capacity in the Netherlands (Petrochemicals Europe, 2014). Large-volume chemical products (Table 4.9, section 4.7) are responsible for 80% of the global chemical industry's energy demand and for 75% of its CO_2 emissions (IEA, 2013). There are two reasons why chemicals with higher-added value (e.g. specialty or fine chemicals) are not included as downstream intermediate or final products. Firstly, their resource substitution potential is relatively small (due to low production volumes) and secondly, there are significant data gaps for existing and future production technologies of these chemicals. As downstream chemical from natural gas, we selected hydrogen.

Besides basic chemicals, those that are modelled explicitly in MARKAL are the intermediates ethylbenzene (EB), ethylene oxide (EO), propylene oxide (PO), terephthalic acid (PTA) and ammonia and the final products polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP) and urea. Intermediate or final chemicals from butadiene are not modelled as most of these are used for the production of synthetic rubber. Together, these products account for 72-81% of ethylene, 56-73% of propylene and 45-53% of aromatics (benzene) consumption in Western Europe in 1994-2013 (Table 4.11, section 4.7).

Selection of bio-based alternatives for the chemical industry
From the broad range of biochemicals and processes that can replace fossil-based chemicals, we selected the following (Table 4.3):

- fermentation-based chemicals (direct sugar to chemicals): lactic acid (and polymerisation to PLA), 1,3-propanediol (PDO) and succinic acid (SA);
- ethanol-based chemicals: ethylene and butadiene;
- thermochemical-based feedstock and chemicals: naphtha from gasification and FT-synthesis (feedstock), hydrogen (for ammonia), aromatics and ethylene;
- methanol-based chemicals: olefins (ethylene, propylene);
- catalytic pyrolysis-based chemicals: olefins and aromatics;
- catalysis-based chemicals: polyethylene furanoate (PEF) from sugars.

4.2.2.2 Structure of the aviation sector and model representation
Kerosene from crude oil refining is the reference fossil jet fuel. RJF conversion technologies are selected based on de Jong et al. (2015).[27] The conversion technologies able to produce a RJF fraction are hydroprocessed esters and fatty acids (HEFA), hydrothermal liquefaction (HTL), pyrolysis (PYR), gasification and FT-synthesis (Figure 4.3). Hydrotreated renewable diesel (HRD), a closely related but slightly less complex compound than HEFA diesel, is also included as it is currently reviewed as a blend with fossil kerosene (Boeing, 2014). Only used cooking oil (UCO) and woody biomass are included as feedstocks, which is in line with the industry's intention to use only non-food biomass (Sustainable Aviation Fuel Users Group, 2014). The RJF conversion technologies, besides RJF may produce additional

[27] Alcohol-to-jet and direct sugars to hydrocarbons have been excluded due to high production costs (de Jong et al., 2015).

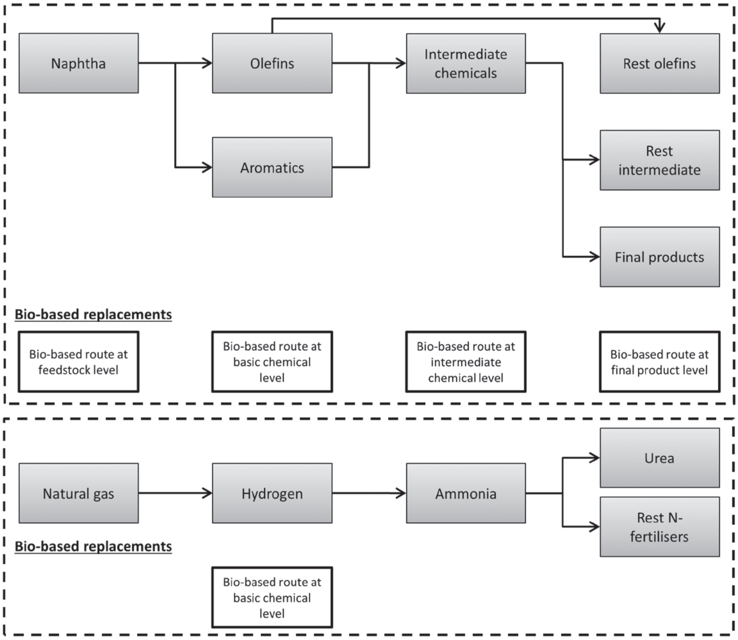

Figure 4.2 Representation of fossil-based and bio-based chemical routes and products in MARKAL-NL-UU (a detailed flowchart is provided in Figure 4.19)

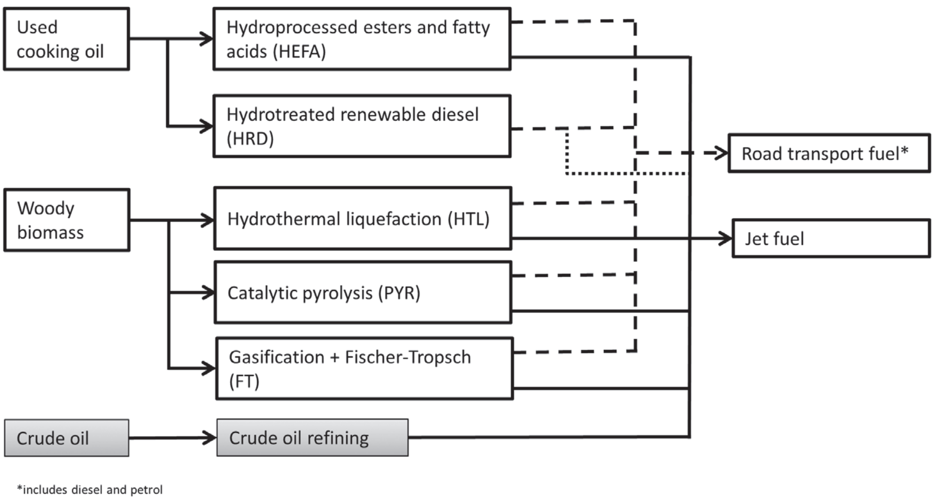

Figure 4.3 Representation of aviation sector in MARKAL-NL-UU along with other biomass conversion technologies

products, such as diesel and LPG (in the model, diesel and LPG are assumed as petrol). These products can be used in the road transport sector. Depending on the technology and the chemical composition of the products, maximum blending constraints (blend walls) for RJF are established by the American Society for Testing and Materials (ASTM) and are included in the model (Table 4.4).

4.2.2.3 Other technologies

Feedstock costs make up a large share of the total production costs of fermentation or enzymatic processes. The sustainability performance of lignocellulosic sugar production is promising, which is why we included cost-structures for solid biomass conversion to C5 and C6 sugars, thus providing demand sectors with the option to use or import raw sugar or starch from crops or invest in domestic lignocellulosic sugar production capacity. In addition, we enriched the technology portfolio of the transport fuel sector in MARKAL-NL-UU with cost-structures for conversion processes of 2nd generation ethanol, starch-based ethanol and biodiesel production from vegetable and UCO. In addition, we implemented market constraints for biomass conversion technologies to fuels. Individual 2nd generation technologies were assumed to supply no more than 5% of fuel demand in 2020 and 10% in 2030, based on de Wit et al. (2010). Furthermore, we included cost-structures for the anaerobic digestion of sewage water, manure co-digestion and wet organic waste. Biogas can then be utilised in small-scale CHP plants for the production of electricity and heat. We also included the option of upgrading biogas to green gas, which can replace fossil natural gas in heat applications. Electricity production from synthetic natural gas (SNG) was included only as a co-production option of the specified technology (Table 4.3). Finally, we included biomass boilers as an alternative to natural gas heat supply in the industry sector (Laurijssen et al., 2012; Saygin et al., 2014). Heat from wood stoves was included, assuming a constant supply of fuelwood for domestic heating from wood stoves because no substantial growth was expected (Rijksoverheid, 2010).

4.2.2.4 Multiple process outputs

Several technologies produce multiple outputs, which can be used across the different sectors. For example, a CHP plant generates both electricity and heat, delivering to the electricity and heat sector. Biorefining is the sustainable processing of biomass into a spectrum of marketable products and energy. Several biomass conversion processes fall under this definition. In this chapter, we refer to advanced biochemical biorefineries if enzymatic processes are used to convert solid biomass to lignocellulosic sugar and ethanol, and we refer to advanced thermochemical biorefineries if biomass is gasified to syngas for further conversion to products such as FT-fuels (de Jong et al., 2012a). Lignocellulosic sugar biorefineries convert solid biomass to C5 and C6 sugars and lignin. Lignin is supplied as solid biomass feedstock to the electricity or heat sector. Another example is biomass gasification and FT-synthesis, which generate fuels (diesel and petrol, used as transport fuels), electricity (supplied to the grid) and FT-naphtha (feedstock for the chemical industry). Conventional coal gasification and FT-synthesis to road transport fuels is excluded from the model. In MARKAL-NL-UU, co-production has been taken into account by linking all process outputs to the sector modules involved, either as feedstock or as end-products, thus achieving valorisation of biomass constituents and improving overall efficiency. Cascading of biomass

Table 4.3 Overview of conventional and biomass conversion technologies to chemicals included in MARKAL-NL-UU

Feedstock level	Basic chemical level	Intermediate chemical level[b]	Final product level[b]
Conventional conversion technologies			
Refining of crude oil to naphtha[a]	Steam cracking of naphtha to olefins and aromatics Fluid catalytic cracking (FCC) of crude oil fractions to propylene[a] Catalytic reforming of reformate to aromatics[a] Steam reforming of natural gas to hydrogen	Oxidation of ethylene to ethylene oxide (EO) Hydrolysis of ethylene oxide to ethylene glycol (EG) Oxidation of aromatics to terephthalic acid (PTA) Oxidation of aromatics to phthalic anhydride (PA) Isomerisation, hydroformylation and hydrogenation of propylene oxide (PO)to 1,4-butanediol (BDO) Synthesis of hydrogen to ammonia	Polymerisation of ethylene to polyethylene (PE) Polymerisation of propylene to polypropylene (PP) Esterification of PTA and EG to polyethylene terephthalate (PET) Dehydrogenation of EB[c] to styrene Oxidation and dehydration of propylene and EB to styrene and propylene oxide (PO) Synthesis of ammonia to urea
Biomass conversion technologies			
Fermentation-based chemicals			
		Fermentation of sugar to succinic acid (SA) Hydrogenation of SA to BDO[d]	Fermentation of sugar to lactic acid followed by polymerisation to polylactic acid (PLA) Fermentation of sugar to 1,3-propanediol (PDO) followed by esterification of PTA to polytrimethylene terephthalate (PTT)
Ethanol-based chemicals			
	Catalytic dehydration of ethanol to ethylene Catalytic dehydration of ethanol to butadiene		
Thermochemical-based feedstock and chemicals			
Gasification of biomass followed by FT synthesis to FT-fuels and FT-naphtha	Steam cracking of FT-naphtha to olefins Gasification and water-gas shift to hydrogen for ammonia synthesis Gasification and separation of aromatics, ethylene[e]		

Table 4.3 (continued)

Feedstock level	Basic chemical level	Intermediate chemical level[b]	Final product level[b]
Biomass conversion technologies			
Methanol-based chemicals			
	Catalytic conversion of methanol to ethylene and propylene		
Catalytic pyrolysis-based chemicals			
	Catalytic conversion of pyrolysis oil to olefins and aromatics		
Catalysis-based chemicals			
			Catalytic conversion of sugar to 2,5-Furandicarboxylic acid (FDCA) and polyethylene furanoate (PEF)

[a]Refinery operation, modelled as import commodity. [b]In this table, downstream conversion technologies at an intermediate and final product level that are reported under conventional conversion technologies are common for chemically equivalent bio-based feedstocks such as olefins and aromatics (see section 4.2.2). [c]Also modelled as import commodity. [d]SA can also be used as a direct phthalic anhydride substitute. As no conversion technology is assumed for this process, it is not listed in this table. [e]Co-produced SNG is used for electricity generation.

Table 4.4 Blending constraints of renewable jet fuel and renewable diesel with total jet fuel

Technology	Fuel	Maximum share blended with jet fuel[a]
HEFA	Renewable jet fuel (HEFA jet)	50%
HEFA	Renewable diesel (HEFA road)	10%
HRD	Renewable diesel (HRD road/jet)	10%
HTL	Renewable jet fuel (HTL jet)	30%
PYR	Renewable jet fuel (PYR jet)	30%
FT	FT jet fuel (FT jet)	50%

[a]Only the blending constraints for HEFA jet and FT jet are defined according to the specifications of the ASTM, as these are the only RJF with an official specification (ASTM, 2015). The blending constraints for HRD, HTL and PYR jet are anticipated blending constraints.

from high-value applications such as materials to energy recovery is another efficient option of biomass utilisation, but this has not been included in this analysis.

4.3 INPUT DATA AND SCENARIOS

4.3.1 Current and future techno-economic performance

4.3.1.1 Production costs

Competition between technologies is based on the performance characteristics of the energy and non-energy products they deliver. Their cost-structures consist of capital investments for a given capacity, fixed costs (supplies and administrative costs) and variable costs (feedstock, labour, other material inputs and utilities). The discount rate is set at 7%. All nominal costs are deflated to real costs in € 2010 terms, based on exchange rates and price indices (European Central Bank, 2014; Eurostat, 2014a; U.S. Department of Labor, 2014). The production costs of technologies included in MARKAL-NL-UU prior to the model expansion have been described in literature (Brouwer et al., 2015; van den Broek et al., 2011; van Vliet et al., 2011). Cost-structures of technologies that have been introduced in the model's update are based on the following method:

- *capital investment costs* are the aggregate of Inside Battery Limits (ISBL; e.g. key process components), Outside Battery Limits (OSBL; e.g. utilities, control systems, buildings, storage) and contingency. We use data from available literature and company announcements. For technologies for which only ISBL costs are provided, OSBL costs are assumed as 35% of ISBL and contingency as 25% of ISBL and OSBL costs. To estimate the capital investment costs of technologies at different scales, we apply 0.7 as the scaling factor in the formula:

$$\frac{\mathrm{Cost}_{base}}{\mathrm{Cost}_{scaled}} = \left(\frac{Capacity_{base}}{Capacity_{scaled}}\right)^{Scaling\ factor}$$

  Equation 4.1

  In addition, location factors are used for capital investment costs that refer to regions other than Europe (Table 4.14, section 4.7)
- the *fixed costs* for technologies that are not provided by the data source are estimated based on Hermann and Patel (2007) (Table 4.16, section 4.7). If fixed costs are provided by the data source, these are used instead. Labour costs are either included in reported operational expenditure or are estimated based on an annual full-time salary of € 56,210.[28] If labour hours or labour costs are not provided by the data source, they are

[28] Based on a wage of 28.7 €/h and 2,080 h/yr for industry in the Netherlands in 2005. Converted to 2010 wages using labour cost indices for the Netherlands (Eurostat, 2014b).

assumed as 5% of variable costs. Labour costs are scaled using Equation 4.1 and a scaling factor of 0.2 (similar to Patel et al. (2006), where a 0.25 scaling factor is applied);

- biomass feedstock and energy are typical major cost components of *variable costs*. Based on process efficiencies, feedstock costs are calculated using feedstock prices (Table 4.17, Table 4.23, Table 4.24, section 4.7). For technologies that require external energy input (electricity, heat), the model takes the additional demand into account and supplies it by conversion technologies based on the system's least-cost pathways. Variable costs of technologies that are self-sufficient on energy are indirectly accounted for as additional capital investments (e.g. CHP, boiler). Other variable costs taken into account are cellulase in lignocellulosic sugar and ethanol technologies, catalyst costs in pyrolysis technologies and acetic acid costs in PTA production.

Production costs are related to the n^{th} plant, thus excluding the potentially higher costs of the first unit installed due to operation at a low utilisation rate. The cost components described above are disclosed in the section 4.7 (Table 4.29, Table 4.30).

4.3.1.2 Technological development and scenarios of bio-based technologies

Future technology costs linked with endogenous learning rates are associated with operational experience, design and construction of the technology (Junginger et al., 2010). Unit costs decline by a percentage (learning rate) for each doubling of installed production capacity. This method has been extensively studied and applied for bioenergy systems (de Wit et al., 2010; Junginger et al., 2006; van den Wall Bake et al., 2009). Learning rates are associated with installed capacities on a global scale (Junginger et al., 2006). Compared to the rest of the world, the Netherlands does not represent a sizeable market for most technologies, which is a limitation for applying endogenous technological learning in this study. Therefore, we rely on estimations of future costs from bottom-up engineering studies and expert judgements on potential improvements of various technology components such as yield and energy efficiency. In addition, we take into account scaling of technologies that may achieve cost reduction through economy of scale. Furthermore, as several technologies are in different developmental phases, the moment of commercialisation (start year) becomes highly relevant. Technical improvements, scale-up, start year and technology availability are determined exogenously and are applied to capture the cost development of technologies. These improvements and the consequent cost reductions over time depend on various factors, such as research and development (R&D) efforts and stimulating policies, and involve substantial uncertainties regarding development pathways.

To capture the uncertainty in technology development, we incorporate two scenarios: low and high technology development (LowTech and HighTech).

- *LowTech* takes into account technologies that are commercially available today. Based on existing or announced capacities, it assumes the capacities of the technologies to be relative small. There is a low rate of incremental improvements in process yields and autonomous efficiency improvements (Table 4.5). This scenario describes a business-as-usual case;

- *HighTech* assumes that more technologies may be implemented and on a larger scale. There is a high rate of incremental improvement in process yields and autonomous efficiency improvements (Table 4.5). Preconditions for this scenario are strong R&D efforts, investment at the early stages of development, support of technologies to pass the valley of death (e.g. required for gasification), and commercial success of existing installed capacities, such as the 2nd generation ethanol facilities of POET-DSM, Abengoa, and DuPont, the fast pyrolysis unit of BTG and other facilities (Janssen et al., 2013).

In this study, technology developments based on technology improvement, scale-up, start years and availability are incorporated in the different scenarios as follows:

- *Technology improvements*. We assume technical improvements for bio-based and fossil-based processes for conversion yields and process energy efficiency. Assumptions are made exogenously regarding the improvement in production costs, yields and efficiencies from the start year t_0 of a technology to year t_{0+n}. For technologies that are mature (e.g. esterification of vegetable oils, downstream technologies, industrial biomass boilers) or where there is no available information on future performance, the cost and efficiency data remain constant throughout the modelling period. Increases in capital costs due to process or energy efficiency improvements have been ignored. For technologies that were included in MARKAL-NL-UU prior to the present study, developments have been based on literature and no differentiation has been made between LowTech and HighTech scenarios (e.g. technologies for the power sector) (Brouwer et al., 2015; van den Broek et al., 2011; van Vliet et al., 2011). Table 4.5 presents different improvement rates assumed per technology in each scenario. Table 4.21 in section 4.7 presents the different assumptions in detail.
- *Scale-up*. For biofuels and biochemicals, the LowTech scenario assumes small scale production capacity, depending on the technology. For example, for pyrolysis fuels up to 91 $MW_{feed\ in}$ can be deployed and for lignocellulosic sugar and ethanol up to 400 $MW_{feed\ in}$ (e.g. lignocellulosic sugar ethanol). Scale-up of bio-based chemical technologies occurs in time-steps of 10 years. By contrast, the HighTech scenario assumes that technologies may be scaled up to 2 $GW_{feed\ in}$ (e.g. gasification, 2nd generation ethanol) and the scale increases every 5 years. To determine the costs of technologies at different scales we apply Equation 4.1.
- *Start year and technology portfolio*. The 'technology readiness level' can be used to determine the start year of technologies. However, it is uncertain how fast a technology can advance through the different development levels (from concept to commercial deployment; see e.g. Peisen et al. (1999)). Furthermore, even technologies at an advanced readiness level will not always pass the valley of death. In addition, for biochemicals that are not chemically identical with their fossil counterparts, but that have different technical functions and complex supply chains, it could take more than 20 years to emerge at large production scales (de Jong et al., 2012a). Therefore, start year and technology portfolio are important parameters to be varied in the two scenarios. Different statuses and ranges of technology development phases are mentioned in literature (Gerssen-Gondelach et al., 2014; TKI-BBE, 2015b). Figure 4.4 and Figure 4.5

present the start year, technologies and the capacities assumed in the two scenarios for the chemical and transport fuel conversion technologies. The start years of biomass gasification that supply fuels in earlier versions of MARKAL-NL-UU (van Vliet et al., 2011) are aligned with the start years of biomass gasification technologies, as can be seen in Figure 4.5.

Table 4.5 Incremental yield and autonomous annual energy efficiency improvements in chemical conversion technologies of the low and high technology development scenarios (Table 4.21 in section 4.7 presents the different assumptions in detail)

Product (process)	**Improvement**	**Range in scenarios**	
		LowTech	HighTech
Lignocellulosic sugar/ethanol	Sugar extraction	1% p.a.	2-3% p.a.
(various pretreatment methods/fermentation)	Fermentation	0.05-1% p.a.	0.1-2% p.a.
Biochemicals (fermentation and catalytic conversion of methanol)	Yield improvement	0.25% p.a.	0.5% p.a.
	Process energy efficiency	0.5% p.a.	1% p.a.
FT-naphtha (gasification and FT-synthesis)	Yield improvement	0.25% p.a.	0.5% p.a.
Basic chemicals (steam cracking)	Process energy efficiency	1% p.a.	1.8% p.a.
Ammonia and hydrogen (steam reforming)	Yield improvement	0.25% p.a.	0.5% p.a.
	Process energy efficiency	0.5% p.a.	1% p.a.

4.3.2 Biomass cost-supply

In this study we incorporate cost-supply curves of biomass from domestic resources (i.e. the Netherlands) and intra-EU resources that could be exported[29] based on projections of the economic potential of biomass from 2010 to 2030 of the Intelligent Energy Europe (IEE) project Biomass Policies (Table 4.23, section 4.7) (Elbersen et al., 2015). From the total EU28 potential, we allocate a specific share available to the Netherlands based on the share of the Dutch total primary energy supply relative to the EU28 (OECD/IEA, 2014). For 2010-2030, this has been estimated at approximately 5%. For simplification purposes, the different feedstocks have been aggregated to broader categories (Table 4.23, section 4.7). In addition to domestic and European biomass, we assume an extra-EU supply of liquid biofuels (1st and 2nd generation ethanol, biodiesel), vegetable oil, sugar and solid biomass (wood pellets). The amount of imported wood pellets is constrained to 400 PJ_{prim} (23.4 Mt_{wpe}; wood pellet equivalents assuming 17.1 MJ/kg lower heating value (LHV)). Imported biofuels, vegetable oil and sugar amount to 50 PJ_{prim}, which is sufficient to meet the 10% blending target of road transport in 2020. Each feedstock category is associated with specific conversion technologies included in MARKAL-NL-UU as shown in Figure 4.20 (section 4.7). Table 6 presents the available domestic and imported biomass potential in MARKAL-NL-UU.

[29] Excluded from export are low-quality biomass, liquid and solid manure which are assumed to be utilised in their country of origin, and household waste, with the exception of paper, wood and UCO. Types excluded from the domestic potential do not contribute significantly to the total potential.

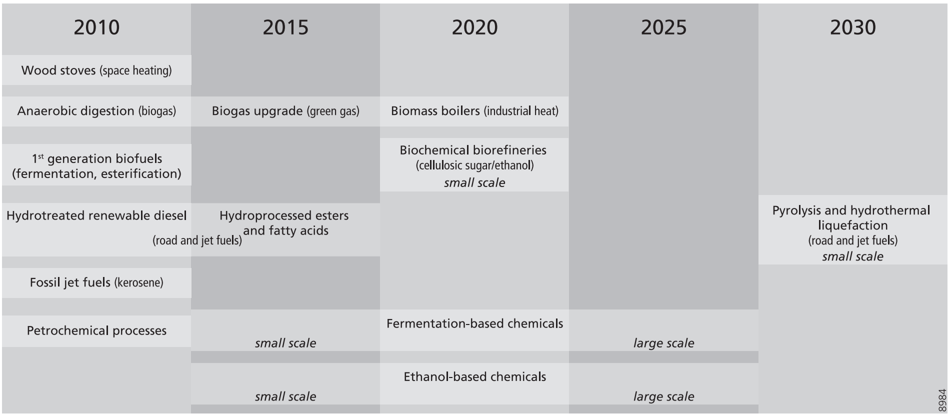

Figure 4.4 Low technology development scenario (LowTech) for conversion technologies added in MARKAL-NL-UU

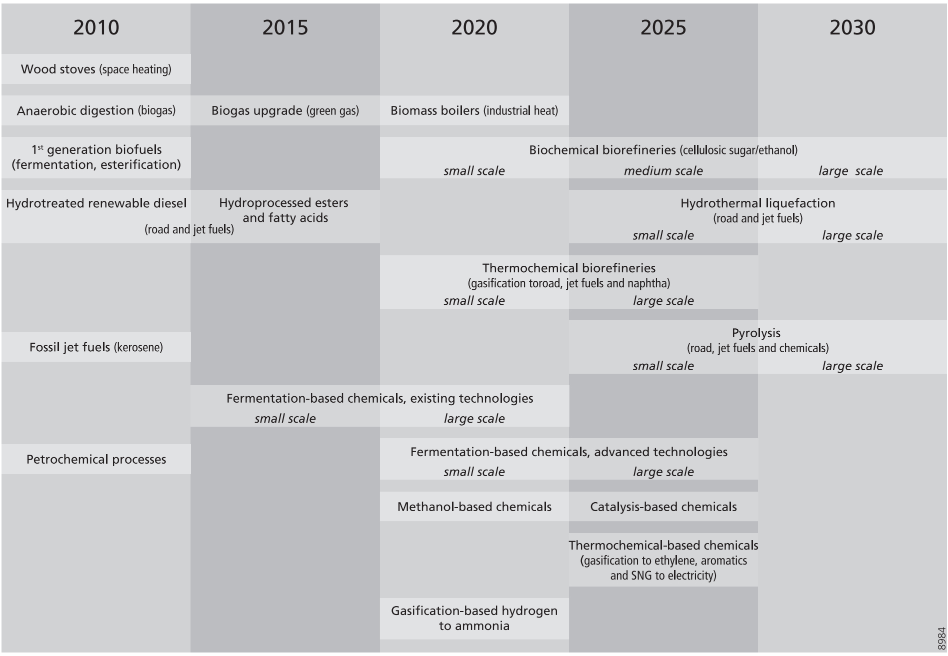

Figure 4.5 High technology development scenario (HighTech) for conversion technologies added in MARKAL-NL-UU

Table 4.6 Available domestic and imported biomass potential in MARKAL-NL-UU for the Netherlands (NL) in 2010-2030 (rounded figures)

[PJ]	2010		2020		2030	
	NL	EU	NL	EU	NL	EU
Crops	2	32	13	89	22	101
Crop residues	8	52	7	50	7	51
Wood crops	0	0	1	15	2	16
Forestry products and residues[a]	46	235	52	235	59	254
Waste domestic	88		80		83	
Used cooking oil EU		5		5		5
Extra-EU imports solid biomass			400			
Extra-EU imports liquid biomass			50			
Total domestic	144		153		172	
Total import	772		843		878	

[a]Fuelwood for wood stoves is added ad hoc to the total domestic potential. It is 15, 18, and 20 PJ for 2010, 2020 and 2030, respectively, and is reported under forestry products and residues.

4.3.2.1 Logistic costs of imported biomass

The cost-supply curves for the intra-EU feedstock categories in MARKAL-NL-UU are estimated based on the following approach:

- the logistic costs of EU supply are calculated from NUTS2[30] regions to Rotterdam, assuming transport as wood chips (Hoefnagels et al., 2014a, 2014b). These costs are added to the cost of biomass feedstocks derived from the IEE project Biomass Policies; they are assumed to be constant throughout the modelling period using 2015 oil prices (Elbersen et al., 2015). Note that advanced pre-processing, e.g. pelletisation, could reduce the cost of long-distance supply chains, but has not been included in this study;
- the national cost-supply potential from the NUTS2 level is estimated by aggregating the biomass supply potential of each country's NUTS2 regions. The biomass cost of supply to the Netherlands is their weighted average;
- the regional cost-supply potential per biomass feedstock for 4 EU regions (North, South, East, West, according to the United Nations' classification; Table 4.22, section 4.7) is estimated by aggregating the national potentials of each biomass feedstock. Regional costs are determined based on the weighted average of the national feedstocks;
- each region's cost-supply potential per biomass category is estimated by aggregating the supply potential of the different feedstock types of the same category. Their costs are determined based on the weighted average of these types.

The cost of supply of extra-EU categories is estimated based on the following:

[30] Nomenclature of territorial units for statistics.

- wood pellet price is based on average free-on-board biomass prices over a 10-year period (2006-2015), while other literature sources were used to estimate the cost-price development of the extra-EU feedstocks (Table 4.24, section 4.7);
- transport costs to the Netherlands are assumed, based on fossil fuel price and fossil fuel consumption in the logistics chain. These were determined based on Hoefnagels et al. (2014a, 2014b). For extra-EU sugar, transport is assumed to be similar to wood pellets. Transport costs of extra-EU ethanol are added to the cost-price of 1st and 2nd generation ethanol produced in Brazil, based on fuel consumption in the chain (Cardoso et al., 2012). Extra-EU vegetable oil and biodiesel transport costs are included, based on shares of transport costs over the import values of the commodities to Rotterdam according to OECD/FAO (2015).

Costs of sugar from inside and outside the EU are assumed to be identical and are based on the Food and Agriculture Organisation of the United Nations (OECD/FAO, 2014), as prices are expected to converge after the abolition of the sugar quota in Europe.

Table 4.23 in section 4.7 presents the costs of the aggregated biomass categories and costs of transport to the Netherlands in 2010-2030, and Table 4.24 (section 4.7) presents the costs of globally traded commodities. The transport costs assumed in this study are conservative, as they are based on wood chip logistics for long-distance supply chains, leading to overall conservative costs of biomass supply and ignoring the development of fossil fuel prices up to 2030. Literature indicates that significant cost gains can be achieved in biomass transport if biomass is processed to pellets in the sourcing region. Furthermore, biomass densification leads to higher efficiencies on the conversion side. Such improvements can lead to more cost-effective biomass supply chains than assumed in this study, thus improving the cost-competitiveness of biomass deployment in the energy system (Batidzirai et al., 2014; Uslu et al., 2008).

We account for CO_2 emissions of produced and mobilised biomass, using the same method as for the cost-supply estimates (i.e. weighted average of NUTS2 regions to four geographic regions). Domestic emissions from domestically produced biomass contribute to the national CO_2 emissions; regional (and global) emissions are addressed separately. Emission factors are presented in Table 4.26 (section 4.7) (Giuntoli et al., 2014).

4.3.3 Final energy and non-energy demand

4.3.3.1 Energy demand

The final energy demand for electricity and heat in all sectors of the Dutch economy is based on the latest projections made by the Energy Research Centre of the Netherlands (ECN, 2015), taking into account policies established in 2012. Demand for road transport fuel (vehicle-kilometers) is described in van Vliet et al. (2011). Demand for aviation fuel has been derived from the average growth projections of the PRIMES model (baseline scenario) and literature for Europe, assuming they are the same for the Netherlands (Chèze et al. (2011), EC (2003); Figure 4.6; Table 4.19, section 4.7). The demand for aviation fuels

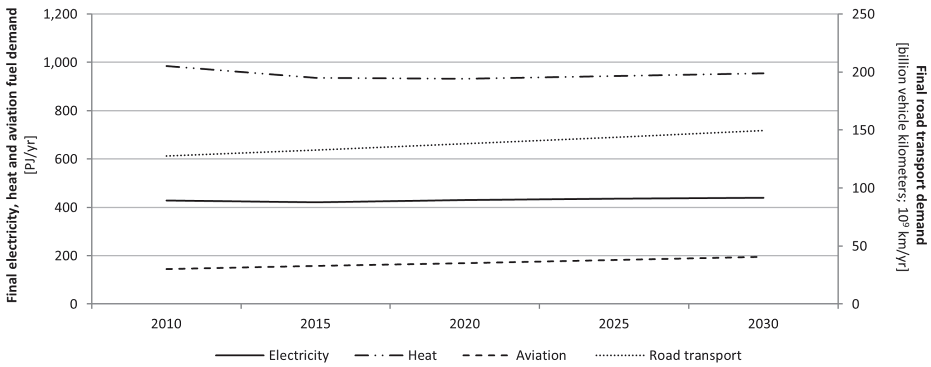

Figure 4.6 Final demand of energy sectors in MARKAL-NL-UU

is based on jet bunker fuel consumption in the Netherlands. This includes jet fuel mainly consumed for international flights, as domestic consumption is negligible. By contrast, the final demand for electricity, heat and road transport fuel is based on domestic consumption in this model (imports-exports of electricity and road transport fuels are ignored).

4.3.3.2 Chemicals demand

The level of investment either as expansion of existing capacity or as deployment of biomass conversion technologies depends on the production of chemicals in the Netherlands up to 2030 to meet domestic and export demand. We base the demand on the production volume of chemicals in the Netherlands rather than on domestic consumption, as the trade flows of chemical commodities are too complex to take them into account in the model (e.g. re-exports, conversions to different commodities). To determine the production volume of chemicals, we use publically available information on production capacities in 2006-2011 (Chemweek (2000-2009), ICIS (2006), Lako (2009), Neelis et al. (2007a, 2007b), Neelis et al. (2003), OGJ (2012); Table 4.13, section 4.7). These are extrapolated to 2030 based on high growth rates (Saygin et al. (2009); Figure 4.7; Table 4.10, section 4.7).[31] To estimate the production volume, we assume capacity utilisation rates of 85% (Neelis et al., 2005). The volume of basic chemicals that are not used for the production of intermediate or final products is defined as the residual demand for each basic chemical (Figure 4.2). This demand is based on the total volume of basic chemicals produced in the Netherlands minus the demand for the production of intermediate and final products according to capacity and process yields (Table 4.13, Table 4.15, section 4.7). The residual capacity, which in turn defines the residual demand, is presented in Table 4.12 (section 4.7).

[31] Future demand for chemicals is a significant input parameter as it determines the level of capacity investments required, assuming that existing steam crackers will be decommissioned in the modelling timeframe. However, future production demand for chemicals within the Netherlands is uncertain, considering that production capacity in other regions may increase due to competitiveness, uncertain fossil fuel prices and so on. Therefore, lower or even negative growth rates may be expected.

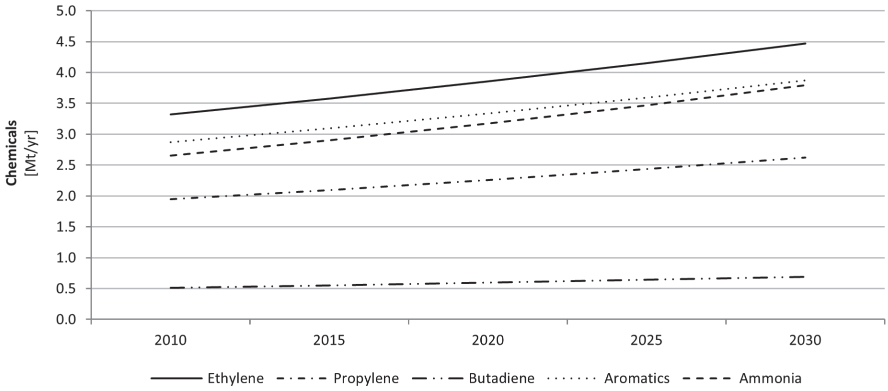

Figure 4.7 Production demand for basic chemicals and ammonia assumed in MARKAL-NL-UU for the Netherlands in 2010-2030

4.3.4 Policy development

Under the EU Renewable Energy Directive (RED), each member state has an obligation to meet country-specific targets to achieve the Union's target of 20% renewable energy share in the final energy demand by 2020 (EC, 2009b). For the Netherlands this corresponds to a minimum of 14% renewable energy in the country's final energy demand (electricity, heat and transport fuels) by 2020. In addition, 10% of the final energy demand in road transportation must be of renewable origin (biofuels, renewable electricity). Biofuels from wastes, residues, non-food cellulosic material and lignocellulosic material contribute twice and renewable electricity in transport contributes 2.5 times to the blending target. This study excludes the contribution of RJF to the renewable energy share and blending target of the EU RED if these RJFs exclusively supply the aviation sector, although this is allowed according to the directive and implemented by the Netherlands (EC, 2009b). In addition, the Dutch Energy Agreement (SER, 2013) outlines specific goals regarding the use of biomass for co-firing in coal power plants, the deployment of on-shore and off-shore wind turbines and a renewable energy share in the final energy demand beyond 2020 (Table 4.18, section 4.7). More specifically, the renewable energy share, according to the Dutch Energy Agreement should be 16% in 2023. In this study, these targets are incorporated in all scenarios. In addition, we have included CO_2 emission tax based on the International Energy Agency's World Energy Outlook New Policies scenario (OECD/IEA, 2015). The CO_2 tax levels across the years are presented in Table 4.17 (section 4.7).

4.4 RESULTS

4.4.1 Final energy consumption

Figure 4.8 shows the final energy consumption in the Netherlands in 2010-2030. The renewable target is the key driver for the deployment of renewable energy resources (scenarios do not exceed the renewable energy share target; black rectangular markers in Figure 4.8), and limited variation is observed across the scenarios. Therefore, greater efforts are required to achieve diffusion of renewables beyond policy targets. Biomass plays a key role in meeting the targets early in the time horizon (black circular markers in Figure 4.8). The relative contribution will decrease towards 2030, because of the increase in the contribution of other renewables, especially wind power. In 2020, renewable electricity and heat generation technologies will be contributing most to the renewable energy target. By 2030, this pattern will be continued only in LowTech (see also results per sector in sections 4.4.1.1 - 4.4.1.4). In HighTech the contribution of biomass heat will decrease, as renewable transport fuels grow due to the availability of more efficient technologies, compared to LowTech. Accounting for the non-energy sector, we find that in 2030 and in HighTech, biochemicals will contribute more to the shares of renewable energy and non-energy than in 2020.

4.4.1.1 Electricity

Figure 4.9 shows electricity output by source in the Netherlands in 2010-2030. Coal-based electricity is increasing between 2010 and 2020. By 2030, natural gas and electricity from other renewable sources (primarily wind) will have increased compared to 2020, while coal-based electricity will have decreased to levels lower than in 2010. Reduction in coal is partly due to the gradual phasing out of old coal-fired power plants in the Netherlands. However, coal-based electricity is still supplied as new coal-based electricity capacity was installed in 2015 (3.45 GW_e according to SER (2013)). The decrease in coal-based electricity output is also due to higher levels of CO_2 tax in 2030 compared to 2010-2020. The contribution of renewable energy sources in 2030 is highest under HighTech, primarily due to co-production of electricity in biorefineries as electricity output from non-biomass renewables is similar across the scenarios. The output of non-biomass renewables is driven by the Dutch Energy Agreement (SER, 2013). On-shore wind turbines of 2 GW_e combined capacity will be installed as early as 2020 to meet the renewable energy target. As this will be above the capacity level supported in the Dutch Energy Agreement, on-shore wind will be competitive with electricity from biomass.

4.4.1.2 Heat

Biomass heat will contribute significantly to the renewable energy share (21-37%; Figure 4.8) and will triple between 2010 and 2030 (from 30 PJ_{th} to up to 104 PJ_{th} in 2030; Figure 4.10). The highest contribution will come from biomass heat use in industry (52-79% of total biomass heat output). The remainder of renewable heat will be similar across the scenarios and is primarily the output of bio-CHP, MSWI and wood stoves. Heat from co-firing biomass in power plants will contribute only in 2020.

4.4.1.3 Transport fuels

Table 4.7 Blending shares of biofuels in the transport sector (road transport, aviation) in the different scenarios and time periods

	2010	2020		2030	
		LowTech	HighTech	LowTech	HighTech
Total biofuel blending	2%	6%	6%	5%	11%
biofuel blending road transport (incl. double counting)	4%	10%	10%	10%	27%
biofuel blending road transport	3%	8%	8%	8%	14%
biofuel blending aviation	0%	0%	0%	0%	7%

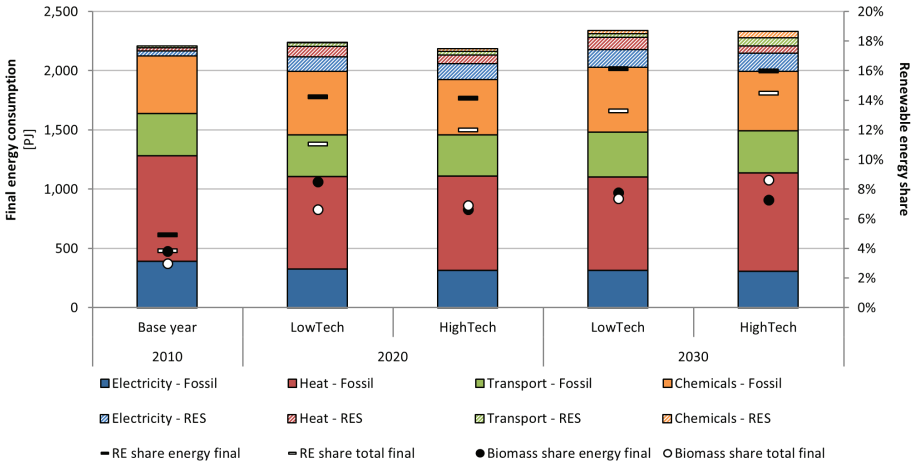

Figure 4.8 Final energy consumption in the Netherlands in 2010-2030 (excluding jet fuel)

Results for the transport sector (road and jet fuels; Figure 4.11, Table 4.7) indicate (a) a significant contribution of fossil fuels (diesel, petrol, kerosene) in the transport fuel mix (89-95% across scenarios) and (b) a diversified technology portfolio of biofuels across the technology development scenarios. In 2020, HRD from vegetable oil and UCO are key in LowTech (combined output of 30 PJ); however, in HighTech the supply changes from renewable diesel from vegetable oil to 1st generation ethanol. No RJFs will have been supplied by 2020 in the two scenarios. In 2030, in LowTech, HRD will still contribute to the biofuel mix. No RJFs will be supplied in LowTech in 2030.

In HighTech, biofuel output is more diverse: compared to LowTech, HRD will be almost completely phased out from road transport and will be supplied to the aviation sector (5.5 PJ). Furthermore, large quantities of FT-diesel and petrol will be supplied to road transport (53 PJ) and a small share will go to the aviation sector (7.5 PJ). In addition, 1st generation ethanol will be supplied in small quantities (3 PJ). The diversification of HighTech in 2030 is due to the access to low-cost feedstocks (imported wood pellets) in combination with technology development (biomass gasification and FT-synthesis), which will make biomass conversion technologies competitive; supply is distributed based on cost competitiveness instead of being driven by the blending target (e.g. HRD is supplied to road transport in the LowTech scenario but to aviation in the HighTech scenario, as in this scenario FT-fuels cover a large part of road transport fuel demand). In LowTech the blending target is the main driver for biofuel production across the modelling period. In HighTech, the road transport sector's blending target will have been exceeded in 2030 to achieve the EU RED renewable energy share.

4.4.1.4 Biochemicals

The output of biochemicals varies significantly across the two scenarios and time periods (Figure 4.12). Biochemical output increases from 2020 to 2030 in both scenarios (a factor 4

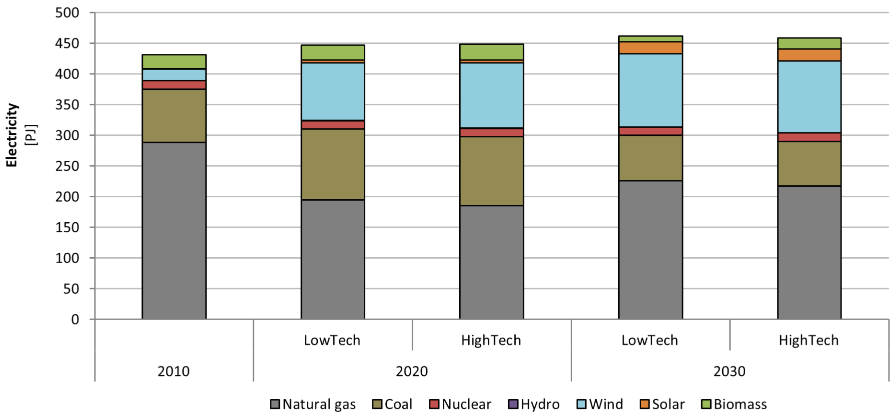

Figure 4.9 Electricity output in the Netherlands by source in 2010-2030

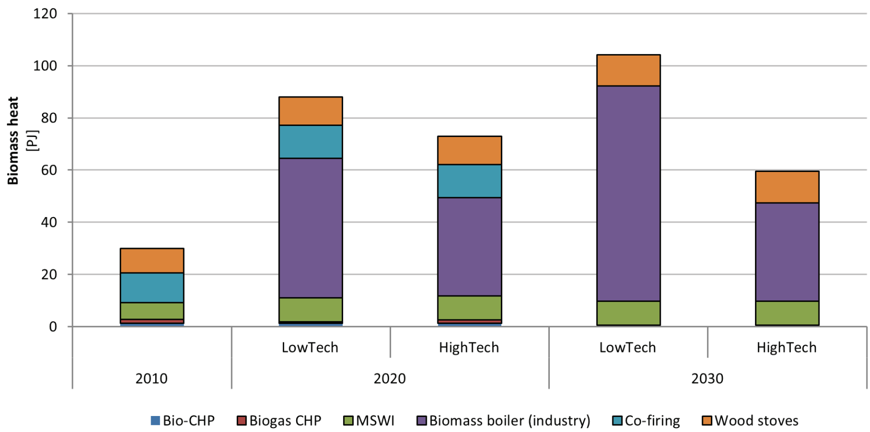

Figure 4.10 Biomass heat production in the Netherlands in 2010-2030

growth in LowTech and almost a factor 2 growth in HighTech). However, in absolute terms, the output in LowTech is 30-70% lower than in HighTech. The biochemical that appears to be competitive in both scenarios and time periods is PLA. In 2030 and in HighTech, biochemicals will compete in the same market: PLA output will remain the same from 2020 to 2030 while PEF and ethylene from ethanol will also be supplied. These developments are due to take place without policy incentives or support schemes for biochemicals. Their emergence is an outcome of cost competition with fossil-based chemicals as driven by biomass and oil price, the high growth rates of the chemical industry assumed in this study, but also because part of the steam cracker capacity is decommissioned (approximately 3 Mt ethylene, which was installed before 2000). Figure 4.13 puts bio-based energy use for

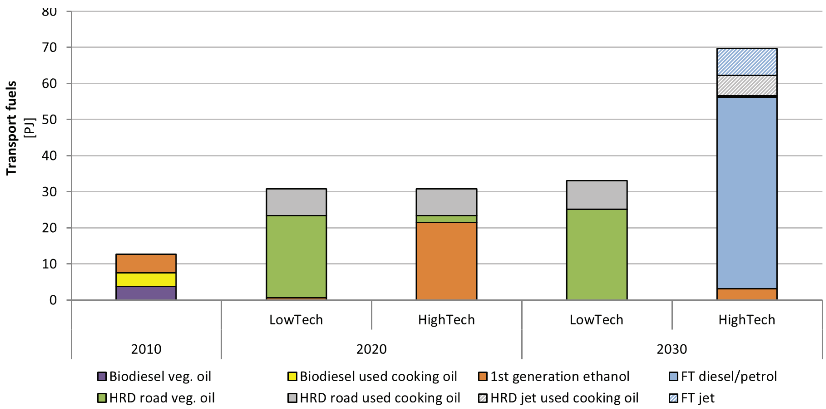

Figure 4.11 Bio-based transport fuels in the Netherlands in 2010-2030

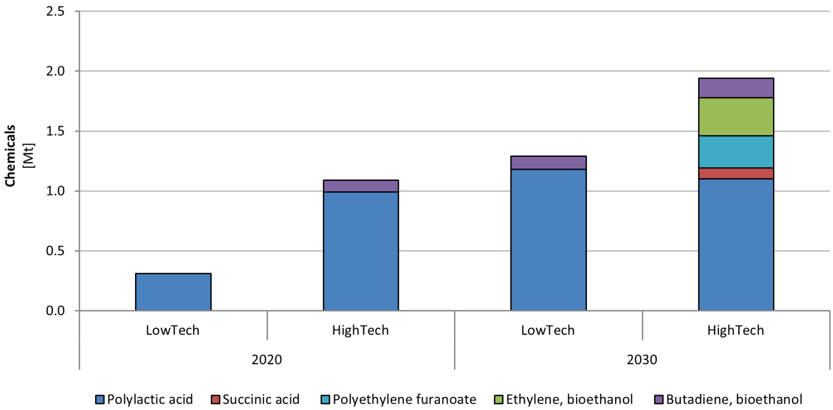

Figure 4.12 Biochemicals production in the Netherlands in 2020-2030

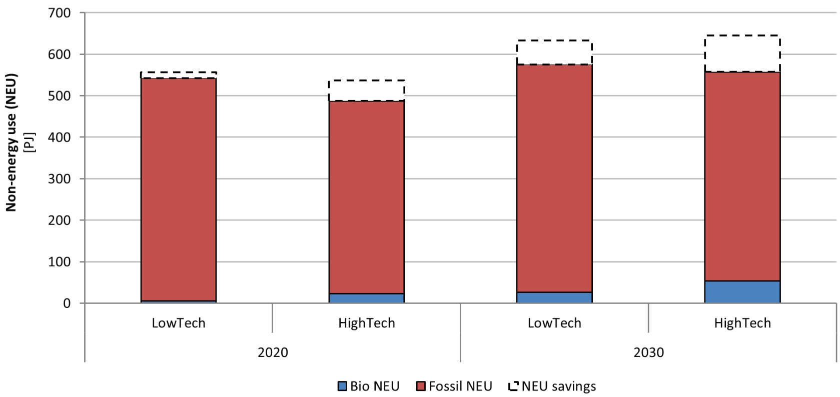

Figure 4.13 Fossil and bio-based energy use for chemicals in the Netherlands in 2020-2030. Fossil energy savings are also estimated

non-energy purposes into perspective by comparing it to the total non-energy use in the Netherlands. The share of the biochemicals over the total chemicals ranges from 1-5% in 2020 to 5-10% in 2030.[32]

4.4.2 Biomass consumption

Figure 4.14 shows primary energy consumption of fossil and renewable resources. Total primary energy increases by 1-2% in the two scenarios. However, fossil energy decreases by 6-8%. The reduction in fossil energy is compensated by an increase in biomass and other renewables. Biomass consumption reaches 230 PJ in LowTech and 300 PJ in HighTech (CARG of 2.7% and 4%, respectively between 2010-2030), which illustrates the influence of technology development scenarios in biomass consumption. Other renewables are not influenced by the scenarios, as we do not vary technology improvement of non-biomass renewables between LowTech and HighTech and because their deployment is primarily driven by targets.

Figure 4.15 and Figure 4.16 show biomass flows from different sourcing regions, namely the Netherlands, intra-EU and extra-EU in primary energy terms[33] in 2030 (section 4.3.2). The figures also include consumption per sector[34] and final production of bio-based energy and non-energy for 2030 (sections 4.4.1.1 - 4.4.1.4). Domestic biomass accounts for approximately 35-38% of total consumption and the remaining volumes are imported. Extra-EU resources account for slightly more than half of total biomass consumption. Biomass consumption for heat is high in both scenarios (highest consuming sector in LowTech, second highest consuming sector in HighTech). Technology development significantly increases the production of fuels and chemicals in terms of consumption and production. Large quantities of biomass are consumed in HighTech in advanced biorefineries, biochemical and thermochemical (160 PJ or approximately 9 Mt_{wpe}). In LowTech, consumption is significantly lower as thermochemical biorefineries are not part of the scenario's technology portfolio due to slower technical progress, and only small-scale lignocellulosic sugar biochemical refineries are deployed (7 PJ or approximately 0.4 Mt_{wpe}). Furthermore, some biomass flows are directed to other sectors. For example, a comparison of Figure 4.15 and Figure 4.16 makes clear that forestry products and residues used for heat in the LowTech scenario are shifted towards fuels in the HighTech scenario. Forestry products and residues are the most important resources as they account for approximately 60% of total biomass consumption in both scenarios. Rapid technology development makes more use of lignocellulosic feedstocks, such as crop residues. The use of these biomass sources is limited in LowTech.

[32] The estimation of the bio-based non-energy use is based on the LHV of final products. Fossil non-energy use savings are determined in the same manner (as opposed to deploying a counterfactual scenario where no bio-based chemicals production is allowed).

[33] Imported biofuels from global markets are accounted for in final energy terms. Estimation of bio-based chemicals in energy terms is based on the LHV of bio-based chemical output.

[34] Biomass consumed in multi-output processes such as biorefineries is allocated based on the LHV of outputs.

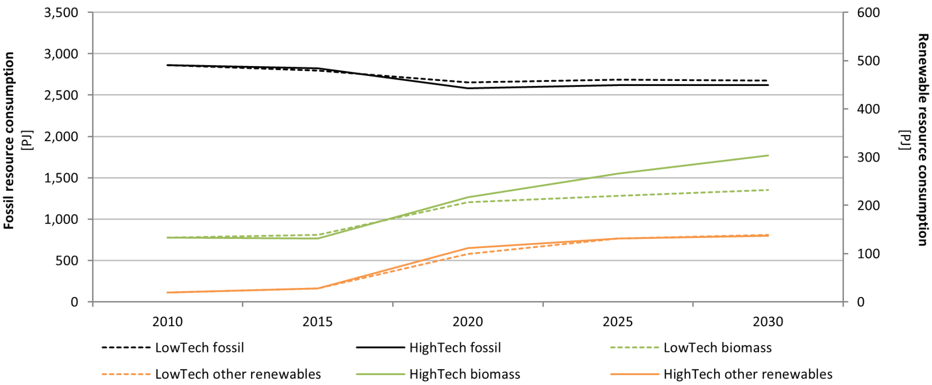

Figure 4.14 Primary energy consumption. Left axis: fossil; right axis: biomass and other renewable energy in the Netherlands in 2010-2030 (wind, solar and hydro $PJ_{prim}=PJ_{final}$)

4.4.3 CO_2 emissions

Figure 4.17 presents direct CO_2 emissions, i.e. domestic emissions that occur within the geographical boundaries of the Netherlands, including domestic biomass production and transport (grouped under 'Heat and other sectors' in Figure 4.17). Figure 4.18 presents indirect emissions, i.e. emissions that occur in the supply chain outside the Netherlands, including imported biomass production and transport. Direct CO_2 emissions decrease over time and across the technology development scenarios. Sectors that contribute most to the reduction are *electricity*, due to the deployment of wind and the switch from coal to natural gas, *industry*, due to the use of biomass heat, *heat*, due to the decrease in demand, and *transport* (only in the HighTech), due to the large biofuel supply. However, CO_2 targets are not met in any of the scenarios. This indicates that with the assumed fossil fuel prices, CO_2 tax, and technology portfolio greater efforts will be required to achieve reduction targets. Due to imported biomass, there are significant indirect emissions due to biomass production and transport (approximately 3.5 $MtCO_2$), whereas indirect emissions due to import and extraction of oil and gas are approximately 5.5 $MtCO_2$. In 2020, indirect emissions will counterweigh the reduction achieved in the Netherlands, but by 2030 there will be a net reduction in the range of 8-16 $MtCO_2$ (or 5-10%) compared to 2010. Emissions from land use change have not been taken into account. Emissions from jet fuels are not included in Figure 4.17 and Figure 4.18, because they are primarily associated with international flights and are not allocated to the Netherlands. These are in the range of 13.4 $MtCO_2$ for both scenarios in 2020. In LowTech, where only kerosene is used, the emissions will reach 15.5 $MtCO_2$ in 2030, while RJF blending will lead to savings of 1 $MtCO_2$ in HighTech during the same year.

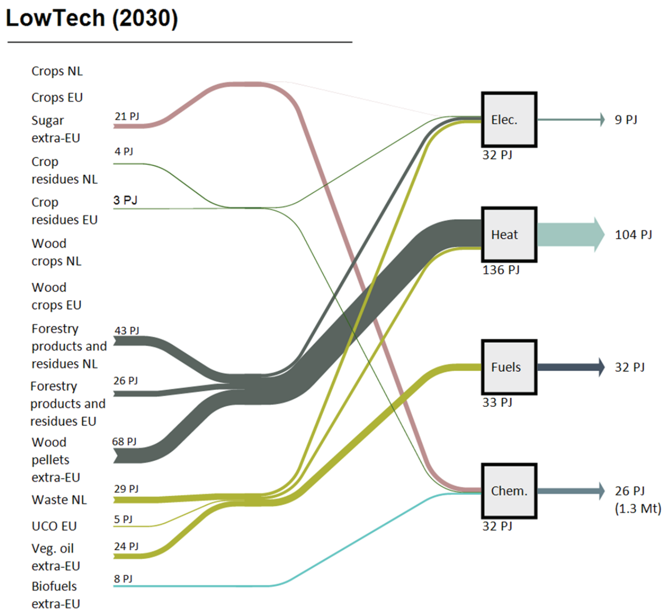

Figure 4.15 Biomass flows in the Netherland in 2030 under low technology development scenario assumptions

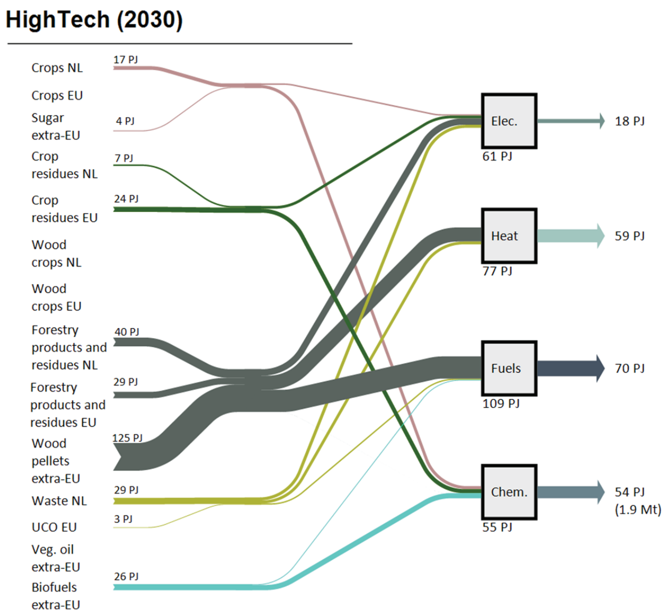

Figure 4.16 Biomass flows in the Netherlands in 2030 under high technology development scenario assumptions

4.5 DISCUSSION

The results of this chapter should be interpreted in the context of the input assumptions and the method used. In the following sections we discuss the influence of the modelling approach, technology selection, data limitations, sensitivity and uncertainty analysis.

4.5.1 Modelling approach

Firstly, in MARKAL-NL-UU there are no market constraints that limit the deployment of conversion capacity for the most cost-effective technologies. To some extent, we can account for this limitation by introducing supply constraints in the conversion capacities of advanced biofuels (section 4.2.2.3). However, this was not applied to the conversion capacities of biochemicals. On the one hand, the aim of this chapter is to demonstrate optimal pathways of biomass to end-use sectors, taking chemicals into account. Limiting the production capacity of biochemicals would deviate from this goal. On the other hand, such constraints may be relevant for specific routes, such as PLA or ethylene from ethanol, which were found to reach production volumes of up to 1.95 Mt by 2030. In comparison, today's single plant capacities reach 155 kt/yr of lactic acid (for PLA) and 200 kt/yr of ethanol-based ethylene (nova-Institut, 2012). Therefore, it could be argued that similar constraints should be applied for these processes. However, while all technologies compete for the same biomass (section 4.3.2) the biomass potential available to the Netherlands is not fully utilised. Secondly, technology deployment and energy supply do not increase gradually or smoothly; rather, they expand drastically and switch from one source to another within a 5-year period if found cost-effective. This can be illustrated by looking into 5-year instead of 10-year outcomes. Thirdly, consumers and producers may have different criteria for preferred technologies. For transportation this has already been discussed in van Vliet et al. (2011). Similar issues are relevant for biochemicals, especially for those that are not chemically identical to their assumed fossil-based counterpart. An example is PLA, which has different barrier properties than PET and PE (Groot and Borén, 2010). End-use consumers (e.g. brand owners) may not encourage such a large-scale shift, which may delay the market penetration of the technologies. Drop-in biochemicals (i.e. bio-based replacements identical to fossil-based chemicals such as ethylene from ethanol) are likely to be less subjected to this (de Jong et al., 2012a). However, other sustainability criteria (e.g. labour conditions, genetically modified feedstocks) may still form a barrier to large-scale market diffusion.

The existing modelling framework could benefit from decoupling the domestic demand for chemicals from the overall production in the Netherlands, as this would align the demand across all end-uses of energy. It would also enable better representation of organic waste flows from bio-based materials consumed in the Netherlands into MSWI and would allow incorporating end-of-life policies such as recycling and/or incineration in the model. Closely related are the higher biomass efficiency gains that can be obtained if cascading of biomass is applied to the system, namely the prioritised consumption of biomass for high-value applications such as materials and chemicals, followed by reuse and recycling before being finally consumed for energy (Keegan et al., 2013). Given the regional boundaries of this study, detailed material flows are required to capture prospective domestic consumption of bio-based materials, reuse practices, recycling and end-of-life practices, as well

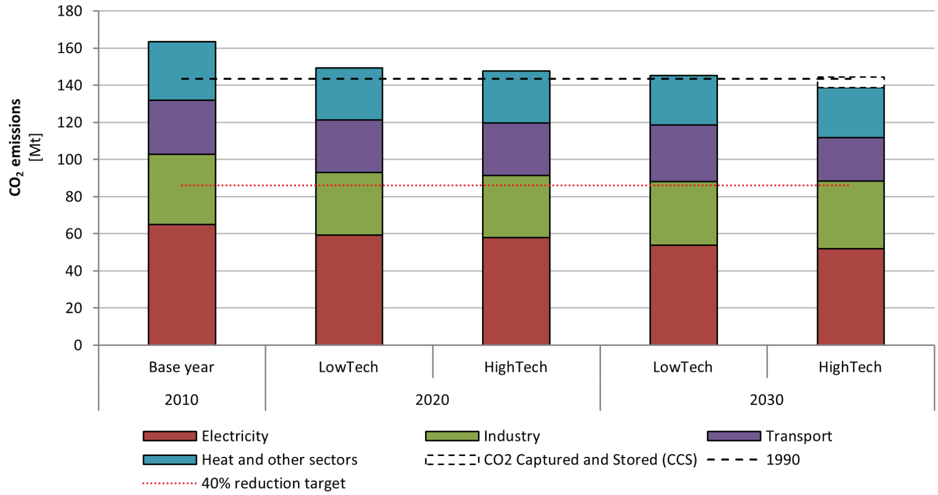

Figure 4.17 Direct CO_2 emissions in the Netherlands in 2010-2030

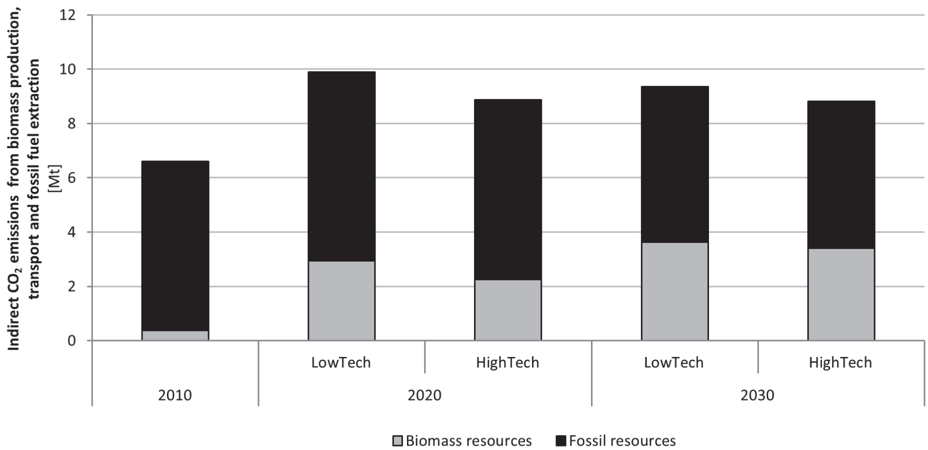

Figure 4.18 Indirect CO_2 emissions from biomass and fossil resources imported and consumed in the Netherlands in 2010-2030

as the corresponding policies. While these challenges have not been addressed by this study, they are recommended for future research and model improvement.

Another aspect that has been excluded from the modelling framework is implementation and competitiveness of energy efficiency measures that can reduce energy demand, especially in sectors where biomass was shown to play a key role, e.g. industrial heat. To some extent, this has been addressed by autonomous efficiency improvements in the chemical industry and the decrease in the demand for energy due to existing efficiency measures. However, more stringent energy efficiency may contribute beyond to what is implicitly included in this study and may influence the cost-competitiveness of renewable energy supply technologies.

Furthermore, oil refineries have not been explicitly modelled. Crude oil is assumed an imported commodity and is converted to diesel, petrol, kerosene and naphtha by ignoring their production co-dependency in refineries. A reduction in fossil fuels, as the results of this study suggest, would entail a reduction in crude oil refining with a consequent reduction in the output of naphtha or other refinery chemicals, and potentially also lower prices due to lower demand. It is recommended as an improvement to the model to better represent the interrelation between on the one hand fuels and naphtha from refineries and on the other hand crude oil and price correlations. This suggestion involves intensive data collection on refinery cost-structures and stock in the Netherlands, which is a rather complex task.

4.5.2 Technology selection and data limitations

The technologies included in this study are not exhaustive regarding the several bio-based chemical conversion pathways that exist or are being developed (section 4.2.2.1). However, the selected routes can be considered most representative as they are currently produced on a large scale (e.g. PLA and ethanol-based ethylene), are in the ramp-up phase (e.g. SA) or are promising for the future with an expected CARG to 2020 of over 10% (Dammer et al., 2013). Besides high-value/low-volume biochemicals (section 4.2.2.1), chemicals have been excluded if they were in the early phases of development, e.g. algae-based or fatty acid-based chemicals and lignin-based aromatics. An exception is the production of polyhydroxyalkanoates (PHA), which are already at an early commercial stage. However, the available data for PHAs are limited. PHAs have an estimated high growth potential and may lead to significant emission reduction. However, similar to many other pathways of biomass to materials, they have higher production costs than other polymers (de Jong et al., 2012a).

One shortcoming of the technologies included in the present study is the limited representation of conversion routes to aromatics. Model outcomes suggest the supply of aromatics from oil refineries at assumed cost-prices. As a result, if future biomass conversion routes to aromatics have lower production costs than refinery chemicals, additional fossil fuel replacement can take place. One option would be to incorporate isobutanol from fermentation or aqueous phase reforming and subsequent conversion to paraxylene. However, literature shows that the production costs of paraxylene are 2,885-4,121 \$/t, which is approximately 3-4 times more expensive than fossil-based aromatics (Lin et al., 2014). Therefore this technology was not included, as it would not compete with alternatives. Gasification and conversion of syngas to FT-fuels was shown to be a key technology in the model results of HighTech, demonstrating synergies between the transport and electricity sector. Supply of FT-naphtha to the chemical sector was not shown as a cost-effective option, due to the additional capacity of steam cracking required for the conversion of naphtha to olefins. Direct conversion of syngas to olefins could potentially offer benefits by avoiding this intermediate step (Torres Galvis et al., 2012). Lignin valorisation technologies, although early in technology readiness, could become an interesting alternative beyond 2030 as large-scale lignocellulosic sugar and ethanol production is included in the technology portfolio, where lignin is produced as a co-product. Potential technologies go beyond the ones addressed above. A key constraint to extend the technology portfolio is data availability. In this study we were faced with difficulties in estimating reliable and verifiable cost-structures. To be able to assess bio-based chemical conversion technologies in a systems analysis framework, more data should become available. This requires action from bioeconomy stakeholders, including the industry, which typically hold such information. If more data is available, expanding the temporal scope of the model beyond 2030 can be deemed feasible and more insights can be derived.

Additional technologies do not only relate to the bio-based chemical sector. More technology options could be explored, such as the production of synthetic natural gas from biomass gasification in different locations (e.g. Ukraine). However, in a cost-optimisation model, such options could dominate the supply since they may potentially have lower production costs than natural gas (Batidzirai, 2013). Nevertheless, infrastructure, technology deployment (gasification), markets and so on are not expected to be fully operational in 2030. In addition, other industrial sectors where biomass can be utilised are excluded from this study: synthetic fibres, composite materials, natural rubbers and traditional users of biomass such as pulp and paper, construction, and charcoal use in iron and steel industries.

A major determinant of the deployment of biomass conversion technologies is the cost-supply of biomass. In this study biomass was disaggregated to feedstocks and regions to define detailed biomass cost-supply curves; biomass costs are a major determinant of production costs, and therefore of the deployment of conversion technologies. Nevertheless, modelling of biomass cost-supply could be further improved. For example, feedstock-specific logistics can be applied as a proxy instead of wood chips, thus improving the representation of transport costs from the sourcing regions to the Netherlands. Furthermore, pretreatment methods such as pelletisation or torrefaction could also take place in the sourcing regions, which will increase biomass production costs but may well significantly reduce transport costs to the Netherlands, thus increasing cost-efficiency and stimulating the deployment of biomass conversion technologies (Batidzirai et al., 2014; Uslu et al., 2008).

4.5.3 Uncertainty and sensitivity analysis

The primary goal of this study was to design and apply a modelling framework, which accounts for competitive and synergetic uses of biomass for energy and non-energy applications. To assess potential deployment pathways of biomass conversion technologies, we developed two scenarios that account for a key future uncertainty: the rate of technology development.

However, there are several exogenous parameters that may influence model outcomes, which need to be assessed prior to providing robust directions to policy making. Key uncertainties that have not been assessed by this study include variation in fossil fuel prices (Table 4.17, section 4.7) and their impact on biomass costs, variation of CO_2 emission mitigation policies such as high CO_2 taxation or an emission cap, constrained biomass supply, with the Netherlands having access to only intra-EU resources, or conversely access to low-cost biomass feedstocks from regions outside the EU (such as Ukraine), stricter sustainability constraints and specific support to technologies. Furthermore, complete closure of coal-based power plants based on a government decision can significantly influence the fuel mix for electricity generation and the CO_2 emission performance of the system. Additional scenario and sensitivity analysis may provide greater insight if applied to the modelling framework of this study.

4.6 CONCLUSIONS

This paper describes the design and application of a modelling framework that accounts for synergistic and competitive uses of biomass conversion technologies in the energy system. A cost-optimisation model was extended with emerging sectors and conversion technologies of the bioeconomy, namely biochemicals and RJF. We incorporated detailed cost-supply curves of several types of biomass production and transport, included a substantial number of prospective technologies for biofuel, biochemical and RJF production, and linked biorefineries that valorise different biomass constituents and other multi-output processes with different end-use sectors. This model can assess the cost-efficient deployment of biomass in the energy system of a country, in competition with other renewable energy and CO_2 emission mitigation technologies (e.g. CCS). Parameterisation includes other aspects such as renewable energy (e.g. EU RED) and climate policies (e.g. CO_2 taxation), which help to assess the required biomass volumes, the contribution of different renewable energy technologies and sectors to meet targets, their impact on CO_2 reduction and the lack of a level playing field, which biochemicals may face due to mandatory policies in the energy sectors.

Two scenarios were applied to address uncertainty in future technology progress. The modelling framework was applied to the Netherlands; it takes targets into account from national (Dutch Energy Agreement) and European (EU RED) policies up until 2030.

So far the non-energy sector, and in particular the chemical industry, has been omitted from most mid-term, cost-optimisation energy systems modelling. Nevertheless, its importance to chemicals, in view of emerging biomass conversion technologies, the potential synergies with other sectors through valorisation of different biomass constituents, and also the competition with other renewables across the energy sectors make this study one of the first endeavours to shed light on cost-effective uses of biomass in the bioeconomy. Multi-output processes (biorefineries) have been incorporated, by linking the supply of biomass constituents to different uses (e.g. sugars to ethanol, lignin to heat/electricity). Biochemical refinery products, such as lignocellulosic sugar and ethanol, or thermochemical refinery products such as FT-fuels or naphtha, can supply multiple sectors. For example, ethanol can be supplied to road transport or to the chemical industry, and FT-fuels can be supplied to the road transport or aviation sector. In this manner, uses are highlighted that are cost-effective from a systems perspective.

The results show that policy targets are a key driver for the deployment of renewable energy technologies. In meeting targets, biomass contributes significantly, especially in sectors where there are limited renewable alternatives, such as heat and fuels. In addition, electricity from wind was found to be a key contributor. Under low technology development and with policy targets as the primary driver for renewable energy deployment, low value applications of biomass play a major role. In contrast, rapid technological progress enables bio-based growth of the transport sector, primarily through the supply of FT-fuels, and of the chemical sector, through the supply of diverse biochemicals. For co-produced electricity and a supply of biofuels beyond the EU RED's blending mandate, lower value conversion routes such as biomass heat are limited when compared to low technology development scenarios. This indicates that if technology development is accelerated, biomass can offer cost-competitive alternatives without support from policies on biochemicals.

Furthermore, accelerated technology development enables the production and supply of RJF through gasification and FT-synthesis pathways beyond 2020; under slow technology development, this would require incentives (such as a blending target or subsidies). Biochemicals are found cost-competitive from 2020 onwards. Scenario comparison reveals different volumes across the time horizon. Sugar imports and advanced biorefineries for lignocellulosic sugar production supply the necessary low-cost feedstocks to bio-based chemical conversion technologies, which produce significant volumes of chemicals (up to 1.95 Mt). Without strong technology development, biochemicals depend primarily on the cost-supply of imported sugar. Therefore, preconditions for the deployment of biochemicals are technology development and access to low-cost feedstocks. Moreover, non-energy use from biomass may in the long term make a significant contribution to final energy consumption despite the uneven playing field created by binding renewable energy targets. The current policy framework does not include bio chemicals, thus possibly delaying early deployment. Biorefineries are shown to drive biomass consumption and bio-based energy and chemical supply, as together they consume approximately one third of total biomass consumption in HighTech in 2030.

The dependency of the Netherlands on imports indicates that supply chains need to be developed that can ensure access to low-cost biomass feedstocks. Depending on the technology development scenario, the growth in biomass consumption and biomass types varies; high technology development consumes large quantities of biomass equivalent to 18 Mt_{wpe}, which would require significant efforts in infrastructure and logistics. At the same time, different types and volumes of feedstock such as agricultural residues were valorised by biorefineries in the HighTech scenario.

This study revealed that renewable energy deployment (primarily wind) and biomass (through heat, biofuels and CCS in gasification technologies) reduce CO_2 emissions over time in comparison to the base year (2010). However, CO_2 mitigation targets are not met under the assumed scenario conditions (fossil fuel prices, CO_2 taxation, national and European renewable energy targets). Therefore, greater efforts are required to achieve emission reduction targets and to highlight the potential contribution of biomass, e.g. a mandatory cap or higher CO_2 emission taxation. The representation of biomass supply chains can be improved by assuming biomass densification (pelletisation, torrefaction), which improves cost-efficiency of biomass value chains due to lower logistic costs and increased conversion efficiencies. In addition, the technology portfolio could be enriched as there are several conversion technologies, especially for biochemicals, that can offer alternatives. The product portfolio can also be extended to include high-value chemicals as opposed to the bulk products that are mainly included in this study. Furthermore, other systemic aspects such as biomass cascading, competition induced by other policies (e.g. on energy efficiency) and technical aspects such as process integration of technologies, could lead to significant improvements of the model presented in this study. Despite these limitations, this is one of the first endeavours to address cost-efficient value chains of biomass in a regional model for the medium term. Finally, the study revealed important cross-sectoral synergies in meeting policy targets and shifts of biomass feedstocks in different sectors based on the technology development scenarios. However, further research is required as to which are the most important parameters, in order to provide concrete directions for competitive uses of biomass in the medium term together with recommendations for policy makers.

4.7 APPENDIX

Box 4.1 Methods and tools for bioeconomy assessment

The petrochemical industry in MARKAL-NL-UU is structured as follows. Refineries supply to the petrochemical industry oil naphtha, propylene and aromatics. In the model, they are all assumed as import commodities at set cost-prices. In addition, some intermediate chemicals (e.g. ethylbenzene; EB) are assumed to be partly imported for further downstream processing. That is because the production capacity of these chemicals in the Netherlands does not suffice to cover the production capacity of their direct derivatives (e.g. styrene). Therefore they are also assumed as traded –and in this case imported– commodities.

- Naphtha is supplied to steam crackers and is converted to olefins and aromatics. Through downstream technologies they synthesise intermediate chemicals such as EO and ethylene glycol (EG), which in turn are used to produce final products (e.g. polyethylene terephthalate (PET), styrene). As the intermediate EB is also produced within the boundaries of the Dutch petrochemical industry, together with the imported flow they make up the total volume available for downstream processing in the country. Also, final products such as polyethylene (PE) and polypropylene (PP) are directly produced by olefins without requiring intermediate chemicals.
- Alternative conversion technologies produce Fischer-Tropsch (FT) naphtha from biomass, which competes with oil naphtha at the feedstock level. A number of alternative conversion technologies (thermochemical, biochemical, catalytic) can supply olefins either from coal (excluded from this study) or biomass. The olefins produced by these technologies compete with production chains of basic chemicals from naphtha crackers and propylene and aromatics supplied from refineries; for further downstream processing they use the same technologies as conventional basic chemicals. As intermediate chemical process we include the fermentative production of succinic acid (SA). SA apart from being considered as a platform chemical it can replace chemical intermediates such as phthalic anhydride (PA) and 1,4-butanediol (BDO). This entails that SA can substitute the production chain of PA and BDO starting from feedstock and including all downstream processing steps to these chemicals. Finally, at the product level, we include fermentation of sugars to polylactic acid (PLA) and 1,3-propanediol (PDO). PLA is assumed to have similar functionality with PET and PE. PDO after esterification with terephthalic acid (PTA) produces polytrimethylene terephthalate (PTT), which is assumed to have the same functionality as PET. Therefore, PLA and PDO can replace the production chain of PE and PET. This also implies that the demand for aromatics, required to produce PET, is reduced. Sugar is also used to produce furandicarboxylic acid (FDCA), which replaces PTA for PET production.
- Natural gas is converted to hydrogen via steam reforming and is synthesised to ammonia via the Haber-Bosch process. We include alternative routes for hydrogen production through biomass gasification. Finally, ammonia is converted to urea while the remaining fraction is used for production of other nitrogen fertilisers.

Box 4.2 Methods and tools for bioeconomy assessment

The conventional supply chain of polyethylene terephthalate (PET) requires import of oil naphtha from refineries to produce ethylene by steam cracking. Ethylene is then converted to ethylene oxide (EO), which is hydrolysed to ethylene glycol (EG). In addition, aromatics either from refineries or from steam crackers are used to produce purified terephthalic acid (PTA), which is esterified with EG to PET. The alternative conversion technologies included in the model introduce the following possibilities: (a) produce Fischser-Tropsch (FT) naphtha instead of oil naphtha being supplied by refineries and use the existing downstream routes to PET, (b) produce ethylene from ethanol instead of ethylene by oil or FT-naphtha and then use the existing downstream routes to PET, (c) produce methanol which is catalytically converted to ethylene and use the existing downstream routes to PET, (d) ferment sugars to 1,3-Propanediol (PDO) and then produce polytrimethylene terephthalate (PTT) as PET substitute thus avoid the existing downstream routes to PET (i.e. EO and EG production), (e) catalytically convert pyrolysis oil to ethylene and xylene and use the existing downstream routes to PET, (f) ferment sugars to PLA as a PET substitute thus avoid the existing downstream routes to PET (g) Produce furandicarboxylic acid (FDCA) from sugars thus avoiding PTA production (polyethylene furanoate (PEF) requires EG) (h) Produce aromatics and ethylene through gasification. In this example, routes (a)-(d) still require production of fossil PTA, route (f) produces a fully bio-based PET replacement, route (g) produces bio-based PTA substitute but still requires production of ethylene (fossil-based or bio-based), while route (e) does not require PTA production.

Table 4.8 Non-energy use of fossil energy in the Netherlands in 2012 (CBS, 2014)

[PJ]	Coal and coal products	Crude and petroleum products	Natural gas	Electricity
Food and beverages	0.2			
Paper and printing		1.3		
Organic basic chemicals	3.0	463.0	8.8	
Fertilisers			64.2	
Other basic chemicals		0.7	11.0	
Other inorganic chemicals	0.6	11.3		6.1
Chemical and pharmaceutical products		2.3		
Iron and steel	60.5			
Non-ferrous metals				7.8
Metal products, machinery		14.9		
Other manufacturing and repair				0.0
Transport		1.9		
Building materials			0.2	
Construction		8.4		
Non-specified[a]	2.6	3.1		

[a]Estimated by deducting from the total non-energy use of each fuel type the non-energy use per fuel type and sector as reported by CBS (2014).

Table 4.9 Top energy consuming and CO_2 emitting chemical products based on global production volumes according to IEA (2013)

Chemical	Modelled explicitly in MARKAL-NL-UU	Chemical	Modelled explicitly in MARKAL-NL-UU
Ethylene		Aromatics	
Ethylene	Yes	Benzene	Yes
Ethylene glycol	Yes	Caprolactam[a]	No
Ethylene oxide	Yes	Cumene[a]	No
Polyethylene	Yes	Mixed xylene	Yes
Vinyl chloride monomer	No	Phenol[a]	No
Propylene		Paraxylene	Yes
Acrylonitrile	No	Styrene[a]	Yes
Polypropylene	Yes	Terephthalic acid[a]	Yes
Propylene	Yes	Toluene	Yes
Propylene oxide	Yes	Other	
		Ammonia	Yes
		Methanol	No[a]

[a]These chemicals are primarily produced by aromatic compounds, although for their synthesis olefins are also required. In this table they are listed under aromatics for categorisation purposes. [b]Methanol is modelled as potential transport fuel and feedstock for olefins production.

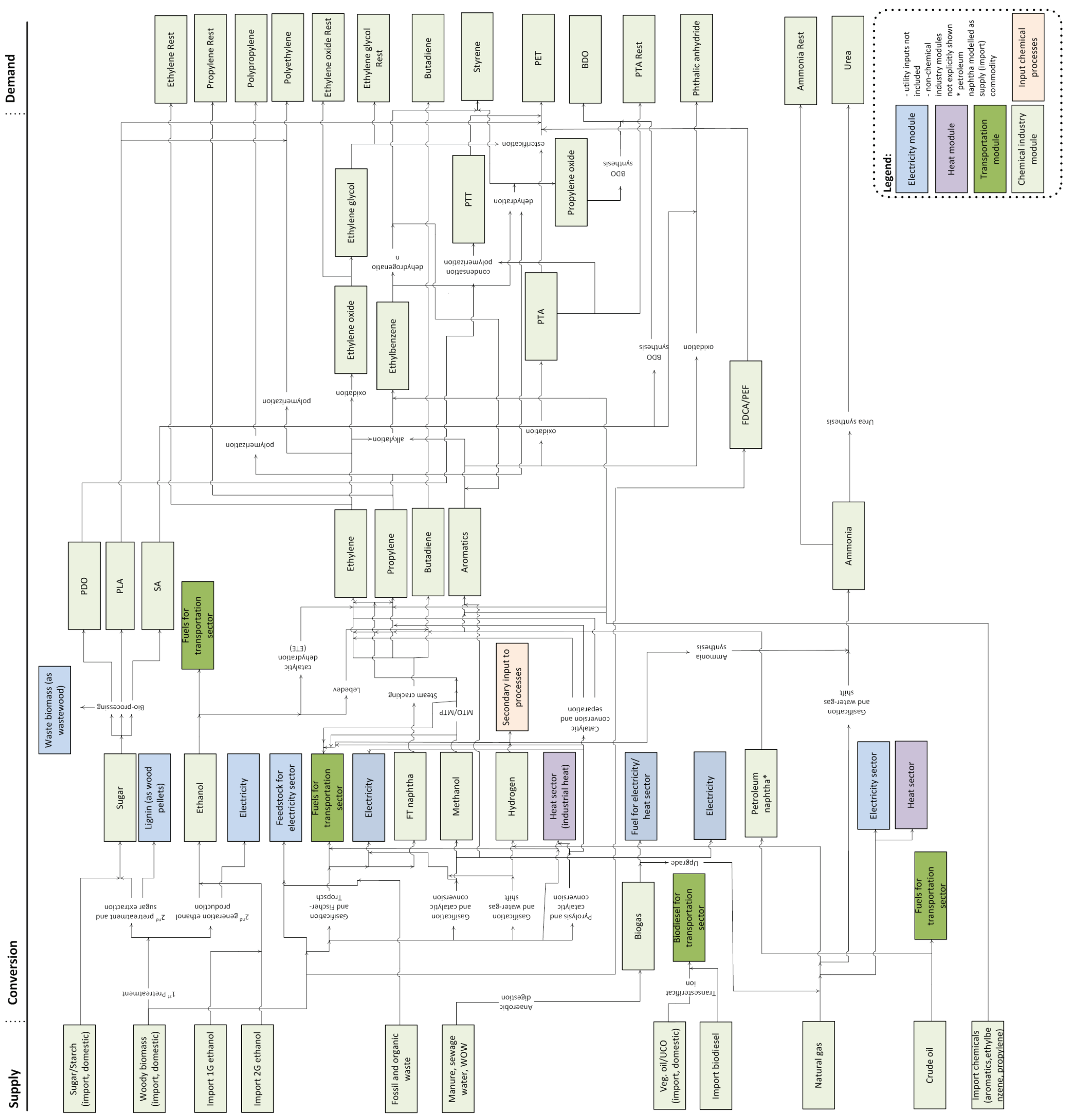

Figure 4.19 Detailed chemical industry module in MARKAL-NL-UU (version August 2015)

Table 4.10 Growth rates of chemicals assumed in this study estimated from (IEA, 2009)

	2006-2030
Olefins/aromatics	1.50%
Other chemicals[a]	1.60%
Ammonia	1.80%

[a]In IEA (2009) the growth rate represents methanol and is estimated only for Germany.

Table 4.11 Basic chemical consumption by chemical intermediate in Western Europe between 1994 and 2013 (Petrochemicals Europe, 2014)

Chemical[a]	Minimum [kt/yr]	Maximum [kt/yr]	Minimum[b]	Maximum[b]
Ethylene				
Ethylbenzene[c]	1,105	1,659	6%	8%
Ethylene dichloride	2,526	3,000	13%	16%
Ethylene oxide[c]	1,658	2,447	9%	12%
Other	947	2,210	5%	11%
Polyethylene[c]	9,868	12,710	56%	61%
Vinyl acetate monomer	237	474	1%	3%
Propylene				
Acrylonitrile	649	1,298	0%	11%
Cumene	767	1,180	6%	8%
Other	1,457	3,069	10%	21%
Oxo alcohols	413	1,298	3%	10%
Polypropylene[c]	5,548	9,325	47%	60%
Propylene oxide[c]	1,062	1,948	9%	13%
Aromatics (Benzene)[d]				
Alkylbenzene	195	684	3%	9%
Cumene	1,368	2,215	21%	28%
Cyclohexane	684	1,140	9%	14%
Ethylbenzene[c]	3,192	4,430	45%	53%
Nitrobenzene	521	1,205	8%	13%
Other	0	228	0%	3%

[a]Raw data were not available. Data presented in this table were extracted from the figures presented in Petrochemicals Europe (2014). [b]Minimum and maximum share estimated based on the lowest and highest consumption by weight of basic chemical for the respective chemical intermediate between 1994 and 2013.[c]Modelled explicitly in MARKAL-NL-UU. Residual downstream chemicals are modelled indirectly as rest demand for basic chemicals. [d]Information on toluene and xylene were not available.

Table 4.12 Residual capacity of basic chemicals and biofuels in the Netherlands

Basic chemical	Residual capacity in 2010 [Mt]
Ethylene	1.17
Propylene	0.63
Butadiene	0.6
Ethylene glycol	0.09
Terephthalic acid	0.17
Ammonia	2.38
Transesterification, biodiesel	1.67
HEFA	0.8
Fermentation, ethanol	0.72

Table 4.13 Production capacity of chemicals in the Netherlands

Chemical[a]	Process	Capacity [Mt]	Year	Reference
Ethylene				
Ethylene	Naphtha cracking	3.97	2011	OGJ (2012)
Ethylene glycol	Hydrolysis	0.16	2006	ICIS (2006)
Ethylene oxide	Oxidation	0.47	2008	Chemweek (2000-2009)
Polyethylene	Polymerisation	1.92	2009	Chemweek (2000-2009)
Propylene				
BDO	Propylene oxide isomerisation, hydroformylation, hydrogenation	0.13	2006	ICIS (2006)
Polypropylene	Polymerisation	0.78	2006	ICIS (2006)
Propylene	Naphtha cracking	2.06	2011	Estimate[b]
Propylene	Gas oil FCC	0.26	2008	Estimate[c]
Butadiene				
Butadiene	Naphtha cracking	0.61	2011	Estimate[b]
Aromatics				
Aromatics	Naphtha cracking	1.18	2011	Estimate[b]
Aromatics	Catalytic reforming	2.16	2008	Estimate[d]
Ethylbenzene	Alkylation	1.21	2000	Neelis et al. (2007a, 2007b), Neelis et al. (2003) Estimate[e]
PET	Esterification	0.22	2006	Chemweek (2000-2009)
Phthalic anhydride	Catalytic conversion of ortho-xylene	0.08	2006	ICIS (2006)
PTA	Paraxylene oxidation	0.35	2006	ICIS (2006)
Styrene	Ethylbenzene dehydrogenation	0.64	2006	ICIS (2006)
Styrene	Ethylbenzol dehydration	1.56	2006	ICIS (2006)
Other				
Ammonia	Steam reforming and Haber-Bosch	2.91	2006	Lako (2009)
Urea	Ammonia and CO_2 synthesis	1.23	2006	ICIS (2006)

[a]Chemicals are categorised under the key basic chemical they are produced from. Note, however, that most intermediate chemicals or final products synthesised from different basic chemicals. [b]Based on ultimate yields of naphtha steam crackers in Neelis et al. (2005). [c]Based on FCC capacity (OGJ, 2009) assuming a $10.5\%_{wt,fresh\ feed}$ (Couch et al., 2007). Production volume estimated based on 80% capacity utilisation (VNPI, 2015). [d]Based on catalytic reforming capacity (OGJ, 2009), 0.85 t reformate/t heavy naphtha, 2 t reformate/t of catalytic reforming products and 68% aromatics yield (Neelis et al., 2007a, 2007b; Neelis et al., 2003). [e]Production capacity of 2000 assumed the same for 2006.

Table 4.14 Location factors (Broeren et al., 2014)

Location factor	Region
118.2	Asia Pacific (Japan, Korea)
100	Europe (France, Germany, Italy, Netherlands, Poland, UK)
90.9	North America (United States and Canada)
81.8	China and India
113.6	Middle East and Africa (Saudi Arabia, Iran, Kuwait, Egypt)
113.6	Other Developing Asia (Indonesia, Thailand, Malaysia, Singapore, Taiwan)
113.6	South America (Brazil, Venezuela, Argentina, Colombia, Trinidad, Chile)
113.6	Transition economies (Russia, Ukraine)

Table 4.15 Process yields of basic chemicals to derivatives based on Neelis et al. (2007a, 2007b) and Neelis et al. (2003)

t basic chemical/t product	[t ethylene]	[t propylene]	[t aromatics]
Polyethylene	1.035	0	0
Ethylene oxide	0.88	0	0
Ethylene oxide for PET	0.24	0	0
Ethylene glycol	0.71	0	0
Ethylene glycol for PET	0.24	0	0
PTA	0	0	0.68
PTA for PET	0	0	0.58
PET	0.24	0	0.4
Polypropylene	0	1.01	0
Ethylbenzene	0.27	0	0.76
Stryrene	0.29	0.21	0.78
Ethylbenzene for styrene	0.31	0	0.79
Propylene and ethylbenzene for styrene	0.32	0.34	0.9
Propylene oxide (by-product)		Formed as by-product[a]	
Propylene oxide for BDO		0.84 t PO/t BDO	
Phthalic anhydride	0	0	0.92

[a]Based on 0.39 t PO/t EB via the ethyl benzol route.

Table 4.16 Fixed cost components based on Hermann and Patel (2007)

Fixed cost component		
Operating supplies	10%	of operating labor
Maintenance supplies	1.50%	of ISBL
Maintenance labor	2.50%	of ISBL
Laboratory labor	13%	of operating labor
Taxes and insurance	3%	of ISBL and OSBL
Plant overhead	80%	of total labor
G&A	6%	of plant gate cost (variable other fixed costs)

Table 4.17 Feedstock, energy and material prices used in MARKAL-NL-UU

Product	Cost/Price	2010	2015	2020	2025	2030	Reference
Crude oil	€/GJ_{LHV}	10.19	11.53	9.51	11.47	13.43	IEA (2015)
Coal	€/GJ_{LHV}	3.3	2.4	2.8	3.0	3.1	OECD/IEA (2015)
Natural gas	€/GJ_{LHV}	5.331	6.02	5.048	6.148	7.248	OECD/IEA (2015)
Naphtha	€/GJ_{LHV}	11.2	12.7	10.5	12.6	14.8	Estimate[a]
CO_2 – New policies	€/t	5	10.03	15.07	20.2	25.3	OECD/IEA (2015)
Propylene (FCC)	€/t	606	685	565	681	798	Estimate[c]
Refinery aromatics	€/t	606	684.7	564.7	681.1	797.6	Estimate[c]
Ethylbenzene (import)	€/t	1,144	1,144	1,144	1,144	1,144	ICIS (2008)
Uranium oxide	€/GJ	1.93	2.05	2.17	2.31	2.45	MARKAL-NL-UU

[a]Asche et al. (2003) mention that changes in oil prices are only partly reflected in naphtha price as opposed to other fuel grade refinery products. This is also inferred by the large variation in naphtha refining margins, which based on Argus Media (2012), is 1-20% compared to oil price. In the present study we assume a price margin for oil refining to naphtha of approximately 85 $/t, which is equivalent to 10% of price difference between oil and naphtha. However, applying a constant crack spread of oil-to-naphtha might not be appropriate for estimating future prices (McKinsey, 2012). [c]Similarly, for refinery chemicals we assume a price margin of 20% compared to oil price.

Table 4.18 Targets affecting energy supply in the Netherlands based on the Dutch Energy Agreement as incorporated in MARKAL-NL-UU (SER, 2013)

Measure	Quantitative target
Renewable energy share on final energy (2023)[a]	16%
Additional capacity of off-shore wind power (2015)	0.13 GW
Additional capacity of off-shore wind power (2016-2020)	1.65 GW
Total capacity of on-shore wind power (2020)	6 GW
Maximum final energy from biomass for co-firing (2020)	25 PJ
Coal-power plants decommissioning (2016)[b]	1.65 GW
Coal-power plants decommissioning (2017) [b]	1.4 GW

[a]In MARKAL-NL-UU implemented as a binding target for 2025. [b]In MARKAL-NL-UU implemented in 2020 (2.7 GW coal capacity was built in the 1980s -difference with Energy Agreement is 300 MW).

Table 4.19 Final electricity and heat demand in MARKAL-NL-UU

Final energy demand [PJ][a]	2010	2020	2030
Electricity[b]	428.5	430.8	439.5
Co-generated heat, agriculture[c]	61	74	87
Co-generated heat, industry[c]	141	144	133
Co-generated heat, residential and commercial[c]	24	26	29
Rest heat, agriculture[d]	26.3	18.9	10.2
Rest heat, industry[e]	269.5	255.9	292.1
Rest heat, residential and commercial[f]	462.1	412.9	403.4

[a]If additional electricity and heat input is required by deployed technologies this demand is added in the respective sectors. Demand for electricity operates in isolation in order to reduce distortion due to price effects (i.e. imports and exports of electricity are not accounted for. According to ECN (2015) imports and exports of electricity balance beyond 2020. [b]To avoid double counting of electricity for conventional industrial operations which are modelled explicitly in MARKAL-NL-UU (steam crackers, ammonia, downstream processes) we deduct 10 PJ_e from final electricity demand in 2010-2030. [c]Based on Brouwer et al. (2015). [d]Based on final heat use in agriculture, excluding co-generated heat in agriculture. [e]Based on final heat use in industry and water and waste sector, excluding co-generated heat in industry. To avoid double counting of heat for conventional industrial operations (downstream processes) we deduct 60 PJ_{th} from final industrial heat demand in 2010-2030. [f]Based on final heat use in households and services, excluding co-generated heat in households and services and district heating.

Table 4.20 Total available biomass in MARKAL-NL-UU

[TJ]	2010	2020	2030
The Netherlands[a]			
Total Biomass Policies[b]	129,045	134,870	151,912
Total (this study)	128,939	134,692	151,756
Coverage	99.9%	99.9%	99.9%
Allocated to the Netherlands from EU28 (excluding the Netherlands)			
Total Biomass Policies[b]	428,042	499,866	546,526
Total (this study)	322,338	392,721	427,421
Coverage	75%	79%	78%

[a]Fuelwood for wood stoves is added ad hoc to the total domestic potential in addition to the total potential mentioned in this table. [b]Elbersen et al. (2015).

4

Table 4.21 Incremental yield and autonomous annual energy efficiency improvements in chemical conversion technologies of the low and high technology development scenarios

Product	Process	Scenario	
		LowTech	**HighTech**
Cellulosic sugar[a]	Dilute acid	C5-C6 yield: 1% p.a. (max C6: 80%, max C5: 70%)	C6 yield: 2% p.a. (max: 91%), C5 yield: 3% p.a. (max: 90%)
	Steam explosion	C5-C6 yield 1% p.a. (max C6: 80%, max C5: 65%)	C6 yield: 2% p.a. (max: 93%), C5 yield: 3% p.a. (max: 90%)
	Liquid hot water	-	C6 yield: 2% p.a. (max: 98%), C5 yield: 3% p.a. (max: 93%)
Cellulosic ethanol[a]	Dilute acid	C5-C6 yield: 1% p.a. (max C6: 80%, max C5: 70%); C5 fermentation efficiency 1% p.a., C6 fermentation efficiency 0.05% p.a. (max 94%)	C6 yield: 2% p.a. (max: 91%), C5 yield: 3% p.a. (max C5: 90%); C5 fermentation efficiency 2% p.a., C6 fermentation efficiency 0.1% p.a. (max 94%)
	Steam explosion	C5-C6 yield 1% p.a. (max C6: 80%, max C5: 65%); C5 fermentation efficiency 1% p.a.; C6 fermentation efficiency 0.05% p.a., (max 94%)	C6 yield 2% p.a. (max: 93%), C5 yield 3% p.a. (max: 90%); C5 fermentation efficiency 2% p.a., C6 fermentation efficiency 0.1% p.a. (max: 94%)
	Liquid hot water	-	C6 yield 2% p.a. (max: 98%), C5 yield 3% p.a., (max: 93%); C5 fermentation efficiency 2% p.a.; C6 fermentation efficiency 0.1% p.a. (max: 94%)
Ethylene[b]	Catalytic dehydration of ethanol	Annual yield improvement 0.25% (max based on theoretical yield from stoichiometry for each conversion route) Annual energy efficiency improvement 0.5% (max: 10%)	Annual yield improvement 0.5% (max based on theoretical yield from stoichiometry for each conversion route) Annual energy efficiency improvement 1% (max: 20%)c
Succinic acid[b]	Direct crystalisation today		
	Direct crystalisation future		
PLA[b]	Fermentation today		
	Fermentation future		
PDO[b]	Fermentation batch today		
	Fermentation continuous today		
	H_2O pervaporation future		
Olefins[d]	Methanol to olefins (UOP)		
	Methanol to olefins (Exxon)		
Propylene[d]	Methanol to propylene		
Butadiene[b]	Two-stage process ethanol fermentation		

Table 4.21 (continued)

Product	Process	Scenario	
		LowTech	**HighTech**
Basic chemicals[d,e]	Steam cracking	1.0% annual energy efficiency improvement (business as usual)	1.8% annual energy efficiency improvement (best available technology)
Naphtha[d]	Fischer-Tropsch synthesis	Annual yield improvement 0.25%	Annual yield improvement 0.5%
Ammonia	Steam reforming	Annual yield improvement 0.25% (max 20.4 GJ/t, based on future best available technology)	Annual yield improvement 0.5% (max 20.4 GJ/t, based on future best available technology)
		Annual energy efficiency improvement 0.5% (max: 10%)	Annual energy efficiency improvement 1% (max: 20%)
		Maximum yield 20.4 GJ/t (based on future BAT)	
Hydrogen		Annual yield improvement 0.25% (max based on theoretical yield)	Annual yield improvement 0.5% (max based on theoretical yield)
		Annual energy efficiency improvement 0.5% (max: 10%)	Annual energy efficiency improvement 1% (max: 20%)

[a]All improvements assumed from 2015 onward. Core technology components are scaled due to higher sugar output based on the Excel model from Hamelinck et al. (2005). [b]For today technologies, improvements are assumed from 2015 onward. For future technologies improvements are assumed from 2020 onward. [c]This is in line with the Dutch Energy Agreement in which the annual savings in energy consumption are expected to be 1.5% (SER, 2013). [d]Improvements assumed from 2010 onward. [e]Estimate based on Saygin et al. (2013).

Table 4.22 EU28 region classification for biomass supply in MARKAL-NL-UU based on United Nations (UN, 2014)

Northern Europe	**Southern Europe**
Denmark	Croatia
Estonia	Greece
Finland	Italy
Ireland	Malta
Latvia	Portugal
Lithuania	Slovenia
Sweden	Spain
United Kingdom of Great Britain and Northern Ireland	Cyprus
Eastern Europe	**Western Europe**
Bulgaria	Austria
Czech Republic	Belgium
Hungary	France
Poland	Germany
Romania	Luxembourg
Slovakia	Netherlands (excluded from western Europe's potential)

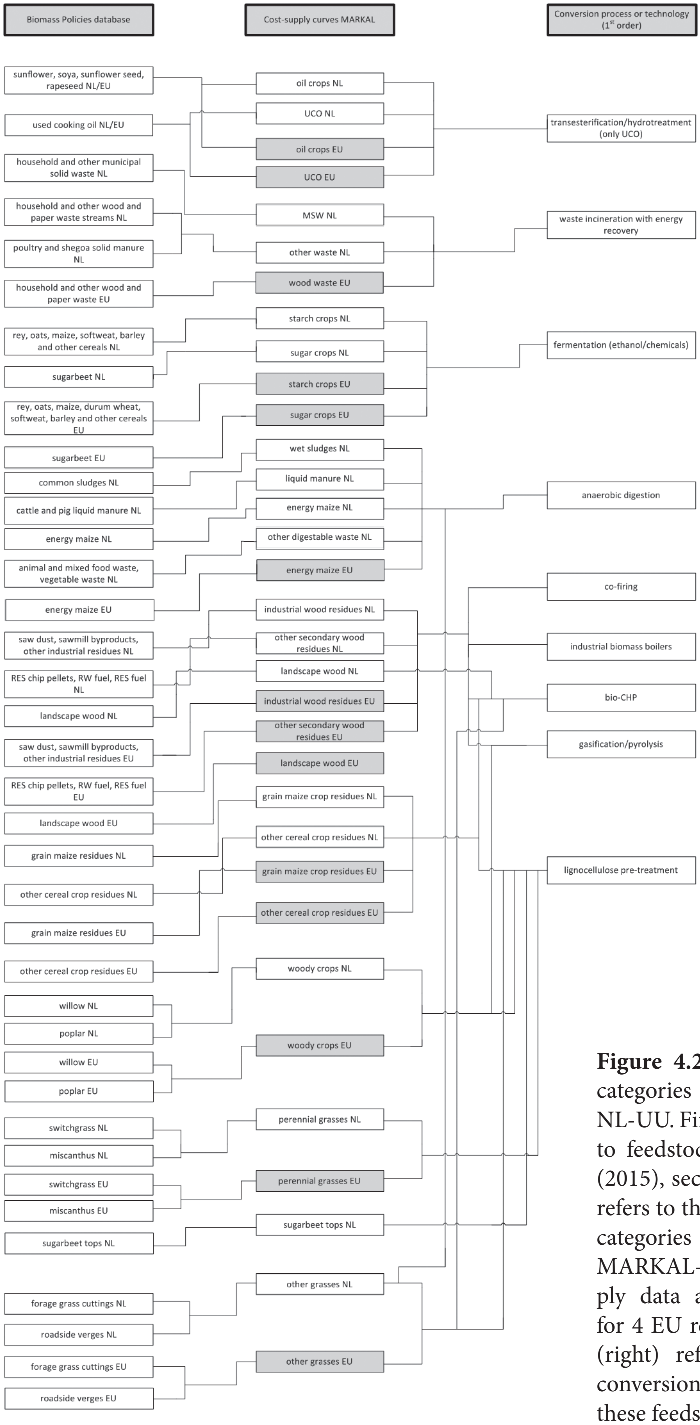

Figure 4.20 Biomass feedstock categories used in MARKAL-NL-UU. First column (left) refers to feedstocks in Elbersen et al. (2015), second column (middle) refers to the aggregation of these categories to feedstocks used in MARKAL-NL-UU (cost-supply data are weighted average for 4 EU regions), third column (right) refers to categories of conversion technologies where these feedstocks can be supplied

Table 4.23 Biomass availability and cost of supply for the Netherlands and four EU28 regions (excluding Netherlands) for energy and non-energy applications used in MARKAL-NL-UU

	Biomass availability [PJ]			Cost of supply [€/GJ]			of which transport costs [€/GJ]		
	2010	**2020**	**2030**	**2010**	**2020**	**2030**	**2010**	**2020**	**2030**
Oil crops NL	0.2	0.1	0.5	16.3	18.2	27.0	1.0	1.0	1.0
Oil crops East EU	3.2	4.7	5.7	14.7	16.7	24.8	5.5	6.1	7.0
Oil crops North EU	1.3	1.4	2	16.2	17.1	26.0	4.8	5.0	5.0
Oil crops South EU	0.8	1.2	1.5	19.4	20.0	26.9	7.0	7.0	7.4
Oil crops West EU	7.8	11.2	11.6	18.2	20.6	20.9	3.6	3.8	4.1
Used Cooking Oil NL	2.1	2.5	2.8	8.5	8.5	8.6	0.7	0.7	0.7
Used Cooking Oil East EU	0.4	0.4	0.4	13.2	13.1	13.6	5.3	5.2	5.7
Used Cooking Oil North EU	0.6	0.6	0.7	12.2	12.2	12.4	4.7	4.7	4.9
Used Cooking Oil South EU	1.7	1.8	1.9	14.0	13.9	14.4	6.3	6.2	6.7
Used Cooking Oil West EU	1.8	1.8	1.9	10.9	10.9	11.1	3.1	3.1	3.3
Sugar crops NL	0.3	2.4	3.6	19.3	19.5	21.1	0.9	0.9	1.0
Sugar crops East EU	0.1	0.5	0.7	22.3	24.0	24.8	3.9	5.3	4.7
Sugar crops North EU	0.1	0.4	1.4	23.1	23.7	24.7	3.6	4.9	4.5
Sugar crops South EU		1.2	2.2	0.0	25.3	27.2	0.0	6.6	7.2
Sugar crops West EU	2.6	2.5	2.6	21.2	21.4	21.7	2.8	2.8	3.0
Starch crops NL	0.6	0.7	0.8	9.7	10.9	15.6	0.9	0.9	1.0
Starch crops East EU	1.6	2.8	4.2	13.5	16.6	22.0	5.4	6.3	6.8
Starch crops North EU	1.2	1.2	2.8	13.6	14.8	20.2	4.5	4.4	4.8
Starch crops South EU	2.7	3.9	4.7	17.1	18.7	21.3	5.9	6.2	6.6
Starch crops West EU	2.6	1.7	1.8	13.4	14.7	15.0	4.2	4.4	4.7
Municipal solid waste NL	10.1	8.1	8.6	0.7	0.7	0.7	0.7	0.7	0.7
Solid manure NL	17.2	17.2	17.7	3.8	3.8	3.8	0.8	0.8	0.9
Liquid manure NL	36.5	35.6	35.1	0.7	0.7	0.8	0.7	0.7	0.8
Common sludges NL	6.6	7.7	8.6	0.7	0.7	0.7	0.7	0.7	0.7
Other digestable waste NL	35.1	9.2	10.3	2.4	2.4	2.5	0.7	0.7	0.7
Energy maize NL	0.8	0.4	0.2	2.7	2.7	2.8	1.4	1.4	1.6
Energy maize East EU	0.5	0.1	0	5.6	6.0	6.4	4.7	4.6	5.0
Energy maize North EU	0.2	0.1	0.1	6.6	5.8	6.0	4.5	4.0	4.2
Energy maize South EU	0	0	0	8.2	8.3	8.7	6.6	5.8	6.3
Energy maize West EU	6.5	4.4	4.6	5.0	5.3	5.6	3.3	3.6	3.9
Secondary forestry residues NL	3.2	3.3	3.3	3.3	3.3	3.3	0.7	0.7	0.7
Secondary forestry residues East EU	8	9.6	12.3	8.4	8.4	8.9	5.5	5.5	6.1
Secondary forestry residues North EU	22.1	24.4	27.5	9.4	9.3	9.6	6.4	6.3	6.7
Secondary forestry residues South EU	3.5	4.2	5.2	10.0	9.8	10.2	6.7	6.6	7.0
Secondary forestry residues West EU	14	15.9	16.6	7.2	7.2	7.5	3.7	3.6	3.9
Primary forestry residues and stemwood NL	4.9	5.3	5.8	4.1	4.0	4.1	0.7	0.7	0.8
Primary forestry residues East EU	30.2	31.7	29.9	8.6	8.8	9.2	6.0	6.3	6.8
Primary forestry residues North EU	47	36.8	38.2	10.7	10.5	10.8	6.3	6.2	6.5
Primary forestry residues South EU	26.1	29.3	29.4	10.5	10.3	10.7	6.5	6.5	6.9
Primary forestry residues West EU	45.4	36.4	37.9	7.7	7.6	7.9	3.8	3.7	4.0

Table 4.23 (continued)

	Biomass availability			Cost of supply			of which transport costs		
		[PJ]			[€/GJ]			[€/GJ]	
	2010	2020	2030	2010	2020	2030	2010	2020	2030
Wood waste NL (incineration)	17.2	17.2	21.3	4.2	4.3	4.4	0.7	0.7	0.7
Wood waste East EU (incineration)	2.8	2.8	5	9.0	8.8	9.4	6.0	5.8	6.4
Wood waste North EU (incineration)	5.5	5.5	9.5	2.8	2.9	3.0	1.6	1.7	1.8
Wood waste South EU (incineration)	2	2	3	8.2	8.1	8.5	4.6	4.6	4.9
Wood waste West EU (incineration)	2.4	2.4	2.9	5.4	5.3	5.5	2.4	2.3	2.5
Landscape wood NL	5.8	5.8	8	3.3	3.3	3.4	0.7	0.7	0.8
Landscape wood East EU	6.3	7.3	9.9	7.7	7.7	8.2	5.8	5.7	6.3
Landscape wood North EU	6.5	7.9	10.4	8.3	8.3	8.6	5.9	5.9	6.2
Landscape wood South EU	5	5.6	7.6	9.0	8.9	9.3	6.7	6.6	7.1
Landscape wood West EU	7.5	8.7	9	6.6	6.5	6.9	4.4	4.4	4.7
Agricultural residues NL	3.3	2.7	2.6	6.7	6.3	5.8	0.9	0.9	0.9
Agricultural residues East EU	14	13.9	13.4	8.4	8.1	8.5	5.6	5.5	6.1
Agricultural residues North EU	8.2	7.7	8.1	8.6	8.2	8.2	4.7	4.7	5.0
Agricultural residues South EU	6	5.4	5.5	9.0	8.9	9.2	6.8	6.7	7.2
Agricultural residues West EU	17.8	18	18.8	7.1	6.8	7.1	3.5	3.5	3.8
Short rotation forestry NL		0.9	1.7		9.8	9.5		0.8	0.9
Short rotation forestry East EU	0	7.3	8.4	7.0	8.5	9.0	4.9	6.2	6.7
Short rotation forestry North EU	0	2.7	3	9.0	8.8	9.0	4.6	5.4	5.7
Short rotation forestry South EU	0	2.6	3	13.2	17.4	17.3	6.1	6.9	7.3
Short rotation forestry West EU	0	1.9	2	7.3	8.5	8.8	3.5	3.8	4.1
Perennial grasses NL		8.8	16.5		6.9	6.8		0.9	0.9
Perennial grasses East EU	0.1	20.4	21.7	6.2	8.4	9.0	4.9	7.3	7.9
Perennial grasses North EU	0	1.1	1.3	9.4	9.3	9.3	4.6	4.4	4.6
Perennial grasses South EU	0.1	18.1	20	9.3	11.8	12.0	5.6	6.9	7.3
Perennial grasses West EU	0.3	11.8	12.3	7.1	7.3	7.6	3.6	3.9	4.2
Other grasses NL	2.43	2.54	2.6	1.3	1.2	1.3	0.8	0.8	0.8
Other grasses East EU	0.39	0.39	0.4	5.9	5.8	6.3	5.4	5.3	5.8
Other grasses North EU	0.72	0.77	0.82	5.7	5.6	5.8	5.2	5.2	5.4
Other grasses South EU	0.83	0.88	0.94	7.2	7.1	7.5	6.7	6.6	7.1
Other grasses West EU	3.55	2.54	2.62	4.0	4.0	4.3	3.2	3.3	3.6
Sugarbeet stubble NL	1.76	1.44	1.48	1.6	1.6	1.6	0.9	0.9	0.9

Table 4.24 Extra-EU costs of biomass resources for energy and non-energy applications used in MARKAL-NL-UU

Product	Cost-Price	2010	2015	2020	2025	2030	Reference
Raw sugar	€/t	345.96	273.07	243.78	268.98	275.39	OECD/FAO (2014)
Vegetable oil	€/GJ	19.6	16.1	16.5	16.3	16.6	OECD/FAO (2014)
Biodiesel	€/GJ	16.7	16.7	25.7	32.2	32.2	OECD/FAO (2014)
Ethanol (1G) – HighTech	€/GJ	18.2	18.2	18.2	18.2	18.3	Cardoso et al. (2012), Jonker et al. (2015)
Ethanol (2G) – HighTech	€/GJ	36.9	34.0	21.5	20.5	19.8	Cardoso et al. (2012), Jonker et al. (2015)
Ethanol (1G) – LowTech	€/GJ	29.4	27.3	25.4	24.3	23.6	Cardoso et al. (2012), Jonker et al. (2015)
Ethanol (2G) – LowTech	€/GJ	36.9	34.1	32.4	31.0	30.1	Cardoso et al. (2012), Jonker et al. (2015)
Wood pellets	€/GJ	7.3	7.4	7.4	7.5	7.6	Argus Media, (2015), Hoefnagels et al. (2014a, 2014b)

Table 4.25 Aggregation of biomass types of the IEE project Biomass Policies for the MARKAL-NL-UU biomass cost-supply module

Biomass resource MARKAL-NL-UU	Biomass resource	
	Netherlands	**EU28**
Oil crops	Biodiesel_rape_seed, Biodiesel_soya, Biodiesel_sunflower_seed, osr_sunflower	Biodiesel_rape_seed, Biodiesel_soya, Biodiesel_sunflower_seed, osr_sunflower
UCO	NACE_UFO	NACE_UFO
Sugar crops	Bioethanol_sugarbeet	Bioethanol_sugarbeet
Starch crops	Bioethanol_barley, Bioethanol_durum_wheat, Bioethanol_maize, Bioethanol_oats, Bioethanol_other_cereals, Bioethanol_rey, Bioethanol_softwheat	Bioethanol_barley, Bioethanol_durum_wheat, Bioethanol_maize, Bioethanol_oats, Bioethanol_other_cereals, Bioethanol_rey, Bioethanol_softwheat
MSW	HH_MSW, NACE_MSW	-
Solid manure	shegoa_solid, poultry_solid, cattle_solid	-
Incineration wood waste	HH_Paper, HH_Wood, NACE_Paper, Nace_Wood	HH_Paper, HH_Wood, NACE_Paper, Nace_Wood
Common sludges	NACE_Comslud	-
Liquid manure	cattle_liquid, pig_liquid	-
Other digestable waste	NACE_AnMixfood, HH_AnMixfood, NACE_Vegetal	-
Energy maize	Energy_maize	Energy_maize
Secondary forestry residues	other_industrial_residues, saw_dust, sawmill_byprod	other_industrial_residues, saw_dust, sawmill_byprod
Primary forestry residues and stemwood	Res_chips_pellets, Res_fuel, RW_fuel	Res_chips_pellets, Res_fuel, RW_fuel
Lanscape wood	Landscape_care_wood	Landscape_care_wood
Agricultural residues	all_cereals, grain_maize	all_cereals, grain_maize
Woody crops (SRF)	Poplar, Willow	Poplar, Willow
Perennial grasses	Miscanthus, Switchgrass	Miscanthus, Switchgrass
Other grasses	Forage_grass_cutting, Road_side_verges	Forage_grass_cutting, Road_side_verges
Sugarbeet stubble	sugarbeet	-

Table 4.26 Emission factors used for domestic and regional (EU) biomass production per feedstock type based on JRC analysis according to the method set in COM(2010) 11 and SWD(2014) 259 (JRC, 2014)

Biomass policies feedstock	Proxy emission factor for	Emission factor [gCO_{2eq}/MJ_{feed}]	Biomass policies feedstock	Proxy emission factor for	Emission factor [gCO_{2eq}/MJ_{feed}]
all_cereals	straw pellets	5.1	NACE_AnMixfood		
Biodiesel_rape_seed	rapeseed biodiesel[a]	29	NACE_Comslud		
Biodiesel_soya	rapeseed biodiesel	29	NACE_MSW		
Biodiesel_sunflower_seed	rapeseed biodiesel	29	NACE_Paper		
Bioethanol_barley	wheat ethanol[b]	23	NACE_UFO		
Bioethanol_durum_wheat	wheat ethanol	23	NACE_Vegetal		
Bioethanol_maize	wheat ethanol	23	Nace_Wood		
Bioethanol_oats	wheat ethanol	23	osr_sunflower	rapeseed biodiesel[a]	29
Bioethanol_other_cereals	wheat ethanol	23	other_industrial_residues	wood chips from industry residues	0.3
Bioethanol_rey	wheat ethanol	23	pig_liquid	manure[d]	45
Bioethanol_softwheat	wheat ethanol	23	pig_solid		
Bioethanol_sugarbeet	sugarbeet ethanol[c]	12.4	Poplar	wood chips from SRC (poplar, fertilised)	3.9
cattle_liquid	manure[d]	45	poultry_solid		
cattle_solid			Res_chips_pellets	Average of woodchips and wood pellets (case 2a) from forest residues	7.1
Energy_maize	maize digestate	1.42	Res_fuel	wood chips from forest residues	1.6
Forage_grass_cutting	agri residues	0.9	Road_side_verges	agri residues (high/low density)	0.9
grain_maize	agri residues	0.9	RW_fuel	wood chips from stemwood	1.4
HH_AnMixfood			saw_dust	wood chips from industry residues	0.3
HH_MSW			sawmill_byprod	wood chips from industry residues	0.3
HH_Paper			shegoa_solid		
HH_Vegetal			sugarbeet		
HH_Wood			Switchgrass	average of range for miscanthus	5
Landscape_care_wood	woodchips forest residues	1.6	Willow	wood chips from SRC (poplar, fertilised)	3.9
Miscanthus	average of range for miscanthus	5			

[a]Converted to feedstock emissions using 1.7 MJ_{feed}/MJ_{fuel} efficiency, excluding allocation for byproducts [b]Converted to feedstock emissions using 1.9 MJ_{feed}/MJ_{fuel} efficiency, excluding allocation for byproducts [c]Converted to feedstock emissions using 1.3 MJ_{feed}/MJ_{fuel} efficiency, excluding allocation for byproducts [d]Emissions assumed equal to credit given to manure treatment.

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Table 4.27 Cost-structures of electricity production technologies (Brouwer et al., 2015; van den Broek et al., 2011)

Cost parameter	Technology	2010	2020	2030
Investment costs [€/kW]	NGCC	722	650	650
	PC	1708	1589	1547
	IGCC	2142	1921	1807
	NGCC-CCS	1225	1084	1002
	PC-CCS	2721	2488	2255
	IGCC-CCS	2959	2537	2276
	Wind on-shore	1230	1165	1159
	Wind off-shore	3431	2425	2040
	Nuclear	2834	2834	2834
	PV	2080	1415	1117
	NGCC	20	18	17
Fixed O&M costs [€/kW]	PC	82	77	71
	IGCC	76	71	64
	NGCC-CCS	35	26	24
	PC-CCS	102	87	80
	IGCC-CCS	98	81	75
	Wind on-shore	18	18	18
	Wind off-shore	74	62	52
	Nuclear	71	71	71
	PV	21	20	18
Variable O&M costs [€/GJ]	NGCC	0.0	0.0	0.0
	PC	0.4	0.4	0.4
	IGCC	0.3	0.3	0.2
	NGCC-CCS	0.4	0.4	0.4
	PC-CCS	1.4	1.3	1.2
	IGCC-CCS	0.5	0.4	0.3
	Wind on-shore	0.0	0.0	0.0
	Wind off-shore	0.0	0.0	0.0
	Nuclear	0.7	0.7	0.7
	PV	0.0	0.0	0.0
Efficiency [%]	NGCC	62	64	67
	PC	49	52	56
	IGCC	49	53	57
	NGCC-CCS	52	56	60
	PC-CCS	38	43	47
	IGCC-CCS	41	47	51

Table 4.28 Cost-structures of road transport fuel technologies (van Vliet et al., 2011)

Designation in MARKAL	Feedstock	Outputs [fuel: GJ_{out}/GJ_{in}]	Investment [€/kW_{in}]	O&M [€/kW_{in}/yr]	Emissions [$kgCO_2/GJ_{in}$] to air	Emissions [$kgCO_2/GJ_{in}$] to CCS
Fischer–Tropsch						
F_BF_10	Biomass	FT diesel: 0.429, electricity: 0.064, FT petrol: 0.076	997	43	69	0
F_BF_20	Biomass	FT diesel: 0.425, electricity: 0.091, FT petrol: 0.076	690	31	55	0
F_BF_20_CC	Biomass	FT diesel: 0.425, electricity: 0.074, FT petrol: 0.076	707	31	7	48
Hydrogen						
F_BH_20	Biomass	H_2: 0.683, electricity: 0.048	641	29	90	0
F_BH_20_CC	Biomass	H_2: 0.683, electricity: 0.017	671	30	1	89
F_NH_00	Natural gas	H_2: 0.7, electricity: 0.037	483	22	53	0
F_NH_20_CC	Natural gas	H_2: 0.7, electricity: 0.019	504	22	2	51
F_NH_10_N	Natural gas	H_2: 0.737	592	387	56	0
F_NH_10_DC	Natural gas	H_2: 0.728	663	404	25	33
Methanol						
F_BM_20	Biomass	methanol: 0.54, electricity: 0.092	583	24	53	0
F_BM_20_CC	Biomass	methanol: 0.54, electricity: 0.076	583	24	7	46
DME						
F_BD_20	Biomass	DME: 0.554, electricity: 0.095	598	25	53	0
F_BD_20_CC	Biomass	DME: 0.554, electricity: 0.077	609	25	7	46
Oil refining						
F_OP_00	Crude oil	petrol: 0.914			7	0
F_OD_00	Crude oil	diesel: 0.897			8	0

Table 4.29 Cost-structures of road transport and jet fuels

Technology	**Feedstock**	**Capacity** [kt feed/yr]	**Outputs** [fuel: t fuel/t feed in]	**Investment** [M€]	**Fixed O&M** [M€/yr]	**Variable O&M, excl. feedstock, utilities** [€/t feed]
Road[a]						
Fermentation	Starch	425	ethanol: 0.479	149	5	9.4
Transesterification	Vegetable oil	356	biodiesel: 0.969, glycerin: 0.08	81	5	54
2nd generation ethanol dilute acid	Solid biomass	665	ethanol: 0.198, electricity: 0.1 (GJ_{el})	313	11	16
2nd generation ethanol steam explosion	Solid biomass	1,664	ethanol: 0.205, electricity: 0.24 (GJ_{el})	493	17	16
2nd generation ethanol liquid hot water	Solid biomass	3,327	ethanol: 0.208, electricity: 0.26 (GJ_{el})	908	32	1
Road/Aviation[b]						
Hydroprocessed esters and fatty acids (HEFA)	Vegetable/ used cooking oil	968	RJF: 0.13, renewable diesel: 0.68, naphtha: 0.02, propane[c]: 0.06	650	49	126
Hydrotreated renewable diesel (HRD)	Vegetable/ used cooking oil	968	renewable diesel: 0.83, propane[c]: 0.05	650	49	126
Hydrothermal liquefaction (HTL)	Solid biomass	657	RJF: 0.07, renewable diesel: 0.2, gasoline: 0.09	406	33	64
Catalytic pyrolysis (Pyr)	Solid biomass	657	RJF: 0.04, renewable diesel: 0.11, gasoline: 0.12	470	38	64
Refining	Crude oil		kerosene: 0.929 (GJ_{out}/GJ_{in})			0.0025 (€/GJ_{in})

[a]Fermentation and transesterificaiton based on Bain (2007), 2nd generation ethanol based on Hamelinck et al. (2005) [b]Based on de Jong et al. (2015). [c]propane/LPG of HEFA/HRD assumed as gasoline. Production costs of gasification and Fischer-Tropsch to jet are the same with road transport fuels, assuming that maximum of 25% of Fischer-Tropsch diesel can be upgraded to jet standards according to ASTM specification.

Table 4.30 Cost-structures of chemical production and other technologies[a]

	Technology	Capacity	Conversion efficiency	Co-products	Utilities	Capital costs (at specified scale)	Fixed costs	Labor costs (part of fixed costs)	Variable costs (excluding feedstock and process energy)	Data/ Study year
1	Cellulosic sugars DA	400 MW feed HHV	C6: 75%, C5: 60% in 2015; 40.2 GJ feed/t sugar	0.09 t lignin	electricity: 34.8 MWe, process heat: 0.5 kg steam/kg ds, sugar concentration: 7.2 GJ/t sugar	234.1 M€	81.8 € /t sugar	3.1 € /t sugar, 55.6 fte, 56,206 € 2010/fte	0	2005/ 2011
2	Cellulosic sugars SE	1000 MW feed HHV	C6: 75%, C5: 55% in 2015; 38.6 GJ feed/t sugar	0.1 t lignin	electricity: 42 MWe, process heat: 0.3 kg steam/kg ds, sugar concentration: 7.2 GJ/t sugar	256.7 M€	41.5 € /t sugar	3.7 € /t sugar, 67.3 fte, 56,206 €/fte	29.6 € /t sugar (in 2015)	2005/ 2011
3	Cellulosic sugars LHW	2000 MW feed HHV	C6: 75%, C5: 60% in 2015; 38.2 GJ feed/t sugar	-	electricity: 72 MWe, process heat: 1.2 kg steam/kg ds, sugar concentration: 7.2 GJ/t sugar	469.4 M€	39.4 € /t sugar	4.3 € /t sugar, 67.3 fte, 56,206 €/fte	12.3 € /t sugar (in 2015)	2005/ 2011
4	2nd generation ethanol DA	400 MW feed HHV	C6: 75%, C5: 60% in 2015 Fermentation efficiency: xylose 86%, arabinose 86%, glucose 93%, galactose 86%, mannose 86%; 3.2 GJ feed/GJ ethanol	0.1 GJel	No external energy requirement	313 M€	2.82 € /GJ ethanol and electricity out	Included in fixed costs	2.65 € /GJ ethanol and electricity out	2005
5	2nd generation ethanol SE	1000 MW feed HHV	C6: 75%, C5: 55% in 2015 Fermentation efficiency: xylose 86%, arabinose 86%, glucose 93%, galactose 86%, mannose 86%; 3.1 GJ feed/GJ ethanol	0.24 GJel	No external energy requirement	493 M€	1.71 € /GJ ethanol and electricity out	Included in fixed costs	2.57 € /GJ ethanol and electricity out	2005
6	Lignocellulosic ethanol LHW	2000 MW feed HHV	C6: 75%, C5: 60% in 2015 Fermentation efficiency: xylose 86%, arabinose 86%, glucose 93%, galactose 86%, mannose 86%; 3.06 GJ feed/GJ ethanol	0.26 GJel	No external energy requirement	908 M€	1.55 € /GJ ethanol and electricity out	Included in fixed costs	0.13 € /GJ ethanol and electricity out	2005
7	Catalytic dehydration of ethanol to ethylene	100 kt/yr	1.74 t ethanol/t ethylene	-	Based on information proprietary to data suppliers. Please contact the authors for more details	186 M€	336 €	98 €/t ethylene (included in fixed costs)	0	2009 onwards
8	Succinic acid from sugars - direct crystalisation today	30 kt/yr	1.13 t sugar/t succinic acid	Based on information proprietary to data suppliers. Please contact the authors for more details		100 M€ Other 2005 onwards	339 €	104 €	0	Capital costs 2013
9	Succinic acid from sugars - direct crystalisation future	100 kt/yr	Based on information proprietary to data suppliers. Please contact the authors for more details			232 M€ Other 2005 onwards	276 €	72 €	0	Capital costs 2013
10	Succinic acid to phthalic anhydride	-	1 t succinic acid/t phthalic anhydride	-	-	-	-	-	-	

Table 4.30 (continued)

	Technology	Capacity	Conversion efficiency	Co-products	Utilities	Capital costs (at specified scale)	Fixed costs	Labor costs (part of fixed costs)	Variable costs (excluding feedstock and process energy)	Data/ Study year
11	Succinic acid to BDO	50 kt/yr	Based on information proprietary to data suppliers. Please contact the authors for more details						0 (Hydrogen production included from the model)	2000
12	PLA from sugars, today	30 kt/yr	Based on information proprietary to data suppliers. Please contact the authors for more details	0.04 t waste biomass ds	Based on information proprietary to data suppliers. Please contact the authors for more details	124.25 M€	181.4 €	83 €	0	
13	PLA from sugars, future	100 kt/yr	Based on information proprietary to data suppliers. Please contact the authors for more details	1 kg PLA	Based on information proprietary to data suppliers. Please contact the authors for more details	288.5 M€	229 €	50 €	0	
14	PDO from sugars - today, batch	30 kt/yr	2.4 kg sugar/kg PDO	0.2 kg waste biomass	electricity: 9.1 GJe, heat: 14.5 GJ	68 M€	302 €	79 €	0	
15	PDO from sugars - today, continious	30 kt/yr	2.4 kg sugar/kg PDO	0.2 kg waste biomass	electricity: 2.9 GJe, heat: 14.5 GJ	60 M€	234 €	54 €	0	
16	PDO from sugars - future, H2O pervaporation	100 kt/yr	1.9 kg sugar/kg PDO	0.1 kg waste biomass	electricity: 1.6 GJe, heat: 22.4 GJ	95 M€	112 €	20 €	0	
17	Methanol to Olefins (UOP)	100 kt/yr ethylene	C2H4: 26%, C3H6: 33%, Fuel gas: 2%; 0.18 t ethylene/t methanol	0.23 t propylene, 0.01 t fuel gas	heat: 13.2	104 M€	41.8 €	131 € (not included in fixed costs)	0	2009
18	Methanol to Olefins (Exxon)	100 kt/yr ethylene	C2H4: 14%, C3H6: 18%, Gasoline: 29% Fuel gas: 0.1%; 0.10 t ethylene/t methanol	0.12 t propylene, 0.20 t gasoline, 0.001 t fuel gas	heat: 32.8 GJ	188 M€	78.3 €	246 € (not included in fixed costs)	0	2009
19	Methanol to Propylene (Lurgi)	100 kt/yr propylene	C3H6: 46%, Gasoline: 20% Fuel gas: 6%; 0.32 t propylene/t methanol	0.14 t gasoline, 0.04 t fuel gas	heat: 12.2 GJ	64 M€	23.2 €	76 € (not included in fixed costs)	0	2009
20	Two stage ethanol to Butadiene	100 kt/yr	2.77 t ethanol/t butadiene	0.05 t hydrogen, 0.01 GJth	electricity: 0.56 GJel, heat: 4.01 GJ	79.6 M€	188 €	36	0	2012
21	Ammonia from Hydrogen	733 kt/yr	0.178 t hydrogen/t NH3 (21.35 GJ/t NH3)	-	electricity: 1.04 GJe	457 M€	67 €	18 €	0	2008
22	Ammonia from steam reforming natural gas	1,200 kt/yr	21.35 GJ/t NH3	-	heat: 9.55 GJ, electricity 0.1 GJe	527 M€	60 €	12 €	0	
23	Naphtha-based steam cracker	660 kt/yr (assumed)	1.993 t light naphtha/t ethylene	0.52 t propylene, 0.15 t butadiene, 0.32 t aromatics	heat: 24.6 , electricity: 0.19 GJe	645 €/t	175 €	39 €	0	2000 onwards
24	Fischer-tropsch pellets to fuels and bio-naphtha	400 MW feed LHV	14.35 GJ in/GJ naphtha	0.106 GJ petrol, 6.15 GJ diesel, 1.01 GJe	No external energy requirement	723 M€	878 €	43 €	0	2011
25	Fischer-tropsch TOPS to fuels and bio-naphtha	2000 MW feed LHV	14.5 GJ in/GJ naphtha	0.106 GJ petrol, 6.15 GJ diesel, 1.44 GJe	No external energy requirement	2,444 M€	635 €	45 €	0	2011

Table 4.30 (continued)

	Technology	Capacity	Conversion efficiency	Co-products	Utilities	Capital costs (at specified scale)	Fixed costs	Labor costs (part of fixed costs)	Variable costs (excluding feedstock and process energy)	Data/ Study year
26	Fischer-tropsch coal to fuels and naphtha	2000 MW feed HHV	14 GJ in/GJ naphtha	0.106 GJ petrol, 6.15 GJ diesel, 1.1 GJe	No external energy requirement	2,540 M€	699 €	48 €	0	2011
27	Fischer-tropsch TOPS to fuels and bio-naphtha with CCS	2000 MW feed HHV	14.5 GJ in/GJ naphtha	0.106 GJ petrol, 6.15 GJ diesel, 1.1 GJe	No external energy requirement	2,505 M€	650 €	45 €	0	2011
28	Fischer-tropsch coal to fuels and naphtha with CCS	2000 MW feed HHV	14 GJ in/GJ naphtha	0.106 GJ petrol, 6.15 GJ diesel, 0.84 GJe	No external energy requirement	2,601 M€	714 €	48 €	0	2011
29	Pyrolysis gasoline from woodchips	91 MW	4.13 GJ in/GJ petrol	0.35 GJe, 0.01 t hydrogen, 0.35 GJ steam	No external energy requirement	80 M€	7.4 €	-	0.55 €	2012
30	Pyrolysis gasoline and chemicals from woodchips	91 MW	4.13 GJ in/GJ petrol	0.35 GJe, 0.35 GJ steam, 5.7 kg aromatics, 4.85 kg ethylene, 8.16 kg propylene, 2.15 kg LPG Uassumed as petrol equivalent)	No external energy requirement	95.7 M€	7.4 €	-	0.55 €	2012
31	Bio FT-Naphta steam cracker	660 (assumed)	1.9 t light FT-naphtha/t ethylene	0.58 t propylene, 0.29 t butadiene	heat: 24.6 , electricity: 0.19 GJe	645 €/t	175 €	39 €	0	2000 onwards
32	Coal FT-Naphtha steam cracker	660 (assumed)	1.9 t light FT-naphtha/t ethylene	0.58 t propylene, 0.29 t butadiene	heat: 24.6 , electricity: 0.19 GJe	645 €/t	175 €	39 €	0	2000 onwards
33	Manure co-digestion to biogas	505 Nm3/h	0.29 t/GJ biogas	-	No external heat energy requirement (met by biogas), electricity: 0.12 kWh/Nm3 biogas	2.4 M€	1.42 €/GJ	Non-specified (part of fixed costs)	6.4 €/GJ	2010 onwards
34	Sewage water treatment to biogas	100 Nm3/h	0.045 t/GJ biogas	-	No external heat energy requirement (met by biogas), electricity: 0.02 kWh/Nm3 biogas	0.1 M€	0.27 €/GJ	Non-specified (part of fixed costs)	0	2010 onwards
35	Wet organic waste to biogas	950 Nm3/h	0.29 t/GJ biogas	-	No external heat energy requirement (met by biogas), electricity: 0.12 kWh/Nm3 biogas	4 M€	1.1 €/GJ	Non-specified (part of fixed costs)	4.8 €/GJ	2010 onwards
36	Biogas to CHP from manure co-digestion	505 Nm3/h	0.28 GJe/GJ biogas	0.142 GJth	No external energy requirement	3.4 M€	7.98 €/GJe	Non-specified (part of fixed costs)	25 €/GJe	2010 onwards
37	Biogas from co-digestion manure to green gas	505 Nm3/h	0.67 GJ green gas/GJ biogas	-	No external energy requirement	3.6 M€	3.68 €/GJ	Non-specified (part of fixed costs)	11.8 €/GJ	2010 onwards
38	Biogas from wet organic waste to green gas	950 Nm3/h	0.67 GJ green gas/GJ biogas	-	No external energy requirement	5.8 M€	2.91 €/GJ	Non-specified (part of fixed costs)	9.4 €/GJ	2010 onwards

Table 4.30 (continued)

	Technology	Capacity	Conversion efficiency	Co-products	Utilities	Capital costs (at specified scale)	Fixed costs	Labor costs (part of fixed costs)	Variable costs (excluding feedstock and process energy)	Data/Study year
39	Polymerisation of ethylene to polyethylene	270 kt/yr								2000
40	Ethylene to ethylene oxide	270 kt/yr								2000
41	Ethylene oxide to ethylene glycol	170 kt/yr								2000
42	Xylene (aromatics) to terephthalic acid	500 kt/yr								2000
43	PET pellets production	180 kt/yr								2000
44	PTT pellets production	250 kt/yr								2000
45	Ethylbenzene production	780 kt/yr	Based on information proprietary to data suppliers. Please contact the authors for more details							2000
46	Styrene production route dehydrogenation ethylbenzene	-								2000
47	Styrene/propylene oxide production	-								2000
48	BDO production from propylene oxide	50 kt/yr								2000
49	Polymerisation of propylene to polypropylene	250 kt/yr								2000
50	Phthalic anhydride from orthoxylene	100 kt/yr								2000
51	Urea production	450 kt/yr								2000

[a]Bartels (2008), Broeren et al. (2014), Burla et al. (2012), Ereev and Patel (2012), Hamelinck et al. (2005), Hermann and Patel (2007), Humbrid et al. (2011), ICIS (2008), IEA-ETSAP/IRENA (2013), Kermeli et al. (2013), Lako (2009), Lensink et al. (2014), Patel (2014), Ren and Patel (2009), Ren et al. (2009), Worrell and Biermans (2005).

5

The role of bioenergy and biochemicals in CO_2 mitigation through the energy system – A scenario analysis for the Netherlands

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ABSTRACT

Bioenergy and bioenergy and carbon capture as well as storage are shown to be key options to embark on cost-efficient trajectories that realise climate targets. However, most studies have not yet assessed the influence on these trajectories of emerging bioeconomy sectors such as biochemicals and renewable jet fuels. To support a systems transition, there is also need to demonstrate the impact on the energy system of technology development, biomass and fossil fuel prices. We aim to close this gap by assessing least-cost pathways to 2030 for a number of scenarios applied to the energy system of the Netherlands, using a cost-minimisation model. The type and magnitude of biomass deployment is highly influenced by the level of technology development, fossil fuel prices and ambitions to mitigate climate change. Across all scenario variants, biomass consumption ranges between 180 and 760 PJ. High technology development leads to additional 100-270 PJ of biomass consumption compared to low technology development counterparts. Traditional sectors, namely industrial biomass heat and biofuels, supply 61-87% of bioenergy across scenarios, while wind turbines are the main supplier of renewable electricity. Low technology pathways show lower biochemical output by 50-75%, do not supply renewable jet fuels and do not utilise additional biomass. High technology development leads to lower emissions than low technology development counterparts, primarily due to bioenergy and carbon capture and storage. However, in most scenarios the emission reduction targets for the Netherlands are not met. Thus stronger climate policy is required, especially in view of fluctuating fossil fuel prices, which are shown to be a key determinant of bioeconomy development. Nonetheless, high technology development is a no-regrets option to realise deep emission reduction as it also ensures stable growth for the bioeconomy even under unfavourable conditions.

5.1 INTRODUCTION

At the Conference of Parties in Paris in 2015 it was agreed to intensify efforts on limiting global temperature to well below 2 °C, while aiming at 1.5°C, above pre-industrial levels by 2100 to prevent most of the negative consequences of man-induced global warming (UNFCCC, 2015a). To that end, nations submitted their Intended Nationally Determined Contributions (INDCs) for climate action, which state their greenhouse gas (GHG) emission reduction targets to 2030 (UNFCCC, 2015b). Since 2009, the European Union (EU) set out to reach 20% emission reduction by 2020 compared to 1990 through increased renewable energy production and energy efficiency (EC, 2009b). A 40% reduction target by 2030 is published in the EU INDC and is consistent with its intention to reach 80-95% domestic emission reduction by 2050 compared to 1990 (EC, 2015a, 2012c). While the EU is on track to meet its 2020 targets (EC, 2015b), the United Nations Framework Convention on Climate Change concluded that the global aggregate GHG emission reduction pledged in INDCs falls short by about 35% to remain in the trajectory of least-cost 2 °C scenarios by 2030 (UNFCCC, 2015b).

To reach long-term climate goals, many models rely on large-scale bioenergy deployment, carbon capture and storage (CCS) and their combination (bioenergy with carbon capture and storage that may lead to negative emissions; BECCS; IPCC (2014a)). Biomass is the largest source of renewable energy today, contributing to 10% of total global primary energy supply (Chum et al., 2011) and its share is projected to increase up to 50% by 2100 in scenarios that account for climate targets (Rose et al., 2014). Biomass conversion technologies, CCS and BECCS have been proposed as technical solutions that reduce emissions in the most cost-optimal way (Fuss et al., 2014; Matthews et al., 2015; Winchester and Reilly, 2015). While BECCS reduces the need for near-term mitigation (Rose et al., 2014), stimulating the bioeconomy early in the time horizon could be pivotal in embarking on pathways that realise long-term climate goals. However, sectors such as aviation and chemicals, which have few or no other renewable alternatives than biomass, have not yet been included in most models. Thus, the expected contribution of biomass to meet emission reduction targets does not account for emerging sectors of bioeconomy within energy systems.

There is sufficient evidence to suggest that emerging bioeconomy sectors on the one hand and CCS and BECCS on the other can play a significant role in the future. Based on mid-term demand projections, biochemicals and bioplastics (frequently referred to as non-energy uses of biomass) may consume 9-24% of biomass demand by 2050 (Piotrowski et al., 2015). Other studies show 15-17% of total biomass to be used for non-energy applications (18-27 EJ/yr) and to supply approximately 7-11 EJ/yr of non-energy biomass products (Daioglou et al., 2015). In other sectors, such as aviation, the EU has the ambition to reach 88 PJ (2 Mt, assuming 44 GJ/t heating value) renewable jet fuel (RJF) consumption, which is about 3.7% of its projected jet fuel demand by 2020 (EC, 2011b, 2003). Regarding emission reduction, at a global level, BECCS would need to contribute between 2 and 10 $MtCO_2$/yr in 2050 in order to ensure compliance with the 2 °C target (4-22% of the 1990 baseline; Fuss et al. (2014)). Based on Rose et al. (2014), modern bioenergy supply may reach 37% share (or up to about 250 PJ) over total primary energy supply by 2050 and is largely combined with BECCS. Despite these expectations, comprehensive assessments of extended bioeconomy sectors in energy system models and interactions with other renewable energy sources (RES; e.g. wind or solar) and mitigation technologies (CCS, BECCS) are scarce.

To assess the potential for bioeconomy and its contribution to GHG emission reduction within the energy and non-energy system, an integrated systems assessment is required that takes into account emerging sectors and addresses key factors of uncertainty. Our earlier study incorporated the chemicals and aviation sector in a national energy system model of the Netherlands and demonstrated that biomass conversion technologies may be cost-competitive compared to other fossil and renewable alternatives by 2030 to achieve renewable energy goals (Chapter 4). With respect to biomass conversion, industrial heat from biomass, lignocellulosic sugar production, biochemicals from sugar fermentation and Fischer-Tropsch (FT) road transport fuels from solid biomass gasification were shown to be most promising options. These findings are in line with other research (e.g. Ren & Patel (2009), Ren et al. (2009), Saygin et al. (2013, 2014), Gerssen-Gondelach et al. (2014)). However, our earlier study also showed that while the renewable energy technology portfolio was stimulated by renewable energy policies, emission reduction targets of 40% emission mitigation by 2030, compared to 1990 were not met. Therefore, additional insights are needed as to the required preconditions to meet those targets. One limitation of the abovementioned study is that it only assessed the influence of technology development as a factor of future uncertainty, while other crucial parameters such as varying fossil fuel prices and availability of low-cost biomass in combination with technological progress may also affect bioeconomy developments and the pathways to emission reduction. In addition, in view of climate targets, these uncertainties need to be assessed under a technology-neutral setting, with climate policy such as a CO_2 tax being the only driver for the deployment of a cost-optimal technology portfolio.

Such an assessment is performed in the present study by using a national cost-minimisation linear programming model developed for the Netherlands (MARKAL-NL-UU; Chapter 4) that apart from technology characterisation of the fossil energy system also includes key biomass conversion technologies, other renewables and mitigation options (CCS, BECCS). Using scenario assessment for a combination of uncertainty factors on technology development, biomass cost-supply and fossil fuel prices we estimate the achieved GHG emission reduction, the required technology portfolio, the demand for biomass and supply the of bioenergy and biochemicals in each case.

With such insights, the risks and opportunities for bioeconomy development can be highlighted, thereby supporting informed decision-making that improves the competitiveness of biomass supply chains and other renewables and mitigation technologies to reach emissions reduction targets. These may on the one hand include support to stimulate early development of renewable energy technologies to the benefit of cost-efficiency through learning and economies of scale. On the other hand, investments can take place earlier in the supply chain to improve agricultural productivity or pretreatment that can reduce supply costs of feedstocks for downstream conversion. Also, by assessing bioeconomy developments within the uncertain environment shaped by fossil fuel price, their magnitude of influence can be demonstrated. This way bioeconomy can be supported in a direction that makes it less vulnerable to external fossil fuel price volatility.

5.2 MATERIALS AND METHODS

To determine the impacts of bioeconomy development and its contribution to GHG emission mitigation to 2030, we translate key parameters of future uncertainty (technology development, biomass cost-supply, fossil fuel prices) to scenarios and then perform scenario analysis by comparing outputs derived from a cost-minimisation linear programming energy system model developed for the Netherlands.

5.2.1 Model

The MARKAL-NL-UU applied in this study, uses cost-minimisation linear programming techniques to define the technology portfolio required to meet demand for energy (electricity, heat, fuels) and chemicals that lead to least total system costs. The model can be described by three core modules: energy supply, energy and chemicals conversion, and energy and chemicals demand.

For a detailed description of the MARKAL family of models see Loulou et al. (2004). The electricity sector and the CCS technology portfolio for the Netherlands are described in van den Broek et al. (2008, 2011). The model's extension to the road transport sector is included in van Vliet et al. (2011). Finally, emerging bioeconomy sectors have been included in Chapter 4. The technology portfolio of MARKAL-NL-UU is described in Chapter 4.

5.2.1.1 Energy supply

In the energy supply module, cost-supply trajectories of fossil, nuclear and biomass resources are included. For fossil fuels, the price develops according to the International Energy Agency World Energy Outlook 2015 (IEA-WEO) New Policies Scenario (OECD/IEA, 2015), unless stated otherwise. Fossil fuel price variation is a key aspect of future uncertainty, which is taken into account in the scenario assessment (section 5.2.2.3).

Biomass cost-supply curves are estimated based on the sourcing region (domestic, European, global) and are specified for different biomass types. Road-side costs and potentials for biomass are determined for 2010-2030, based on the Intelligent Energy Europe (IEE) project Biomass Policies (Elbersen et al., 2015). Costs refer to market prices for already traded biomass types and to road-side costs for biomass markets that are not developed (Elbersen et al., 2015). Traditional biomass uses such as food, feed and fibres are prioritised over bioenergy and biochemicals in determining the available potential. To these costs, we add transport costs to the Netherlands using a geographic explicit model on Biomass Intermodal Transportation (BIT-UU, described in Hoefnagels et al. (2014a, 2014b) assuming the energy density of wood chips for transported biomass). Biomass and transport costs are estimated a NUTS2 regional level[35] of EU28 countries and are aggregated based on the weighted average for 4 EU regions as described in Chapter 4. From the regional biomass

[35] Nomenclature of territorial units for statistics (NUTS).

supply potential it is assumed that approximately 5% may be available for export to the Netherlands, based on the share of the Dutch total primary energy supply over the EU to 2030.[36]

These assumptions may lead to conservative biomass cost-supply estimates for two reasons. Firstly, transport costs are based on wood chip logistics thereby ignoring cost-efficiency gains that can be achieved if biomass is densified at the sourcing region, e.g. to wood pellets. Secondly, each country may supply larger potential than the 5% we allocated if markets are well developed. Both of these factors are addressed in scenarios (section 5.2.2.2).

Next to biomass from EU sources, five types of commodities form extra-EU sources are included, namely raw sugar, wood pellets, 1st and 2nd generation ethanol, vegetable oil, and biodiesel. Ultimately, it depends on the total production system costs, which include feedstock and conversion, to indicate the cost-optimal use of intra-EU or extra-EU resources.

A total of 400 PJ of solid biomass and 50 PJ of liquid biomass is assumed to be available for imports by the Netherlands. Such potential is approximately 26 Mt in wood pellet equivalent, which is rather large considering that it corresponds to global wood pellet consumption in 2015 (about 25.5 Mt; AEBIOM (2015)). However, there is sufficient evidence that suggests that these volumes may be available (Chum et al., 2011; Ganzevles, 2014; Smeets, 2014). The influence of extra-EU import is assessed in a separate scenario, which assumes that only domestic and intra-EU biomass is available (section 5.2.2.2).

5.2.1.2 Technologies for energy and chemicals conversion

The model includes a large portfolio of fossil (natural gas, oil, coal), nuclear and renewable energy (e.g. biomass conversion, wind turbines, photovoltaics) technologies that convert the primary resources to electricity, heat, fuels for the energy system, and feedstocks or end-products for the fossil-based and bio-based chemical industry. Technologies are characterised based on their cost-structure at a specific year and scale (capital investments, fixed and variable costs) and technical parameters (process energy input, process efficiency).

Biorefineries (biochemical and thermochemical) are also included in the model.[37] Similar to other multi-output processes such as combined heat and power plants, biorefineries deliver outputs to several sectors (e.g. to fuels and electricity) as opposed to e.g. a wind turbine, which delivers only to the electricity sector. This enables different biomass constituents to be used in different sectors and provide to biorefineries access to demand markets in direct competition with other technologies thus reducing total system costs.

An overview of technologies is presented in Chapter 4. Their cost-structures and the cross-sectoral flows are described in Chapter 4. Technology development is rather uncer-

[36] In OECD/IEA (2014) the EU demand is 61 EJ under the 450 ppm scenario in 2030. For comparison, the Dutch demand is 3.2 EJ and the assumed biomass in 2030 is about 430 PJ or 13% of the country's total primary energy supply.

[37] Conventional coal gasification and FT-synthesis to fuels is excluded as an option. Thermochemical refineries refer to biomass gasification.

tain and therefore assessed by scenarios in the present chapter (section 5.2.2.2).

5.2.1.3 Energy and chemicals demand

The final energy demand for electricity, heat, and the production volume of chemicals and jet fuels are fixed, are exogenously determined and specified for the Netherlands based on demand projections from EC (2003), Saygin et al. (2009), Chèze et al. (2011) and ECN (2015) as described in Chapter 4. The final demand for road transport fuels is endogenously calculated based on fixed demand for vehicle-kilometres (van Vliet et al., 2011). While projections for key energy applications such as heat and electricity are relatively stable over the course of years, for non-energy uses future demand poses higher uncertainties. This in turn can determine to a large extent the deployment potential of biochemicals. In an additional scenario we assume that the chemical sector follows a negative growth rate trajectory (section 5.2.2.4). The demand projections are provided in Chapter 4.

5.2.2 Scenarios

We use scenario analysis to assess the role of bioenergy and biochemicals in mid-term emission reduction pathways to 2030, within the national energy system of the Netherlands that takes into account cost-competition with fossil, renewable alternatives and other mitigation technologies (CCS, BECCS). Scenario analysis of “if-then” propositions is shown to be useful to the extent that it provides insights that improve strategic management by better understanding uncertainties and robustness of decisions under a wide range of possible futures. These can be stirred by strategies but can also be influenced by uncontrolled variables (Moss et al., 2010; Schwartz, 1996).

Baseline scenarios in this chapter give a plausible indication on how the energy and non-energy system may develop if no focus is placed on renewable energy and climate goals beyond 2020. We then deploy a set of scenarios that assess the effect of climate policies, namely a CO_2 tax that corresponds to meeting the 2 °C (OECD/IEA, 2015), in combination with bioeconomy strategies focused on the conversion and supply side.

5.2.2.1 Policy context of scenarios

Scenario parameters to 2020

To assess the cost-efficient contribution of biomass and other RES to CO_2 mitigation pathways, conversion technologies should compete on a level playing field. The scenarios that address sectors within the model’s system boundary need to be technology-neutral to avoid distortion caused by policies or support schemes (e.g. subsidies on specific technologies). However, up to 2020, binding policy goals at the EU level and national measures are already agreed and implemented. They include support to electricity, heat and road transport fuels up to 2020 and are assumed to be achieved in all scenarios. More specifically, these include:

- the renewable energy share (14% for the Netherlands) and the biofuel target of 10% including double-counting of biofuels from waste and residues (EU RED; EC (2009b));
- the retirement of old coal-fired power plants built before 1990 and wind deployment

as part of national plans to meet the EU RED targets (SER, 2013);

- maximum of 25 PJ electricity produced by co-firing biomass in coal power stations (SER, 2013).

In addition, we assume an emission tax as part of the climate policies to 2020 (i.e. 15 €/tCO_2 in 2020), based on the IEA-WEO 2015 New Policies scenario. In this study the tax applies to emissions from all sectors (i.e. including transport and residential heat).

Scenario parameters from 2020 onwards

Beyond 2020, all sectors compete on a level playing field. Therefore, cost-competitiveness of secondary energy carriers and chemicals is the only determinant of technology deployment, biomass contribution to demand, and achieved GHG emission reduction.

As set by the EU, long-term emission (2050) mitigation needs to be at least 80% compared to 1990 to remain in the trajectory of global temperature rise up to maximum 2 °C by 2100 (EC, 2015a, 2012c). In the medium term (2030), which is the focus of this study, it entails achieving emission reduction of 40% compared to 1990, in line with the EU INDC (EC, 2015a). We use CO_2 tax as the only policy instrument that stimulates emission reduction based on the IEA-WEO 2015 450 ppm scenario. Tax levels are 42 €/t CO_2 in 2025 and 69 €/tCO_2 in 2030 (OECD/IEA, 2015). These levels are part of the IEA-WEO scenario policy portfolio in achieving climate goals (OECD/IEA, 2015). The CO_2 tax applies to generated emissions in the Netherlands, as opposed to the carbon content of fossil feedstocks used. This somehow may compromise the level playing field of biochemicals as large part of the carbon in biomass feedstock remains embedded in final products. We assess fossil fuel prices and climate policy scenarios separately. The policy context beyond 2020 as described above is used in the *reference, biomass cost-supply* and *fossil fuel price scenarios* (section 5.2.2.2; Table 5.1).

5.2.2.2 Scenario definitions

The emission mitigation pathways are based on key strategies for development of biomass production systems across the supply chain, from feedstock to conversion. These may affect the cost-competitiveness of biomass in the energy system against other mitigation options.

Rate of technology development and technology diffusion

Technology development based on learning and subsequent cost reductions can considerably influence the competitiveness of biomass conversion technologies. Technology costs decline by a constant factor with each doubling of cumulative capacity (BCG, 1968). However, this occurs at a global level, which is outside of the regional scope of the present study. Incremental improvements over time such as in efficiency may also affect conversion costs. These factors are not endogenised in MARKAL-NL-UU. Therefore, we capture the uncertainty of technical progress on cost reduction of biomass conversion technologies and the role of BECCS to 2030 by using two technology pathways that follow *low* and *high technology development* progress. These pathways vary technology parameters and assume different learning rates for biomass conversion. More specifically, the two scenarios differ in technology portfolio, rate of incremental improvements, year of technology availability and scales as described in detail in Chapter 4. Low technology development assumes that little support is provided to conversion technologies by means of stimulating research and development (R&D), fast deployment of 1st-of-a-kind plants, support to technologies to go beyond the valley of death, rapid scale up and so forth. On the other hand, high technology development assumes that these conditions are met through coordinated action of business, industry and governments.

To avoid supply of all demand in the transport sector by a single technology we apply market constraints on 2nd generation technologies based on de Wit et al. (2010). Individual technologies can supply up to 10% of demand in 2030.

Biomass cost-supply

Low-cost biomass

The extra-EU and intra-EU cost-supply of biomass in baseline and reference scenarios are conservative for two reasons. Firstly, the price of extra-EU wood pellets is based on mill-gate costs of around 6 €/GJ (section 5.2.1.1). However, studies indicate that these can be as low as 3.9 €/GJ (Uslu et al., 2008) or 2.3-3.1 €/GJ by 2030 (Batidzirai et al., 2014) when low-cost biomass is used for pellets. These could be achieved, for example, by using surplus or abandoned agricultural land for energy crops production and intensification of agricultural productivity (de Wit and Faaij, 2009; Wicke, 2011). Secondly, the cost-price of intra-EU biomass delivered to the Netherlands, as assumed in this study, is conservative because transport costs are estimated based on wood chip logistics (section 5.2.1.1). Nonetheless, the cost-competitiveness of biomass chains can improve if efforts focus on biomass densification to reduce transport and handling costs (e.g. torrefaction and pelletisation). Such efforts are assumed to take place in the low-cost biomass supply scenario resulting to lower upstream cost-supply of solid biomass. More specifically:

- For extra-EU sources we assume that road-side costs of harvested biomass are 2.5 €/GJ. Such feedstocks costs are attainable. For example, global cost-supply curves assume 1.7 €/GJ of process residues (Gregg and Smith, 2010). Other studies that take into account the vast biomass potential of regions neighbouring to the EU (e.g. Ukraine) mention large potential at cost-prices lower than 2.5 €/GJ for short rotation forestry and lower than 3.2 €/GJ for grass plantations (de Wit and Faaij, 2009). We then apply techno-economic data from Batidzirai et al. (2014) to estimate pellet costs at mill-gate (4.3 €/GJ) and add transport costs to the Netherlands based on fuel costs consumption in the transport chain as described in section 5.2.1.1. This supplies 5.6-5.8 €/GJ wood pellets to the Netherlands, which are lower by around 25% compared to baseline and reference scenarios. Such cost-prices for wood pellets are attainable based on Hoefnagels et al. (2014a).
- For intra-EU biomass, we assume that Eastern European countries focus on ligno-cellulosic crop production (wood, herbaceous crops) thus may supply large volumes

of low-cost biomass to the Netherlands (50 PJ each[38]). We focus on Eastern Europe because it is a region with low cost-supply of biomass from short rotation forestry and perennial grasses compared to other EU regions (road-side costs of Eastern European biomass from wood and perennial grass crops are around 2.4 €/GJ and 1 €/GJ, respectively). To these cost we add transport costs based on the BIT-UU model (Hoefnagels et al., 2014a, 2014b). This results in wood pellet costs of 6.1-6.2 €/GJ from grass crops and 6.6-6.7 €/GJ from wood crops in 2020-2030 and are 26-32% lower compared to the cost-price of the same biomass assumed in the baseline and reference scenarios.

Under these assumptions we define a *low-cost biomass* scenario that is used to assess a situation under which investment decisions focus upstream in biomass supply chains. Note that assuming torrefied pellets (TOPS) instead of wood pellets has the potential to reduce supply costs even further by roughly 15% based on Uslu et al. (2008). Additional cost gains can be realised further downstream due to higher process efficiencies. On the other hand, the choice of torrefaction would restrict the supply of biomass within the chemical sector, which requires all biomass constituents. The incorporation of TOPS as biomass feedstock and efficiency gains on the conversion side requires more detailed analysis and therefore is excluded from the present study.

High-cost biomass

As a consequence of worldwide increase in biomass demand, it is expected that global biomass trade will continue in the future thereby allowing cost-efficient distribution of biomass from supply to demand regions. However, it is uncertain how trade and markets will develop. If the EU is the only region that supports bioeconomy developments then EU demand regions like the Netherlands will have access only to intra-EU resources. The *high-cost biomass* cost scenario assumes that extra-EU import of biomass is not possible, which decreases the total potential by 450 PJ compared to the reference. This effectively leads to increased costs of solid biomass as a large potential below 7.5 €/GJ for solid biomass becomes unavailable.

The cost-supply curves of solid biomass used across the different scenarios is presented in Figure 5.1. The cost-supply curves exclude energy maize, solid waste, fuelwood, landscape wood and road-side grasses, which unlike the solid biomass feedstocks included in Figure 5.1, they are linked with specific end-use applications (e.g. energy maize with co-digestion, solid waste with energy incineration and energy recovery and so forth).

5.2.2.3 Fossil fuel prices

Fossil fuel prices are uncertain and are subjected to change over time (OECD/IEA, 2015) and are key determining, but uncontrolled factors of the success or failure of bioeconomy development strategies. Furthermore, they affect the cost-competitiveness of other renewable alternatives. To capture the uncertainty that such variables may have on emission

[38] Compared to biomass availability from the IEE project Biomass Policies this represents 11% and 30% of available perennial grasses and wood crops in Eastern Europe, respectively.

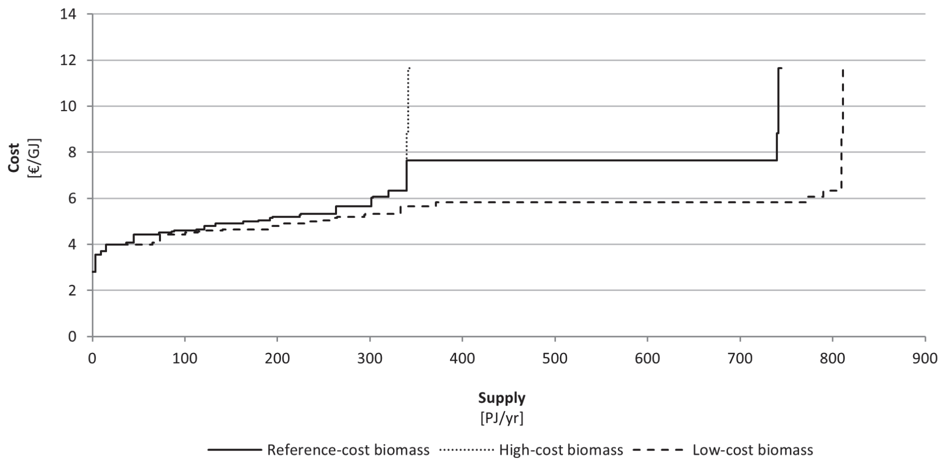


Figure 5.1 Solid biomass cost-supply curves in the different scenarios of this study in 2030 (note: the cost-supply curves exclude energy maize, solid waste, fuelwood, landscape wood and road-side grasses)

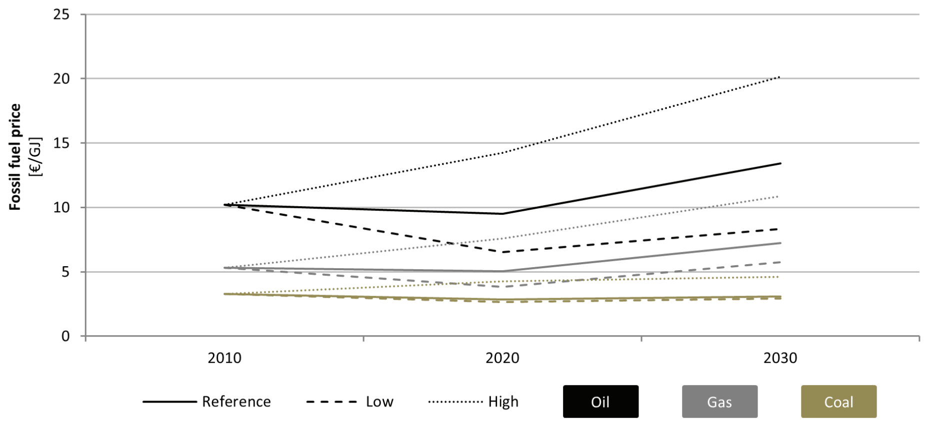


Figure 5.2 Fossil fuel price in the reference scenarios and their fossil fuel price variants

mitigation we deploy the following scenarios:

- *High fossil fuel price*: To obtain insights on the magnitude of biomass and other RES deployment under favourable conditions, a 50% higher fossil fuel prices is assumed compared to those of the New Policies Scenario reported at IEA-WEO 2015 (OECD/IEA, 2015);
- *Low fossil fuel price*: To obtain insights on the magnitude of biomass and other RES development in unfavourable environment prices we use those reported in IEA-WEO 2015 Low oil price scenario (OECD/IEA, 2015). Compared to the New Policies scenario these prices are lower approximately 35% for oil, 20% for natural gas and 6% for coal compared to the reference fossil fuel prices.

The variation of fossil fuel prices is presented in Figure 5.2.

5.2.2.4 Other sensitivity scenarios

Several drivers, such as contraction of the economy or competition from other regions (Broeren et al., 2014), may saturate or even decrease the production demand for chemicals assumed for the Netherlands over time. Future reduction in the demand for chemicals in combination with no decommissioning of existing steam cracking capacity in the Netherlands is assessed as an additional sensitivity scenario. A 10% reduction in demand for chemicals in 2030 compared to 2010 is assumed based on the reduction of the size of the Dutch petrochemical industry according to van Meijl et al. (2016).

Furthermore, in the EU, several governments consider reducing support on, divesting in or even dismantling coal-fired power plants as this may compromise the diffusion of other RES and CO_2 emission reduction goals (Nicola and Andresen, 2015; Pieters, 2016; Yeo, 2015). We assess this possible future in a scenario, which assumes that electricity from coal cannot be supplied in the energy system after 2020.

Other studies show that the role of biomass in the energy system varies, depending on the electricity mix. With exogenously determined electricity supply from other RES ranging between 17-80% and strong climate policy, biomass use for power generation in Europe ranges between 2.5-33% (0.4-2.1 EJ) of total fuel use in 2050 (Brouwer et al., 2016), without however taking competition by other sectors into account. Furthermore, improvements on energy efficiency could reduce heat demand in the industry and residential sector. This suggests that a large number of additional scenarios can be defined to investigate the sensitivity of the system and competition for biomass, which, however are excluded from this study.

Table 5.1 summarises the scenarios that are used in this study.

5.2.3 Indicators and overview of the modelling framework

For each scenario in Table 5.1 we assess:

- the final production output from RES per sector in 2030. For the energy sectors (electricity, heat, fuels) production output is expressed in final energy terms, while for the chemical sector it is estimated based on the lower heating value of the biochemical output. For multi-output processes such as biorefineries and CHP plants, the renewable energy output is allocated based on the energy content of products;

- the contribution of renewable energy in each sector is also assessed in relative terms, i.e. the contribution of renewable energy on the total final energy produced by each sector;
- the renewable energy share (i.e. excluding the non-energy use of the chemical sector);
- biomass demand that reflects total biomass consumption in primary energy terms, same as in Chapter 4;
- the direct CO_2 emissions in the Netherlands. Direct CO_2 emissions are those emitted in the Netherlands, i.e. they exclude emissions from production or extraction and transport of resources (biomass, fossil) to the Netherlands, consistent with IPCC (2006);
- total annual system costs in 2030, compared to the high technology development baseline scenario.

Table 5.1 Overview of the scenarios assessed in this study. Baseline scenarios assume CO_2 tax up to 2020. All other variants assume CO_2 tax up to 2030

		Scenario variable: biomass cost-supply				
		Low-cost biomass	Reference-cost biomass	High-cost biomass		
Baseline (no CO_2 tax beyond 2020)	High technology development	n.a.[a]	HighTechBase	n.a.	Reference fossil fuel price	**Baseline (no CO_2 tax beyond 2020)**
	Low technology development	n.a.	LowTechBase	n.a.	Reference fossil fuel price	
Scenario variable: technology development	High technology development	n.a.	HighTech (RefBio_HighFos)	n.a.[a]	High fossil fuel price	**Scenario variable: Fossil fuel price**
		HighTech (LowBio_RefFos)	HighTechRef[b,c]	HighTech (HighBio_RefFos)	Reference fossil fuel price	
		n.a.[a]	HighTech (RefBio_LowFos)[b]	n.a.[a]	Low fossil fuel price	
	Low technology development	n.a.[a]	LowTech (RefBio_HighFos)	n.a.[a]	High fossil fuel price	
		LowTech (LowBio_RefFos)	LowTechRef[b, c]	LowTech (HighBio_RefFos)	Reference fossil fuel price	
		n.a.[a]	LowTech (RefBio_LowFos)[b]	n.a.[a]	Low fossil fuel price	

[a]Combination of scenarios is not assessed in the present study. [b]Scenario variables used to assess the sensitivity of the biochemical sector in low chemical demand and delayed decommissioning of steam crackers. [c]Scenario variables used to assess the impact of complete closure of coal-fired power plants on CO_2 emissions.

Figure 5.3 presents an overview of the framework used in this study.

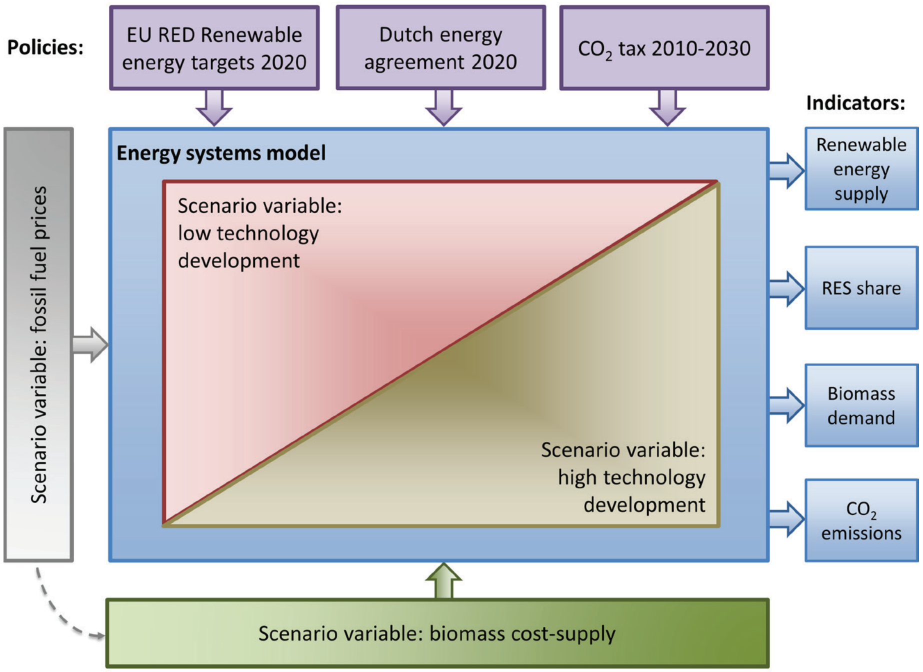


Figure 5.3 Modelling framework used in the present study

5.3 RESULTS

Final production from biomass and other RES, and their contribution to each sector is shown for the reference scenario in combination with the two technology development variants by bars, while the range of outcomes based on the biomass cost-supply and fossil fuel price scenarios is indicated with whiskers (Figure 5.4). Outcomes for the baseline situation in combination with the technology development scenarios are presented with markers. Results are presented for 2030.

For the indicators renewable energy share (Figure 5.5), biomass consumption (Figure 5.6) and CO_2 emissions (Figure 5.7), we present the influence of technology development in reference conditions (i.e. CO_2 tax) in comparison to the baseline for the period 2010-2030. As the results for the two technology development variants do not differ significantly in 2020, we only show the 2010-2030 trajectory of the high technology development scenario for the biomass cost-supply and the fossil fuel price variants. For comparison, we include results for the low technology development scenario in 2030. Apart from the range due to the variation of scenario parameters, results also include the consumption under baseline and reference conditions. For all scenarios, the difference of their total annual system costs from the high technology development baseline is plotted against the corresponding difference in total direct CO_2 emission reduction in 2030. Results with sector specific assumptions are presented in section 5.3.5 (Figure 5.10 - Figure 5.11).

All results per scenario and sector are presented in Table 5.2 through Table 5.5.

5.3.1 Renewable energy

Final production from renewable resources lies between 460-510 PJ in 2030 and does not differ significantly across technology development scenario variants (Figure 5.4a), thereby indicating that under reference assumptions (CO_2 tax, fossil fuel price) technology development is not the only underlying driver for cost-efficient supply of renewable energy. The renewable energy output (Figure 5.4b) corresponds to a 23-24% share on final energy consumption excluding chemicals or to a 18-20% share on final energy consumption including chemicals. More than two thirds (73-79%) of renewable energy output are attributed to biomass (Figure 5.4a, Figure 5.4b), which is higher than the anticipated contribution of biomass in the energy system based on the EU RED targets for 2020 (Rijksoverheid, 2010; Stralen et al., 2013). The remainder mainly represents renewable electricity by other renewable resources (wind and solar).

At a sector level and in absolute terms, technology development affects the supply of biomass heat, biofuels and biochemicals (bars in Figure 5.4a). These are also found to be the sectors with the largest bio-based output. Under low technology development assumptions, heat output from biomass is produced at the expense of biofuels, while under high technology development the reverse occurs; the trade-off between biomass heat and biofuel output is confirmed also when using different policy assumptions beyond 2020 as shown in Chapter 4. Electricity from biomass remains small (20-50 PJ; primarily from biorefineries, co-firing and municipal solid waste incineration), as wind is the key supplier of renewable electricity. Biochemicals are produced even under baseline assumptions as a result of retirement of steam-cracker capacity (20-50 PJ; 5-10% of the sector's output; Figure 5.4). While the CO_2 tax only affects the process emissions of the chemical sector, Figure 5.4 shows that in the high technology development scenario the output of biochemicals almost doubles (about 100 PJ; 17%). This is a result of multi-output technologies that produce both chemicals and road transport fuels, with the latter being affected by the CO_2 tax.

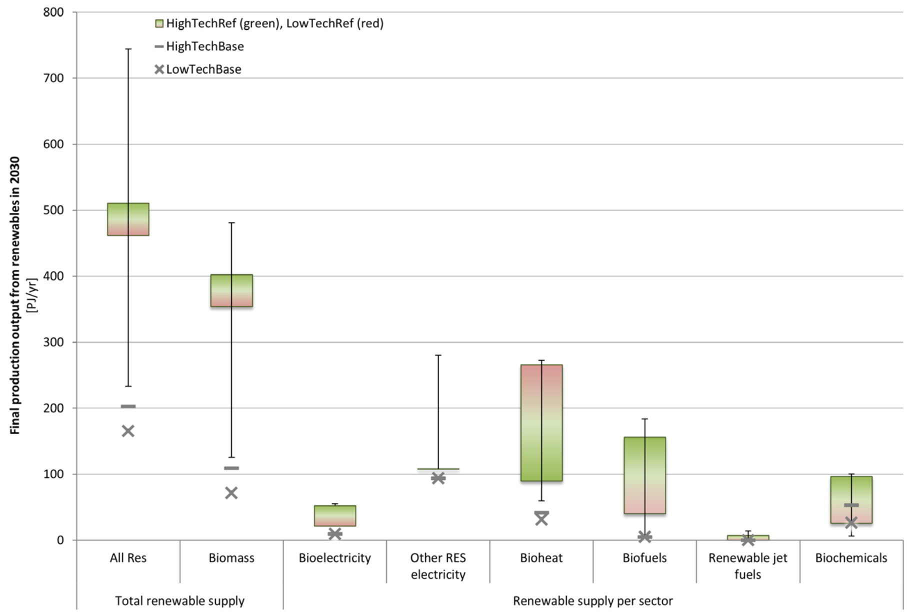
The electricity sector is most sensitive in fossil fuel price assumptions. High fossil fuel prices can lead up to a factor 2.5 increase of electricity from RES compared to reference scenarios. Electricity from other RES is not affected by the other scenario variants, e.g. low fossil fuel prices, as most of the wind capacity is installed by 2020 to conform with Dutch EU RED targets. Nonetheless, other assumptions, such as higher CO_2 emission taxes, or higher targets of RES for low-carbon power systems could lead to other outcomes regarding other RES (Brouwer et al., 2016). Furthermore, Brouwer et al. (2016) suggest that low biomass prices, could place electricity generation from biomass earlier in the merit order than electricity from natural gas; thus, biomass could have a larger role in the electricity sector.

At a sector level other scenario variants, namely biomass cost-supply and fossil fuel prices, do not have significant influence, as ranges are found to be comparable with those of technology development scenarios (whiskers in Figure 5.4a). However, a combination of scenarios may lead to ±50% variation of final production from renewables compared to the reference (Figure 5.4a). The high technology development scenario combined with high fossil fuel prices leads to 745 PJ renewable energy output. Low technology development combined with low fossil fuel prices leads to 230 PJ, which is comparable to baseline scenarios (i.e. without CO_2 tax beyond 2020). Outcomes of low fossil fuel price scenarios are comparable with high-cost biomass scenarios for the two technology development scenario variants (Table 5.2). On the other hand, high fossil fuel prices drive renewable energy output more than low-cost biomass scenarios (Table 5.2). However, this occurs primarily in the electricity sector as renewable electricity is supplied by other RES, i.e. non-biomass sectors, namely wind and solar.

While in absolute terms, final energy supply from biomass per sector is comparable across all scenario variants (Figure 5.4a), in relative terms, its contribution to the sector's final energy varies (Figure 5.4b). Most notable is the contribution of biofuels to road transport fuels, which goes beyond 60% under high technology development combined with high fossil fuel price assumptions. This occurs due to the increased biofuel supply (20% higher than in the reference scenario) and due to reduced fuel demand by the sector (roughly 1/3 or 130 PJ decrease compared to the reference scenario, as more efficient vehicles are deployed). These are primarily wheel motor hybrid vehicles with 76% higher efficiency compared to regular petrol cars found in reference scenarios (i.e. 927 compared to 526 km driven/GJ van Vliet et al. (2011)). The market constraint on individual 2nd generation biofuel technologies (section 5.2.2.2) in high technology development assumptions, limits the production output of FT-fuels in the reference and low-cost biomass scenarios. In the high fossil fuel price variant it limits 2nd generation ethanol production due to the deployment of hybrid petrol engines for which ethanol is the substitute. The share of 1st generation biofuels over the total transport fuel demand is 1-13% and 1-12% in low and high technology development scenario variants, respectively. The share of 2nd generation biofuels over the total transport fuel demand is 0-3% and 10-29% in low and high technology development scenario variants, respectively. With massive efforts to improve the thermal performance of buildings, similar benefit effects could be demonstrated for heat. Nonetheless, such measures are not included in the present study.

Low technology development combined with low fossil fuel price scenarios can reduce the contribution of biofuels to the sector down to only 1%. Similarly, in the high technology development scenario, while the output of RJF and biochemicals to total final energy supply is relatively small, their contribution to their sectors is up to 7% and 17%, respectively. This is quite significant considering that today's output is limited and that within a 15-year timeframe such developments can obtain a large market share.

Figure 5.5 shows the renewable energy share and the contribution of biomass in more detail. Biomass contributes more than two thirds to the share of renewable energy in reference (Figure 5.5a) and biomass cost-supply scenarios (Figure 5.5b). Access to low-cost biomass



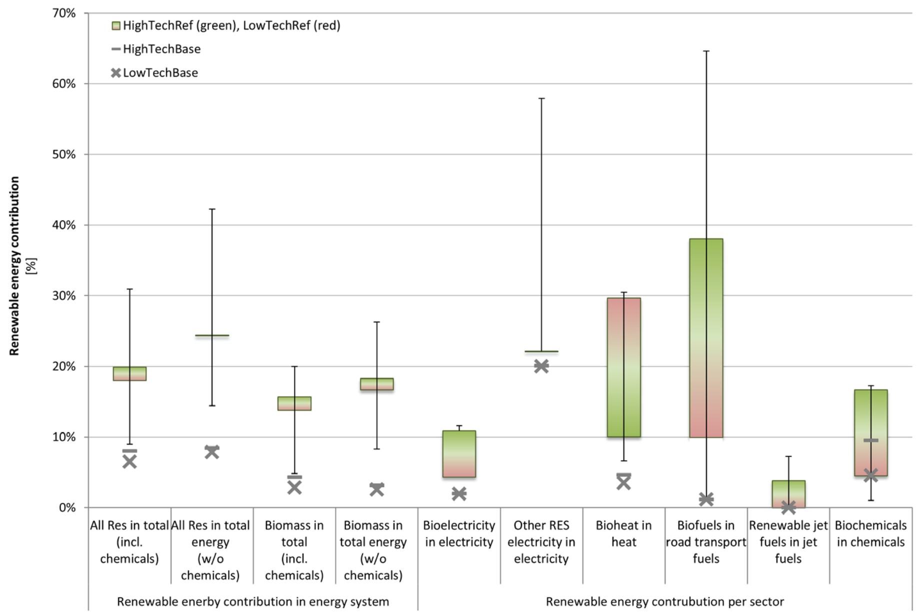


Figure 5.4 (a) Final renewable energy and non-energy supply renewable resources in the energy system and per sector (top figure) and (b) contribution of renewable energy and non-energy in the energy system and per sector (remainder is fossil fuels) in the Netherlands in 2030 (bottom figure). Bars indicate ranges of reference scenarios, whiskers indicate range of biomass cost-supply and fossil fuel price scenarios (see scenario descriptions in Table 5.1)

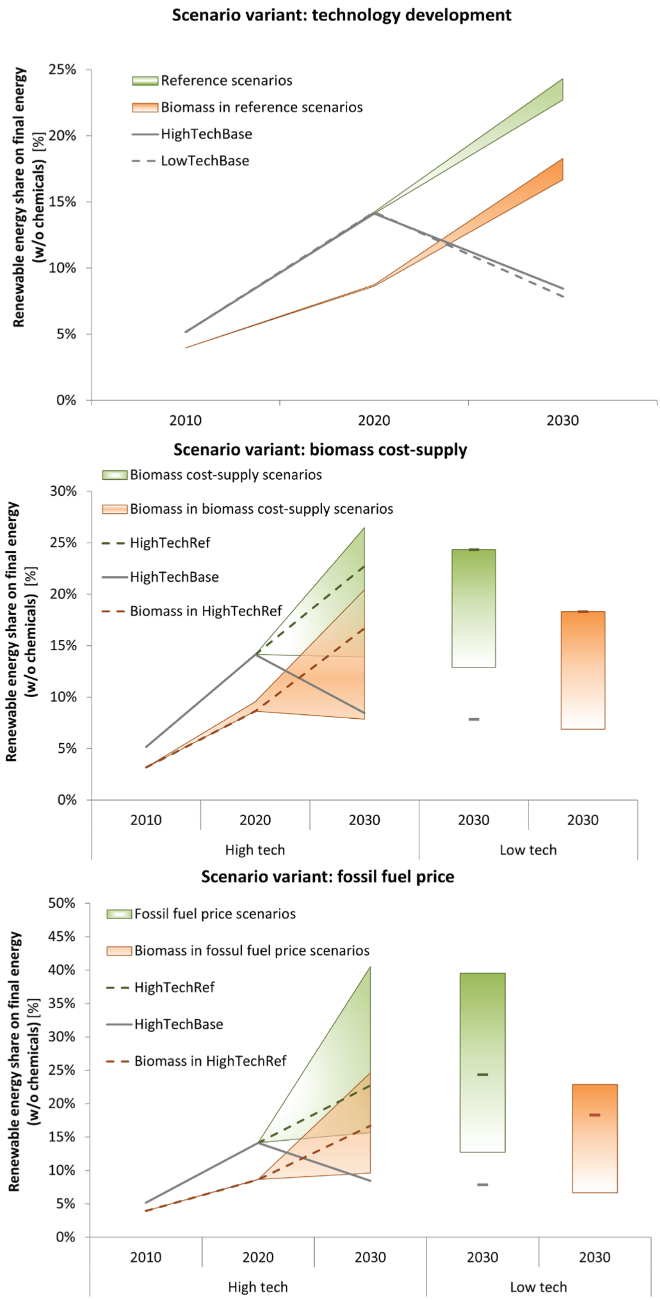


Figure 5.5 Renewable energy share on final energy and biomass contribution in the Netherlands in 2010-2030 under high technology development compared to low technology development in 2030 for (a) technology development (b) biomass cost-supply and (c) fossil fuel price scenarios. In the Low tech variant grey markers indicate the baseline and green markers the reference result (see scenario descriptions in Table 5.1)

increases the renewable energy share and the contribution of biomass by approximately 17% but only under high technology development scenarios. Limited access to low-cost biomass maintains its contribution in 2030 similar to 2020 levels, i.e. 14% share from RES, where biomass supplies approximately 50% of renewable energy. Under the low technology development scenario, access to low-cost biomass does not increase the RES share or its contribution. This entails that in terms of final energy supply. Restricted access to low-cost supply coupled with low technology development can pose barriers to cost-efficient deployment of biomass in the Netherlands.

Fossil fuel price variation leads to wider ranges (Figure 5.5c). First and foremost, under high fossil fuel prices the renewable energy share almost doubles compared to reference scenarios; biomass contribution does not follow the same relative growth due to the increase of electricity from wind turbines in the energy system. Under low fossil fuel prices, RES and biomass contribution remain in 2020 levels for both high and low technology development scenarios, similar to low-cost biomass scenarios.

5.3.2 Biomass consumption

In reference scenarios, biomass consumption in primary energy terms ranges between 450-620 PJ in 2030. Early in the time horizon (2020), consumption driven by technology development is relatively small, at approximately 200 PJ in both technology development scenarios and is comparable to baseline projections. Nevertheless a factor 2 growth is observed compared to 2010. By 2030, due to the CO_2 tax, biomass consumption is 330-460 PJ higher than the baseline (Figure 5.6a).

Access to low-cost biomass shows that additional 100 PJ are used by the energy system (Figure 5.6b), however only under high technology development assumptions. Total biomass consumption exceeds 700 PJ, which as seen in (Figure 5.6b) also increases by roughly 4% the contribution of RES and biomass to the energy system. On the other hand, high biomass costs can reduce biomass consumption levels significantly (to slightly above 300 PJ) even under high technology development assumptions. The range found in biomass cost-supply scenarios combined with low technology development by 2030 is significantly smaller than in high technology development, i.e. 240 PJ compared to 400 PJ. The high technology development scenario shows growth in biomass consumption between 2020-2030 but in the low technology development scenario consumption levels remain fairly constant. The above indicates that low technology development could impede long-term bioeconomy growth as indicated by biomass consumption.

Biomass consumption is also highly sensitive to the assumed development of fossil fuel prices to 2030 (Figure 5.6c). For high and low technology development scenarios, biomass consumption is more sensitive to low than high fossil fuel price assumptions. A 50% increase in fossil fuel prices leads to approximately 25% increase in biomass consumption in high and low technology development scenarios compared to their reference (indicated in Figure 5.6c by the area above the dotted line and the upper marker in high and low technology development, respectively). Low fossil fuel prices lead to 50-60% reduction of biomass consumption found in reference scenarios. For the high technology development scenario, high and low fossil fuel prices lead to similar consumption levels with low-cost and high-cost biomass, respectively (Figure 5.6b, Figure 5.6c). Therefore, even under unfavourable conditions induced by low fossil fuel prices, high technology development scenarios demonstrate small but stable growth in biomass consumption. On the contrary, low technology development scenarios, consume maximum 560 PJ under most favourable conditions induced by high fossil fuel prices, which are comparable to reference consumption levels of the high technology development scenario (the upper range of the bar is comparable to the upper range of the dotted line in Figure 5.6c).

5.3.3 CO_2 emissions

In Figure 5.7a it is shown that the CO_2 tax leads to emission reduction in the range of 35-43% for low and high technology development scenarios, respectively, compared against projected baseline emissions. In absolute terms, additional emission reduction due to high technology development is 15 $MtCO_2$ compared to low technology development. The decreasing trend in emissions is steeper in 2020-2030 enabled by higher tax levels and additional technological options. Nevertheless, even under the high technology development scenario additional 20 $MtCO_2$ emission reduction is required to reach the 40% emission reduction target, compared to 1990. Low-cost biomass supply leads to additional 5 $MtCO_2$ reduction only under the high technology development scenario, partly bridging the gap with the target (Figure 5.7b). As low technology development scenarios do not utilise additional low-cost biomass compared to reference outcomes (Figure 5.6b), they offer no additional emission reduction (Figure 5.7b). High-cost biomass supply leads to emission reduction levels comparable with those achieved in the reference low technology development scenario (i.e. approximately 35% compared to the baseline or direct CO_2 emissions in the range of 120 $MtCO_2$). Similar reduction is observed in the high technology development scenario under low fossil fuel prices (Figure 5.7c). However, under such circumstances the distance to the 40% emission reduction target is 30% (or 40 $MtCO_2$). To remain in cost-efficient emission reduction trajectories, high technology development seems to be a no-regret solution even under unfavourable conditions shaped by high-cost biomass or low fossil fuel prices as they offer significant potential for deeper emission reduction. More specifically, results indicate that under high fossil fuel prices, the 40% emission reduction target is reached in the high technology development scenario. In the low technology development scenario, however, CO_2 mitigation is 12 $MtCO_2$ behind the target (Figure 5.7c). Note that these emissions exclude those that occur outside the geographic boundaries of the Netherlands from production and transport of biomass, land use change, extraction and transport of fossil fuels and jet fuels (section 5.2.3).

Figure 5.8 shows the amount of carbon captured and stored by CCS and BECCS across the different scenarios (19-41 $MtCO_2$). The contribution of CCS and BECCS in emission reduction is significant (42-60% compared to the baseline). The remainder of emission reduction is primarily achieved through biomass (20-40 $MtCO_2$), as with the exception of high fossil fuel price scenarios, the capacity of wind power and other RES does not increase significantly compared to the baseline.

CCS is stimulated by the high CO_2 tax while in baseline scenarios no CCS is deployed. The

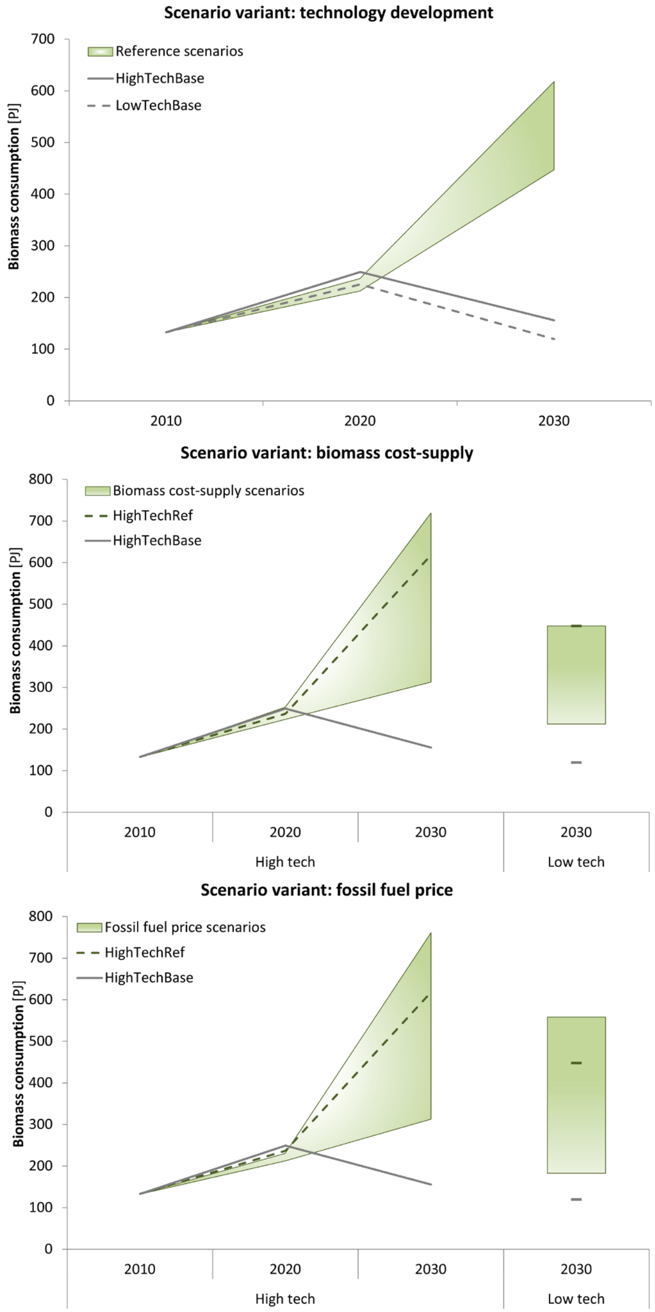


Figure 5.6 Biomass consumption in the Dutch bioeconomy in 2010-2030 under high technology development compared to low technology development in 2030 for (a) technology development (b) biomass cost-supply and (c) fossil fuel price scenarios (see scenario descriptions in Table 5.1). In the Low tech variant grey markers indicate the baseline and green markers the reference result

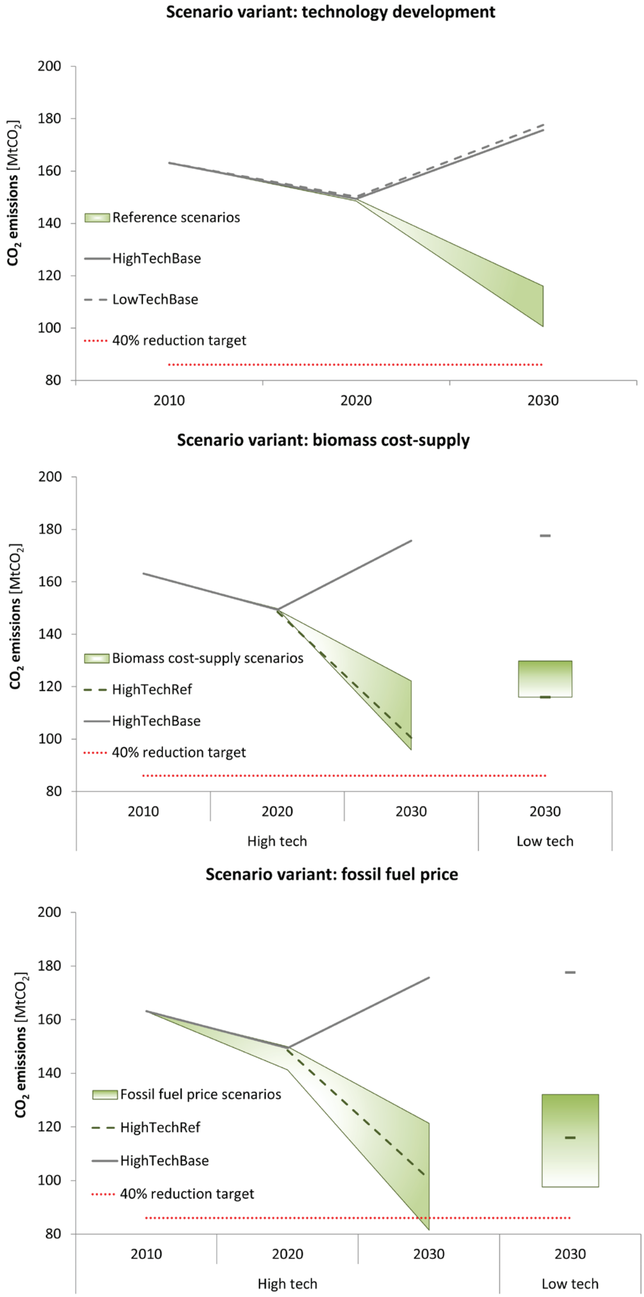


Figure 5.7 CO_2 emissions in the Netherlands in 2010-2030 under high technology development compared to low technology development in 2030 for (a) technology development (b) biomass cost-supply and (c) fossil fuel price scenarios (see scenario descriptions in Table 5.1). In the Low tech variant grey markers indicate the baseline and green markers the reference result

key difference across the technology development scenarios is the deployment of BECCS in gasification technologies that supply FT-fuels to the transport sector. These technologies are assumed not to be available in low technology development scenarios. The carbon capture by the power sector is primarily associated with retrofitted coal-based power plants. It represents more than 65% of the emissions captured and stored by the sector. The remainder is associated with gas-based capacity and is similar across the technology development scenarios. In high technology development scenarios BECCS represent 16-50% of the emissions captured and stored.

In scenarios that assume high fossil fuel prices, CCS and in addition BECCS in high technology development scenarios, represents 10-20% of the emission reduction achieved compared to the baseline (10-20 $MtCO_2$ is stored). In these scenarios significant emission reduction is achieved through other RES, as the output of wind electricity increases by approximately a factor 3, compared to the baseline (emission reduction from bioenergy and other RES is 70-75 $MtCO_2$). In addition, less coal capacity is projected to be used. Due to the decrease in demand for transport fuels by deployment of efficient vehicles the transport sector also contributes to emission reduction.

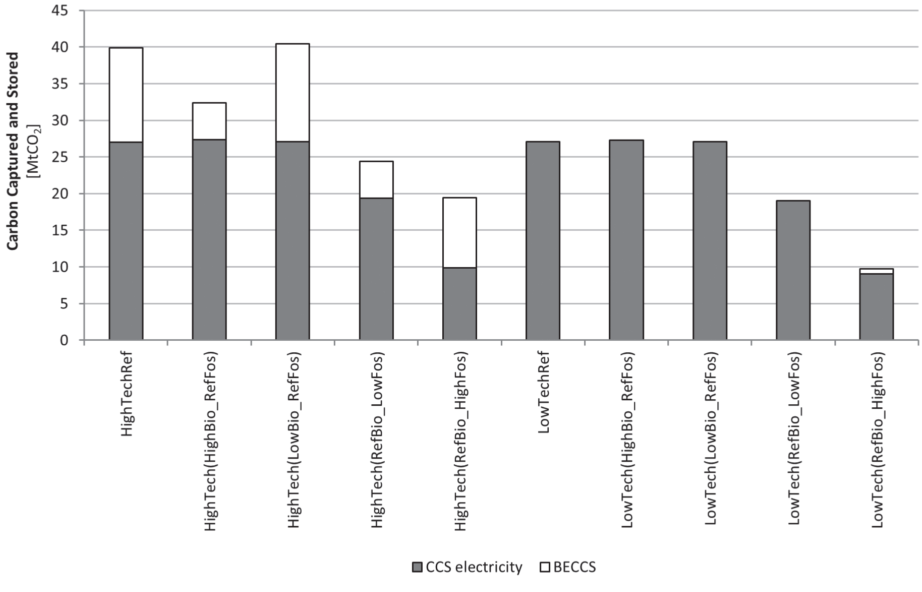


Figure 5.8 Carbon captured and stored across different scenario variants in the Netherlands in 2030

5.3.4 System costs

We compare total annual system costs and total direct CO_2 emissions in 2030 between all scenario variants and the baseline scenario of high technology development (Figure 5.9). This reveals that total system costs in most scenarios increase from 0.6-13.1 bn€/yr compared to the high technology development baseline. An exception are the low fossil fuel price scenario variants which show lower costs of about 6.5 bn€/yr. Annual system costs are most sensitive to fossil fuel price variation. A 35% decline in oil prices (section 5.2.2.3) reduces annual system costs by about 9% and a 50% increase in oil prices increases annual system costs by approximately 18-19% in 2030, compared to the high technology development baseline (Table 5.4). High technology development scenarios consistently show lower system costs and CO_2 emissions in 2030 and cumulative system costs and CO_2 emissions over the period 2010-2030 (Table 5.5) when compared to low their low technology development counterparts. This illustrates that high technology development is a no-regrets solution also when costs are taken into account. Note, though that external costs such as support to R&D or to 1st-of-a-kind plants, which are required to support high technology development are not taken into account.

5.3.5 Other scenarios

5.3.5.1 Low demand for chemicals

Figure 5.10 shows that assuming decline in demand for chemicals over time in combination with delayed decommissioning of old steam cracking capacity in the Netherlands affects the production output of biochemicals. This is noticed early in the time horizon (2020), when under low technology development assumptions, no production of biochemicals takes place, and under high technology development assumptions the production output

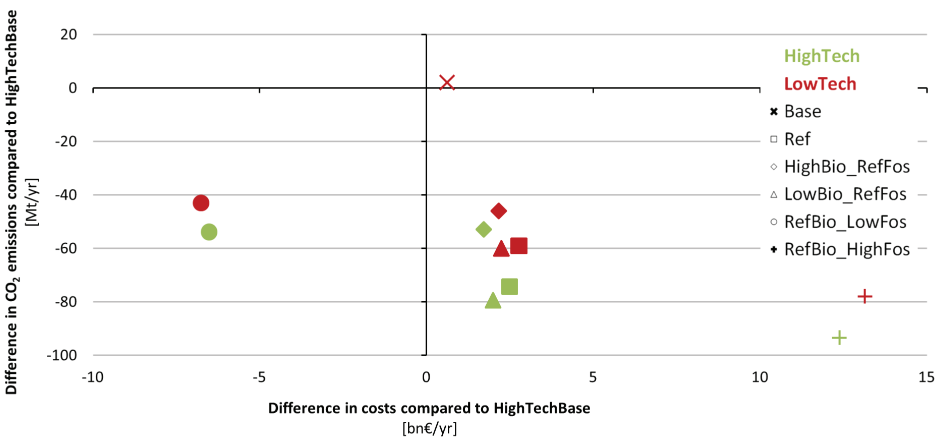


Figure 5.9 Difference of total annual system costs and total direct emissions in 2030 between HighTechBase and the other scenario variants

is reduced by 75% compared to the reference. The difference in production output between scenarios becomes smaller by 2030, when under reference fossil fuel prices lower demand for petrochemicals leads to a 16-37% reduction of output in low and high technology development compared to their reference. However, assuming low fossil fuel prices creates an uncompetitive environment for biochemicals throughout the modelling period. This may be also an outcome of the limited number biochemicals that are assumed in this study combined with the fact that the CO_2 tax does not affect non-energy use. As Figure 5.11 shows, assuming lower demand for chemicals does not affect the direct CO_2 emissions of the energy system. Compared to their reference scenarios, the low chemical demand scenarios lead to 4-5% lower CO_2 emissions, primarily due to less process energy emissions (electricity, heat) as a result of decrease in industrial demand.

5.3.5.2 Decommissioning of coal-based power capacity

Decommissioning coal-fired power stations in the Netherlands after 2020 leads to increase in wind electricity by 55-75% (48-65 PJ; off-shore wind turbine capacity increase of 5.2 GW_e and 3.9 GW_e in low and high technology development, respectively, compared to the reference) and 13-18% (30-39 PJ) in natural gas-based electricity. By 2030, off-shore wind turbines are is expected to become more cost-efficient than other options leading to higher renewable energy share and contribution in the electricity sector. Deployment of on-shore wind turbines reaches constraint levels (8 GW_e) across all scenarios with high CO_2 tax by 2030. Despite the significant deployment of wind power, direct CO_2 emissions remain at levels comparable with reference scenarios. Direct CO_2 emissions are decreased by 1 $MtCO_2$. That is because CCS combined with coal power plants is no longer an available mitigation option and wind turbines compete with CCS. Overall the total carbon removed and stored by CCS is lower by 15 $MtCO_2$ and 12.5 Mt in low and high technology development, respectively. This scenario requires additional 6.7-9 M€/yr from 2020 onward, and increases CO_2 mitigation costs by 12% and 14% in high and low technology development, respectively (Table 5.4 and Table 5.5).

5.4 DISCUSSION

This study compared multiple scenario outcomes of an energy system model to gain insights in CO_2 emission reduction that can be achieved by renewable energy, CCS and BECCS deployment when driven by cost-competition with fossil fuel alternatives. We used the MARKAL-NL-UU model, which includes a representation of modern and emerging biomass conversion technologies, other renewable and fossil fuel conversion technologies. We did not incorporate any policy assumptions beyond 2020 to allow for free competition between all options. Using CO_2 tax as the only instrument for emission reduction, we assessed the achievement of or the distance to the EU's 40% emission target in 2030 compared to 1990.

We incorporated different biomass cost-supply curves to assess how deployment of biomass conversion technologies at a sectoral level and emission reduction at a systems level can be affected. We assessed how dependent the national bioeconomy and renewable energy system is on fossil fuel price variation. By combining biomass cost-supply and fossil fuel price

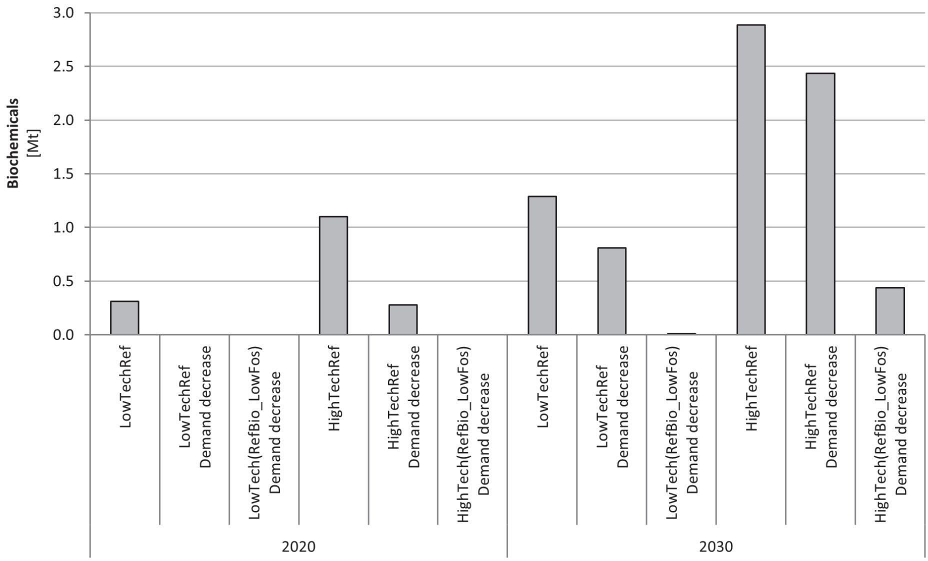


Figure 5.10 Biochemical output in reference and low fossil fuel price scenarios assuming different growth rates for the chemical industry in the Netherlands in 2020-2030

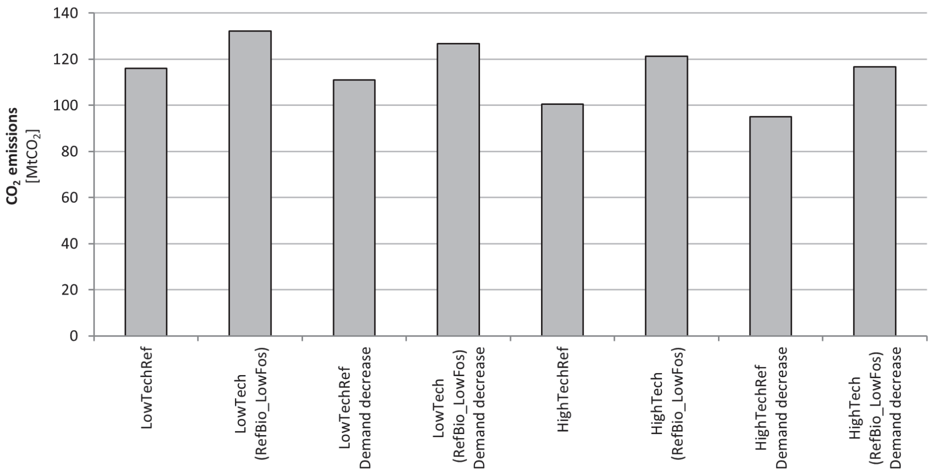


Figure 5.11 Direct CO_2 emissions in reference and low fossil fuel price scenarios assuming different growth rates for the chemical industry in the Netherlands in 2020-2030

scenarios with different assumptions on technology development, which vary in learning progress and technical parameters of technologies, we captured key uncertainties of mid-term bioeconomy development. This is one of the first studies that demonstrates how emission reduction targets are affected when new bioeconomy sectors are included in the energy system next to CCS and alternative RES sources such as wind and solar.

Nonetheless, there are important considerations that should be taken when interpreting the outcomes. Firstly, we used CO_2 emission pricing as the instrument to stimulate emission reduction as opposed to applying a cap on national emissions. The level of the CO_2 price was determined exogenously, aligned with longer-term climate targets (OECD/IEA, 2015). In most scenarios the 40% emission reduction target is not reached albeit significant reduction is realised (46-97 $MtCO_2$ across scenarios compared to baseline; Figure 5.7). The CO_2 price assumptions of IEA-WEO 2015 reflect the EU, and not the required level for an individual country such as the Netherlands to achieve the target. Evidently, a higher CO_2 price would be required for the Netherlands. In addition, the assumed CO_2 price is an outcome of simulation where other policy measures and technologies such as energy efficiency are taken into account. Such measures are not included in our model. It could be argued that the assumed CO_2 price would be adequate to achieve the target in all scenarios had low-cost efficiency measures such as insulation of buildings been included. Then, the abatement achieved by biomass, other RES, CCS and BECCS could be lower. Related to the above, is that the CO_2 tax is assigned on sectoral emissions and not on the fossil carbon they consume. This is relevant for the chemicals sector, which consumes large volumes of fossil carbon as feedstock that remains embedded in the products. Applying the tax on fossil carbon consumption similar to other studies (e.g. Daioglou et al. (2014)) could lead to different system dynamics because the benefit of avoiding CO_2 emissions from waste management would be taken into account. However, for a national model, this entails an improved representation of end-of-life phase of products, where cascading uses and exports of chemicals are taken into account (Chapter 4). This study finds that significant volumes of biochemicals could potentially be produced by 2030 (5-20% of total chemical output in final energy terms; Figure 5.4b), even under baseline assumptions (5-10% of total chemical output in final energy terms; Figure 5.4b). This entails that there may be a high potential for cascading uses of biomass from higher to lower value applications (Keegan et al., 2013). While this is not modelled in this study, it is important to point out that cascading uses would lead to increase in efficient biomass use in the energy system, and possibly to an increased output of biomass heat and electricity. A combination of factors such as delayed retirement of steam cracking capacity, low fossil fuel price environment and decline in demand have an impact on the competitiveness of biochemicals as the output becomes negligible. Furthermore, the outcomes represent only domestic emissions that occur in the Netherlands. The emissions related with production and transport of fossil fuels and biomass outside the Netherlands are not included. Consequently, neither are emissions from direct and indirect land use change. Emissions from indirect land use change are rather uncertain (Wicke et al., 2012). However, this study demonstrates that large volumes of biomass may be consumed in the Netherlands in the medium term and indirect land use change emissions may influence global CO_2 emission reduction efforts. In Chapter 4 we showed that emissions from production and import of biomass from regions outside the Netherlands were approximately

4 $MtCO_2$, which are 4-9% compared to the emission reduction achieved across scenarios from the baseline of this study. These emissions do not affect the main conclusions drawn in this study, as they do not affect domestic emissions and distance to target. However, they are relevant when emissions at a larger geographical scope are assessed.

While the above are important to consider, this study shows results that are well-aligned with other efforts. A study that assessed lowest-cost complementarity of integrating fossil-based capacity with predetermined RES diffusion to achieve low-carbon power systems, illustrates the significance of wind turbines and CCS to achieve emission reduction (Brouwer et al., 2016). While there are key differences in scope (geographical, temporal) and modelling techniques between the present study and Brouwer et al. (2016), they both show that lowest system costs are achieved with a mix of RES and CCS in the power sector. Similar outcomes are supported by van den Broek et al. (2011) in scenarios which take ambitious climate policies into account. A key difference between the outcomes of these studies compared to the results presented here is the deployment of CCS in gas-fired instead of coal-fired plants. An explanation to this can be the recent instalment of coal-based capacity in the Netherlands, which remains operational until 2030. An additional explanation could be that in the present study more sectors are included in the energy system. As van Vliet et al. (2011) showed, when accounting for the transport sector in the energy system the role of BECCS in biomass-based FT-fuel production is prominent. This finding, as confirmed by the present study, is also relevant when emerging bioeconomy sectors are included in the energy system. Therefore, the significance of BECCS is demonstrated not only as a longer-term emission mitigation option, which many studies support (Fischedick et al., 2011; Fuss et al., 2014), but also in earlier in the time horizon, provided that the technology can be commercialised within the assumed timeframe.

Regarding biochemicals, to our knowledge there are limited studies that provide future estimates at a systems level, as for example Daioglou et al. (2014). According to their study, biomass has the potential to supply up to 40% of total demand for non-energy in 2100 (or about 45 EJ/yr; Daioglou et al. (2014)). Other studies have also performed assessments of future biochemical potential (e.g. Dornburg et al. (2008), Ren & Patel (2009), Ren et al. (2009); nova-Institut (2013), Saygin et al. (2013, 2014), Gerssen-Gondelach et al. (2014), Piotrowski et al. (2015)) without, however, taking systems dynamics into account. European Bioplastics estimate that global production capacity of bioplastics will reach 7.85 Mt in 2019 (EuBP, 2016). Our study estimates that production output of biochemicals may reach up to 1.1 Mt in the Netherlands in 2020 depending on scenario conditions. While results of these studies cannot be directly comparable with the output of this study, they all confirm that biochemical increases over time.

Against this background the most important observations can be summarised in the following:

The size of bioeconomy depends on developments across the supply chain and the fossil fuel price. By 2030, the contribution of biomass in renewable energy supply is higher than the approximately 50% that is anticipated according to other studies by 2020 (Rijksoverheid, 2010; Stralen et al., 2013). It ranges between 52-77% and corresponds to biomass consumption volumes of 183-760 PJ, depending on scenario assumptions. Biomass supply depends on intra-EU and extra-EU biomass and based on literature it is deemed available (Chum et al., 2011; Ganzevles, 2014; Smeets, 2014). The supply from RES observed in the decade 2020-2030 is due to technological growth and increase in the CO_2 emission tax. Other RES remain fairly constant to 2020 levels, while the bioeconomy grows. Investments across the supply chain both on the supply side, as modelled by the low-cost biomass scenario, and on the conversion side, as modelled by the high technology development scenario, lead to increased contribution of biomass in the system (Figure 5.2a). Low fossil fuel prices do not lead to contraction of the RES share compared to 2020 and reduce total system costs by about 6.5 bn€/yr in 2030, however, even under high technology development no growth is observed. In the face of low fossil fuel prices, mechanisms are required to ensure bioeconomy growth such as a CO_2 tax higher than 69 €/tCO_2 by 2030.

A mixed technology portfolio is required to achieve deep emission reduction. A wide technology portfolio is required to achieve emission reduction in the medium term, to realise long-term climate goals. In particular, the role of wind in the electricity sector, bioenergy in road transport and industrial heat, but also CCS and BECCS are significant. This finding is widely supported by literature (IPCC, 2012; Matthews et al., 2015; Winchester and Reilly, 2015). Introducing new bioeconomy sectors in the energy system, namely biochemicals and RJF does not alter it. As other RES do not increase significantly in the scenario outcomes, except when high fossil fuel prices are assumed, the post-2020 emission reduction can be attributed to biomass (20-40 MtCO_2 or 40-60% compared to the baseline) and CCS (19-41 MtCO_2 or 42-60% compared to the baseline). In high technology development scenarios that, among other options, include BECCS, emission reduction is higher by 6-17% (7.5-20 Mt) compared to low technology development scenarios (BECCS contributes 47-83%). BECCS can have a significant role earlier in the time horizon than most studies indicate (Fischedick et al., 2011; Fuss et al., 2014), if the technology is commercialised. With demand-side improvements (e.g. on industrial and residential energy efficiency), the role of biomass heat may diminish in the longer term. This could create opportunities for other bioeconomy sectors to grow. Such an assessment requires incorporation to the model of energy efficiency measures or a longer term temporal scope (e.g. 2040).

Sector-specific assumptions do not compromise the potential emission reduction. A decrease in demand for chemicals in combination with other factors such as delayed retirement of steam cracking capacity and low fossil fuel prices affects the size of the biochemical sector. The latter reduce the output of biochemicals by about 70% compared to the reference, while combined with the former assumptions the reduction ranges between 85-99%. However, the systems' CO_2 emissions are not affected. Furthermore, dismantling all coal-based power generation capacity, leads to an increase of RES (wind) and natural-gas power generation. While coal is effectively phased out entirely from the energy system of the Netherlands, the emission levels in 2030 remain the same as CCS capacity compared to reference scenarios is lower and the emission reduction is offset by wind turbines.

High technology development is a no-regrets option to achieve deep emission reduction. Post-2020, high technology development uses 313-760 PJ of biomass depending on scenario assumptions. Compared to the low technology development counterparts, it offers additional opportunities to utilise biomass in the energy system as indicated by the additional 100-270 PJ. High technology development combined with the low-cost biomass scenario use approximately 100 PJ more compared to the reference. Assuming low-cost biomass does not lead to increased consumption in low technological growth scenarios. Thus, improvements early in the supply chain increase the size of the bioeconomy only under high technological growth. Furthermore, high technology development consistently leads to lower emissions and cumulative system costs than low technology development in 2030. At the same time, high technology development creates a more resilient bioeconomy even if fossil fuel prices remain low as there is continuous growth to 2030. However, this observation excludes external costs, which are required to achieve high technological growth, such as in R&D or support to 1st-of-a-kind plant. Nonetheless, to achieve deeper levels of emission reduction required to embark on low-cost trajectories that meet long-term climate targets high technology development is needed.

5.5 APPENDIX

Table 5.2 Final renewable energy consumption per sector and scenario and total final renewable energy consumption (including chemicals) per scenario in the Netherlands in 2030

		Final renewable energy						
		Electricity Other RES	Electricity Biomass	Heat Biomass	Road transport Biofuels	Aviation Renewable jet fuels	Chemicals Biomass	Total RES
		[%]						[PJ]
Baseline	LowTechBase	57%	6%	19%	3%	0%	16%	165
	HighTechBase	46%	5%	21%	2%	0%	26%	203
Reference	LowTechRef	23%	5%	58%	9%	0%	6%	462
	HighTechRef	21%	10%	18%	31%	1%	19%	510
Biomass cost-supply	LowTech (HighBio_RefFos)	43%	9%	38%	3%	0%	8%	250
	HighTech (HighBio_RefFos)	36%	7%	20%	20%	5%	13%	302
	LowTech (LowBio_RefFos)	23%	5%	58%	9%	0%	6%	462
	HighTech (LowBio_RefFos)	19%	9%	26%	28%	1%	17%	582
Fossil fuel prices	LowTech (RefBio_LowFos)	46%	9%	40%	2%	0%	3%	233
	HighTech (RefBio_LowFos)	34%	10%	28%	16%	4%	7%	315
	LowTech (RefBio_HighFos)	40%	4%	39%	12%	0%	4%	693
	HighTech (RefBio_HighFos)	35%	7%	23%	25%	1%	9%	745

Table 5.3 Final energy consumption per sector and scenario and total final energy consumption (including chemicals) per scenario in the Netherlands in 2030

		Final energy					
		Electricity	Heat	Road transport	Aviation	Chemicals	Total
				[%]			[PJ]
Baseline	LowTechBase	18.4%	35.2%	16.1%	7.7%	22.6%	2,544
	HighTechBase	18.5%	35.4%	16.2%	7.7%	22.1%	2,524
Reference	LowTechRef	19.1%	34.9%	15.9%	7.6%	22.5%	2,562
	HighTechRef	18.8%	34.9%	16.0%	7.6%	22.6%	2,561
Biomass cost-supply	LowTech (HighBio_RefFos)	18.9%	34.6%	15.9%	7.5%	23.1%	2,586
	HighTech (HighBio_RefFos)	19.0%	35.0%	16.0%	7.6%	22.3%	2,556
	LowTech (LowBio_RefFos)	19.1%	34.9%	15.9%	7.6%	22.5%	2,561
	HighTech (LowBio_RefFos)	18.9%	34.9%	16.0%	7.6%	22.6%	2,565
Fossil fuel prices	LowTech (RefBio_LowFos)	18.3%	34.4%	15.8%	7.5%	23.9%	2,598
	HighTech (RefBio_LowFos)	18.4%	34.6%	15.9%	7.6%	23.5%	2,582
	LowTech (RefBio_HighFos)	19.8%	36.9%	12.5%	8.0%	22.8%	2,428
	HighTech (RefBio_HighFos)	19.8%	37.2%	11.8%	8.1%	23.0%	2,405

Table 5.4 Primary biomass consumption, direct CO_2 emissions and undiscounted total system costs per scenario in the Netherlands in 2030

		Biomass consumption	Direct CO_2 emissions	Annual costs
		[PJ]	[$MtCO_{2eq}$]	[bn€]
Baseline	LowTechBase	120	178	70.3
	HighTechBase	156	176	69.7
Reference	LowTechRef	447	117	72.4
	HighTechRef	618	101	72.2
Biomass cost-supply	LowTech(HighBio_RefFos)	212	130	71.8
	HighTech(HighBio_RefFos)	313	123	71.4
	LowTech(LowBio_RefFos)	448	116	71.9
	HighTech(LowBio_RefFos)	720	96	71.7
Fossil fuel prices	LowTech(RefBio_LowFos)	183	133	62.9
	HighTech(RefBio_LowFos)	313	122	63.2
	LowTech(RefBio_HighFos)	558	98	82.8
	HighTech(RefBio_HighFos)	762	82	82.0
Other scenarios[a]	LowTech(LowChem_NoCrack)	435	112	69.7
	HighTech(LowChem_NoCrack)	617	95	69.8
	LowTech(LowChem_NoCrack_LowFos)	177	127	61.5
	HighTech(LowChem_NoCrack_LowFos)	301	117	61.2
	LowTech(RefBio_RefFos_NoCoal)	418	116	72.8
	HighTech(RefBio_RefFos_NoCoal)	618	99	72.5

[a]These scenario variables are defined in section 5.2.2.4 and results are presented in section 5.3.5. Scenarios using the abbreviation “LowChem” assume decreasing chemical demand. Scenarios using the abbreviation “NoCrack” assume no decommissioning of steam crackers. Scenarios using the abbreviation “NoCoal” assume dismantling of coal-power plants beyond 2020. All other scenario variants are same as defined in Table 5.1.

Table 5.5 Cumulative total system costs and cumulative total system direct CO_2 emissions in the Netherlands in 2010-2030

		Cumulative costsa 2010-2030	Cumulative emissions 2010-2030
		bn€	[$MtCO_{2eq}$]
Baseline	LowTechBase	629	3.412
	HighTechBase	627	3.397
Reference	LowTechRef	633	3.111
	HighTechRef	631	3.027
Biomass cost-supply	LowTech(HighBio_RefFos)	632	3.164
	HighTech(HighBio_RefFos)	631	3.111
	LowTech(LowBio_RefFos)	632	3.054
	HighTech(LowBio_RefFos)	631	2.948
Fossil fuel prices	LowTech(RefBio_LowFos)	599	3.154
	HighTech(RefBio_LowFos)	598	3.120
	LowTech(RefBio_HighFos)	676	3.046
	HighTech(RefBio_HighFos)	673	2.981
Other scenariosb	LowTech(LowChem_NoCrack)	623	3.068
	HighTech(LowChem_NoCrack)	623	2.986
	LowTech(LowChem_NoCrack_LowFos)	592	3.103
	HighTech(LowChem_NoCrack_LowFos)	591	3.077
	LowTech(RefBio_RefFos_NoCoal)	633	3.139
	HighTech(RefBio_RefFos_NoCoal)	631	3.047

[a]Future costs are discounted to 2010 assuming a discount rate of 7%. [b]These scenario variables are defined in section 5.2.2.4 and results are presented in section 5.3.5. Scenarios using the abbreviation "LowChem" assume decreasing chemical demand. Scenarios using the abbreviation "NoCrack" assume no decommissioning of steam crackers. Scenarios using the abbreviation "NoCoal" assume dismantling of coal-power plants beyond 2020. All other scenario variants are same as defined in Table 5.1.

6

On the macro-economic impact of bioenergy and biochemicals – Introducing advanced bioeconomy sectors into an economic modelling framework with a case study for the Netherlands

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ABSTRACT

Advanced uses of biomass for bioenergy and biochemicals are being gradually introduced and are expected to grow considerably in regional economies, thus raising questions on their mid-term macro-economic impacts. To assess these impacts, we use a computable general equilibrium model and a regional energy systems model side-by-side. The former is extended with new sectors of lignocellulosic biofuels, bioelectricity, biochemicals, lignocellulosic biomass supply and tradeable pellets. Next to 1st generation biofuels and other renewable energy supply, the economic impacts of bioeconomy are assessed for technology development and trade openness scenarios. We demonstrate the macro-economic model by assessing developments of the Dutch bioeconomy in 2030. Under rapid technical growth and trade openness, the models consistently show increased biomass consumption and supply of bioenergy and biochemicals from lignocellulose through large-scale deployment of advanced biomass conversion technologies. Traditional fossil-based sectors are replaced by biomass, which brings additional macro-economic benefits on the gross domestic product (0.8 bn€/yr) and value added (0.7 bn€/yr). Furthermore, it reduces the projected decline in trade balance (0.7 bn€/yr) and employment (2.5-4.5%) compared to low technology development. Extending the temporal scope to beyond 2030 may demonstrate additional macro-economic benefits of bioeconomy. This requires assessing the influence of improvements in the agricultural sector that may lower biomass prices, learning and other developments of promising biomass conversion technologies in the longer term. Uncertain fossil fuel and CO_2 price developments necessitate additional sensitivity analysis.

6.1 INTRODUCTION

The role of biomass in today's economies extends beyond traditional sectors such as food, feed, materials (e.g. plant fibres, lumber, paper and pulp) and traditional uses for energy (inefficient heating and cooking). While more than half of current global biomass use for energy is traditional (48-54 PJ, 76-79% in 2008, Chum et al., (2011)), advanced and efficient supply of bioelectricity, biomass heat and biofuels is growing rapidly and is expected to continue so in the future (Chum et al., 2011). Lignocellulosic biofuels and advanced biomaterials are being gradually introduced in some regions (BRASKEM, 2016; Janssen et al., 2013; Voegele, 2016). Aviation and shipping rely exclusively on biofuels to partly decarbonise their energy use (Aviation Transport Action Group, 2012; Florentinus et al., 2012). Biochemicals and biochemical products (e.g. bioplastics) are already produced globally and consume about 4.5% of total biomass used for energy and biochemicals (Piotrowski et al., 2015). In 2016, global production capacity of bioplastics reached 2 Mt and based on the industry's projections it is expected to quadruple before 2020 (EuBP, 2016).

These expectations raise questions on the impacts of bioeconomy developments, possible synergies, and conflicts of biomass supply to different sectors, especially when in competition for biomass from emerging uses such as biochemicals, and on their role in climate change mitigation. Long-term projections with the global energy system simulation model TIMER show that bioenergy can contribute about 20% in emission reduction with carbon taxes above 130 \$/tCO_2 by 2100 (Daioglou et al., 2015). The largest greenhouse gas (GHG) emission reductions come from biofuels in road transport and bioelectricity in combination with carbon capture and storage (BECCS) whilst the emission reduction from biochemicals remains relatively small. Nevertheless, about 18% (19 EJ) of the chemical sector's secondary energy use may come from biomass in 2100 (Daioglou et al., 2015). Such outcomes demonstrate the importance of bioeconomy on a global scale and in the long term.

However, increase of biomass consumption for bioenergy and biochemicals in the medium term may already entail large and rapid changes in the structure of regional economies with possible effects on their gross domestic product (GDP), value added and trade balance. Under this perspective, applied economics are uniquely suited to understand the impacts of various policy and technical trajectories (Zilberman, 2013). Hoefnagels et al. (2013) assessed macro-economic impacts of bioeconomy developments in the Netherlands by combining a computable general equilibrium model (CGE; LEITAP) with a bottom-up Excel tool. Hoefnagels et al. (2013) demonstrated that substitution of fossil energy carriers by bioenergy and biochemicals may come with economic benefits and contribute to GHG emission reduction under preconditions of enhanced technology development and imports of sustainable biomass resources. However, as the authors indicate their approach was faced with limitations. Firstly, LEITAP did not include a detailed representation of lignocellulosic feedstocks such as agricultural residues thereby ignoring cost-efficiency improvements that can be achieved assuming densification and trade of solid biomass (e.g. pelletisation) and the impact of by-products. The availability of low-cost biomass can be pivotal for the competitiveness of biomass conversion technologies and can also have potential impacts on other sectors and land requirements. Secondly, bio-based and fossil-based conversion technologies were aggregated at a high level. As the authors indicate, the macro-economic model could benefit by improved cost-structures, especially on capital-intensive 2nd generation technologies that utilise low-cost biomass feedstocks (Hoefnagels et al., 2013). Thirdly, LEITAP did not treat other renewable energy sources (e.g. wind, solar). Sub-sectoral changes, however, were found to have major influence on the macro-economic impacts (Hoefnagels et al., 2013). Finally, the biochemical sector in LEITAP was modelled implicitly and its biochemicals product portfolio was limited. Therefore, the higher level of disaggregation and improved representation of competing resources, technologies and sectors are needed to shed light on underlying elements that can be critical for the bioeconomy.

Conversion of different biomass feedstocks to an array of food, feed, material, energy and biochemicals have created complex dynamics through which biomass participates in different sectors of the economy (van Meijl et al., 2015). The bioeconomy affects not only farmers, but also material, energy and chemical industries, the well-being of consumers, balance of trade, and the government budget. Understanding the impacts of the bioeconomy on the overall economy requires an improved modelling framework that accounts for the feedback mechanisms between bioeconomy and other markets, that takes direct and indirect effects of biomass use into account and covers the global dimension of supply, trade and sustainability that are inherent to biomass. CGE analysis' is considered most suitable (Sadoulet and de Javry, 1995; van Meijl et al., 2015). Partial equilibrium and input-output economic models do not capture the whole economy or include price effects, respectively. For example, CGE models have been used to address implications of biofuel policies on agricultural markets, land use change and related emissions. This led to improved endogenous modelling of land markets in CGE models (Wicke et al., 2012). Recent efforts also focused on improving modelling of biofuels by introducing ethanol, biodiesel and their by-products (Banse et al., 2011; Laborde, 2011), and prospective biomass feedstocks for advanced biofuels production such as corn stover, energy crops, palm oil residues (Taheripour and Tyner, 2013; van Meijl et al., 2012). To date, the biochemical sector is too small and there is no clear distinction in statistics and databases (e.g. Global Trade Analysis Project (GTAP)), which are used by CGE models. Furthermore, the chemical industry sector is aggregated at a high level, while in reality the sectoral flows of the industry are much more complex. As biomass conversion technologies to biochemicals may offer renewable alternatives at different levels (Chapter 4), disaggregation of the chemical industry sector in CGE models is required. Choumert et al. (2006) have presented a method on how to improve the oil-refining sector in the CGE model Emissions Prediction and Policy Analysis (EPPA); however, chemical products still remain at an aggregate level.

By using a global, multi-region, multi-sector CGE model, this article addresses the key limitations of the study conducted by Hoefnagels et al. (2013). To this purpose, we expanded the Modular Applied GeNeral Equilibrium Tool (MAGNET), the successor of LEITAP. MAGNET is expanded with advanced and emerging bioeconomy sectors and in particular lignocellulosic fuels, electricity, heat and chemicals from biomass. Furthermore, we improved the representation of biomass supply including agricultural residues, forestry residues and pretreatment of those feedstocks to pellets for international biomass trade. We extended the model with regional renewable energy policies that are crucial for current bioenergy developments. To overcome a key limitation of CGE models on technology representation, we improved technology details in MAGNET by collaborating with a cost-minimisation linear programming energy system model of the Netherlands (MARKAL-NL-UU). Recently, advanced biomass conversion technologies, biochemicals and renewable jet fuels (RJF), have been incorporated in MARKAL-NL-UU (Chapter 4, Chapter 5). The analysis shows that renewable electricity from wind turbines, biofuels, biomass heat and carbon capture and storage (CCS and BECCS) may play a crucial role by 2030 (Chapter 5). Biochemicals are expected to become cost-competitive with fossil-based chemicals as they are produced even when no drivers such as a CO_2 emission tax are assumed (5-10% of the sector's supply; Chapter 5). RJF, on the other hand, are produced only under specific assumptions that assume high technology development rates. Factors such as the rate of technical change, the cost-supply of biomass and fossil fuel price projections affect the level of biomass deployment in the energy system. Nonetheless, whether to meet renewable energy targets (Chapter 4) or to embark on cost-efficient emission mitigation pathways (Chapter 5), advanced and emerging biomass uses in the Netherlands need to grow substantially, from about 140 PJ in 2015 to up to 760 PJ in 2030 (Chapter 5). Model collaboration can take place as alignment and harmonisation of input data, detailed model comparison and model linkage (Wicke et al., 2015). Following Zilberman (2013), we apply a framework where the energy system model MARKAL-NL-UU (Chapter 4) is used side by side with MAGNET and supplies it with insights on technology trajectories to 2030. We compare results obtained by MAGNET and MARKAL-NL-UU and highlight points of interaction that can lead to improved representation of bioeconomy in CGE models that are required to assess its macro-economic impacts.

6.2 MATERIALS AND METHODS

To improve technology details of existing sectors and expand MAGNET with new bioeconomy sectors, we develop a modelling framework in which, the cost-minimisation linear programming energy system model, MARKAL-NL-UU is also used. MAGNET is a multi-regional, recursive-dynamic, applied general equilibrium model based on neo-classical microeconomic theory (Woltjer et al., 2014). MAGNET contains a number of advanced features pertinent to modelling the impact of technological and policy developments within the bioeconomy where land use is a crucial production factor. These features include factor market representations of imperfectly substitutable types of land, a land use allocation structure, segmented labour and capital markets, and a new land supply curve to address large reductions in the amount of available land for agriculture (Dixon et al., 2016; van Meijl et al., 2006). Biofuel production is included by introducing the production and use of ethanol and biodiesel and their by-products (Banse et al., 2008; Smeets et al., 2014). Blending targets are included in the model via an end-user tax on road transport fuels that is used to subsidise biofuel production and stimulate production up to the level implied by the blending target. MARKAL-NL-UU is a model of the Dutch energy system that has been recently expanded to assess techno-economic impacts of bioeconomy developments in the Netherlands to 2030 (Chapter 4, Chapter 5). Following a total system cost-minimisation paradigm MARKAL-NL-UU is suitable to highlight key technologies per sector under different scenarios in the medium term (Chapter 5). An overview of MAGNET is presented in section

6.6.1 An overview of MARKAL-NL-UU is presented in Chapter 4 and in section 6.6.2.

6.2.1 Overview of the modelling framework

We select key fossil and renewable energy conversion technologies (e.g. natural gas and coal, electricity, biomass co-firing in coal plants, on-shore and off-shore wind turbines), estimate their current and future cost-structures based on the MARKAL-NL-UU input database, and supply them to MAGNET. To select key advanced biomass conversion technologies (advanced biofuels and biochemicals), we define four scenarios that are developed around two axes of uncertainty, namely technology development and openness of market. These scenarios are applied on MARKAL-NL-UU, which is then deployed to assess the cost-optimal technology portfolio in each case, as the model projects the energy and chemical production mix per sector and technology. From MARKAL-NL-UU outputs we obtain the key biomass conversion technology portfolio of existing and new bioeconomy sectors for the Dutch region, we produce their cost-structures and learning pathways and provide them as data inputs to MAGNET to disaggregate key bioeconomy sectors and provide scenario assumptions for technological change for each scenario. This way, MAGNET is enabled to explicitly assess the macro-economic impacts of different technology development scenarios. The model is calibrated to version 9 of the GTAP database (Narayanan et al., 2015), which contains detailed production, bilateral trade, transport and protection data characterising economic linkages within and among regions. All monetary values of the data are in millions of US dollars (M$) and the base year for version 9 is 2007, which is updated to 2015 using macro-economic, yield data and energy data (see section 6.2.3). In the standard GTAP database, the definition of bio-based activities is limited to eight crop and four livestock sectors; eight processed food and beverages sectors, fishing, forestry, textiles, wearing apparel, leather, wood and paper products. Additional sources of bio-based activity (e.g. bioelectricity, biofuels, biochemicals) are subsumed within aggregated parent industries. In MAGNET, 1st generation biofuels were included prior to this study (Banse et al., 2008; Smeets et al., 2014) and updated with data from the Energy Information Administration (EIA, 2014). In this study, the database is extended to explicitly represent additional sources of biomass supply (i.e. residues, plantations and pellets), lignocellulosic biofuels based on thermochemical and biochemical pathways, bioelectricity and biochemicals (see section 6.2.2). A detailed overview of the sectoral and regional aggregation in MAGNET can be found in section 6.6 (Table 6.5). IMAGE-TIMER is used as a data source for estimating biomass cost-supply curves and technology details for the rest of the world. For new bioeconomy technologies that are not represented in international statistics and IMAGE-TIMER, the same cost-structures as obtained from MARKAL-NL-UU are assumed for other regions of the world. This does not influence the results of this study as production without a specific stimulating policy package is almost not existent in other world regions. Figure 6.1 presents the overview of the modelling framework. Section 6.2.1.1 describes the policy and scenario assumptions applicable to both models. A methodological description of the MAGNET's extension, technology selection and sectoral aggregation is described in section 6.2.2. The input data generated by MARKAL-NL-UU are presented in section 6.2.3. Finally a comparison of the model outcomes for bioenergy and biochemicals, and the macro-economic outcomes based on MAGNET are presented in section 6.3.1.

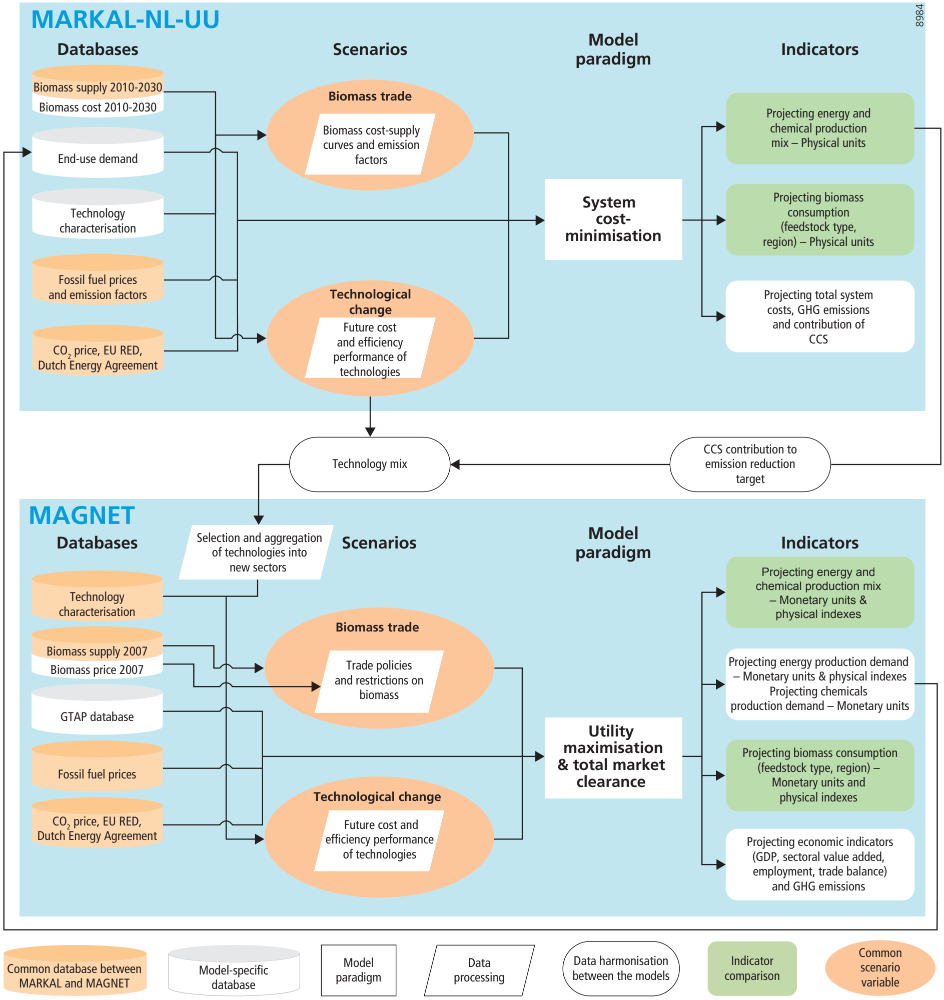

Figure 6.1 Model collaboration framework of MAGNET and MARKAL-NL-UU

6.2.1.1 Policy context and scenario assumptions

Policy context

In the medium term, regional (i.e. the Renewable Energy Directive of the European Union (EU RED), EC (2009b)) and national renewable energy policies (i.e. Dutch energy agreement, SER (2013)) are implemented to promote the deployment of renewable energy in efforts to mitigate climate change. These are incorporated as key policy assumptions in both models as they are expected to influence macro-economic and techno-economic bioeconomy developments in the short term. Firstly, the renewable energy share of electricity, heat and transport fuels for the Netherlands is 14% in 2020 based on the EU RED (EC, 2009b) and 16% in 2030 based on the Dutch energy agreement (SER, 2013). Secondly, the biofuel share in road transport is 10% in 2020 including double-counting of biofuels from waste and residues based on the EU RED (EC, 2009b). Thirdly, a maximum supply of electricity from biomass co-firing (25 PJ) is assumed based on SER (2013). In this study the latter two are continued to 2030. A minimum capacity of on-shore and off-shore wind turbines is supported by the Dutch government (1.8 GW additional capacity of off-shore wind turbines in 2015-2020 and 6 GW total capacity of on-shore wind turbines by 2020) (SER, 2013). Finally, a tax on CO_2 emissions is applied as an additional instrument to stimulate emission reduction (section 6.2.3).

Scenario context

We define and apply four scenarios, similar to Hoefnagels et al. (2013) that are developed around two axes of uncertainty, namely rate of technology development and openness of market. Due to their paradigms, the models incorporate these scenarios in a different manner, as discussed below. However, the models apply the same assumptions, thereby allowing data exchange between and output comparison of MAGNET and MARKAL-NL-UU.

Technology development

To assess the deployment prospects of biomass conversion technologies for the Dutch energy system and chemical industry in the medium term, we incorporate two scenarios designed to assess different technology development pathways. The two variants assume low (*LowTech*) and high (*HighTech*) speed of technology development. The technological assumptions in the two scenarios differ in terms of improvement rates in efficiency, year of commercialisation, scales and technology portfolio. These parameters, ultimately affect the cost-competitiveness of biomass conversion technologies compared to the reference fossil-based system and other renewable energy options. The scenarios are developed around 2nd generation biomass conversion technologies to biofuels and biomass conversion technologies to biochemicals. The technology development scenarios are described in detail in Chapter 4 and van Meijl et al. (2016).

The two variants are applied in MARKAL-NL-UU, which calculates the cost-optimal technology portfolio for each scenario. In the framework of this study, this acts as a criterion for selecting the key technologies in the road transport and chemical industry sector. As Figure 6.1 shows, the cost-structures over time of the selected technologies are supplied to MAGNET. In this manner, MAGNET also incorporates the technology development scenarios. Therefore, the learning rate of advanced biomass conversion technologies to biofuels ad biochemicals is implemented exogenously in MAGNET as cost-efficiency improvements over time. For other biomass conversion technologies such as bioelectricity and other energy technologies such as wind turbines or coal-fired power plants the learning rates are endogenously determined in MAGNET and therefore remain unaffected by the technology development scenarios (see section 6.2.2.1). CCS and BECCS technologies are not modelled explicitly in MAGNET.

Openness of market

The influence of biomass sourcing on technology deployment is assessed using two scenario variants, namely a regional (*Reg*) and a global (*Glob*) trade scenario. The *Reg* scenario assumes that the EU, and in extension the Netherlands, support bioeconomy developments only if EU resources are used (for example to ensure sustainability of supply, to reduce dependency on non-EU countries and to stimulate rural development and employment in the region). This scenario is also plausible if domestic biomass demand from exporting regions such as the USA increases or if the EU applies strict sustainability criteria. It excludes largely traded biomass resources such as primary forestry biomass. In the *Reg* scenario only EU biomass is taken into account for the potential supply. The *Glob* scenario assumes that no trade barriers for biomass are imposed as development and standardisation of sustainability criteria guarantee the sustainable origins of biomass. Furthermore, another precondition is that logistics infrastructure and supply of biomass take place to achieve low biomass supply costs and make biomass a tradable commodity.

In MAGNET trade openness is implemented directly by allowing or disallowing biomass trade between the EU and the rest of the world. Biomass cost-supply curves within the EU are based on Elbersen et al. (2015) and for other regions are based on cost-structures of IMAGE-TIMER (Daioglou et al., 2016) (section 6.2.3). MARKAL-NL-UU uses exogenously determined cost-supply curves from Elbersen et al. (2015). The *Glob* variant as applied in MARKAL-NL-UU assumes ad hoc a maximum supply potential of traded biomass from global markets available to the Netherlands (450 PJ; Chapter 4).

A *NoBiobased* scenario is used as a counterfactual within the MAGNET. In the NoBiobased scenario new bioeconomy developments in the transport, chemical and energy sectors are reduced to low levels to benchmark the macro-economic contribution of new bioeconomy developments in the other scenarios. In this scenario the renewable energy policies are abolished.

The scenario variables and scenario names used in this study are presented in Table 6.1.

Table 6.1 Scenario variables and names used in MAGNET and MARKAL-NL-UU

Scenario variable	Technology development	Openness of market
RegLowTech	Low technology development	EU biomass supply
RegHighTech	High technology development	EU biomass supply
GlobLowTech	Low technology development	Global biomass supply and trade
GlobHighTech	High technology development	Global biomass supply and trade
NoBiobased[a]	No technological development	No new bioeconomy trade

[a]Applicable only in MAGNET.

6.2.2 MAGNET extension

6.2.2.1 New bioeconomy sectors and technology selection in MAGNET

Biomass supply
Agricultural and forestry residues are frequently addressed as key biomass feedstocks for energy and non-energy uses (i.e. the feedstock used as raw material that is not used for fuel purposes or transformed to fuels) (Daioglou et al., 2015; Rose et al., 2014). Energy crops (plantations) on arable land may contribute most to the global total technical potential of biomass, assuming rapid efficiency improvements in agriculture (Chum et al., 2011). Solid biomass from residues and plantations may be used directly by conversion sectors or they can be densified to tradable wood pellets leading to increased cost-efficiency of the value chain due to reduced storage and logistic costs (Batidzirai et al., 2014; Hoefnagels et al., 2014b; Uslu et al., 2008). MAGNET includes eleven primary agricultural, one fishery and one forestry biomass-producing sector. In MAGNET, three new sectors are included to capture the developments on the biomass supply side, namely residues, plantations and pellets:

- the residues sector collects and transports various types of forest harvest residues (forest management, logging, wood processing industry) and agricultural residues (harvesting and processing of agricultural crops). Residues can be used directly by conversion technologies or be supplied to the pellets sector;
- the plantations sector produces dedicated woody or grassy crops. Biomass from plantations can be used directly by conversion technologies or be supplied to the pellets sector;
- the pellets sector may use biomass from the residues or plantation sector.

An overview of the biomass supply sectors linked with downstream economic activities is provided in Figure 6.2. A detailed overview of the biomass supply sectors and their aggregation in MAGNET can be found in section 6.6 (Table 6.5). As shown in Chapter 5, a higher disaggregation of biomass feedstocks enables assessing improvements in agricultural sectors (e.g. by distinguishing woody from grassy energy crops). The combination with improvements in pelletisation and logistics may lead to up to 30% lower cost of biomass supply and thus increasing the cost-effective use of biomass in the regional economy. Nonetheless, in Chapter 5 biomass cost-supply curves are estimated exogenously. In MAGNET, price developments are calculated endogenously and therefore, low-cost biomass supply scenarios require assessments of additional improvements in the agricultural sector (e.g. yield improvements), which are not captured by the scenarios of the present study.

Conversion technologies per sector

Fuels
2nd generation technologies have the potential to produce biofuels at lower levelised costs than fossil fuels and lead to higher avoided GHG emissions compared to 1st generation biofuels by 2030 (Chum et al., 2011; Gerssen-Gondelach et al., 2014). Preconditions for

diffusion of lignocellulosic biofuels are the availability of low-cost biomass feedstocks and commercialisation of 2nd generation technologies. In replacing fossil fuels, biochemical, thermochemical and other thermal or catalytic routes (e.g. pyrolysis) may supply lignocellulosic biofuels. Based on MARKAL-NL-UU outcomes (section 6.6.2.3, Figure 6.13), MAGNET is extended to include two production technologies for lignocellulosic biofuels:

- biochemical, which convert lignocellulosic biomass to ethanol;
- thermochemical, which involve gasification of solid biomass to syngas and subsequent synthesis to Fischer-Tropsch (FT) fuels.

Other promising biofuel pathways include hydrotreatment of vegetable oils and fats, pyrolysis or other thermochemical routes that produce methanol and hydrogen. These are not included in the analysis as most representative routes are chosen based on MARKAL-NL-UU. The selected pathways may also supply feedstocks required by chemical conversion technologies. The thermochemical pathways may supply naphtha and the biochemical pathways lignocellulosic ethanol and sugar for further conversion. An overview of biomass conversion to lignocellulosic biofuels is provided in Figure 6.2. A detailed overview of the biofuel supply sectors and their aggregation in MAGNET can be found in section 6.6 (Table 6.5).

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Electricity
In order to demonstrate substitution effects in electricity supply, the electricity sector in MAGNET needs to be split to several different renewable-based and fossil-based supplying

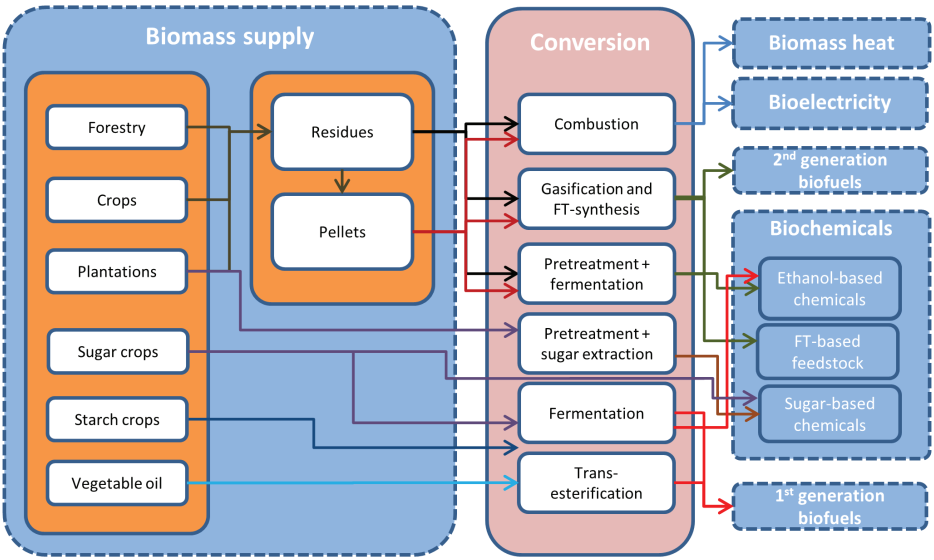

Figure 6.2 Overview of bio-based sectors and linkages in MAGNET

sectors in detail. More renewable options are available at competitive costs and at the same time disaggregation of fossil sectors implies a better representation of available options for GHG emissions mitigation strategies. Therefore, we split the electricity-producing sector in MAGNET into six source sectors for electricity production: from biomass (which includes co-firing of wood pellets in coal-based power plants), wind and solar, hydro and geothermal, coal, natural gas, and nuclear. In addition, an electricity transport and distribution sector is included. An overview of biomass conversion to electricity is provided in Figure 6.2. A detailed overview of the electricity sector and its aggregation in MAGNET can be found in section 6.6 (Table 6.5).

Heat

The contribution of biomass heat to renewable energy demand is expected to be significant and in the EU and globally. For EU27, based on the National Renewable Energy Action Plans biomass heat represents approximately 65% of the final bioenergy demand, and may vary between 62-72% depending on scenario conditions (Stralen et al., 2013). Saygin et al. (2014) mention that 13-14 EJ of biomass can be economically deployed at a global level to supply industrial steam by 2030, especially if low-cost biomass residues are used. In MAGNET, heat is not modelled as a separate sector but implicitly as a direct substitute for natural gas (Kretschmer and Peterson, 2010).

Chemicals

In the GTAP database the chemical sector is represented at an aggregate level by a single category that includes basic chemicals and other chemical products, and uses only petroleum products as input (Narayanan et al., 2013). However, the flows of the petrochemical industry are more complex. Different crude oil or gas products are used and are converted first to basic chemicals that are further converted to several intermediate chemicals before finally being synthesised and processed to consumer goods such as plastics. Most of the chemical industry's output is consumed for polymers (Harmsen et al., 2014), which is also the category we focus on this study. Along the lines of strategies (Bos and Sanders, 2013; Corbey Committee, 2012; Croezen and Bergsma, 2012) that aim at providing renewable alternatives to the chemical industry those discussed the most are drop-in chemicals and new products. Drop-in chemicals could either be alternative feedstocks for steam crackers or directly chemicals that can be processed further downstream to final products. Drop-in chemicals can use existing infrastructure and compete in established markets as they share exactly the same properties with their fossil counterparts. New biochemicals require new infrastructure and may compete in the same or new markets with conventional chemicals (e.g. polylactic acid (PLA)). By 2100, biomass may provide up to 40% of the primary energy required for non-energy uses by supplying primarily basic chemicals (namely olefins and aromatics), refinery products (namely aromatics) and ammonia (Daioglou et al., 2014). Gerssen-Gondelach et al. (2014) demonstrate that fermentation-based chemicals such as PLA are already cost-competitive and that their competitiveness may increase even more compared to their fossil reference products by 2030. In order to capture the emerging sector of biochemicals in MAGNET we account for key strategies and chemicals by splitting the sector and selecting representative pathways based on MARKAL-NL-UU (Chapter 4, section 6.6.2.3, Figure 6.14):

- ethanol to chemicals, which uses 1st generation or lignocellulosic ethanol to produce bioethylene (ethanol-based chemicals; Figure 6.3). This sector uses already existing infrastructure and competes in the same markets with fossil-based alternatives. As final product of this route polyethylene (PE) is selected, which is the largest globally produced polymer (IEA, 2013). Ethanol can be used to produce other drop-in chemicals (e.g. propylene, butadiene) and a variety of other chemicals and could be a potential platform for the future (Posada et al., 2013);
- direct sugar to chemicals, which can use 1st generation or lignocellulosic fermentable sugars (white biotechnology, Patel et al. (2006); sugar-based chemicals; Figure 6.3). Although 1st generation and lignocellulosic ethanol also use fermentable sugars prior to converting it to ethylene, this route produces directly through fermentation products that can replace plastics. Lactic acid and its polymerisation to PLA is chosen as a representative product for using sugar as a platform for biochemicals.

In addition, to assess the option of supplying alternative feedstocks to the industry, we include FT-naphtha as a thermochemical-based feedstock (mixed bio-based and fossil-based chemicals; Figure 6.3). This choice is not supported by MARKAL-NL-UU outcomes, as this was not shown as a prospective option for the chemical sector. However, accounting for it in the structure of MAGNET is essential as other options such as hydrogenated animal or vegetable oils or ethane and propane from ethanol may pick up in the future. Furthermore, the HighTech scenario results of MARKAL-NL-UU show that biomass gasification technologies are promising for the road transport sector (section 6.6.2.3, Figure 6.13).

The chemical sector is split in three sectors, namely various conventional fossil-based chemicals, mixed fossil-based and bio-based chemicals, and biochemicals as shown in Figure 6.3. The petroleum (Petro, in Figure 6.3), conventional sugar and 1st generation ethanol existed in MAGNET prior to this study. The chemical routes and technologies described above

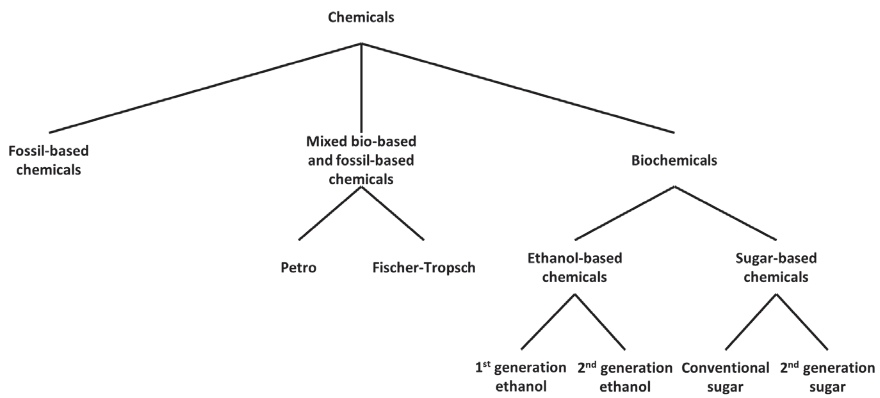

Figure 6.3 Interactions between new fossil-based and bio-based chemical products

are included as four additional new sectors, namely lignocellulosic sugar, ethanol-based chemicals, sugar-based chemicals and a mixed bio-based and fossil-based chemicals sector, next to the remaining fossil-based chemical sector. An overview of biomass conversion to biochemicals is provided in Figure 6.2. The MAGNET chemical sector and data aggregation is presented in section 6.6 (Table 6.5).

6.2.3 Input data

Fossil fuel and CO_2 price developments to 2030 are exogenously determined and fixed for both models. These are common between MAGNET and MARKAL-NL-UU based on the International Energy Agency's World Energy Outlook 2014 (IEA-WEO 2014; Table 6.2), thereby providing consistency in key drivers of both models (OECD/IEA, 2014). Other exogenous data inputs relate only with the macro-economic model. These are GDP and population growth, which are based on the Shared Socioeconomic Pathways 2 scenario of the Intergovernmental Panel on Climate Change (O'Neill et al., 2015). In the calibration stage of MAGNET, region-specific and sector-specific technological change is calibrated by forcing the model to meet the exogenous GDP targets given the exogenous estimates of factor endowments (skilled labour, unskilled labour, capital and natural resources) and population. This level of technological change is translated to the sectoral level using a sector-specific growth ratio of total factor productivity based on Central Planning Bureau figures (CPB, 2003). The technological change, in turn, in the baseline scenario and simulation experiments is exogenous, while GDP becomes endogenous and calibrated values for technological changes are used. Furthermore, as explained in section 6.2.1.1, the two models use different biomass cost-supply curves. These are determined exogenously for MARKAL-NL-UU (Chapter 4) and endogenously for MAGNET. Both models use the same database for biomass supply potential in Europe based on Elbersen et al. (2015). Finally, for MAGNET, the base years, in terms of value added and production values, for different countries vary according to available statistical economic and technical data.

Table 6.2 Exogenous data inputs for MAGNET and MARKAL-NL-UU

		Base year (2007)	2010	2020	2030	Reference
Oil	€/GJ (€/bbl)	9.1 (52)	9.4 (54)	12.6 (73)	13.9 (79.6)	OECD/IEA (2014)
Natural gas	€/GJ (€/MBtu)	5.2 (5)	5.0 (5.1)	6.9 (7.3)	7.4 (8.0)	OECD/IEA (2014)
Coal	€/GJ (€/t)	2.5 (55)	3.1 (68.8)	2.9 (65.7)	3.1 (70.1)	OECD/IEA (2014)
CO_2	€/t	0	4.4	14.6	24.1	OECD/IEA (2014)
Biomass (wood pellets)	€/GJ	6.8	*estimated endogenously in MAGNET*			Chapter 4

The database is refined with new bioeconomy sectors as described in section 6.2.2. Data on production volumes, conversion efficiencies, cost-structures, trade and transport costs are derived from various data sources (Table 6.6 in section 6.6.1). Input data for the Netherlands in 2007 are shown in Table 6.3. The production of 1st generation biofuels, which are produced by sugar, starch and oil crops, is split from the "chemicals, rubbers and plastics" industry in the GTAP database. The cost-structure of bioelectricity generation is similar to conventional electricity generation technologies (e.g. co-firing with coal and combined heat and power). It is assumed that all bioelectricity production is purchased entirely by the electricity sector (i.e. non-traded). The price of lignocellulosic biomass (i.e. plantations, residues and pellets, including transport costs), in 2007 is 6.8 €/GJ, which is the price of imported pellets and is common between MARKAL-NL-UU and MAGNET.

The production volume of the biomass supply sectors is derived from the production of bio-electricity, biochemicals and lignocellulosic biofuels. The bioelectricity sector is introduced and is split from the original GTAP electricity sector. Its cost-structure is based on co-firing biomass with coal. The production volume of lignocellulosic biofuels and biochemicals in the base year is partially assumed, since in reality production volumes are zero or extremely low today. Co-products of biofuels and biochemicals are included as credits that reduce the production costs. The fossil-based and bio-based chemicals sector is the largest sector, since this sector includes the production of plastics from fossil feedstocks.

The production of 1st generation and lignocellulosic biofuels, bioelectricity and biochemicals is not cost-competitive with conventional production technologies in 2007 and is therefore subsidised. The difference in production costs between conventional and bio-based sectors is covered by an input subsidy on biomass, which is paid via an output tax of the bio-based sectors.

Crucial for the cost-competitiveness of the bio-based sectors and for their macro-economic impacts are the development of the efficiency and costs of bio-based conversion technologies. This especially concerns the production of biochemicals and lignocellulosic biofuels. The scenario-specific learning rates and technical developments for the new technologies are based on MARKAL-NL-UU and shown in Table 6.4. The latter information is processed to input saving technical progress in MAGNET to implement the HighTech and LowTech scenario variants. The assumed rate of input saving technical change is based on the cost structures presented in Table 6.4 for 2007 and MARKAL-NL-UU results for 2030. This ensures that the production costs in MAGNET are the same as in MARKAL-NL-UU in 2030. The values show the input per unit output in 2030 compared to input per unit output in 2007 at constant prices. Technical change in the bioelectricity and the 1st generation biofuels sector is partly endogenous in MAGNET and identical between scenarios. The trade openness of biomass markets in MAGNET is introduced by applying prohibitive tariffs between EU and the rest of the world on biomass.

Table 6.3 Production volume, prices and cost-structures of the new bio-based sectors in the Netherlands

	Ethanol (1st generation; grains)[c]	Ethanol (1st generation; sugar)[c]	Biodiesel	Plantations	Residues	Pellets	Fischer-Tropsch fuels[d]	Ethanol (2nd generation)[d]	Lignocellulosic sugar	Fossil and bio-based chemicals[e]	Sugar-based chemicals[f]	Ethanol-based chemicals[g]	Bioelectricity
Production	M€	M€	M€	M€	M€	M€	M€	M€	M€	M€	M€	M€	M€
volume[a]	3	1	48	11	174	9	3	3	0	2	1	1	245
Production cost shares of endowments and intermediate deliveries (percentage in total costs) ↓ [%]													
Land	0	0	0	13	0	0	0	0	0	0	0	0	0
Labour	11	11	2	31	8	20	5	8	8	8	11	7	14
Capital	6	8	9	38	45	59	26	10	16	10	31	16	14
Wheat and grains	56	0	0	0	0	0	0	0	0	0	0	0	0
Sugar crops	0	57	0	0	0	0	0	0	0	0	0	0	0
Vegetable oils	0	0	76	0	0	0	0	0	0	0	0	0	0
Plantations	0	0	0	0	0	0	4	4	2	0	0	0	2
Residues	0	0	0	0	20	15	42	47	45	0	0	0	45
Pellets	0	0	0	0	0	0	11	12	8	0	0	0	8
Ethanol (1st generation)	0	0	0	0	0	0	0	0	0	0	0	48	0
Ethanol (2nd generation)	0	0	0	0	0	0	0	0	0	0	0	9	0
Sugar	0	0	0	0	0	0	0	0	0	0	23	0	0
Lignocellulosic sugar	0	0	0	0	0	0	0	0	0	0	4	0	0
Transport	1	2	4	18	26	0	4	6	0	0	0	0	0
O&M[b]	26	22	9	0	0	6	9	14	21	81	31	20	28
Total	100	100	100	100	100	100	100	100	100	100	100	100	100

[a]Production volumes for non-existing sectors in 2007 are assumed as follows: Fischer-Tropsch and lignocellulosic ethanol are same as 1st generation ethanol from grains; for sugar-based chemicals and ethanol- based chemicals 1 M€ production volume is assumed; production of lignocellulosic sugar, pellets, residues and plantations are derived from production of biofuels, biochemicals and bioelectricity. [b]Operation and Maintenance. [c]Price of ethanol from grains and sugar crops is based on the cost-price of 1st generation ethanol in MARKAL-NL-UU in 2010. [d]Price of Fischer-Tropsch fuel and 2nd generation ethanol is the cost-price of 2nd generation ethanol in MARKAL-NL-UU in 2010. [e]Price is based on weighted cost-price of conventional (fossil-based) polyethylene and bio-based polyethylene assuming a 0.1% share of bio-based polyethylene in total polyethylene production volume. Cost-price of bio-based polyethylene in 2007 is based on the cost-price in MARKAL-NL-UU in 2020. [f]Price is based on weighted cost-price of polylactic acid production from conventional sugar and polylactic acid production from lignocellulosic sugar assuming a 10% share of the production volume coming from 2nd generation ethanol polyethylene. [g]Price is based on weighted cost-price of polyethylene production from ethanol from grains and sugar crops and from lignocellulosic ethanol assuming a 10% share of the production volume coming from 2nd generation ethanol polyethylene.

Table 6.4 Input saving technical change in the biomass conversion sectors in the HighTech and LowTech scenarios (input per unit output in 2030 relative to input per unit output in 2007 at constant prices)

HighTech scenario						
Endowments and intermediate deliveries ↓	Fischer-Tropsch fuel	Ethanol (2nd generation)	Lignocellulosic sugar	Fossil and bio-based chemicals	Sugar-based chemicals	Ethanol-based chemicals
Land	n/a	n/a	n/a	n/a	n/a	n/a
Labour	0.29	0.15	0.47	1.00	0.63	0.83
Capital	0.52	0.36	0.27	1.00	0.70	0.73
Plantations	0.41	0.43	0.56	0.56	n/a	n/a
Residues	0.41	0.43	0.56	0.56	n/a	n/a
Pellets	0.41	0.43	0.56	0.56	n/a	n/a
Ethanol (1st generation)	n/a	n/a	n/a	n/a	n/a	0.76
Ethanol (2nd generation)	n/a	n/a	n/a	n/a	n/a	0.76
Sugar	n/a	n/a	n/a	n/a	0.66	n/a
Lignocellulosic sugar	n/a	n/a	n/a	n/a	0.66	n/a
Transport	0.29	0.15	n/a	n/a	n/a	n/a
Other O&M	0.29	0.15	0.47	1.00	0.63	0.83
LowTech scenario						
Endowments and intermediate deliveries ↓	Fischer-Tropsch fuel	Ethanol (2nd generation)	Lignocellulosic sugar	Fossil and bio-based chemicals	Sugar-based chemicals	Ethanol-based chemicals
Land	n/a	n/a	n/a	n/a	n/a	n/a
Labour	1.00	0.70	0.69	0.69	0.79	0.94
Capital	1.00	0.58	0.70	0.70	0.70	1.00
Plantations	1.00	0.67	0.69	1.00	n/a	n/a
Residues	1.00	0.67	0.69	1.00	n/a	n/a
Pellets	1.00	0.67	0.69	1.00	n/a	n/a
Ethanol (1stgeneration)	n/a	n/a	n/a	n/a	n/a	0.85
Ethanol (2nd generation)	n/a	n/a	n/a	n/a	n/a	0.85
Sugar	n/a	n/a	n/a	n/a	0.66	n/a
Lignocellulosic sugar	n/a	n/a	n/a	n/a	0.66	n/a
Transport	1.00	0.70	n/a	n/a	n/a	n/a
Other O&M	1.00	0.70	0.69	1.00	0.79	0.94

6.3 RESULTS

6.3.1 Comparison between the bottom-up and the top-down results

In this section, model outputs of MAGNET and MARKAL-NL-UU for the road transport, electricity and chemical sector in 2030 are compared against each other. Despite the methodological differences and the different modelling paradigms and techniques of the two models their outcomes are fairly consistent. The comparison is made on the basis of the monetary flows of MAGNET and the physical flows of MARKAL-NL-UU. For that purpose outcomes are indexed in reference to the RegLowTech scenario results in 2030.

6.3.1.1 End-use of biomass

Figure 6.4 shows that the shares of fossil-based and bio-based fuels in road transport are consistent between MAGNET and MARKAL-NL-UU. The blending target in the LowTech scenarios is the key driver of the biofuel output in both models. This implies that mandates of 10% are met 7% by 1st generation and 1.5% by lignocellulosic biofuels. The latter are counted double in meeting the target but this has no (macro) economic implications. Biofuel shares in both models are considerably higher in HighTech scenarios as especially lignocellulosic biofuels become competitive. In GlobHighTech, the biofuel share is 13% in MAGNET and 12% in MARKAL-NL-UU and is mainly lignocellulosic biofuels (90-100%).

Biomass use in electricity production is limited across scenarios (Figure 6.5). In MAGNET, biomass use is slightly higher and constant across scenarios while for MARKAL biomass use is higher in HighTech scenarios due to co-produced electricity by biorefineries. The share of fossil-based power production is comparable between scenarios and models. Coal-based power contributes approximately 20% and gas-based power contributes approximately 45% to total power production. Consequently, the renewable energy share is also consistent between the models and equal to about 35%. In both models, wind is the key renewable energy source. In MAGNET, its share in electricity production is 31% and in MARKAL-NL-UU 25.5% in 2030.

Figure 6.6 shows the biomass and fossil feedstock use in the chemical sector in the Netherlands in 2030. Both models estimate low to moderate shares of biochemicals over total chemicals. Based on MAGNET ethanol-based and sugar-based chemicals contribute 1% in LowTech and almost 4% in HighTech scenarios. Both models show a higher increase in advanced biochemicals production in HighTech compared to LowTech scenarios. The difference in shares can be explained by the broader statistical definition of the chemical sector used by the GTAP classification in MAGNET. Technological change is key for biochemical developments. The expected effects of technological change lead to strong increase in biochemicals production, especially in the HighTech scenarios. In MAGNET the output is about 2 bn€/yr in the HighTech and 0.7 bn€/yr in the LowTech scenarios. The biochemical output in MARKAL is almost 1.8 Mt in GlobHighTech. This is consistent, assuming a plausible price of 1,000 €/t.

Key products are ethanol-based chemicals from 2nd generation ethanol (0.6 bn€/yr in the LowTech and 1.7 bn€/yr in the HighTech scenarios), sugar-based chemicals from ligno-

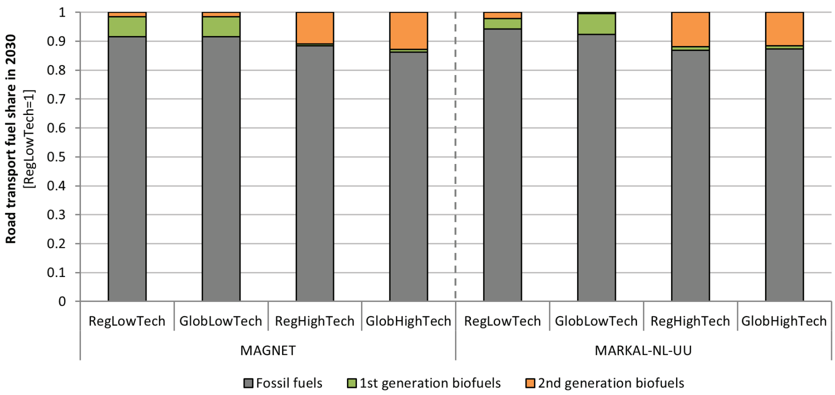

Figure 6.4 Comparison of road transport fuel share between MAGNET and MARKAL-NL-UU in the Netherlands in 2030 (index: transport fuel share RegLowTech=1)

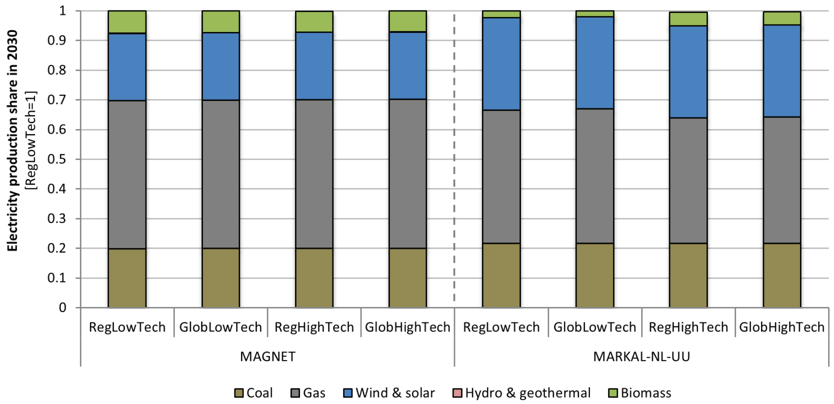

Figure 6.5 Comparison of electricity production by source in the Netherlands in 2030 between MAGNET and MARKAL-NL-UU (index: total electricity production RegLowTech=1)

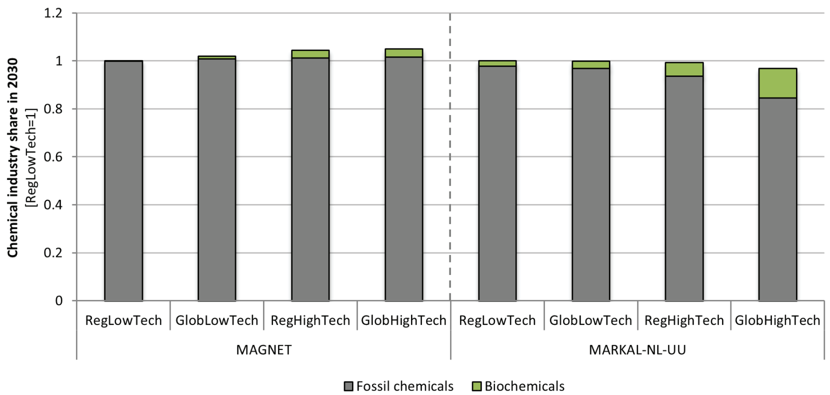

Figure 6.6 Comparison of fossil chemical and biochemical share in the chemical sector in the Netherlands in 2030 between MAGNET and MARKAL-NL-UU (index: total chemicals RegLowTech=1)

cellulosic sugar (0.2 bn€/yr in the HighTech scenarios) and from conventional sugar (0.06 bn€/yr in the HighTech scenarios). Openness of market plays a modest role, as biomass for 2nd generation technologies is available within Europe at competitive prices. Openness of market generates additional biochemical output of 0.03 bn€/yr and 0.08 bn€/yr in the LowTech and HighTech scenarios, respectively. Without technological change the production and use of biochemicals is limited. A methodological difference is that demand in MARKAL-NL-UU is fixed, and both higher technological change and a more diverse technology portfolio lead to lower use of inputs. In MAGNET, demand is endogenous and technological change leads to an increase in the chemical sector and demand for inputs. This explains why the total chemical industry output in MAGNET outcomes increases and in MARKAL-NL-UU outcomes decreases.

6.3.2 Biomass consumption

In 2030, biomass consumption is up to a factor 4 higher relative to no bioeconomy developments ("NoBiobased" scenario in Figure 6.7). Consumption is comparable across the trade openness scenarios, indicating that there will be sufficient biomass available within the Netherlands and Europe at competitive prices to supply domestic bioeconomy developments under the assumed renewable energy targets by 2030. More stringent targets, may lead to imports from extra-EU resources. However, these are not assessed in the present study. Domestic and imported residues play a key role across both technology development scenarios (about 65-75% and slightly above 90% of total biomass consumption in LowTech and HighTech, respectively). In LowTech, 1st generation feedstocks (i.e. sugar, oilseeds and cereals) are primarily imported and supply about 20-30% of the total consumed biomass (from the EU in the Reg and mostly sugar from North and South America in the Glob scenario variants). As in HighTech scenarios lignocellulosic biomass conversion technologies become cost competitive, 1st generation feedstocks make up only 2-4% of total biomass consumption as the supply of domestic and imported residues is higher by about 20% compared to LowTech.

6.3.3 Macro-economic impact

Large-scale deployment of biomass could have a positive impact on the value added of the Dutch economy in the medium term (2030). Across all scenarios only RegLowTech has a negative GDP effect of -0.2 bn€/yr in 2030 (Figure 6.8). Open markets and investments in technology development lead to a positive GDP effect of 0.8 bn€/yr in 2030 (GlobHighTech). High technology development and global markets add up to 1 bn€/yr to GDP from 2030 onwards.

In 2015-2020, the main effect that drives the outcomes is the compliance with the EU RED targets, which leads to negative GDP effects in all scenarios compared to a NoBiobased scenario in which the EU RED targets are not achieved, as bioenergy technologies are not competitive with their fossil substitutes. The negative effect is stronger in the Reg scenarios (-0.8 bn€/yr) than in the Glob scenarios (-0.2 bn€/yr) as the EU (and the Netherlands) cannot import (relatively cheap) biofuels from South and North America. The positive effect of technology advances becomes visible in the 2020-2030 period and is larger for HighTech

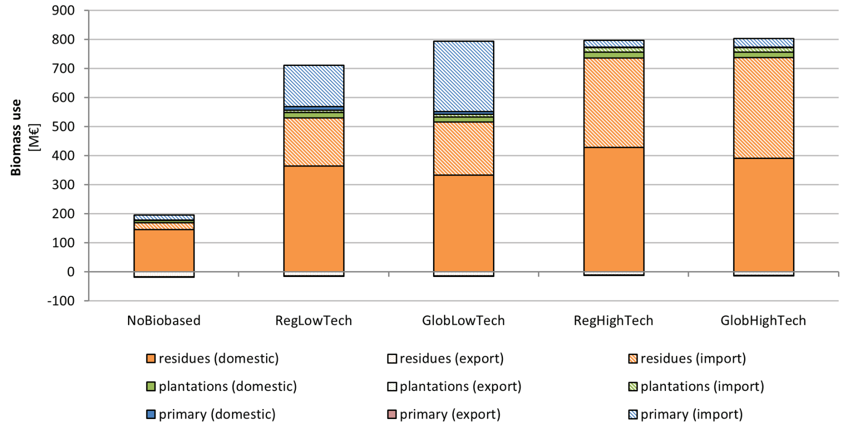

Figure 6.7 Biomass consumption per feedstock for bioenergy and biochemicals in the Netherlands in 2030

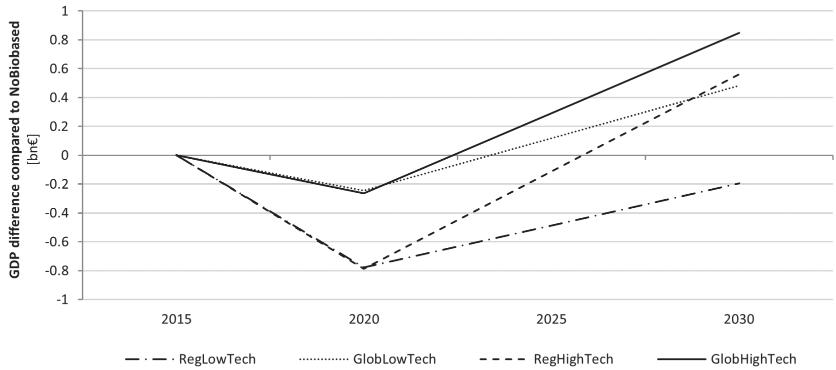

Figure 6.8 GDP impact of the different scenarios relative to the NoBiobased scenario in the Netherlands in 2015-2030 (absolute difference, bn€)

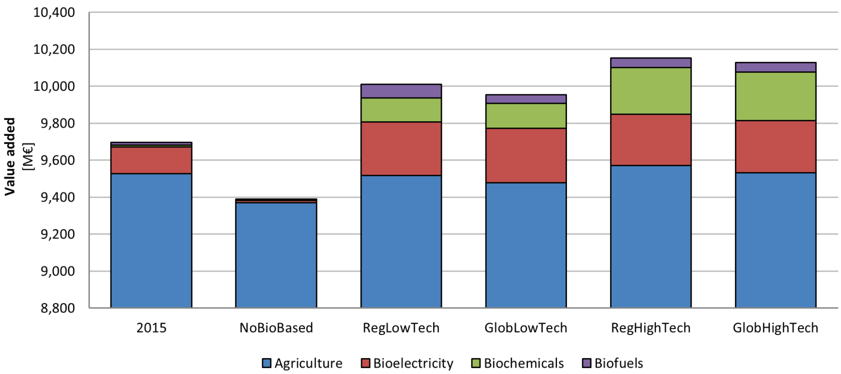

Figure 6.9 Value added in different scenarios in the Netherlands in 2030 compared to reference (No-Biobased) and 2015 (excluding food processing, forestry and pulp and papers sectors)

(1 bn€/yr) than LowTech (0.6 bn€/yr) scenarios (i.e. it is increase in GDP between 2020 and 2030).

Figure 6.9 shows the value added of key bioeconomy sectors in the Netherlands in 2015 and for the scenarios of this study in 2030. The increase in value added of the selected bioeconomy sectors relative to the NoBiobased scenario is higher in HighTech than in LowTech scenarios by about 0.74-0.76 bn€/yr and 0.56-62 bn€/yr, respectively. The growth of the bioeconomy has a positive impact on value added of all its sectors. Within the bioeconomy, agriculture is the key sector. In the NoBiobased scenario the value added slightly decreases (by 0.160 bn€/yr) between 2015 and 2030. In all other four scenarios the value added of agriculture in the Netherlands increases. The value added in the biochemical sector due to new biochemicals increases by 0.13 bn€/yr in LowTech scenarios and by 0.25 bn€/yr in HighTech scenarios (compared to a NoBiobased scenario in 2030). Value added of biofuels is modest (0.05 bn€/yr) and similar across the scenarios. In RegLowTech it is slightly higher (0.075 bn€/yr). Bioelectricity use is largely driven by policies and its value added is fairly constant across the scenarios. Relative to the NoBiobased scenario the value added of bioenergy is about 0.28 bn€/yr higher in 2030.

In the NoBiobased scenario the employment in agriculture, bioenergy and biochemical sectors is lower in 2030 compared to 2015 as a result of employment decline in agriculture, which is in compliance with the long-term trend (Figure 6.10). It is important to realise that we assume full employment in the long run and jobs created in the bioeconomy sectors are drawn from other sectors. Increased employment induced by bioeconomy developments as indicated by the four scenarios in 2030 compared to the NoBiobased scenario in 2030 only partially mitigates the decline in employment. More specifically, without bioeconomy developments employment decreases by 6.5% in 2030 compared to 2015 ("NoBiobased" scenario in Figure 6.10). With bioeconomy developments, employment decreases by 5.9-6.1% in 2030 compared to 2015 (all other scenario variants in Figure 6.10). The highest impacts in employment come from the production of biochemicals in the case of the HighTech scenarios and to a smaller extent in the GlobLowTech scenario. Employment in the biofuel sector is linked to 1st generation and lignocellulosic biofuels in LowTech and HighTech scenarios, respectively.

Biomass imports and replacement of fossil fuels have several impacts on trade balance (Figure 6.11). Relative to the NoBiobased scenario, the trade surplus of the Netherlands decreases by about 1.5 bn€/yr. In the short term (2020), the total trade balance in the Netherlands is projected to deteriorate relative to a NoBiobased baseline (not shown in Figure 6.11). The overall negative impact is caused by the introduction of the EU RED targets, which requires substitution of fossil technologies with more costly biomass conversion technologies, especially in electricity production. The fossil and total energy trade balance improves, but this is more than offset by increased biomass imports and especially the deterioration of the trade balance of other industries and services. However, after 2020 biomass conversion technologies become more cost-efficient and Dutch export of lignocellulosic biofuels and especially biochemicals increases. Technical change has a positive impact on the overall trade balance and this reduces the decline of the overall trade balance due to

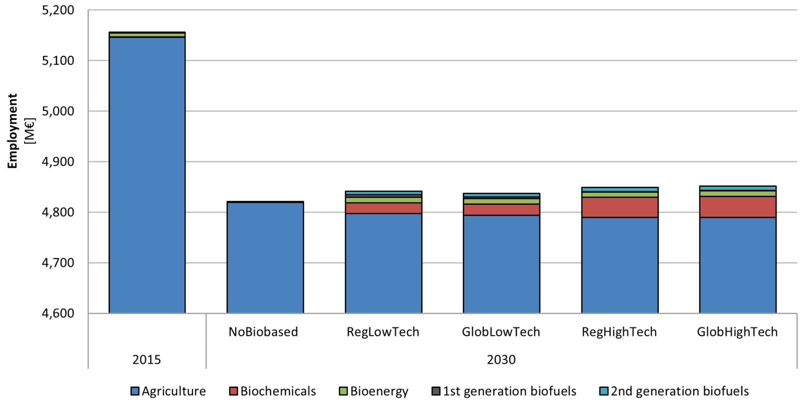

Figure 6.10 Employment in agriculture and new bioeconomy sectors across different scenarios in the Netherlands in 2030 compared to NoBiobased in 2030 and the initial situation 2015

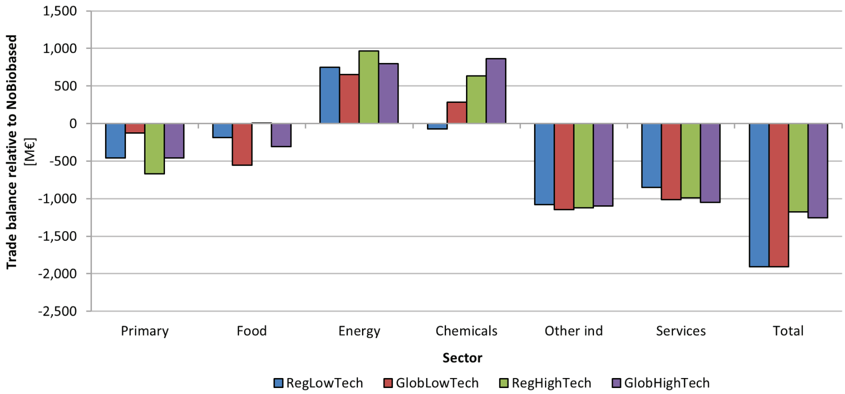

Figure 6.11 Trade balance difference relative to NoBiobased scenario in 2030

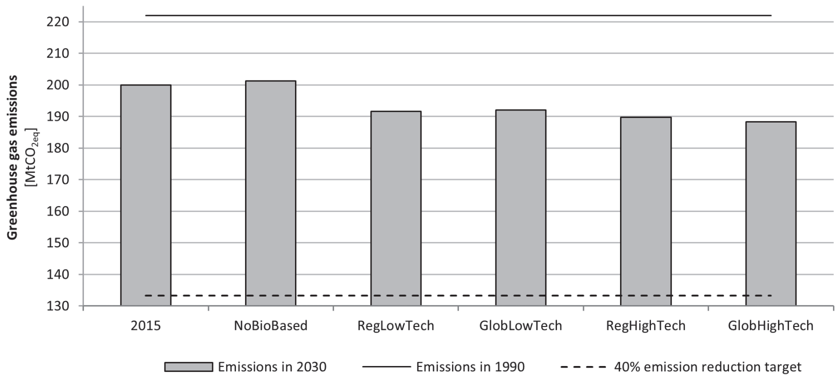

Figure 6.12 Total greenhouse gas emissions in the Netherlands in 2030

the EU RED. With technology change, the negative impact becomes lower. The reduction of trade surplus is about 1.2 bn€/yr in the HighTech scenarios and about 2 bn€/yr in the LowTech scenarios.

The energy trade balance improves in the LowTech scenarios, as fossil energy is substituted by 1st generation biofuels, which are partly produced domestically. The energy trade balances improves slightly more in the HighTech scenarios as the production of bio-based and conventional fuels and chemicals increases partly due increased export demand. The exports of bio-based products substitute fossil energy use and reduce GHG emissions in other countries.

6.3.4 Greenhouse gas emissions

Figure 6.12 shows the total GHG emissions across all scenarios in the Netherlands in 2030 compared to 2015. The introduction of the renewable energy policies and technological change reduce emissions by 4-6%. Relative to 2015, the NoBiobased scenario leads to slightly higher emissions as biomass use in fuels and electricity is suppressed. In the LowTech scenarios, emissions decline by 8 Mt (4% compared to 2015) mainly due to the introduction of the renewable energy directives for biofuels and electricity. In HighTech scenarios, technological change induces the substitution of fossil-based by bio-based technologies and this contributes additionally 1% and 2% in the regional and global scenario, respectively. Figure 6.12 also shows that while emissions are reduced by about 15% compared to 1990, there is still significant emission reduction of about 50 MtCO_{2eq} required to achieve the 40% emission reduction target set by the EU in 2030.

6.4 DISCUSSION

6.4.1 Modelling framework

The framework and the extension of bioeconomy sectors in MAGNET presented in this article, stand as an improvement of previous efforts that aimed at addressing future impacts of the bioeconomy using CGE models. Firstly, the incorporation of policy targets (EU RED) and the explicit representation of advanced biomass conversion technologies next to other renewable conversion technologies, allowed us to assess the sectoral deployment of biomass and other renewable energy as a model output, in contrast to fixed and scenario-dependent deployment levels as assumed in Hoefnagels et al. (2013). Secondly, instead of relying on ad-hoc technology comparison, the cost-competitiveness of technologies within the energy system and the chemical industry was taken into account prior to MAGNET's extension. The selection of technologies, their cost-structures and cost-efficiency improvements over time were based on key technologies projected to be deployed within the energy system of the Netherlands by 2030 as a result of a cost-optimal technology portfolio of biomass conversion technologies next to other renewable resources and fossil fuel conversion technologies. This enabled us to incorporate biochemicals in MAGNET by splitting the chemical industry sector of the GTAP database according to the method applied in the chemical industry module of MARKAL-NL-UU (Chapter 4). This is a significant improvement of biochemicals' representation in the CGE family of models, which to date was scarce. MAG-

NET can now address the macro-economic impact of biochemicals deployment in line with strategies that aim at providing renewable alternatives to the chemical industry. Furthermore, similar to other economic models, we included in MAGNET 2nd generation biofuels and lignocellulosic feedstocks. Additionally, we included the option to densify woody biomass from residues and energy crops to tradeable pellets. MAGNET is one of the few CGE models that can take, next to liquid biofuels, also effects of global solid biomass trade into account. Lastly, this framework harmonises key elements of two very distinct models enabling their soft-linking. For example, MAGNET can also provide data based on macro-economic interactions such as developments of demand and prices to MARKAL-NL-UU (e.g. as in van den Broek et al. (2011) or van Meijl et al. (2016)).

The models' outcomes are consistent based on the strong model collaboration, harmonisation of input parameters and improved sectoral representation in MAGNET from insights received by MARKAL-NL-UU. Both models show that high technology development and open trade stimulate large-scale biomass deployment in the economy. They both indicate that biomass conversion to modern energy and chemicals is larger when compared to low technology development counterparts. Furthermore, both models show that technology development and trade scenarios affect the growth of biochemicals and influence the type of biofuel supply (1st generation in LowTech as opposed to 2nd generation in HighTech scenarios). Bioelectricity production is fairly constant as wind turbines are the main supplier of renewable electricity.

Linking monetary flows of MAGNET with physical indicators of MARKAL-NL-UU can enable comparison of the models' outputs also on an absolute basis as opposed to the relative basis that was performed in this study. Biomass price levels between the models are different. Global and domestic biomass prices in MAGNET are estimated endogenously based on equilibrium of supply, while MARKAL-NL-UU uses fixed and exogenously defined cost-supply curves. To harmonise further input data between the models we recommend to use biomass prices from MAGNET and apply them to the biomass cost-structures of MARKAL-NL-UU and iterate the model runs. Furthermore, biomass for industrial heat was shown to be an important pathway in the cost-minimisation results. This is expected to remain the largest bioenergy sector to 2030 by other studies (Stralen et al., 2013). The framework of this study could be used to improve the representation of the heat sector in MAGNET. However, increase in industrial energy efficiency and efficiency improvements in the built environment, may lead to a decline in heat demand. Including this in models requires more sophisticated data on the heat sector regarding technologies, efficiency measures and their development over time.

Finally, the framework and the models presented in this study could benefit by soft-linking the models' input and outputs. For example, in MAGNET, international trade of produced commodities (electricity, fuels, chemicals) is calculated endogenously and it affects the equilibrium prices of commodities and the regional demand, based on supply-demand elasticities, costs and market shares. On the other hand, the demand in MARKAL-NL-UU is fixed, exogenously determined and international trade in produced commodities is not included. By supplying demand projections, for example, of chemicals from MAGNET estimates to MARKAL-NL-UU and iterating the model runs, more insights could be obtained. As shown in Chapter 5, biochemical output in MARKAL-NL-UU decreases when assuming lower demand for chemicals.

6.4.2 Sensitivity

The macro-economic outcomes, similar to Hoefnagels et al. (2013), show that high technology development and open trade come with significant macro-economic benefits for the Netherlands in 2030. Nonetheless, in order to provide robust directions that can support policy making, additional analysis is required. First and foremost, as demonstrated in Chapter 5 fossil fuel prices are a key determinant of technology competitiveness (range of 183-762 PJ of biomass consumption in 2030). Similarly, variation in fossil fuel prices can affect the macro-economic developments assessed by MAGNET, as they are exogenous parameters. Lower fossil fuel prices are expected to reduce the macro-economic benefits derived from of bio-based technologies. Higher fossil fuel prices on the other hand may lead to an increase of their macro-economic benefits. A local sensitivity analysis on fossil fuel prices is therefore recommended.

Secondly, the scenarios were developed around technology development and biomass trade. However, more than 90% of value added lies in the agricultural sector (Figure 6.9). Therefore, yield developments may impact the results. Assuming a 10% increase in the Dutch crop and forestry sectors, leads to significant positive effects on GDP (0.3% compared to 0.12% in GlobHighTech) and value added, due to the increase of the market share of Dutch export-oriented agri-food industries. Nonetheless, as shown by MAGNET outcomes (Figure 6.7), only a minor part of domestic primary agricultural products are consumed by the Dutch bioenergy and biochemical sectors. An increase of biomass supply to global markets is noticed, as the global agricultural feedstock prices do not decline significantly from the increased supply of the Netherlands. A more detailed local sensitivity analysis on improvements in agriculture is therefore recommended.

Finally, production costs of bioenergy and biochemical and their development over time are a determinant of the size of bioeconomy and its impacts. Literature indicates a wide range of production costs depending on the technology type, their techno-economic parameters (e.g. size, efficiency) and location (Gerssen-Gondelach et al., 2014). The broad coverage of MAGNET, requires simplification of the technology portfolio that is included, thereby ignoring other options that may be available. For the selected technologies, the dependency of production cost on the technology size and efficiency is addressed by different input assumptions in the technology development scenarios. However, for currently non-commercially existing technologies, MAGNET applies the same technological assumptions on cost-structures and development over time for all regions of the world. These could be improved, if region-specific cost-structures are defined for the advanced biomass conversion technologies that were introduced in this study. Robustness of outcomes could be further assessed by conducting a sensitivity analysis on production costs in line with ranges found in literature.

6.4.3 Emission reduction pathways

The assumed tax on emissions and cost-competitiveness of renewable technologies reduce CO_2 emissions in the Netherlands by about 15% in the medium term compared to 1990 (Figure 6.12). This, however, is not adequate to achieve the EU wide 40% emission reduction target in 2030 compared to 1990 (EC, 2014b). For the Netherlands, this outcome is also supported by MARKAL-NL-UU results (Chapter 4 and Chapter 5). In the present study, renewable energy deployment is primarily driven by the assumed renewable energy targets. Stronger climate policy such as a CO_2 tax of about or above 100 \$/tCO_2 in 2030 (based on IEA-WEO 2014), is required to embark upon least-cost trajectories that achieve the 2 °C climate target that was agreed at the Conference of Parties in Paris in 2015 (COP21; UNFCCC (2015a)). The modelling framework presented in this article did not assess such ambitious CO_2 mitigation scenarios. To demonstrate their implications on the economy, improvements are required. Firstly, a plethora of models show that the role of CCS and BECCS is critical to achieve emission reduction in the medium and long term (Rose et al., 2014). Without CCS, mitigation costs may increase significantly (Fischedick et al., 2011). Large investments would already be required from 2030, which may also induce socio-economic implications. In addition, the contribution of CCS and BECCS in CO_2 mitigation may also influence the levels of bioenergy in modern bioeconomy sectors and of other renewable resources. Similar to many CGE models that include CCS (e.g. EPPA, ReMIND; Luderer et al. (2011), McFarland et al., (2002)), MAGNET should also incorporate in power and liquid fuel sectors CCS technologies endogenously. Secondly, ambitious emission reduction scenarios entails large biomass consumption volumes from regions also outside the Netherlands, and therefore a larger regional-scope is more relevant. MAGNET being a global multi-region model can address this, provided that region-specific cost-structures are introduced. Along these lines, the temporal scope of the assessment needs to be extended beyond 2030, as learning is limited by the time horizon of the present study, not only on the conversion side but also on the supply side (e.g. feedstock supply). Extending the time-horizon may create different dynamics driven by the mitigation efforts and could possibly demonstrate a larger deployment of modern bioenergy and biochemicals in the economy. This also entails that additional technologies may need to be included in the portfolio of the models used in this framework.

6.5 CONCLUSIONS

We presented a modelling framework, which uses a technology-explicit and technology-rich cost-minimisation linear programming energy system model (MARKAL-NL-UU) to improve the representation of bioeconomy in a global, multi-region, multi-sector CGE model (MAGNET). The framework harmonises model inputs (technology cost-structures, scenario assumptions, exogenous parameters) and is used to extend bioeconomy sectors in MAGNET by selecting key biomass conversion technologies for production of advanced biofuels and biochemicals. The models' outputs are compared against each other to complement and validate the collaboration of the two models. Finally, MAGNET is used to assess mid-term bioeconomy impacts on GDP, value added, employment and trade balance, in the Netherlands in 2030 under different scenarios of technology development and biomass trade.

The incorporation of lignocellulosic biomass supply, its conversion to tradeable pellets, and of advanced biomass conversion technologies are an important improvement of MAGNET, as they are key prerequisites for assessing the macro-economic impacts of bioeconomy developments. We found that high technology development is a no-regrets option in terms of macro-economic impacts. GlobHighTech scenario results of the two models consistently show deployment of advanced biomass conversion technologies for road transport fuels and chemicals, increased output of 2nd generation biofuels and biochemicals, and biomass consumption. The prevalence of high technology development scenarios by 2030 is also supported by the macro-economic indicators of this study. More specifically, by 2020 our findings indicate a temporary deterioration of GDP and trade balance due to the support required in order to meet EU RED targets. However, investing early in the time horizon to support large-scale bioeconomy developments may revert these trends by 2030. Under high technology development and international biomass trade scenarios may come with macro-economic benefits to the Netherlands on GDP (0.8 bn€/yr), value added (0.7 bn€/yr), reduce projected decline in trade balance (by about 0.7 bn€/yr or 36% compared to LowTech) and employment (by about 2.4-4.5% compared to LowTech). Total agriculture is still a key sector within the total bioeconomy, high technology development increases value added from agriculture by about 0.05 bn€ compared to low technology development. It is also shown that because policies are the key drivers of biofuels and bioelectricity, their value added and contribution remains fairly constant across the scenarios. The largest difference comes from the biochemical sector, which can increase both its absolute and relative contribution if high technology development is supported. Nonetheless, in view of these developments emission reduction to 2030 is modest (i.e. 15% or about 30 $MtCO_2$ compared to 1990). To meet climate goals, more ambitious scenarios need to be assessed (e.g. with CO_2 tax of at least 100 \$/tCO_2). Pursuing ambitious emission mitigation goals is anticipated to increase the biomass share in the economy, trigger fast deployment of technologies and additional technological learning leading, by also replacing fossil-based capacity, thereby increasing the significance of bioeconomy. Especially for the Netherlands, which is expected to become a net natural gas importer beyond 2030 effects on trade balance could be major.

While the present study led to a key improvement in the representation of new bioeconomy sectors in a CGE model, we acknowledge that adaptations and improvements are required so that it may be used to provide concrete directions to policy making. The first set of improvements requires a thorough sensitivity analysis on factors and uncertainty parameters that can influence bioeconomy developments. These are fossil fuel prices, CO_2 prices and agricultural management practices. The second set of improvements calls for an improved linkage between the bottom-up and top-down models by supplying outputs of MAGNET to MARKAL-NL-UU, such as biomass price or future energy demand and iterate the model runs. This will lead to improved consistency between the model results and to complementary conclusions based on macro-economic and physical indicators, from two very distinct models. The latter also requires a consistent approach of converting monetary to physical outputs or vice-versa. Finally, by incorporating CCS and BECCS technologies in the power and liquid fuel sectors of MAGNET, and by extending the temporal and regional scope of the assessment more ambitious climate change mitigation scenarios can be assessed.

6.6 APPENDIX

6.6.1 Overview of the MAGNET model

MAGNET is a recursive dynamic, multi-regional, multi-commodity CGE model, covering the entire global economy (Woltjer et al., 2014). It is built upon the standard GTAP model (Hertel, 1997) and is the successor to LEITAP, which has been previously used to assess bioeconomy (Hoefnagels et al., 2013) and other policy analysis studies (Banse et al., 2008; van Meijl et al., 2006). MAGNET assumes perfect competition, namely producers are price takers whereas in order to produce output they choose the cheapest combination of imperfectly substitutable labour, capital, land, natural resources and intermediates. The core of MAGNET is an input–output model, which links industries in value added chains from primary goods, over continuously higher stages of intermediate processing, to the final assembly of goods and services for consumption. MAGNET uses a general multilevel sector-specific nested constant elasticity of substitution (CES) production function to allow for substitution between primary production factors and intermediate production factors and for substitution between different intermediate inputs such as land and animal feed. Input and output prices are endogenously determined by the markets so as to achieve supply and demand equilibrium. Factor markets are competitive between sectors but not between regions. Households are assumed to distribute income across savings and (government and private) consumption expenditures according to fixed budget shares (Cobb-Douglas (CD) expenditure function). Private consumption expenditures are allocated across commodities by introducing a richer representation of income effects in the demand system. In particular marginal budget shares may vary with the expenditure level (non-homothetic constant differences of elasticity (CDE) expenditure function)[39]. Government consumption is allocated across commodities by fixed budget shares (CD expenditure function). Labour, capital and natural resources (primary production factors) are fully employed in each region and the aggregated supply of each factor equals its demand (equilibrium). MAGNET assumes that that products traded internationally are differentiated by country of origin (Armington assumption). This assumption generates smaller and more realistic responses of trade to price changes than implied by models of homogeneous products (Armington, 1969). It includes an improved treatment of agricultural sectors with emphasis on land (imperfectly substitutable types of land, land use allocation structure, land supply function), feed components, fertilisers (N, P, K), agricultural policies and biofuel policies (Banse et al., 2008; Eickhout et al., 2009; Nowicky et al., 2009; van Meijl et al., 2006; Von Lampe et al., 2014). Segmentation and imperfect mobility between agriculture and non-agriculture labour and capital are introduced in the modelling of factors markets.

[39] On the consumption side, a dynamic CDE expenditure function is implemented which allows for changes in income elasticities when purchasing power parity (PPP)-corrected real GDP per capita changes (Woltjer et al., 2014).

6.6.1.1 MAGNET extension

Table 6.5 MAGNET sector and data aggregation

Sectoral disaggregation (63 commodities):

Primary agriculture (11 commodities): pdr (paddy rice); wheat (wht); other grains (grain); oilseeds (oils); raw sugar (sug); vegetables, fruits and nuts (hort); other crops (crops); cattle and sheep (cattle); pigs and milk (milk); crude vegetable oil (cvol) ;poultry (pigpoul); raw

Food and beverages (6 commodities): meat (cmt); meat product (omt) dairy (dairy); sugar processing (sugar); vegetable oils and fats (vol); other food and beverages (ofd);

Other 'traditional' bio-based activities (2 commodities): fisheries (fish); forestry (frs);

Bio-mass supply (10 commodities): plantations (plan); residue processing (res); pellets (pel); agricultural residues (r_pdr, r_wht, r_grain, r_oils, r_crops, r_hort); forestry residues (r_frs);

Bio-based energy (9 commodities): first-generation biodiesel (biod); first-generation bioethanol (biog); bioelectricity (bioe); second-generation thermal technology biofuel (ft_fuel); second-generation biochemical technology biofuel (eth); biofuel feedstock grains (bf_g); biofuel feedstock molasses (bf_m); biofuel feedstock oils (bf_o); biofuel feedstock sugar (bf_s);

Bio-based chemicals (4 commodities): lignocellulose sugar (lsug); polylactic acid (pla); polyethylene (pe); mixed bio/fossil chemicals (f_chem);

Bio-based and non-bio-based animal feeds (4 commodities): bioethanol by-product distillers dried grains and solubles (ddgs); biodiesel by-product oilcake (oilcake); animal feed (feed);

Fertilizer (3 commodities): fertilizer nutrient nitrogen (fert_N), phosphorous (fert_P), potassium (fert_K);

Fossil fuels (5 commodities): crude oil (c_oil); petroleum (petro); gas (gas); gas distribution (gas_dist); coal (coa);

Electricity (6 commodities): electricity from gas (ely_g); electricity from coal (ely_c); electricity from nuclear (ely_n); electricity from wind and solar (ely_w); electricity from hydro and thermal (ely_h); electricity transport (ely);

Other sectors (4 commodities): chemicals, rubber and plastics (chem); transport (trans); other industry (OthInd); services (serv);

Regional disaggregation (8 regions):

EU members (4 regions): Netherlands (NLD); Germany (DEU); Belgium (BEL); Rest Europe (REU);

Non-EU regions (4 regions): North America (NA); South and Central America (SCA); Africa (AF); Asia and Oceania (ASIA).

Table 6.6 Overview of data sources for the new bio-based sectors in MAGNET

Biomass supply sectors: residues, plantations, pellets	
	Database and references
Production volume	Derived from the production and consumption of bioenergy and biochemical and biomass conversion efficiency
Sustainable potential of residues	Europe: Biomass Policies project (Elbersen et al., 2015)
	Rest of the world: IMAGE (Integrated Model to Assess the Global Environment) (Daioglou et al., 2016)
Cost-structure	IMAGE (Integrated Model to Assess the Global Environment) (Daioglou et al., 2016)
Trade	Trade of biomass from plantations and residues is assumed to be zero. Pellet trade data is taken from UN Comtrade database (UNSD, 2015)
Transport costs	Targets IMage Energy Regional simulation model (IMAGE-TIMER) (Stehfest et al., 2014)
Biomass conversion sectors: bioelectricity, 2nd generation biofuels and biochemicals	
	Database and references
Production	Biofuels: assumed; based on Targets IMage Energy Regional simulation model (IMAGE-TIMER) (Stehfest et al., 2014)
	Bioelectricity: Energy Information Administration (EIA) database (EIA, 2014)
	Biochemicals: assumed; based on MARKet ALlocation model (MARKAL-NL-UU) (Stehfest et al., 2014; Chapter 4)
Cost-structure	Biofuels: MARKet ALlocation model (MARKAL-NL-UU) (van Vliet et al., 2011; Brouwer et al., 2015; Chapter 4)
	Bioelectricity: International Energy Agency (IEA) database and the MARKet ALlocation model (MARKAL-NL-UU) (IEA, 2010; Brouwer et al., 2015; Chapter 4)
	Biochemicals: MARKet ALlocation model (MARKAL-NL-UU) (Chapter 4)
Trade	By assumptions: no trade of bioelectricity, trade of second generation biofuels equal to 5% of production
Transport costs	Targets IMage Energy Regional simulation model (IMAGE-TIMER) (Stehfest et al., 2014)

6.6.2 Overview of the MARKAL-NL-UU model

The bottom-up linear cost-minimisation linear programming energy system model of the Netherlands MARKAL-NL-UU is used to assess the cost-optimal technology portfolio of biomass conversion, other renewables, and fossil fuel technologies and related GHG emissions to supply an exogenously determined demand for electricity, heat, transport, and chemicals to 2030 (Chapter 4). MARKAL-NL-UU uses fossil fuel and CO_2 prices to 2030 based on the New Policies scenario of the International Energy Agency's World Energy Outlook 2014 (OECD/IEA, 2014). Furthermore, the model assumes that targets of the EU Renewable Energy Directive (EU RED) on biofuel blending and renewable energy share on final energy consumption are met by 2020 (EC, 2009b); these are extended based on the Dutch Energy Agreement to 2030 (SER, 2013).

6.6.2.1 Biomass cost-supply

MARKAL-NL-UU includes detailed cost-supply curves of biomass supply from EU28 countries. Biomass feedstocks include sugar and vegetable oil crops, agricultural residues, forestry and primary wood processing residues, energy crops, short rotation forestry and organic waste. The supply potential in 2010-2030 is based on the Intelligent Energy Europe project Biomass Policies (Elbersen et al., 2015) and is consistent between MARKAL-NL-UU and MAGNET. In addition, MARKAL-NL-UU uses cost-prices of biomass feedstocks from the same project, while in MAGNET these are determined endogenously. Finally, MARKAL-NL-UU includes cost-supply curves for wood pellets, sugar, biofuels (bioethanol, biodiesel) and vegetable oil from global markets (Chapter 4).

6.6.2.2 Technology cost-structure and development

MARKAL-NL-UU includes detailed cost-structures of fixed and operating costs for a wide range of technologies and their development to 2030. These are presented in detail in Chapter 4. The cost-structures for electricity production (from fossil fuels, biomass and other renewable resources), 1st generation biofuels, and biomass heat are directly used as inputs to MAGNET.

6.6.2.3 MARKAL-NL-UU results

Results from the bottom-up model for the four scenarios are used to provide insights and relevant data on technologies to MAGNET. Below, we summarise key outcomes for the bioeconomy sectors.

- bioelectricity is limited across the scenario variants (about 9% of total electricity output), and is produced mainly from incineration of organic solid waste and co-produced by biorefineries, in particular in the high technology development scenarios. While biomass co-firing is not found to be a cost-attractive conversion pathway, it is selected as an input to MAGNET to model electricity production from biomass;
- industrial heat from biomass is found as the most cost-optimal solution to meet EU RED targets when compared against other fossil and renewable alternatives. In the Netherlands, biomass heat in industry replaces primarily natural gas, which remains the key fuel for heat in other sectors. This shows the importance of including biomass heat as a natural gas substitute in MAGNET;
- 1st generation biofuels are supplied in LowTech scenarios, while advanced 2nd generation fuels (FT-fuels, 2nd generation ethanol) are supplied in HighTech scenarios (Figure 6.13). Note that compared to Chapter 4, MARKAL-NL-UU in the present study excludes the aviation sector, to enable consistency and comparison with MAGNET;
- oil naphtha remains the primary feedstock to produce chemicals. The production of sugar fermentation chemicals and bulk ethanol-based chemicals is significant. Sugar and ethanol supplied to the biochemical conversion technologies are both from 1st generation and lignocellulosic resources (Figure 6.14).

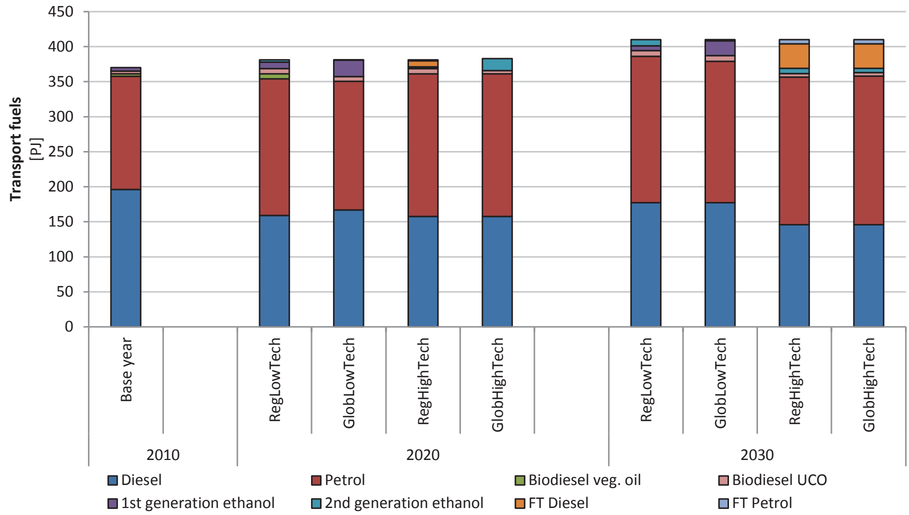

Figure 6.13 Road transport fuel consumption per fuel type in the Netherlands in 2010-2030 based on MARKAL-NL-UU (van Meijl et al., 2016)

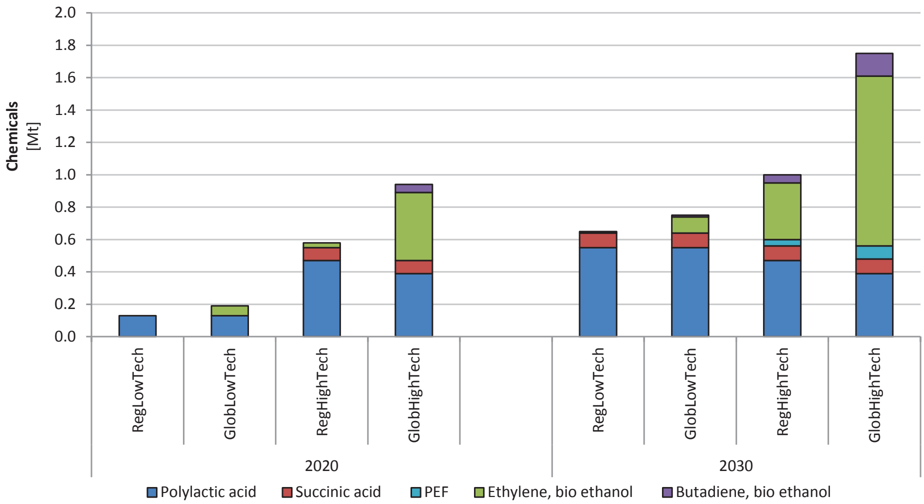

Figure 6.14 Biochemical production per biochemical product in the Netherlands in 2020-2030 based on MARKAL-NL-UU (van Meijl et al., 2016)

7

Summary and conclusions

7.1 RESEARCH CONTEXT

The foreseen change of climate, owing to the rising atmospheric concentration of anthropogenic Greenhouse Gases (GHGs), is to a large extent a result of the industrialised society's heavy reliance on a fossil fuel-based economy. Without fundamental changes in energy supply and demand aimed at deep GHG emission reduction over the next decades (about 40 to 70% in 2050 compared to 2010; IPCC (2014b)), humanity and the planet are threatened by irreversible impacts. Realising such targets requires the substitution of fossil fuels by renewable energy sources, increase in energy and resource efficiency and the deployment of Carbon Capture and Storage (CCS). Biomass is currently the largest supplier of renewable energy (about 10% of global Total Primary Energy Supply (TPES) or about 60 EJ (REN21, 2016)). Should climate change targets be pursued, biomass for bioenergy (i.e. electricity, heat and fuels from biomass) may reach up to 300 EJ by 2050, as it may reduce significantly the use of fossil resources, GHG releases and may even lead to negative emissions (if combined with CCS, i.e. Bioenergy and Carbon Capture and Storage; BECCS).

Today, bioenergy is mainly traditional (i.e. inefficient fuelwood combustion for heating, cooking and lighting). Transitioning from a fossil-based economy towards a bioeconomy requires also a transition from traditional to modern, efficient supply of bioenergy and materials. Biomass is increasingly used for the production of biochemicals, thereby offering a renewable alternative to the chemical industry (about 0.6 EJ; Saygin et al. (2014)) with ethanol-based bulk chemicals (i.e. ethylene and ethylene derivatives) accounting for about 50% of the global total bioplastics production capacity today (IfBB, 2015). By 2050, biomass may substitute up to about 30% of the global total primary energy required for non-energy uses by the chemical industry (i.e. the feedstock used as raw material that is not used for fuel purposes or transformed to fuels) (Daioglou et al., 2014).

Bioenergy and biochemical production, however, does not come without challenges. The GHG emission savings and cost performance of 1st and 2nd generation bioenergy and biochemical supply chains are highly dependent on the characteristics of the production system. Among others these include the type of feedstock used, its location, the agricultural management practice, the induced Land Use Change (LUC) and associated direct (dLUC) and indirect (iLUC) emissions, and the performance of the conversion technologies. Furthermore, deployment levels of bioenergy and biochemicals are highly uncertain. Future supply potentials and their regional distribution will depend on developments in biomass producing sectors (e.g. agriculture, forestry). Demand regions may invest further in their domestic biomass conversion capacity if technologies become more competitive, thereby further increasing the demand for biomass. Feedstock supply to the demand regions will depend on the progress of international biomass trade. Such developments might be accelerated or delayed depending on the level of policy ambitions to mitigate GHG emissions and future fossil fuel prices.

It is not straightforward to assess the influence of these characteristics because biomass deployment in the economy also depends on several dynamic factors based on the interaction between biomass supply and the technologies in the energy system (including the chemical industry). If bioeconomy sectors grow, potential conflicts may arise from end-use sectors when in competition for the same feedstock. At the same time, cross-sectoral synergies may surface if multi-output conversion technologies (e.g. biorefineries) are deployed. Furthermore, other renewable energy conversion and carbon mitigation technologies (CCS, BECCS) compete with biomass in the supply of renewable energy and in their potential contribution to emission reduction. However, there are uncertainties on the readiness and speed of technical change of biomass conversion, other renewable energy and carbon mitigation technologies that ultimately determine their economic competitiveness against the fossil energy system.

Bioeconomy developments are typically assessed by a variety of tools and methods, which have fundamental differences in the activity they evaluate, their geographical detail, their temporal scope, the impact categories and their interactions with the human and natural system (Wicke et al., 2015). Each tool and method is designed to answer specific types of questions and so far none can assess bioeconomy in its entirety. This entails that the available toolkit demonstrates strengths in providing specific insights but limitations in representing the complexity of bioeconomy and its potential impacts (Figure 7.1). The main trade-off across the different bottom-up models and analyses (e.g. Life Cycle Assessment; LCA), Partial Equilibrium Models (PEM), Computable General Equilibrium (CGE) models, and integrated assessment models that are commonly used, is between the level of bottom-up detail that they incorporate and their comprehensiveness to describe the dynamic interactions of the human and natural system (Wicke et al., 2015).

Against this background, the most relevant knowledge gaps in bioeconomy assessments, though not limited to, are on: (a) the environmental impacts of emerging bioeconomy products across established supply chains, (b) the potential role and contribution of biomass within a national context in the transition towards a more sustainable energy system and (c) the macro-economic impacts of emerging bioeconomy developments.

7.2 AIM AND RESEARCH QUESTIONS

This dissertation addresses some of the key knowledge gaps related with current and mid-term developments of emerging bioeconomy products and sectors. It provides different insights in the performance of bioenergy and biochemical production systems and the synergies and the conflicts between bioeconomy sectors by using bottom-up and top-down modelling approaches developed for and applied in specific supply chains and national context. To achieve this aim, it assesses the environmental performance of established biofuel and biochemical supply chains with core process steps occurring in developing and emerging economies (India and Brazil) by addressing methodological implications of LCA. Furthermore, it improves the representation of bioenergy and biochemical production chains in an energy systems and a macro-economic CGE model, paying particular attention to biomass supply and advanced conversion technologies. To obtain insights in the structure and key parameters that are critical for the models' extension, it places its focus on the energy system of the Netherlands. The country has an efficient agricultural sector, it has large refining and petrochemical production capacity, and competitive logistics infrastructure; therefore, it can support large-scale bioeconomy developments. By improving the understanding on environmental and macro-economic impacts that emerging bioeconomy products and sectors may have, this dissertation seeks to demonstrate what the role and implications of large-scale bioeconomy deployment might be in the medium term as required in order to transition to a sustainable system that meets long-term societal challenges. To this end, the following research questions are addressed:

I. What are current and potential mid-term environmental implications of biofuels and biochemical applications?
II. What is the potential size and contribution of bioeconomy in a national energy system in pursuing mid-term climate change mitigation targets?
III. What are the potential mid-term economic implications of bioeconomy developments at a national level?
IV. How can tools and methods for assessing current and mid-term bioeconomy developments be improved?

This dissertation addresses the research questions I to IV in Chapter 2 through Chapter 6 (Figure 7.1, Table 7.1). Chapter 2 examines the environmental impacts of sugarcane ethanol production in Brazil and India. It provides insights in environmental impacts depending on methodological choices and systems used. Based on these findings, Chapter 3 extends the

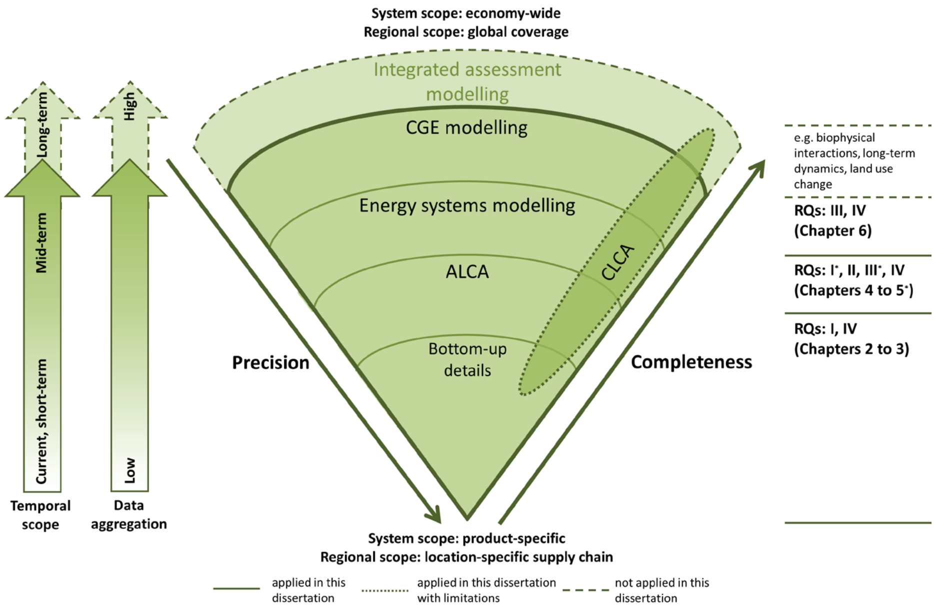

Figure 7.1 Methodological framework used to assess the implications of bioeconomy developments in this dissertation. ALCA stands for Attributional LCA, CLCA stands for Consequential LCA, CGE stands for Computable General Equillimrium (adapted from Creutzig et al. (2012))

assessment to bioplastics from sugarcane ethanol and compares their environmental performance to the European supply of petrochemical plastics. It also focuses on technological improvements and other interventions across the supply chain that may reduce the environmental implications of bioplastics production in the near future. Chapter 4 improves the representation of biomass supply and of advanced biomass conversion technologies to biofuels and biochemicals in a cost-minimisation linear programming model of the energy system of the Netherlands (MARKAL-NL-UU). The model is applied assuming current policy targets and technology development scenario variants to 2030. Chapter 5 deploys the model under different scenario combinations of technology development, biomass cost-supply, fossil fuel prices and sector-specific assumptions to assess their influence on mid-term GHG mitigation at a national level. Chapter 6 presents a modelling framework that uses MARKAL-NL-UU to improve the representation of bioeconomy in a CGE model (MAGNET). The extended MAGNET model is used to demonstrate potential macro-economic impacts of large-scale bioeconomy developments in the Netherlands to 2030.

Table 7.1 Overview of the topics of this dissertation in relation to the research question they address

Chapter	Topic	Method	Regional scope	System scope	Temporal scope	Research question			
						I	II	III	IV
2	Life cycle assessment of sugarcane ethanol production in India in comparison to Brazil	LCA	India, Brazil	Product-specific	Current	X			X
3	Life cycle impact assessment of bio-based plastics from sugarcane ethanol	LCA	India, Brazil, Europe	Product-specific	Current, short-term	X			X
4	Emerging bioeconomy sectors in energy systems modelling – Integrated systems analysis of electricity, heat, road transport, aviation and chemicals: a case study for the Netherlands	Cost-optimisation	The Netherlands	Energy system	Mid-term		X		X
5	The role of bioenergy and biochemicals in CO_2 mitigation through the energy system – A scenario analysis for the Netherlands	Cost-optimisation	The Netherlands	Energy system	Mid-term	X	X	X	X
6	On the macro-economic impact of bioenergy and biochemicals – Introducing advanced bioeconomy sectors into an economic modelling framework with a case study for the Netherlands	CGE modelling	Global, focus on the Netherlands	Total economy	Mid-term			X	X

7.3 SUMMARY OF RESULTS

Chapter 2 presents a cradle-to-gate LCA of ethanol production from sugarcane molasses in Uttar Pradesh, India (henceforth referred to as India) and compares it to ethanol production from sugarcane juice in South-Central Brazil (henceforth referred to as Brazil). Consumption of ethanol in India is gradually growing in road transport and is also used for production of advanced biochemicals. The multi-functionality (or allocation) problem occurs in both Indian and Brazilian ethanol production systems. The product-systems are assessed using the same methodological approach by applying variants of system expansion and economic allocation.

Despite the higher GHG emission profile of Indian sugarcane, ethanol shows lower emission profiles (0.09-0.3 $kgCO_{2eq}/kg_{etOH}$) compared to Brazilian ethanol (0.5-0.6 $kgCO_{2eq}/kg_{etOH}$) when system expansion is applied. This outcome is influenced by two factors. Firstly, the credits associated with surplus electricity in India (about 0.3-0.6 $kgCO_{2eq}/kg_{etOH}$) are higher than those of surplus electricity and bagasse in Brazil (about 0.04-0.2 $kgCO_{2eq}/kg_{etOH}$) as a result of a more CO_2 intensive fuel mix assumed as displaced. Secondly, the impacts of the Indian product-system are dampened due to the fact that exclusively molasses, which are a co-product of sugar production, are used to produce ethanol. When economic allocation is applied throughout all steps of production, then results for ethanol produced in India and in Brazil are comparable (about 0.6 $kgCO_{2eq}/kg_{etOH}$).

Impacts on human health and ecosystem quality are lower in the Indian compared to the Brazilian product-system, but overall comparable. In both cases, these impacts are primarily associated with sugarcane cultivation. In India these mainly relate to application of fertilisers and pesticides. In Brazil, the impacts are aggravated mainly due to pre-harvesting burning practices. Net water consumption and related impacts on human health and ecosystem quality are only relevant for the Indian product-system, as there are indications for decrease in the regional groundwater table.

The mills' and distilleries' co-generation plants, the utilisation of co-products such as stillage and bagasse for energy purposes, and assumptions on specific pesticide application, pre-harvesting burning practices and stillage treatment are found to contribute significantly to results. The Indian product-system carries risks. If, for instance, distilleries do not treat wastewater and anaerobic conditions prevail the impact on ecosystem quality may be substantial and GHG emissions may range from 2.6 to 3.1 $kgCO_{2eq}/kg_{etOH}$.

Accounting for emissions induced by iLUC may increase the emissions of Brazilian ethanol by 0.08-1.2 $kgCO_{2eq}/kg_{etOH}$. The wide range is based on the variation of LUC change emission factors found in literature. For India, molasses are assumed not be diverted from the feed sector for ethanol production, thereby making displacement effects irrelevant. Nonetheless, if the demand for ethanol increases, for example, due to increase in demand for biofuels due to the national Ethanol Blending Programme or biochemicals, then the demand for feed will need to be compensated by crop production elsewhere. In that case, molasses will also carry an impact equivalent to that of feed crops, and thereby Indian ethanol is expected to have higher impacts than those presented in this dissertation.

Chapter 3 applies LCA in a methodologically consistent manner with Chapter 2 to assess the environmental impacts of two of the currently largest produced bioplastics. These are fully bio-based Polyethylene (bio-PE) from Brazilian sugarcane ethanol produced in Brazil, and partially bio-based Polyethylene Terephthalate (bio-PET) from Indian and Brazilian sugarcane ethanol. The bio-based component of bio-PET (i.e. bio-based Monoethylene Glycol; bio-MEG) is produced in Uttar Pradesh, India and is further synthesised to bio-PET in Europe. Both bioplastics are compared against current European supply of petrochemical PE and PET.

Bio-PE production results in GHG emissions of around -0.75 $kgCO_{2eq}/kg_{PE}$, i.e. about 140% lower than petrochemical PE. This estimate takes into account the large bio-based carbon content of the polymer, which is deducted from the final emission profile. The different methods applied to solve the multi-functionality problem do not influence the results significantly due to the contribution of other steps (catalytic dehydration, polymerisation, transport). This indicates the robustness of outcomes on bio-PE. Only the combination of system expansion that assumes efficient or inefficient electricity co-generation by mills and distilleries leads to a ±50% variation on the outcomes due to the related credits of the Brazilian ethanol product-system. Savings on non-renewable energy use are found to be approximately 65% and variation due to system efficiency in combination with system expansion is approximately ±25%.

Bio-PET shows GHG emission profiles (1.9-2.4 $kgCO_{2eq}/kg_{PET}$) that are comparable with petrochemical PET (2.15 $kgCO_{2eq}/kg_{PET}$). Firstly, the bio-based content of bio-PET (i.e. bio-MEG) relative to PET is only about 28% on a mass basis and does not counterweigh the contribution in GHG emissions of other steps of the production chain (terephthalic acid production, esterification, transport). Emissions from transport contribute more to the production of bio-PET from Brazilian ethanol, as ethanol first needs to be transported to India to be converted to the bio-based monomer. As a result, bio-PET from Brazilian ethanol may lead to higher emissions than petrochemical PET production in Europe. GHG emissions from both product-systems, are relatively high as the production of the bio-based component depends on an energy supply system of high CO_2 emission intensity (under reference conditions, process energy of bio-MEG is based on coal-based steam and grid electricity). For the Indian product-systems, the results are influenced by the approach used to address the multi-functionality problem. The main reason is that the system's electricity consumption is higher than the system's electricity supply by co-production (e.g. at the mills and distilleries). Therefore the assumed CO_2 intensity of grid electricity ultimately determines the GHG emissions estimated by the two system expansion approaches. GHG emission savings of bio-PET may reach up to 30% compared to petrochemical PET if best available technology for bio-MEG production is assumed and steam is supplied from natural gas instead of coal. Co-generation of electricity and heat to supply process energy may also reduce the environmental impact of bio-MEG. However, it is shown that regional policy instruments such as feed-in-tariffs that currently exist in India, may incentivise the producer to supply electricity to the grid.

The analysis shows that there are trade-offs between GHG emissions and other environmental impact categories. Production of bio-PE and bio-PET impairs human health and ecosystem quality to up to 2 orders of magnitude compared to their petrochemical counterparts. While the savings on GHG emissions are global, the impacts on human health and ecosystem quality occur at a regional level. The impacts relate mainly to the production and harvesting of sugarcane as discussed in Chapter 2. Improvements in these areas are shown to partly reduce the impacts, however, they are still higher than petrochemical plastics.

Chapter 4 presents a bottom-up energy system model (MARKAL-NL-UU), which is extended with fossil-based and bio-based chemicals as well as with Renewable Jet Fuels (RJF) to assess the deployment of biomass conversion technologies in the Netherlands until 2030. The model comprises detailed cost-structures and mid-term developments for the fossil energy system and mitigation options. It includes detailed parameterisation and cost-supply curves for biomass, renewable energy conversion technologies and CCS. The model incorporates multi-output processes, such as biorefineries, to address cross-sectoral synergies. To capture the uncertainty in technical progress, different technology development scenarios are used in order to assess cost-optimal biomass utilisation pathways over time. Energy efficiency improvements in the residential and commercial sector are assumed implicitly, in line with the heat demand projections under established policies. Assumed policy targets in 2030 include the 10% blending target for biofuels and the 16% renewable energy share on final energy consumption. Also, moderate climate policy, based on an emission tax (25 €/tCO_2) is assumed in 2030.

Slow technical progress leads to biomass use mainly for heating (105 PJ_{th}; 12% of total heat supply), 1st generation biofuels from hydrotreated oils (33 PJ, 5% of total transport fuel supply) and biochemicals based on 1st generation fermentation systems (26 PJ; 5% of total chemicals supply). Renewable electricity supply is primarily supplied from wind turbines and photovoltaics (140 PJ; 30% of total electricity supply), while bioelectricity is limited (9 PJ; 2% of total electricity supply) to production from organic solid waste. Enhanced technology development allows the production of 2nd generation biofuels from solid biomass (61 PJ; 10% of total transport fuel supply), while the remainder is 1st generation biofuels (9 PJ; 1.5% of total transport fuel supply). Therefore, under assumed policies, in both technology development scenarios, decarbonisation of the road transport sectors relies on biomass and not other options such as electric vehicles. RJF are about 13 PJ (about 7% of total jet fuel supply). Furthermore, large volumes of biochemicals are produced (54 PJ; 10% of total chemical supply). These are primarily sugar-based chemicals from 1st generation and lignocellulosic feedstocks in approximately equal shares. The remainder is ethanol-based chemicals from imported ethanol. Bioelectricity supply is doubled compared to low technology development (18 PJ; 4% of total electricity supply). The additional output is supplied by biorefineries. Renewable electricity from wind turbines and solar panels remains at similar levels (about 140 PJ). The required biomass ranges from 230 to 300 PJ in 2030, supplied primarily from imported resources. Under existing policies, CO_2 emissions are shown to reach to 1990 levels (i.e. 140-145 $MtCO_2$ compared to 160 $MtCO_2$ in 2010), however, fail to meet the 40% emission reduction objective (i.e. 86 $MtCO_2$) as aimed by the European Union (EU) by 2030.

Chapter 5 assesses least-cost pathways to 2030 for a number of scenarios applied to the energy system of the Netherlands, using the cost-minimisation linear programing model developed in Chapter 4. In this chapter, ambitious GHG emission mitigation scenarios are assumed based on a CO_2 emission tax of 69 €/tCO_2 in 2030. Technology-specific targets are assumed until 2020. No targets are assumed beyond 2020, thereby enabling technology-neutral competition of biomass conversion to bioenergy and biochemicals, other renewable technologies, and other mitigation options (CCS, BECCS) with reference fossil fuel conversion technologies.

The type and magnitude of biomass deployment is highly influenced by the level of technology development, biomass cost-supply, fossil fuel prices and ambitions to mitigate climate change. Final production from renewable resources lies between 460-510 PJ in 2030 and does not differ significantly across technology development scenario variants. The renewable energy output corresponds to a 23-24% share on final energy consumption (excluding chemicals) or to a 18-20% share on total final energy consumption (including chemicals). More than two thirds (73-79%) of renewable energy output is attributed to biomass. Electricity from biomass remains small (20-50 PJ; primarily from biorefineries, co-firing and municipal solid waste incineration), as wind is the key supplier of renewable electricity. Biochemicals are produced as a result of retirement of steam cracker capacity (20-50 PJ; 5-10% of the sector's output) even under baseline assumptions, where no CO_2 tax on emissions is assumed beyond 2020. In the high technology development scenario, the output of biochemicals almost doubles (about 100 PJ; 17%) compared to the baseline. This is a result of multi-output technologies that produce both chemicals and road transport fuels, with the latter being affected by the CO_2 tax. The electricity sector is most sensitive to fossil fuel price assumptions. High fossil fuel prices can lead up to a factor 2.5 increase of electricity from renewable energy sources compared to reference scenarios. A combination of scenarios may lead to ±50% variation of final production from renewables compared to the reference. More specifically, the high technology development scenario combined with high fossil fuel prices leads to 745 PJ of total renewable energy supply. Low technology development combined with low fossil fuel prices leads to 230 PJ, which is comparable to baseline scenarios without CO_2 tax. The contribution of biofuels in road transport fuels goes beyond 60% under high technology development combined with high fossil fuel price assumptions. This occurs due to the increased biofuel supply (20% higher than in the reference scenario) and due to reduced fuel demand by the sector (roughly 1/3 or 130 PJ decrease compared to the reference scenario, as more efficient vehicles are deployed). The share of 1st generation biofuels over the total road transport and jet fuel demand is 1-13% and 1-12% in low and high technology development scenario variants, respectively. The share of 2nd generation biofuels over the total road transport and jet fuel demand is 0-3% and 10-29% in low and high technology development scenario variants, respectively. Low technology development combined with low fossil fuel price scenarios can reduce the contribution of biofuels to the sector down to only 1%.

Across all scenario variants, biomass consumption ranges between 180 and 760 PJ. High technology development leads to additional 100-270 PJ of biomass consumption compared to low technology development counterparts. Biomass consumption is highly sensitive to the assumed development of fossil fuel prices to 2030. Low technology development scenarios consume a maximum of 560 PJ biomass under most favourable conditions induced by high fossil fuel prices, which are comparable to reference consumption levels of the high technology development scenario.

The CO_2 tax leads to emission reduction in the range of 35-43% for low and high technology development scenarios, respectively, compared against projected baseline emissions in 2030. In absolute terms, additional emission reduction due to high technology development is 15 $MtCO_2$ compared to low technology development. Even under the high technology development scenario additional 15 $MtCO_2$ reduction is required to reach the 40% emission reduction target, compared to 1990. Low-cost biomass supply reduces additional 5 $MtCO_2$ only under the high technology development scenario, partly bridging the gap to the target. Results indicate that under high fossil fuel prices the 40% emission reduction target compared to 1990 is reached in the high technology development scenario. In the low technology development scenario, however, CO_2 mitigation is 12 $MtCO_2$ below the target. The contribution of CCS and BECCS to emission reduction is significant (42-60%, compared to the baseline). The remainder of emission reduction is primarily achieved through biomass (20-40 $MtCO_2$), as with the exception of high fossil fuel price scenarios, the capacity of wind power and other renewable energy sources does not increase significantly beyond 2020 compared to the baseline.

Under reference fossil fuel prices, assuming lower demand for petrochemicals leads to a 16-37% reduction of output in low and high technology development, respectively compared to their reference. If low fossil fuel prices are assumed then an uncompetitive environment for biochemicals is created as in low technology development scenarios production is almost non-existent and in high technology development scenarios the output is reduced by 85%. Decommissioning coal-based power capacity beyond 2020 does not alter the total direct CO_2 emissions as off-shore wind turbines compete with CCS in coal-fired power plants.

Chapter 6 presents a modelling framework, which uses the energy system model developed in Chapter 4 (MARKAL-NL-UU) to improve the representation of bioeconomy sectors in a global, multi-sector and multi-region CGE model (MAGNET). It harmonises technology cost-structures, scenario assumptions on technology development and openness of trade, and exogenous parameters to expand MAGNET by selecting key biomass conversion technologies for advanced biofuels and biochemicals production, lignocellulosic biomass supply and tradable pellets. The framework is demonstrated by assessing macro-economic developments of the Dutch bioeconomy to 2030.

Biofuel shares in both models are considerably higher in high technology development scenarios as especially 2nd generation biofuels become competitive. In combination with international biomass trade, the biofuel share is 13% in MAGNET and 12% in MARKAL-NL-UU, mainly comprised of 2nd generation biofuels. The share of renewable electricity is about 35%, which is primarily supplied by wind turbines. Biomass use in electricity production is limited across scenarios and comparable between the models. Furthermore, both models estimate low to moderate shares of biochemicals over total chemicals, which is larger in high technology development. Key products are ethanol-based chemicals from 2nd generation ethanol (0.6 bn€/yr in the low technology development scenarios and 1.7 bn€/yr in the high technology development scenarios), chemicals from lignocellulosic sugar (0.2 bn€/yr in the high technology development scenarios) and from conventional sugar (0.06 bn€/yr in the high technology development scenarios). Market openness to extra-European resources plays a modest role, as biomass for 2nd generation technologies is available within Europe at competitive prices. Finally, MAGNET outcomes demonstrate high consumption of domestic and imported agricultural and forestry residues (90% of total biomass consumption in 2030). Imported biomass accounts for almost 50% of total biomass consumption.

High technology development and open trade indicate macro-economic benefits on gross domestic product (0.8 bn€/yr) and value added (0.74 bn€/yr). Furthermore, they reduce projected decline in trade balance (0.65 bn€/yr) and employment (2.5-4.5%) compared to low technology development scenarios.

7.4 MAIN FINDINGS AND CONCLUSIONS

I. What are current and potential mid-term environmental implications of biofuels and advanced biochemical applications?

The performance of biofuel production pathways depends on the characteristics of their regional production system even when they are produced from the same biomass feedstock (i.e. sugarcane). Most influential, in view of their GHG emission performance, is the interaction of co-product supply (e.g. bagasse, electricity) with the regional energy supply system (e.g. CO_2 emission intensity of grid electricity, reference fuel for heat supply), as revealed by the approaches used to solve the multi-functionality problem. This generally leads to more robust results on Brazilian ethanol, despite the fact that Indian ethanol is produced from a co-product and therefore upstream impacts from sugarcane and sugar production are dampened due to allocation. On the other hand, the GHG emission savings of Brazilian ethanol depend on LUC emissions emissions due to sugarcane expansion that may lead to up to a factor 3 increase of its cradle-to-gate GHG emissions. In India, currently, this is less of a concern as production relies on a co-product. If, however, ethanol demand grows due to increase in demand either for biofuels or biochemicals, then molasses carry the risk of being diverted from the feed sector, and may therefore induce LUC and associated emissions.

Similar dependencies are found for the GHG emission performance of biochemicals. Biochemicals do not necessarily lead to favourable environmental performance compared to their petrochemical counterparts, if CO_2 intensive fuels (e.g. coal) are used to supply process energy and if insufficient attention is paid to energy efficiency. Ultimately, the environmental performance of bioplastics supply chains depends on the type of biochemical or bioplastic produced, the conversion efficiency, and on the regional context. Production chains of fully bio-based plastics, such as bio-PE demonstrate higher savings (up to 144% compared to petrochemical PE, produced in Europe), as opposed to partially bio-based plastics such as bio-PET (±10%; for comparison, the GHG emission performance of the bio-based component, i.e. the monomer MEG, ranges ±40%, compared to its petrochemical counterpart in Europe). Supply of Brazilian ethanol instead of Indian ethanol does not lead to reduced GHG emissions of bio-PET, because process energy requirements of ethanol conversion to the biochemical are met by the reference Indian energy supply system, which is highly CO_2 intensive. If Indian ethanol is used, burden-shifting due to co-production occurs between the product-systems that interact with the energy system (on the one hand the Indian ethanol product-system that supplies electricity to the grid and on the other the biochemical product-system that uses electricity from the grid). When improvements are assumed on the biochemical product-system (e.g. on conversion efficiency, process energy supply) the emission savings of bio-PET may be up to 30% compared to petrochemical PET. While the potential benefits on GHG emissions occur at a global level, the impairment of human health, ecosystem quality and water stress range from factors to orders of magnitude higher compared to petrochemical production of PE and PET, and affect the biomass supply region, as they are primarily related with the agricultural practices. Such impacts are associated mainly with agrochemical inputs and harvesting practices and are highly sensitive to the assumptions on the types of pesticides, treatment and disposal methods of wastes and their interaction with the environment.

In the medium term, countries that are remote from biomass supply regions such as the Netherlands are expected to use large volumes of biomass. Therefore in the near future supply and conversion may occur in different regions. The GHG emissions reduction that may result from bioenergy and biochemicals ultimately depends on the deployment levels of conversion capacity in the demand region, which are influenced by several factors such as technology development, cost-supply of biomass and ambitions to meet climate targets (Chapter 5). Ambitious scenarios demonstrate that large volumes of biomass may be consumed for production of 1st and primarily 2nd generation biofuels leading to significant decarbonisation of the road transport sector (up to about 78 gCO_2 avoided/MJ). These emissions exclude the upstream emissions from production and transport of biomass to the Netherlands, which largely relies on imports. Emissions from supply chains increase GHG releases but not significantly (Chapter 4) and depend on the supply region and the production systems characteristics (Chapter 2).

Similar conclusions are drawn for biochemicals. Conversion technologies to biochemicals become cost-competitive and are deployed in the national economy by 2030 even under baseline scenarios that assume no CO_2 tax. This leads to large savings of fossil feedstock. Compared to baseline scenarios, additional 17-23 PJ of fossil feedstock is avoided by the production of biochemicals by 2030 (Chapter 5) leading to about 1-2% avoided direct GHG emissions by the Dutch energy system (i.e. 1.3-1.8 $MtCO_{2eq}$, assuming an emission factor of 73.55 $kgCO_{2eq}$/GJ for the fossil feedstock and complete incineration at the end-of-life; or about 1.8 $kgCO_{2eq}$/kg biochemical). The total avoided GHG emissions attributed to biochemicals are higher, as these are also supplied in the baseline scenarios; however, in the absence of a counterfactual scenario these cannot be quantified. Also, ultimately the savings in emissions will depend on the chemical's application, lifetime, and end-of-life.

II. What is the potential size and contribution of bioeconomy in a national energy system in pursuing mid-term climate change mitigation targets?

Under established policy targets (i.e. the EU's renewable energy supply target and the national support on early deployment of on-shore and off-shore wind turbines), the emissions of the Netherlands in 2030 are comparable to 1990 levels but by about 50 $MtCO_2$ away from the 40% emission reduction target in 2030 (Chapter 4 and Chapter 6). The potential role of bioeconomy as investigated within the energy system of the Netherlands is significant when considering that efforts to meet climate targets by 2030 need to be intensified. Rapid technology development makes biomass conversion technologies cost-competitive early in time and through their deployment, in combination with CCS in the power sector and BECCS in transport, significant steps towards deep emission reduction can be made. Furthermore, rapid development and early deployment of biomass conversion technologies, creates a more resilient energy system, less vulnerable to exogenous parameters such as fossil fuel price volatility, while still contributing to emission reduction. The Netherlands may then contribute to emission mitigation as opposed to offshoring their obligation by purchasing allowances.

More specifically, Chapter 5 assesses developments based on an ambitious emission tax of 69 €/tCO_2 by 2030. It shows that if development and deployment of advanced biomass conversion technologies is delayed, then emission targets are not reached in 2030, even under most favourable conditions shaped by low-cost biomass supply and high fossil fuel prices. High technology development becomes a pre condition for a cost-efficient low carbon intensive economy. Nonetheless, even under rapid technical growth of biomass conversion technologies, their contribution to GHG emission mitigation is dependent on global developments of biomass markets and supply costs, and fossil fuel prices. Overall, reduction in CO_2 emissions achieved in high technology development pathways ranges between 15-43% by 2030 compared to 1990. The lower values are related to an uncompetitive environment created either by low fossil fuel prices or underdeveloped biomass markets (such scenarios fall short by about 35 $MtCO_2$ in meeting the target). Under the reference scenario, however, the achieved emission reduction is 30% (i.e. 15 $MtCO_2$ below the target). In the power sector about 100 PJ of wind and solar power are supplied and 27 $MtCO_2$ of emissions from natural gas and coal-fired power plants are captured and stored. The contribution of biomass is significant, as about 620 PJ/yr are needed to supply about 300 PJ of bioenergy (about 17% bioelectricity, 30% biomass heat, 53% biofuels) and about 100 PJ of biochemicals. This corresponds to a 23% share of renewable energy on final energy consumption (i.e. 7% additional to the national target, in which biomass contributes above 70%). The remainder of the emission reduction is achieved by BECCS applied to biofuel production technologies (13 $MtCO_2$ are captured and stored in the reference scenario). BECCS is required to embark on cost-efficient GHG mitigation trajectories to 2030. Without these options mid-term GHG emission mitigation targets may not be reached. The role of advanced biorefineries is key to supply biofuels and feedstock for further conversion to biochemicals, thereby indicating cross-sectoral synergies.

While the reference scenario fails to achieve the target, high technology development valorises opportunities that are created if biomass markets develop and the cost-efficiency of biomass supply chains increases. This can be achieved, for example, by using surplus or abandoned agricultural land for energy crops production and intensification of agricultural productivity and if efforts focus on biomass densification to reduce transport and handling costs (e.g. pelletisation). Embedding bioenergy and biochemicals in the structure of the Dutch energy system utilises additional 100 PJ of biomass. This leads to an additional 5 $MtCO_2$ reduction thereby closing the gap to the target, through increased biomass heat (by 60%), biofuel (by 4%) and bioelectricity (by 3%) supply, compared to their reference. The additional emission reduction is realised primarily in the industry and transport sector. Under most favourable conditions induced by high fossil fuel prices, the target is exceeded by 3%. While the reduction is primarily attributed to the significant increase of wind power (by a factor 2 compared to reference scenarios) and modernisation of the road transport fleet, biomass heat and biofuel production is also stimulated as they increase by 88% and 18%, respectively.

III. What are the potential mid-term economic implications of bioeconomy developments at a national level?

The economic indicators provided by both models used in this dissertation (MARKAL-NL-UU in Chapter 5 and MAGNET in Chapter 6) point to the direction that investing in advanced biomass conversion technologies early in the time horizon is a no-regrets solution for the Dutch economy to support growth and at the same time pursue climate change mitigation goals.

Chapter 5 shows that high technology development scenarios generate by up to 3 bn€ lower cumulative system costs compared to scenarios with low technological growth in the period 2010-2030. Apart from investments in capital stock, key contribution to the total costs comes from variable parameters such as fossil fuel, biomass and CO_2 emission price. It is also shown that large-scale bioeconomy developments in the Netherlands, can bring savings of up to 0.8 bn€/yr compared to a low technology development pathway under reference biomass cost-prices. In absolute terms, high technology development scenarios increase system costs by about 2 bn€/yr compared to a baseline in which no climate action is taken beyond 2020. By 2030, high technology development scenarios demonstrate lower CO_2 emissions between 11-20 $MtCO_2$/yr, compared to low technology development scenario counterparts. The above lead to the conclusion that investing early in the time horizon on bioeconomy leads to lower costs spent on the national energy system and contributes more to GHG emission reduction. However, such scenarios still fail to meet the 40% GHG emission reduction target compared to 1990. This goal is achieved only under the high fossil fuel price scenario, in which total costs increase to about 12 bn€/yr. For such scenarios macro-economic effects are not assessed in this dissertation.

These trends are at large in agreement with the conclusions drawn based on the MAGNET model (Chapter 6). In the near-term (2020) investing in bioeconomy leads to a temporary deterioration of GDP (about 0.2-0.8 bn€/yr) due to the support required in order to meet the EU Renewable Energy Directive's (EU RED) targets in 2020. The short-term impacts on GDP depend on biofuel imports from South and North America. Nonetheless, investments on domestic biomass conversion capacity are required early in the time horizon, as they are found to be a prerequisite in order to reverse the negative GDP trends by 2030. Scenarios that assume bioeconomy developments either in technologies or in biomass markets have a positive effect on the GDP by up to 1 bn€/yr from 2030 onwards. Highest returns for the Dutch economy are observed in the high technology development scenarios. Firstly, they create added value in the domestic agricultural sector. Secondly, emerging sectors such as biochemicals may generate value of about 250 M€/yr, which is comparable to the value added of bioelectricity generation. However, the latter is supported by policies, whereas no similar stimulus is provided to biochemicals. Thirdly, while the overall employment in bioeconomy sectors (agriculture, bioenergy and biochemicals) declines by 2030, bioeconomy investments partly mitigate this effect, compared to a counterfactual scenario. Finally, bioeconomy developments restrain deterioration of trade balance primarily due to exports of 2nd generation biofuels and biochemicals.

IV. How can tools and methods for assessing current and mid-term bioeconomy developments be improved?

7

In this dissertation, established tools and methods have been used in product-specific and region-specific contexts to improve the representation of aspects relevant to bioeconomy assessments. Chapter 2 through Chapter 6 provide a comprehensive overview of improvements and limitations of this effort, which are discussed below as an answer to this research question.

Product-specific assessment
Improvements: the product-system of bioenergy and biochemicals needs to be broken down in each step of production as opposed to assuming inventory data as background processes. This enables modelling key inputs such as energy supply mix and energy intensive processes at the sub-system level, in relation to the approach used to address the multi-functionality problem (system expansion or allocation). Breaking down the sub-systems is deemed necessary for production in the regional context and is especially relevant when system expansion is applied to avoid double-counting or misleading results due to burden-shifting of impacts between product-systems. Presenting outcomes based on all approaches used to account for co-production avoids forming conclusions that may, for example, credit an inefficient product-system.

Completeness of inventories and reliable and verifiable process information (e.g. based on industrial production data), are of equal importance. The increased precision of outcomes as a result of including bottom-up details in LCA studies, as applied in this dissertation, leads to more conservative estimates on the environmental performance of biochemical applications. At the same time, it also leads to increased understanding on trade-offs across impact categories and highlights opportunities for improvement.

Limitations: The consequential LCA framework as applied in this dissertation is faced with limitations. It fails to properly address longer term impacts of additional deployment of bioenergy or biochemicals or possible changes in the energy supply mix. Such changes may, for example, be induced due to strong climate policy where larger shares of renewable energy will be supplied or by low fossil fuel prices where supply from fossil resources will be dominant. Energy systems models can assess such scenarios as demonstrated in Chapter 4 and Chapter 5. Furthermore, the consequential LCA method is too static to assess direct and indirect impacts of sugarcane expansion should the demand for biofuels or biochemicals increase. Improved feedback mechanisms and iterations with other models can help to address this limitation and improve the precision of product-specific analysis. CGE models can provide these as they model impacts of increased demand on the global land use system, similar to the MAGNET model used in Chapter 6.

Energy systems assessment

Improvements: From an energy systems perspective, the extension of an established model (MARKAL-NL-UU) to include biochemicals requires detailed representation of the reference petrochemical industry. Simplification of the petrochemical industry's process flows is necessary as these are complex and large. The key process steps need to be represented explicitly in the model's structure are feedstocks, basic chemicals, intermediate chemicals and final products (Chapter 4). Focusing on a national economy enables the assessment of other parameters relevant to include in the model's structure (i.e. the existing capital stock, cost and technical performance over time, and parameterisation to enable assessment of fossil fuel price variation). In mid-term assessments, the capital stock and its residual technical lifetime affects the point in time of biochemicals' deployment. Secondly, multiple biorefinery outputs need to be linked to end-use sectors, as opposed to, for example, accounting for production costs of the determining product in a reduced form (e.g. by assigning credits). This is particularly relevant for biomass conversion technologies to advanced biofuels and biochemicals to enable the energy systems model to addresses cross-sectoral synergies and limit possible competition for biomass that may emerge as a result of sector-specific policy targets. Thirdly, disaggregation of biomass supply to different feedstocks and supply regions can be helpful to construct detailed biomass cost-supply curves and link them with specific biomass conversion pathways, thereby enabling complete supply chain assessments. Last but not least, technological development and technological learning are an important determinant of bioeconomy competitiveness. Chapter 4 addresses technological learning based on process efficiency and yield improvements over time in line with bottom-up engineering studies and expert judgments. Furthermore, Chapter 4 estimates future cost performance of technologies by taking scale effects into account. These are introduced as input data to the model and are varied in two different technology development scenarios. Therefore, scenario analysis on technology development can be used to assess a key determinant of the system performance.

Limitations: MARKAL-NL-UU is helpful to address mid-term bioeconomy developments in the energy system of the Netherlands. Nonetheless, several improvements can be incorporated within the model's structure to increase the understanding of other parameters relevant with the representation of bioeconomy in models. Firstly, existing oil refineries are not explicitly modelled. As a result, co-dependency of fuels (diesel, petrol, kerosene) with naphtha production is not accounted for. This can be useful to incorporate in order to assess whether the co-dependency can be a limiting factor to the supply of bioenergy and biochemicals. Secondly, the representation of end-of-life treatment options and cascading uses is only partly addressed. Incineration of organic waste, with or without energy recovery, is not linked with the output of biochemicals. Recycling is also not included. These options can increase the efficiency of biomass use in the economy. Thirdly, due to the large contribution of biomass heat in renewable energy supply, efficiency measures in the industry and the residential and commercial sector need also to be included explicitly. This also entails an improved representation of the heat sector. Other aspects that are highly relevant with the system dynamics is the global competition of the petrochemical industry, use of other fossil feedstocks (e.g. shale gas) and their impact on price dynamics and the production and export demand of the Netherlands. These can be partly resolved with model collaboration.

Macro-economic assessment
Improvements: CGE models are able to capture the complex and dynamic economic relationships of large-scale biomass deployment. Key shortcomings of CGE models are their high sectoral and technology aggregation and the limited representation of future structural changes (e.g. induced by bioeconomy developments that are not yet available). This dissertation addresses this by operationalising the CGE model MAGNET (Chapter 6) next to the energy systems model MARKAL-NL-UU (Chapter 4 and Chapter 5). A key improvement of MAGNET is the split of the single chemical sector to three sub-sectors: namely a conventional fossil-based chemical sector, a mixed bio-based and fossil-based chemical sector (that accounts for fossil and bio-based feedstock supply in the chemical industry), and a biochemicals sector (that accounts for replacements of chemical products through the deployment of biomass conversion technologies). To achieve this split, concessions are required in the selection of the most representative routes, as in MAGNET aggregation is still required. This is achieved by supplying to MAGNET the cost-structures of technologies that are found most cost-efficient in the medium term based on MARKAL-NL-UU. These technologies are also most representative in terms of pathways considered by biochemical deployment strategies. In addition, by splitting MAGNET's electricity and road transport fuel sectors to account for biomass co-firing (next to other renewable energy alternatives), and 1st and 2nd generation biofuel production, respectively, existing and advanced biomass conversion pathways are embedded in the structure of the macro-economic model. At the basis of this structure, lies the production and supply of 1st generation and lignocellulosic biomass feedstocks at a regional level, which are used for conversion by all bioeconomy sectors. An important improvement in the representation of biomass supply is the incorporation of a pelletisation sector. This enables assessments of solid biomass trade, next to liquid biofuels trade. Technology change is assessed by means of scenarios, similar to the assumptions of MARKAL-NL-UU. Finally, the parametrisation and inclusion of national and regional renewable energy targets, is essential as outcomes show that they influence deployment levels of biomass and other renewables in the short term.

Limitations: A significant step towards an increased understanding of the macro-economic impacts based on MAGNET outcomes lies in the conversion of key outputs related with biomass supply, trade, and biomass conversion (bioelectricity, biofuels, biochemicals), which are based on monetary flows, to physical quantities (e.g. PJ, Mt). Currently this is considered a limitation, as it does not allow the direct comparison of biomass supply costs, biomass consumption, bioenergy and biochemicals supply, with MARKAL-NL-UU. A consistent way of converting monetary to physical flows will also improve the soft-linking process between the models as, for example, effects of improvements in agriculture, or final energy demand, can be supplied to MARKAL-NL-UU. Moreover, consequential LCA studies can use such output to assess impacts of market-mediated effects. In addition, a limitation of MAGNET is that it does not incorporate explicitly CCS and BECCS technologies in the power and liquid fuel sectors. As a result, the macro-economic impacts of ambitious climate change mitigation scenarios cannot be assessed.

7.5 RECOMMENDATIONS FOR BIOECONOMY STAKEHOLDERS

Several sector-specific (e.g. energy, agriculture, forestry) or other overarching policies (e.g. on climate change mitigation, environment) may affect deployment levels and impacts of biomass for energy and chemicals. Bioeconomy stakeholders (e.g. policy makers, industry, investors, other market actors) need to have sufficient information on the implications and impacts of their decisions. The tools used and conclusions drawn in this dissertation can be useful to provide such information.

7.5.1 Policy makers

Under current policies, the Netherlands will face difficulty in meeting its own obligation and achieve the necessary GHG emission reduction by 2030 (Chapter 4 and Chapter 6). Stricter targets and stimulating policies on bioenergy and other renewable energy supply should be considered, especially in view of low fossil fuel prices (e.g. higher renewable energy share, tighter sector-specific obligations).

If bioeconomy sectors grow, policy makers will be increasingly faced with the need to ensure sustainable biomass use in the economy and to incentivise improvements in the value chain. Sustainable bioenergy pathways need to be identified and sustainability criteria established (e.g. minimum GHG emission savings) for solid biomass conversion to bioenergy (e.g. bioelectricity, biomass heat (EC, 2014c)) and for biomass conversion to biochemicals. LCA approaches, as referred to in the EU RED (EC, 2009b), can be useful in this direction as the production of biochemicals is faced with sustainability issues similar to biofuels (e.g. variation of environmental impact based on feedstock, production location; Chapter 3), albeit more complex (e.g. longer value chains, temporary bio-based carbon storage that depends on the product's lifetime, different technical performance than their petrochemical counterpart). With the expectation of large volumes of biomass being imported to conversion regions such as the Netherlands (Chapter 4 through Chapter 6), sustainable sourcing and development of biomass markets need to be ensured (e.g. through certification schemes), as minimum GHG savings criteria may not safeguard environmental impacts in supply regions. Policy implications of sustainability criteria on biochemicals across the supply chain can be addressed by consequential LCA.

The energy systems model used for projections under more technology-neutral scenarios in Chapter 5 (MARKAL-NL-UU) can be used to highlight the most promising pathways. Scenario results show large biomass heat supply in the industry. The heat sector is, however, overlooked by policies, with the exception of energy efficiency targets. Furthermore, Chapter 4 to Chapter 6 demonstrate that advanced technologies are a prerequisite for efficient biomass use in the economy to achieve cost-efficient deep emission reduction. Advances in novel biomass conversion technologies (e.g. gasification, pyrolysis) and cross-sectoral synergies between the transport and the electricity sector, and the transport and chemical sector enabled by biorefineries, are two areas where research, development and deployment (RD&D) subsidies are still needed. This will enable both early deployment and rapid growth of advanced technologies over time (Chapter 5). In this dissertation, a CO_2 pricing mechanism on emissions is used as the only instrument to stimulate emission mitigation, which may create an uneven level playing field for biochemicals. Policy makers will need to frame robust policies (beyond RD&D) that will allow for all biomass conversion pathways to compete on a level playing field and that address sectoral coupling enabled by biorefineries. For example a CO_2 tax on the fossil feedstock as opposed on CO_2 emissions could be applied instead.

Finally, Chapter 5 also shows path-dependencies that might be induced by policy decisions beyond bioeconomy. For example, decommissioning coal-power plants might reduce the need for deployment of CCS in the power sector in the medium term and associated challenges (e.g. proof of technology at commercial scale, public perception). At the same time requires significant efforts to increase on-shore and off-shore wind turbine capacity and addressing related issues (e.g. intermittency, grid reliance, connectivity, congestion, storage).

7.5.2 Industry and other market actors

As brand owners are currently the largest consumers of bulk biochemical applications (e.g. bioplastics), it is recommended to establish chains of custody that ensure proper practices throughout the supply chain in the short term. Close monitoring should focus on types of agrochemicals, harvesting practices, discharge and treatment of process waste, and process energy supply of conversion systems from low carbon sources (e.g. through certification schemes, on-site verification). Short-term improvements in biomass production (e.g. agricultural management) and industrial practices (e.g. utilisation of available renewable resources for energy supply, efficiency improvements on biomass conversion) are required in line with best practices and with respect to the regional context.

From a systems perspective, cross-sectoral collaboration across the value chain will maximise the industry's opportunities for growth and provide market access to suppliers. As this dissertation demonstrates, cross-sectoral synergies are created by multi-output processes (e.g. biorefineries) on the demand side, which suggest that industry needs to explore opportunities for collaboration or enter new markets by investing, demonstrating and upscaling new technologies. At the same time, these efforts may only become operational if low-cost high-quality biomass supply is ensured. Collaboration with upstream market actors of the value chain is necessary to ensure low-cost biomass supply (e.g. investments in the agricultural sector on crop developments, yield improvements, collection of residues), quality of supply (e.g. biomass pre-processing facilities) and continuity and security of feedstocks (e.g. on infrastructure, storage and logistics), thereby providing a mobilisable and tradable feedstock to demand markets.

The decade to come might be a decisive period that such opportunities can be valorised and address societal challenges. As Chapter 5 shows, should short-term investments occur in the petrochemical industry in the Netherlands, the competitiveness of biochemical production technologies will be reduced significantly, especially in the face of low fossil fuel prices. Such a direction may lock the supply in the petrochemical industry and impede the development of the bioeconomy. In an uncertain investing environment, industry needs to actively engage in strategic decisions that might not achieve short-term benefits but are expected to do so over time.

7.6 DIRECTIONS FOR FURTHER RESEARCH

The deployment of improved methods and tools for bioeconomy assessments presented in this dissertation reveals important shortcomings that future research can resolve. The timeframe (current to 2030) and the geographic scale (the Netherlands) applied in this dissertation had as a key aim to assess environmental impacts of established bioenergy and biochemical pathways and the intricacies of incorporating emerging bioeconomy sectors into MARKAL-NL-UU and MAGNET. While this is achieved, energy systems and macro-economic studies with longer temporal and larger regional scope are required in order to assess the role and contribution of bioeconomy in deep emission reduction.

Recommendations on product-specific LCA studies
Product-specific and supply chain-specific studies based on attributional LCA are valuable to assess environmental impacts of established supply chains within specific system boundaries. However, the assessments are highly reliant on the quality of the data used. Data quality and validity, whilst an overarching recommendation (e.g. completeness of inventories and reliable and verifiable process information based on industrial production data), is particularly relevant for LCAs that aim at addressing impacts of novel biochemical production chains with precision. Increase in data transparency and sensitivity analysis should accompany all attributional LCAs, especially on generic background inputs such as pesticides that are found to influence human health and ecosystem quality. To reduce uncertainty associated with methodologies on the interaction of the technosphere with the natural system, insights from environmental impact assessments based on the local environment that also monitor local levels of pollutants are recommended. Last but not least, modelling and sub-division of product-systems is recommended to the extent possible, as it can be an underlying reason for discrepancy of results. Regarding consequential LCA, model collaboration is required between bottom-up LCA and CGE models, which is discussed below.

Recommendations on energy systems and economic models
In their structure, models should include the petrochemical industry and its flows, as well as biochemical production technologies at different levels (e.g. feedstock, basic chemicals, intermediate chemicals, final products). In addition, explicit modelling of multi-output sys-

tems (e.g. biorefineries) should address sector-coupling. In view of emission mitigation, other industry sectors relevant for bioeconomy developments need to be incorporated (e.g. paper and pulp, steel, construction). A longer timeframe also requires extending the technology portfolio of biochemicals (e.g. to specialty chemicals, algae-based chemicals, lignin-based chemicals). Specialty chemicals are produced at low volumes and are thus neither expected to substitute significant amount of fossil fuels nor contribute directly to GHG emission reduction. However, they have highly functionalised applications and typically have a high market price. Therefore, they may stimulate the contribution of biomass in the economy even further than indicated in Chapter 6. Due to the high relevance of the non-energy and the heat sector, the technological scope of improved assessments also needs to include demand-technologies and measures (e.g. energy efficiency) and combine them with energy systems assessments. Finally, due to the indications for growth in biochemicals' production (Chapter 4 through Chapter 6), it becomes relevant to assess the implications of policy instruments that support biomass for bioenergy and whether they create an uncompetitive environment for biochemicals. A multi-sectoral CO_2 emission tax scheme, or taxation on the fossil carbon content of feedstocks, as opposed on emissions, could be potentially applied.

Recommendations on model collaboration

With these improvements in place, a more consistent and harmonised framework can be developed in which LCA, energy systems and CGE models are operationalised next to sector-specific models (e.g. power sector). Given the projected potential supply of bioelectricity from biorefineries but also the large supply of intermittent wind electricity, their operational flexibility (e.g. on an hourly basis) requires to be assessed by using a dispatch model. This has been demonstrated in Brouwer et al. (2015). A similar soft-linking approach is recommended, one that takes into account findings based on this dissertation such as the large contribution of 2nd generation biofuels to renewable energy targets and the supply of bioelectricity from biorefineries. Furthermore, developing and applying a consistent method to convert the monetary flows of MAGNET to physical units can improve the soft-linking process between a macro-economic and an energy systems model. This is particularly relevant when longer-term assessments are conducted in which economic linkages within and among regions may lead to production capacity shifts (e.g. relevant for the chemical industry), thereby affecting the regional production demand, energy demand and biomass supply. Finally, the product-based and supply chain-specific assessments based on the LCA method can benefit significantly from the outputs of energy systems and macro-economic CGE models. Consequential LCAs can benefit by incorporating mid-term and long-term energy supply mix as estimated by energy systems models for a specific region. Similarly, CGE models can provide insights in biomass supply, land use and land use change emissions. This can increase the precision and consistency of consequential LCA results, which are currently faced with wide disparity. This way comparative LCA studies can be performed based on different supply chains in line with outcomes derived by improved system-wide and economy-wide models.

SAMENVATTING EN CONCLUSIES

Onderzoekscontext

Klimaatverandering, grotendeels een gevolg van stijgende concentraties van antropogene broeikasgassen (BKG), is voor een groot deel het gevolg van de afhankelijkheid van de samenleving van fossiele brandstoffen. Zonder fundamentele veranderingen in de vraag en aanbod van energie die leiden tot vergaande verminderingen in de emissie van broeikasgassen (ongeveer 40 tot 70% in 2050, vergeleken met 2010 IPCC (2014b)), zal de mensheid en de planeet bedreigd worden door onomkeerbare gevolgen. Het realiseren van zulke doelstellingen vergt een vervanging van fossiele brandstoffen door hernieuwbare energie, een verhoging van grondstof-efficiëntie en de ontplooiing van CO_2-afvang en -opslag (afgekort met CCS van carbon capture and storage). Biomassa is momenteel de belangrijkste bron van hernieuwbare energie, met een aandeel van ongeveer 10% in de globale primaire energievoorziening, ongeveer 60 EJ (REN21, 2016). Het aandeel biomassa voor bio-energie (bijv. elektriciteit, warmte en biobrandstoffen) kan oplopen tot 300 EJ in 2050 als we klimaatdoelstellingen nastreven, aangezien het voor een groot deel fossiele grondstoffen kan vervangen, de uitstoot van broeikasgassen kan verminderen en zelfs kan leiden tot negatieve emissies (in combinatie met CCS (i.e. bio-energie gecombineerd met Carbon Capture and Storage; BECCS).

Op dit moment wordt de inzet van biomassa voor energie gedomineerd door traditioneel gebruik in inefficiënte verbranding van brandhout voor warmte, koken en verlichting. Een transitie van een op fossiele grondstoffen gebaseerde economie (fossiele economie) naar een bio-economie vraagt ook om een transitie van traditionele grondstoffen naar een moderne, efficiënte voorziening van bio-energie en materialen. Biomassa wordt steeds vaker ingezet voor de productie van bio-chemicaliën, en biedt daardoor een duurzaam alternatief voor de chemische industrie (ongeveer 0.6 EJ; Saygin et al. (2014)): 50% van de globale plastics productie is wordt reeds geproduceerd van op bio-ethanol gebaseerde bulk chemicaliën (i.e. ethyleen en afgeleiden), (IfBB, 2015). Tegen 2050 zou biomassa tot wel 30% van de niet-energie gerelateerde vraag naar fossiele grondstoffen in de chemische industrie kunnen dekken (i.e. het aandeel dat niet als brandstof wordt gebruikt, maar als grondstof) (Daioglou et al., 2014).

De productie van bioenergie en biochemicaliën komt echter niet zonder uitdagingen. De besparing van broeikasgasemissies en kosten van eerste- en tweede generatie bio-energie en biochemicaliën zijn zeer afhankelijk van de eigenschappen van het productiesysteem. Belangrijke elementen zijn onder andere het type grondstof, de locatie, landbouwpraktijken, geïnduceerde veranderingen in direct- en indirect landgebruik (dLUC van direct Land Use Change, en iLUC van indirect Land Use Change) en gerelateerde broeikasgasemissies en de prestaties van de conversie technologieën.

Bovendien zijn de hoeveelheden bio-energie en biochemicaliën die geproduceerd zullen worden hoogst onzeker. Het toekomstige potentieel en regionale verdeling zullen afhangen van ontwikkelingen in de sectoren die biomassa produceren (bijvoorbeeld landbouw, bosbouw). Wanneer de kosten van conversietechnologiën afnemen, kunnen regio's verder investeren in lokale biomassa-conversie, waardoor de vraag verder toe zal nemen. Het biomassa aanbod is afhankelijk van de voortgang van de internationale handel in biomassa. Dergelijke ontwikkelingen kunnen worden versneld of vertraagd, afhankelijk van het ambitieniveau van beleid om broeikasgasemissies te verminderen en toekomstige prijzen van fossiele brandstoffen.

Het is niet eenvoudig om de invloed van deze factoren te beoordelen, omdat de inzet van biomassa in de economie ook afhankelijk is van dynamische factoren die bepaald worden door interacties tussen biomassa-voorziening en technologieën in het energiesysteem (inclusief de chemische industrie). Wanneer bepaalde sectoren in de bio-economie groeien, kunnen spanningen ontstaan door concurrentie voor grondstoffen. Tegelijkertijd kunnen sectoroverschrijdende synergiën ontstaan wanneer multi-output conversietechnologieën (i.n. bioraffinaderijen) worden ingezet. Daarbij concurreert bioenergie met andere duurzame energie- en koolstofarme technologieën (CCS, BECCS) om duurzame energie te produceren en broeikasgasemissies te verlagen. Er zijn echter onzekerheden over de technologische gereedheid en ontwikkelsnelheid bioenergie, andere duurzame energiebronnen en emissie reducerende technologieën. Deze onzekerheden bepalen uiteindelijk de economische haalbaarheid van de transitie van het fossiele energiesysteem naar een duurzamer alternatief.

Ontwikkelingen in de bio-economie worden meestal beoordeeld door verschillende methoden en instrumenten, die fundamenteel verschillen in de activiteit die ze beoordelen, de geografische precisie, tijdsspanne en interacties met menselijke en natuurlijke systemen (Wicke et al., 2015). Iedere methode is bedoeld om specifieke vragen te beantwoorden, en tot heden kan geen methode de bio-economie in zijn geheel beoordelen. Dit houdt in dat de beschikbare toolkit sterke inzichten kan leveren op specifieke punten, maar beperkt is in het representeren van de complexiteit van de bio-economie en haar potentiële effecten. De belangrijkste afwegingen tussen de verschillende bottom-up modellen en analyses (bijv. Life Cycle Assessment, LCA), Partial Equilibrium Models (PEM), Computable General Equilibrium (CGE) modellen) en geïntegreerde evaluatiemodellen die vaak worden gebruikt gaan over de hoeveelheid detail waarmee ze werken en de volledigheid waarmee ze dynamische interacties tussen menselijke en natuurlijke systemen beschrijven (Wicke et al., 2015).

Tegen deze achtergrond zijn belangrijke kenniskloven (a) de milieueffecten van de toekomstige bio-producten vergeleken met bestaande productieketens, (b) de potentiële rol en bijdrage van biomassa binnen een nationale context in de transitie naar een duurzamer energiesysteem, en (c) de macro-economische effecten van de ontwikkeling van de bio-economie.

Doel en onderzoeksvragen

Dit proefschrift richt zich op een aantal van de belangrijkste kennishiaten in verband met de huidige en middellange termijn ontwikkelingen in producten en sectoren van de opkomende bio-economie. Het biedt verschillende inzichten in de prestaties van bioenergie en biochemische productiesystemen en de synergieën en de conflicten tussen de bio-economie sectoren, door het gebruik van bottom-up en top-down modellen die ontwikkeld zijn voor specifieke productieketens en nationale context. Om dit doel te bereiken worden de milieuprestaties van de gevestigde biobrandstoffen en biochemicaliën productieketens beoordeeld, door de methodologische implicaties van LCA te onderzoeken, en globale productieketens met fundamentele processtappen te bestuderen die in ontwikkelingslanden en opkomende economieën plaatsvinden (India en Brazilië). Bovendien verbetert het de weergave van bio-energie en biochemische productieketens in energie-gerelateerde macro-economisch CGE-modellen, met specifieke aandacht voor aanbod van biomassa en geavanceerde conversietechnologieën. De focus ligt op het energiesysteem van Nederland om inzicht te verkrijgen in de structuur en de belangrijkste parameters die essentieel zijn voor uitbreiding van de modellen. Het land bezit een efficiënte agrarische sector, heeft een grote petrochemische industrie, en heeft een competitieve logistieke infrastructuur waardoor het een grootschalige bio-economie kan dragen. Door het verbeteren van inzichten in de milieu- en macro-economische effecten die de een bio-economie kan hebben, probeert dit proefschrift aan te tonen wat de rol en het gevolg zou kunnen zijn op de middellange termijn van een grootschalige transitie naar een bio-economie die tevens een oplossing biedt voor de maatschappelijke uitdagingen op lange termijn. Daartoe worden de volgende onderzoeksvragen behandeld:

I. Wat zijn huidige en toekomstige milieueffecten van toepassingen van biobrandstoffen en biochemicalieën?
II. Wat is de potentiële omvang en de bijdrage van de bio-economie in een nationaal energiesysteem aan klimaatdoelstellingen op de middellange termijn?
III. Welke mogelijke economische gevolgen hebben ontwikkelingen in de bio-economie op de middellange termijn op nationaal niveau?
IV. Hoe kunnen instrumenten en methoden voor het verkennen van de huidige en middellange termijn ontwikkelingen in de bio-economie worden verbeterd?

Dit proefschrift richt zich op de onderzoeksvragen I tot en met IV in hoofdstuk 2 tot en met hoofdstuk 6 (zie tabel A). Hoofdstuk 2 gaat in op de milieueffecten van suikerriet productie van ethanol in Brazilië en India. Het geeft inzicht in de gevolgen voor het milieu, afhankelijk van methodologische keuzes en systemen die worden gebruikt. Op basis van deze bevindingen, breidt Hoofdstuk 3 de analyse uit naar bioplastics gemaakt van suikerriet ethanol en vergelijkt hun milieuprestaties met Europese petrochemische kunststoffen. Het richt zich ook op technologische verbeteringen en andere ingrepen in de hele productieketen die milieueffecten van de productie van bioplastics in de nabije toekomst kunnen verminderen.

Hoofdstuk 4 verbetert de weergave van het aanbod van biomassa en de productie van geavanceerde biobrandstoffen en biochemicaliën in MARKAL-NL-UU, een lineair kosten-minimalisatie model van het Nederlandse energiesysteem. Het model wordt toegepast uitgaande van het huidige beleid doelstellingen en technologieontwikkeling scenario-varianten tot 2030. Hoofdstuk 5 onderzoekt verschillende scenario's aangaande de ontwikkeling van technologieën, de kosten en aanbod van biomassa, de prijzen van fossiele brandstoffen en de sectorspecifieke doelen om hun broeikasgasemissies op nationaal niveau op de middellange termijn terug te dringen. Hoofdstuk 6 presenteert een raamwerk dat gebruik maakt van MARKAL-NL-UU om de weergave van de bio-economie in een CGE-model

(MAGNET) te verbeteren. Het uitgebreide MAGNET-model wordt gebruikt om potentiële macro-economische effecten als gevolg van de ontwikkeling van een grootschalige bio-economie tot 2030 in Nederland te laten zien.

Tabel A Overzicht van thema's in dit proefschrift in relatie met de onderzoeksvraag waar ze op ingaan

Hoofdstuk	Thema	Methode	Regionale scope	Systeem scope	Tijd scope	Onderzoeksvraag			
						I	II	III	IV
2	LCA van suikerriet ethanol productie in India vergeleken met Brazilië	LCA	India, Brazilië	Product-specifiek	Huidig	X			X
3	LCIA van bioplastics van suikerriet ethanol	LCA	India, Brazilië, Europa	Product-specific	Huidig, korte termijn	X			X
4	Opkomende sectoren in de bio-economie in energiesysteem modellen – Geïntegreerde analyse van elektriciteit, warmte, wegtransport, luchtvaart en chemicaliën: een case studie voor Nederland	Kost-optimalisatie	Nederland	Energie-systeem	Middellange termijn		X		X
5	De rol van bio-energie en biochemicaliën van de CO_2-beperking door middel van het energiesysteem - Een scenario-analyse voor Nederland	Kost-optimalisatie	Nederland	Energie-systeem	Middellange termijn	X	X	X	X
6	Macro-economische impact van bio-energie en biochemicaliën - De invoering van geavanceerde bio-economie sectoren in een economisch model met een case studie voor Nederland	CGE modelling	Globaal, met focus op Nederland	Totale economie	Middellange termijn			X	X

Samenvatting van resultaten

Hoofdstuk 2 presenteert een cradle-to-gate LCA van de productie van ethanol uit suikerriet melasse in Uttar Pradesh, India (hierna te noemen India) en vergelijkt het met de productie van ethanol van suikerriet sap in Zuid-Centraal Brazilië (hierna te noemen Brazilië). Het verbruik van ethanol in het wegvervoer in India groeit geleidelijk en wordt ook gebruikt voor de productie van geavanceerde biochemicaliën. Het multi-functionaliteitsprobleem (of toekenningsprobleem) doet zich zowel in de Indiase als Braziliaanse ethanol productiesystemen voor. Beide systemen worden beoordeeld op basis van varianten van de systeem-uitbreiding en economische allocatie.

Ondanks de hogere uitstoot van broeikasgassen van Indiase suikerriet, toont ethanol een lagere uitstoot van broeikasgassen (0.09-0.3 $kgCO_{2eq}/kg_{etOH}$) in vergelijking met de Braziliaanse ethanol (0.5-0.6 $kgCO_{2eq}/kg_{etOH}$) bij systeem-uitbreiding. Dit resultaat wordt beïnvloed door twee factoren. Ten eerste, de credits voor de elektriciteitsproductie zijn hoger in India (ongeveer 0,3-0,6 $kgCO_{2eq}/kg_{etOH}$) dan de credits gegenereerd door elektriciteit en bagasse in Brazilië (ongeveer 0,04-0,2 $kgCO_{2eq}/kg_{etOH}$), doordat de huidige brandstofmix in India een hogere CO_2-intensiteit heeft. Ten tweede, de effecten van het Indiase product-systeem worden gematigd door het feit dat uitsluitend melasse (een bijproduct van suikerriet productie) wordt gebruikt om ethanol te produceren. Wanneer economische allocatie in alle stappen van de productie wordt toegepast, zijn de resultaten voor India en Brazilië vergelijkbaar (ongeveer 0,6 $kgCO_{2eq}/kg_{etOH}$).

Gevolgen voor de menselijke gezondheid en de kwaliteit van ecosystemen zijn lager in het Indiase productiesysteem, maar blijven over het algemeen vergelijkbaar. In beide gevallen zijn deze effecten in de eerste plaats geassocieerd met de teelt van suikerriet. In India heeft dit vooral te maken met de toediening van meststoffen en pesticiden. In Brazilië worden de effecten vooral versterkt doordat men het land voor de oogst gereed maakt door middel van verbranding. Het waterverbruik en de daarmee samenhangende gevolgen voor de menselijke gezondheid en de kwaliteit van ecosystemen zijn alleen relevant voor het Indiase productiesysteem, omdat er aanwijzingen zijn voor een daling van het regionale grondwaterpeil.

Belangrijke bijdragen aan het resultaat worden geleverd door warmte-kracht koppeling (WKK) bij de molens en distilleerderijen, het gebruik van co-producten, zoals vinasse en bagasse voor energiedoeleinden, en aannames over specifieke toepassing van pesticiden, oogstpraktijken en de behandeling van afvalstromen. Het Indiase productiesysteem draagt risico's. Als er bijvoorbeeld een distilleerderij haar afvalwater niet behandelt en er anaerobe omstandigheden heersen, kan het effect op de kwaliteit van ecosystemen aanzienlijk zijn en de uitstoot van broeikasgassen kan variëren van 2.6 tot 3.1 $kgCO_{2eq}/kg_{etOH}$.

Wanneer men indirecte emissies (iLUC) meeneemt, kunnen de emissies van Braziliaans ethanol met 0.08-1.2 $kgCO_{2eq}/kg_{etOH}$ toenemen. Deze brede range is gebaseerd op de variatie van LUC emissiefactoren genoemd in de literatuur. In India wordt aangenomen dat molasse niet uit diervoederstromen worden ontnomen, waardoor er geen vervangingseffecten plaatsvinden. Niettemin, als de vraag naar ethanol toeneemt, bijvoorbeeld door een toename van de vraag naar biobrandstoffen of naar biochemicaliën, dan moet de vraag door andere gewassen elders worden gecompenseerd. In dat geval zal melasse een effect gelijk aan dat van voedergewassen met zich dragen, en daardoor kan worden verwacht dat Indiase ethanol hogere effecten dragen dan in dit proefschrift wordt gepresenteerd.

Hoofdstuk 3 past LCA toe op een methodologisch consistente wijze met hoofdstuk 2 om de milieueffecten van twee van de meest geproduceerde bioplastics te beoordelen. Dit zijn volledig bio-based polyethyleen (bio-PE) uit Braziliaanse suikerriet ethanol, en gedeeltelijk bio-based polyethyleentereftalaat (bio-PET) uit Indiase en Braziliaanse suikerriet ethanol. De bio-based component van bio-PET (dat wil zeggen biobased mono glycol; bio-MEG) wordt geproduceerd in Uttar Pradesh, India en is verder gesynthetiseerd tot bio-PET in Europa. Beide bioplastics worden vergeleken met de huidige petrochemische PE en PET, geproduceerd in Europa.

Bio-PE productie leidt tot de uitstoot van ongeveer -0,75 $kgCO_{2eq}$/kg PE aan broeikasgassen, ongeveer 140% lager dan petrochemische PE. Deze berekening houdt rekening met het grote bio-based koolstofgehalte van het polymeer, dat wordt afgetrokken van de uiteindelijke emissies. Verschillende allocatie-methoden hebben geen significant verschil op de resultaten. Dit heeft te maken met de bijdrage van andere stappen (katalytische dehydratatie, polymerisatie, transport). Het onderbouwt de robuustheid van de resultaten met betrekking tot bio-PE. Alleen de combinatie van systeem-expansie en (efficiënte of inefficiënte) elektriciteit co-generatie bij suikermolens en distilleerderijen leidt tot een ± 50% variatie in de resultaten als gevolg van kredieten van de Braziliaanse ethanol productie. Besparingen op de niet-hernieuwbare energiebronnen zijn ongeveer 65% en variatie door systeemrendement in combinatie met systeemuitbreiding ongeveer ±25%.

Bio-PET toont broeikasgasemissies (1.9-2.4 $kgCO_{2eq}/kg_{PET}$) die vergelijkbaar zijn met petrochemische PET (2.15 $kgCO_{2eq}/ kg_{PET}$). Ten eerste is het bio-based gehalte van bio-PET (i.e. bio-MEG) ten opzichte van PET slechts ongeveer 28% op massabasis en is niet voldoende om te compenseren voor de bijdrage van andere stappen van de productieketen (tereftaalzuur productie, verestering, transport). Transport emissies dragen meer bij aan de productie van bio-PET van Braziliaanse ethanol, aangezien ethanol eerst moet worden vervoerd naar India om te worden omgezet naar bio-monomeer. Hierdoor kan bio-PET van Braziliaanse ethanol tot hogere emissies leiden dan petrochemische productie van PET in Europa. Broeikasgasemissies van beide productiesystemen zijn relatief hoog omdat de productie van het bio-deel afhankelijk is van een energievoorzieningssysteem met hoge CO_2-intensiteit (onder de referentieomstandigheden is procesenergie voor bio-MEG gebaseerd op stoom en elektriciteit van het net wat geproduceerd is door kolen).

Voor het Indiase productiesysteem worden de resultaten beïnvloed door de aanpak van het multifunctionaliteitprobleem. De belangrijkste reden is dat het elektriciteitsverbruik van het systeem hoger is dan wat het WKK-systeem kan leveren aan elektriciteit en warmte. Daarom bepaalt de veronderstelde CO_2-intensiteit van de elektriciteit van het net uiteindelijk de uitstoot van broeikasgassen bij gebruik van de twee systeem-uitbreiding benaderingen. Broeikasgasemissiebesparingen van bio-PET kunnen oplopen tot 30% ten opzichte van petrochemische PET wanneer de beste beschikbare technologie voor bio-MEG productie wordt aangenomen met stoom op basis van aardgas in plaats van steenkool. Cogeneratie van elektriciteit en warmte voor proces-energie kan ook de milieueffecten van bio-MEG verminderen. Het is echter aangetoond dat regionale beleidsinstrumenten, zoals feed-in-tarieven die momenteel bestaan in India, de producent kunnen aansporen om elektriciteit te leveren aan het net.

Uit de analyse blijkt dat er afwegingen zijn tussen de uitstoot van broeikasgassen en andere milieu-impact categorieën. De productie van bio-PE en bio-PET schaadt de menselijke gezondheid en het ecosysteem in veel grotere mate (twee ordes van grootte) in vergelijking met hun petrochemische tegenhangers. Terwijl de besparing op de uitstoot van broeikasgassen globaal zijn, spelen de gevolgen voor de menselijke gezondheid en de kwaliteit van ecosystemen op regionaal niveau. De effecten hebben vooral betrekking op de productie en het oogsten van suikerriet zoals besproken in hoofdstuk 2. Verbeteringen op deze aspecten verminderen de effecten gedeeltelijk, maar ze zijn nog steeds hoger dan bij petrochemische plastics.

Hoofdstuk 4 presenteert een bottom-up energiesysteem model (MARKAL-NL-UU), dat is uitgebreid met fossiele- en bio-chemicaliën en met hernieuwbare kerosine (RJF, van Renewable Jet Fuels) om de inzet van biomassa conversietechnologieën in Nederland te verkennen tot 2030. Het model verschaft gedetailleerde koststructuren en ontwikkelingen op de middellange termijn voor het fossiele energiesysteem en mitigatie opties. Het bevat gedetailleerde data over kosten en aanbod van biomassa, duurzame energie technologieën en CCS. Het model is voorzien van multi-output processen, zoals bioraffinaderijen, om sectoroverschrijdende synergieën te kunnen analyseren. Verschillende technologie-ontwikkelingsscenario's worden gebruikt om de onzekerheid in de technologische vooruitgang mee te nemen. Verbeteringen van de energie-efficiëntie in de residentiële en commerciële sectoren worden impliciet meegenomen, in lijn met de prognose voor warmtevraag bij huidig beleid. De volgende beleidsdoelen voor 2030 zijn aangenomen: een bijmengingsverplichting van biobrandstof van 10%, een aandeel hernieuwbare energie van 16% van het eindverbruik van energie, en ook wordt gematigd klimaatbeleid aangenomen, met een emissie belasting van 25 € per ton CO_2 in 2030.

Trage technologische vooruitgang leidt tot het gebruik van biomassa hoofdzakelijk voor verwarming (105 PJ_{th}; 12% van de totale warmtelevering), gehydrogeneerde biobrandstoffen gebaseerd op 1^e generatie oliën (33 PJ, 5% van het totale aanbod van transportbrandstof) en biochemicaliën op basis van fermentatie van 1^e generatie grondstoffen (26 PJ, 5% van het totale chemische grondstofgebruik). Hernieuwbare elektriciteit wordt voornamelijk geleverd door windturbines en zonne-energie (140 PJ, 30% van de totale elektriciteitsvoorziening), terwijl bio-elektriciteit beperkt blijft (9 PJ, 2% van de totale elektriciteitsvoorziening) tot de productie uit organisch afval. Bij versnelde technologieontwikkelingen komt de productie van 2^e generatie biobrandstoffen uit vaste biomassa op gang (61 PJ, 10% van het totale aanbod transportbrandstof); het overige deel komt van 1^e generatie biobrandstoffen (9 PJ; 1,5% van het totale vervoer brandstoftoevoer). Het koolstofarm maken van de transportsector rust daarom in beide technologie-ontwikkelings scenario´s en onder het aangenomen beleid, voornamelijk op biomassa en niet op andere opties, zoals elektrische voertuigen. RJF (Renewable jet-fuel) komt uit op 13 PJ (ongeveer 7% van de totale jet brandstoftoevoer). Bovendien worden grote hoeveelheden biochemicaliën geproduceerd (54 PJ, 10% van de totale vraag). Dat zijn hoofdzakelijk chemicaliën op basis van 1e generatie suikers en lignocellulose grondstoffen in ongeveer gelijke aandelen; het overige deel wordt geproduceerd van geïmporteerde ethanol. Het aanbod van bio-elektriciteit is verdubbeld ten opzichte van het lage technologie-ontwikkelingsscenario (18 PJ, 4% van de totale elektriciteitsvoorziening). Het extra vermogen wordt geleverd door bioraffinaderijen. Duurzame elektriciteit uit windturbines en zonnepanelen blijft op hetzelfde niveau (ongeveer 140 PJ). De hoeveelheid benodigde biomassa varieert van 230 tot 300 PJ in 2030, voornamelijk uit ingevoerde bronnen. Onder het bestaande beleid bereiken CO_2-emissies het niveau van 1990 (dat wil zeggen 140-145 $MtCO_2$ in vergelijking met 160 $MtCO_2$ in 2010), maar halen niet 40% doelstelling in 2030 (dat wil zeggen 86 $MtCO_2$), zoals beoogd door de Europese Unie.

Hoofdstuk 5 verkent laagste-kosten paden naar 2030 voor een aantal scenario's toegepast op het Nederlandse energiesysteem, met behulp van een lineair kosten-minimalisatie model ontwikkeld in hoofdstuk 4. In dit hoofdstuk worden ambitieuze scenario’s aangenomen op basis van een CO_2 belasting van 69 €/tCO_2 in 2030. Technologie-specifieke doelstellingen worden verondersteld tot 2020, maar niet daarna, waardoor technologie-neutrale concurrentie mogelijk wordt tussen fossiele productie en conversie van biomassa tot bio-energie en biochemicaliën, andere hernieuwbare technologieën en andere mitigatie opties (CCS, BECCS).

De aard en omvang van de inzet van biomassa wordt sterk beïnvloed door technologische ontwikkeling, biomassa kostencurve, prijzen van fossiele brandstoffen en de ambitie om klimaatverandering tegen te gaan. Finale productie uit hernieuwbare bronnen ligt tussen 460-510 PJ in 2030 en verschilt niet significant tussen de varianten van de technologische ontwikkelingsscenario´s. Het aandeel hernieuwbare energie komt uit op 23-24% van het eindverbruik van energie (met uitzondering van chemicaliën) of op een aandeel van 18-20% van het totale eindverbruik van energie (inclusief chemische stoffen). Meer dan twee derde (73-79%) van de productie van hernieuwbare energie wordt toegeschreven aan biomassa. Elektriciteit uit biomassa blijft laag (20-50 PJ; vooral uit bioraffinaderijen, meestook en huisvuilverbranding), omdat windenergie de belangrijkste leverancier van duurzame elektriciteit is. Biochemicalieën worden geproduceerd als gevolg van de buitengebruikstelling van stoomkraker capaciteit (20-50 PJ; 5-10% van de productie van de sector), zelfs onder de baseline aannames, waarbij geen CO_2-belasting wordt aangenomen na 2020. In het high-tech ontwikkeling scenario verdubbelt de productie van biochemicaliën bijna (ongeveer 100 PJ; 17%) ten opzichte van de baseline. Dit is een gevolg van multi-outputtechnologieën die zowel chemicaliën als transportbrandstoffen produceren, waarbij de laatste wordt beïnvloed door de CO_2-belasting. De elektriciteitssector is het meest gevoelig voor aannames over de prijs van fossiele brandstoffen. Hoge brandstofprijzen kunnen leiden tot een toename met een factor 2,5 van elektriciteit uit hernieuwbare energiebronnen ten opzichte van referentiescenario's. Een combinatie van scenario's kan leiden tot ±50% variatie van de definitieve productie uit hernieuwbare energiebronnen ten opzichte van de referentie. Het high-tech ontwikkelingsscenario in combinatie met hoge brandstofprijzen leidt tot een hernieuwbaar energieaanbod van 745 PJ. Lage technologieontwikkeling in combinatie met lage fossiele brandstofprijzen leidt tot 230 PJ, wat vergelijkbaar is met de baseline scenario's zonder CO_2-belasting. De bijdrage van biobrandstoffen in transportbrandstoffen komt boven 60% uit onder hoge technologische ontwikkeling in combinatie met hoge fossiele brandstofprijzen. Dit gebeurt als gevolg van de toegenomen aanbod van biobrandstoffen (20% hoger dan in het referentiescenario) en als gevolg van een verminderde vraag naar brandstof door de sector (ongeveer één derde oftewel een 130 PJ daling ten opzichte van het referentiescenario, omdat efficiëntere voertuigen worden ingezet). Het aandeel de 1^e generatie biobrandstoffen in de totale vraag naar brandstof voor wegvervoer en de luchtvaart is respectievelijk 1-13% en 1-12% in lage en hoge technologieontwikkeling scenario-varianten. Het aandeel 2^e generatie biobrandstoffen in het wegvervoer en de luchtvaart is 0-3% en 10-29% in respectievelijk lage en hoge technologieontwikkeling scenario-varianten. Lage technologieontwikkeling in combinatie met een lage fossiele brandstofprijs kan de bijdrage van biobrandstoffen in de sector beperken tot slechts 1%.

In alle scenario varianten varieert het verbruik van biomassa tussen 180 en 760 PJ. Een hoge technologieontwikkeling leidt tot extra verbruik van biomassa van 100-270 PJ in vergelijking met lage technologische ontwikkeling scenario´s. Biomassa consumptie is zeer gevoelig voor de aangenomen ontwikkeling van fossiele brandstofprijzen tot 2030. Lage technologieontwikkeling scenario's verbruiken een maximum van 560 PJ aan biomassa onder de meest gunstige scenario´s veroorzaakt door hoge fossiele brandstofprijzen, die vergelijkbaar zijn met het referentie consumptieniveau van het high-tech ontwikkelingsscenario. De CO_2-belasting leidt tot een emissiereductie in de orde van 35-43% voor lage en hoge technologieontwikkeling scenario's, vergeleken met verwachte emissies in 2030. In absolute termen is de additionele emissiereductie als gevolg van hoge technologische ontwikkeling 15 $MtCO_2$ in vergelijking met lage technologische ontwikkeling. Zelfs onder het high-tech ontwikkelingsscenario is nog eens 15 $MtCO_2$ reductie nodig om de emissiereductiedoelstelling van 40% t.o.v. 1990 te halen. De beschikbaarheid van goedkope biomassa vermindert de emissies met een extra 5 $MtCO_2$, alleen onder het high-tech ontwikkeling scenario, wat gedeeltelijk de kloof dicht naar de doelstelling. De resultaten geven aan dat in het kader van hoge fossiele brandstofprijzen, de emissiereductiedoelstelling van 40% ten opzichte van 1990 wordt bereikt in het high-tech ontwikkelingsscenario. In het low-tech ontwikkelingsscenario is de CO_2-mitigatie echter 12 $MtCO_2$ onder de doelstelling. De bijdrage van CCS en BECCS tot emissiereductie is significant (42-60%, ten opzichte van de baseline). De rest van emissiereductie wordt voornamelijk bereikt door middel van biomassa (20-40 $MtCO_2$), omdat - met uitzondering van het scenario met hoge fossiele brandstofprijzen - de capaciteit van windenergie en andere hernieuwbare energiebronnen niet significant toenemen na 2020 ten opzichte van de baseline.

Met referentie fossiele brandstofprijzen leidt een lagere vraag naar petrochemische producten naar een vermindering van 16-37% in de productie in low-tech en high-tech ontwikkelingsscenario's, respectievelijk, in vergelijking met hun referentie. Wanneer lage fossiele brandstofprijzen worden aangenomen ontstaat er een niet-competitieve omgeving voor biochemicaliën omdat in het low-tech ontwikkeling scenario productie bijna geheel wegvalt, terwijl de output met 85% afneemt in het high-tech ontwikkeling scenario. De ontmanteling van kolencentrales na 2020 zorgt niet voor een CO_2-emissie daling, omdat offshore windturbines concurreren met CCS in kolencentrales.

Hoofdstuk 6 presenteert een raamwerk dat gebruikt maakt van het energiesysteem model ontwikkeld in hoofdstuk 4 (MARKAL-NL-UU) om de weergave van de bio-economie sectoren in een globale, multi-sector en multi-regio CGE-model (MAGNET) te verbeteren. Het harmoniseert kostenstructuren van technologieën, aannames over technologieontwikkeling en openheid van de handel, en exogene parameters om MAGNET uit te breiden door een selectie te maken van de belangrijkste conversietechnologieën voor de productie van geavanceerde biobrandstoffen en biochemicaliën, de voorziening van lignocellulose biomassa en verhandelbare pellets. Het raamwerk wordt toegepast in een verkenning van de macro-economische ontwikkelingen in de Nederlandse bio-economie tot 2030.

Het aandeel biobrandstof in beide modellen zijn aanzienlijk hoger in een high-tech ontwikkeling scenario doordat voornamelijk 2^{e} generatie biobrandstoffen concurrerend worden. In combinatie met internationale handel van biomassa, komt het aandeel biobrandstoffen op 13% in MAGNET en 12% in MARKAL-NL-UU, voornamelijk op basis van tweede generatie biobrandstoffen. Het aandeel hernieuwbare elektriciteit is ongeveer 35%, dat vooral door windmolens wordt geleverd. Biomassa gebruik bij de productie van elektriciteit wordt beperkt in de scenario's en is vergelijkbaar in beiden modellen. Daarnaast schatten beide modellen lage tot matige volumes biochemicaliën in de totale chemische productie, waarbij dit aandeel iets groter is in de high-tech ontwikkelingssenario’s. Belangrijkste producten zijn biochemicaliën op basis van tweede generatie ethanol (0,6 miljard €/jaar in de low-tech ontwikkeling scenario's en 1,7 miljard €/jaar in de high-tech ontwikkelingsscenario's), biochemicaliën uit lignocellulose suiker (0,2 miljard €/jaar in de high-tech ontwikkelingsscenario's) en van conventionele suiker (0,06 miljard €/jaar in de high-tech ontwikkelingsscenario's). Openstelling van de markt naar grondstoffen van buiten de EU speelt een bescheiden rol, omdat biomassa voor de tweede generatie technologie beschikbaar is in Europa tegen concurrerende prijzen. Tot slot tonen resultaten uit MAGNET hoge consumptie van binnenlandse en ingevoerde land- en bosbouw residuen (90% van het totale verbruik van biomassa in 2030). Geïmporteerde biomassa is goed voor bijna 50% van de totale vraag naar biomassa.

Hoge technologieontwikkeling en open handel leveren macro-economische voordelen op in het bruto binnenlands product (0,8 miljard €/jaar) en toegevoegde waarde (0,74 miljard €/jaar). Bovendien verlagen ze de verwachte daling van de handelsbalans (0,65 miljard €/jaar) en werkgelegenheid (2,5-4,5%) in vergelijking met low-tech ontwikkelingsscenario’s.

Hoofdbevindingen en conclusies

I. Wat zijn huidige en potentiele middellange-termijn milieueffecten van toepassingen van biobrandstoffen en biochemicalieën?

Die prestatie van biobrandstof productiepaden zijn afhankelijk van regio specifieke kenmerken, zelfs als ze worden geproduceerd uit dezelfde biomassa (bijvoorbeeld suikerriet). Op het gebied van broeikasgasemissies is het leveren van een co-product (bijvoorbeeld bagasse, elektriciteit) en de gerelateerde interactie van met de regionale energievoorziening (bijvoorbeeld de CO_2-emissie-intensiteit van de elektriciteit van het net, en de referentiebrandstof voor warmtelevering) de meest bepalende factor. Dit blijkt uit de gebruikte benaderingen om het multifunctionaliteit probleem op te lossen. Cogeneratie van elektriciteit leidt over het algemeen tot meer robuuste resultaten voor ethanol uit Brazilië, ondanks het feit dat ethanol in India wordt geproduceerd uit een bijproduct. Als gevolg van allocatie worden de upstream effecten van de productie van suiker en suikerriet gematigd. Het emissiereductiepotentieel van ethanol uit Brazilië hangt echter ook sterk af van LUC gerelateerde emissies. Deze kunnen de cradle-to-gate broeikasgasemissies van ethanol uit Brazilië met een factor 3 verhogen. Dit probleem is minder van belang in India, waar ethanol wordt produceerd uit het bijproduct melasse. Indien de vraag naar ethanol groeit, hetzij voor biobrandstoffen of biochemicaliën, dan ontstaat echter het risico dat melasse voor ethanolproductie wordt onttrokken aan de diervoedersector wat kan resulteren in LUC met de bijbehorende broeikasgasemissies.

Vergelijkbare systeemafhankelijke prestaties zijn gevonden voor de uitstoot van broeikasgassen bij biochemicaliën. Wanneer CO_2-intensieve brandstoffen (bijvoorbeeld kolen) worden gebruikt voor de energielevering aan het productieproces, zijn de milieuprestaties van bio chemicaliën niet noodzakelijk beter in vergelijking met hun petrochemische tegenhangers. Uiteindelijk is de milieuprestatie van productieketens afhankelijk van het type biochemicaliën of bioplastic dat wordt geproduceerd, de conversie-efficiëntie en de regionale context. Productieketens van volledige bioplastics, zoals bio-PE tonen hogere broeikasgasbesparingen (tot 144% in vergelijking met petrochemische PE, geproduceerd in Europa), in vergelijking met gedeeltelijke bio-based plastics zoals bio-PET (±10%; Ter vergelijking, de emissies van de bio-based component, dat wil zeggen het monomeer MEG, is ongeveer 40%, in vergelijking met zijn petrochemische variant in Europa). Het gebruik van ethanol uit Brazilië in plaats van ethanol uit India leidt niet tot verminderde broeikasgasemissies van bio-PET. Procesenergie voor de conversie van ethanol naar biochemicaliën wordt voorzien door referentie Indiase energievoorziening, die zeer CO_2 intensief is. Wanneer ethanol uit India wordt gebruikt, verschuiven de lasten tussen het productiesysteem en het Indiase energiesysteem door coproductie (enerzijds het Indiase ethanol productie systeem dat elektriciteit levert aan het net en anderzijds het biochemische productiesysteem dat stroom verbruikt van het net). Wanneer verbeteringen worden aangenomen in het biobased productiesysteem (bijvoorbeeld het conversierendement, de voorziening van procesenergie) kan de emissiereductie van bio-PET oplopen tot 30% in vergelijking met petrochemische PET. Hoewel op mondiaal niveau potentiële voordelen zijn te behalen door vermindering van broeikasgassen, zijn factoren als het effect op menselijke gezondheid, de kwaliteit van ecosystemen en water stress, substantieel hoger in vergelijking met petrochemische productie van PE en PET. Deze factoren beïnvloeden de regio waar de biomassa wordt verbouwd, omdat deze voornamelijk gerelateerd zijn aan landbouwpraktijken. Dergelijke effecten worden hoofdzakelijk geassocieerd met agrochemische producten en oogstpraktijken en zijn zeer gevoelig voor de aannames van het type pesticiden, methoden van behandeling en verwijdering van afval en hun interactie met de omgeving.

Op de middellange termijn wordt verwacht dat landen die ver verwijderd zijn van biomassa aanbodgebieden zoals Nederland, grote hoeveelheden biomassa gaan gebruiken. Daarom zal voorziening en conversie van biomassa in de nabije toekomst in verschillende regio's optreden. De besparing van broeikasgasemissies die kunnen voortvloeien uit het gebruik van bio-energie en biochemicaliën hangen uiteindelijk af van de omvang van de inzet van conversiecapaciteit in de vraag regio. Deze wordt beïnvloed door verschillende factoren, zoals de ontwikkeling van de technologie, aanbodcurve van biomassa en ambities om aan klimaatdoelstellingen te voldoen (hoofdstuk 5). Ambitieuze scenario's laten zien dat grote hoeveelheden biomassa kunnen worden ingezet voor de productie van eerste en tweede generatie biobrandstoffen die leiden tot aanzienlijke emissiereducties in het wegvervoer (tot ongeveer 78 gCO_2 vermeden/MJ). Deze emissies zijn exclusief de upstream-emissies van de productie en transport van biomassa naar Nederland, die grotendeels afhankelijk is van import. Emissies van productie en aanvoerketens verminderen het emissiereductiepotentieel van biobrandstoffen, maar niet significant hoofdstuk 4) en zijn afhankelijk van het voorzieningsgebied de eigenschappen van de productiesystemen (hoofdstuk 2).

Vergelijkbare conclusies worden getrokken voor biochemicaliën. Conversie technologieën voor biochemicaliën kunnen kostenconcurrerend worden ingezet in Nederland tegen 2030, zelfs in de baseline scenario's waar geen CO_2-belasting wordt toegepast. Dit leidt tot een grote besparing van fossiele grondstoffen. Ten opzichte van de baseline scenario's, wordt 17-23 PJ aan fossiele grondstoffen vermeden door de productie van biochemicaliën 2030 (hoofdstuk 5), wat leidt tot een vermindering van broeikasgasemissies van ongeveer 1-2% in het Nederlandse energiesysteem (i.e. 1.3-1.8 MtCO_{2eq}, uitgaande van een emissiefactor van 73.55 kgCO_{2eq}/GJ voor fossiele brandstof, en complete verbranding bij end-of-life; of ongeveer 1.8 kgCO_{2eq}/kg biochemicaliën). De totale vermeden broeikasgasemissies die toegewezen kunnen worden aan biochemicaliën zijn hoger, omdat deze ook in het baseline scenario worden geproduceerd. Deze besparingen kunnen echter niet worden gekwantificeerd zonder scenario exclusief biochemicaliën. Daarbij zijn de uiteindelijke emissiebesparingen afhankelijk van het gebruik, en de levenscyclus van chemische producten.

II. Wat is de potentiële omvang en de bijdrage van de bio-economie in een nationaal energiesysteem in het nastreven van klimaatdoelstellingen op de middellange termijn?

Onder bestaande beleidsdoelen (de EU-doelstelling voor hernieuwbare energie en het nationale ondersteuningsbeleid voor de vroege inzet van onshore en offshore windturbines), zijn de emissies van Nederland in 2030 vergelijkbaar met het niveau van 1990, maar ongeveer 50 MtCO_2 boven de emissiereductiedoelstelling van 40% in 2030 (hoofdstuk 4 en

6). Om de klimaatverandering doelstellingen tegen 2030 te halen moeten inspanningen daarom worden geïntensiveerd en de bio-economie speelt daarbij een belangrijke rol in het energiesysteem van Nederland. Versnelde technologische ontwikkeling vervroegt de commerciële beschikbaarheid van biomassa conversietechnologieën. Door hun inzet, in combinatie met CCS in de energiesector en BECCS in het vervoer, kunnen belangrijke stappen in de richting van diepe emissiereductie worden gemaakt. Bovendien, zorgen een snelle ontwikkeling en vroege inzet van biomassa conversietechnologie voor een veerkrachtiger energiesysteem, dat minder kwetsbaar is voor exogene parameters zoals volatiele fossiele brandstofprijzen. Nederland kan op deze manier bijdragen aan haar eigen mitigatiedoelen, in plaats van uit te besteden door de aankoop van emissierechten.

Meer specifiek verkent hoofdstuk 5 de ontwikkelingen op basis van een ambitieuze CO_2-belasting van 69 €/tCO_2 in 2030. Het toont aan dat als de ontwikkeling en implementatie van geavanceerde technologie voor biomassaconversie wordt vertraagd, de emissiedoelstellingen in 2030 niet worden gehaald, zelfs niet onder de meest gunstige omstandigheden met een aanbod van goedkope biomassa en hoge fossiele brandstofprijzen in combinatie met een ambitieus klimaatbeleid.

Hoogwaardige technologieontwikkeling is een voorwaarde voor een kostenefficiënte koolstofarme intensieve economie. Niettemin, zelfs in het kader van de snelle technologische ontwikkeling van biomassa conversietechnologieën, zal hun bijdrage aan emissiemitigatie afhankelijk zijn van mondiale ontwikkelingen van biomassa markten en leveringskosten, en de prijzen van fossiele brandstoffen. De gerealiseerde CO_2-reductie in high-tech ontwikkelingstrajecten varieert van 15 tot 43% in 2030 ten opzichte van 1990. De lagere waarden hebben betrekking op een niet-competitieve omgeving hetzij door de lage fossiele brandstofprijzen of onderontwikkelde biomassa markten (in deze scenario's komt de emissiereductie ongeveer 35 miljoen tCO_2 lager dan de doelstelling uit). In het referentiescenario is de behaalde emissiereductie echter 30% (dat wil zeggen 15 MtCO_2 onder de doelstelling). In de energiesector worden ongeveer 100 PJ door wind- en zonne-energie geleverd en worden 27 MtCO_2 van elektriciteitsproductie uit aardgas en kolen afgevangen en opgeslagen. De bijdrage van biomassa is belangrijk, want ongeveer 620 PJ/jaar is nodig om ongeveer 300 PJ aan bio-energie te leveren (ongeveer 17% via bio-elektriciteit, 30% biomassa warmte, 53% biobrandstoffen) en ongeveer 100 PJ aan biochemicaliën. Dit komt overeen met een 23% aandeel hernieuwbare energie op het eindverbruik (dat wil zeggen 7% bovenop de nationale doelstelling, waarin biomassa meer dan 70% bijdraagt). De rest van de emissiereductie wordt bereikt door het toepassen van BECCS op de productie van biobrandstoffen (13 MtCO_2 worden opgevangen en opgeslagen in het referentiescenario). BECCS is nodig om een kosten-efficiënt mitigatietraject te voeren tot 2030. Op middellange termijn kunnen de broeikasgas-emissiereductie doelstellingen niet worden bereikt zonder deze opties. Geavanceerde bioraffinaderijen spelen een sleutelrol bij de productie van biobrandstoffen en grondstoffen voor verdere omzetting tot biochemicaliën, wat sectoroverschrijdende synergieën aantoont.

Terwijl in het referentiescenario de emissiedoelstelling niet wordt gehaald, valoriseren high-tech ontwikkeling de mogelijkheden die worden gecreëerd als biomassa markten zich ontwikkelen en de kostenefficiëntie van biomassatoeleveringsketens toeneemt. Dit kan bijvoorbeeld worden gerealiseerd door inzet van overschot of verlaten landbouwgrond voor cultivatie van energiegewassen, door intensivering van landbouw en door voorbewerking van biomassa (bijv. pelletiseren) waardoor transport kosten kunnen worden verlaagd. Het integreren van bio-energie en biochemicaliën in de structuur van het Nederlandse energiesysteem vraagt naar een extra 100 PJ aan biomassa. Dit leidt tot een extra 5 $MtCO_2$ aan emissiereductie en draagt bij aan het behalen van de doelstelling, door meer biomassa warmte (60%), biobrandstoffen (4%) en bio-elektriciteit (3%) te leveren in vergelijking met het referentiescenario. De extra emissiereductie wordt vooral gerealiseerd in de industrie en in de transportsector. Onder de meest gunstige omstandigheden - veroorzaakt door hoge fossiele brandstofprijzen - wordt de doelstelling overschreden met 3%. Terwijl de vermindering in broeikasgasemissies in de eerste plaats het resultaat is van de aanzienlijke toename van windenergie (met een factor 2 in vergelijking met referentiescenario's) en de modernisering van het wegtransport, stijgen biomassa warmte en de productie van biobrandstoffen ook aanzienlijk met respectievelijk 88% en 18%.

III. Wat zijn de mogelijke economische gevolgen op middellange termijn van ontwikkelingen in de bio-economie op nationaal niveau?

De economische indicatoren die door beide modellen die in dit proefschrift (MARKAL-NL-UU in hoofdstuk 5 en MAGNET in hoofdstuk 6) worden gegeven, laten zien dat vroegtijdig investeren in geavanceerde biomassa conversietechnologieën een No-Regret oplossing is om groei van de Nederlandse economie te ondersteunen en tegelijkertijd klimaatdoelstellingen na te streven.

Hoofdstuk 5 toont dat in de high-tech ontwikkelingsscenario's cumulatieve systeem kosten in de periode 2010-2030 tot 3 miljard € lager uitpakken dan in low-tech scenario's. Afgezien van investeringen in kapitaal, komen de belangrijkste bijdragen aan de totale kosten uit variabele parameters zoals fossiele brandstofprijzen en biomassa- en CO_2-prijzen. Ook wordt aangetoond dat grootschalige ontwikkelingen in bio-economie in Nederland besparingen kunnen opleveren van maximaal 0,8 miljard €/jaar in vergelijking met een low-tech ontwikkelingstraject met referentie biomassa kostprijzen. De systeemkosten in high-tech ontwikkelingsscenario’s met klimaatbeleid zijn in absolute ongeveer 2 miljard €/jaar hoger in vergelijking met een baseline waarin geen klimaatbeleid wordt gevoerd na 2020. Tegen 2030, is de CO_2-uitstoot van high-tech ontwikkelingsscenario's tussen de 11 en 20 $MtCO_2$/jr lager in vergelijking met low-tech ontwikkelingsscenario’s. De bovenstaande resultaten leiden tot de conclusie dat een vroege investering in de bio-economie leidt tot lagere kosten in het nationale energiesysteem en een bijdrage levert aan emissiereducties. Dergelijke scenario’s voldoen echter nog steeds niet aan de beoogde doelstelling van 40% emissiereductie ten opzichte van 1990. Dit doel wordt alleen gehaald in het scenario waarin hoge fossiele brandstofprijzen zijn aangenomen. In dit scenario stijgen de totale kosten tot ongeveer 12 miljard €/jaar. Macro-economische effecten van dergelijke scenario’s zijn niet onderzocht in dit proefschrift.

De trends uit hoofdstuk 5 zijn in het algemeen in overeenstemming met de conclusies gebaseerd op resultaten van het MAGNET model (Hoofdstuk 6). Op korte termijn (2020), leiden investeren in bio-economie tot een tijdelijke vermindering van het BBP (ongeveer 0,2-0,8 miljard €/jaar) als gevolg van de vereiste investeringen om de nationale doelstelling uit de EU-richtlijn hernieuwbare energie (EU RED) in 2020 te behalen. De korte-termijn effecten op het BBP hangen af van de import van biobrandstoffen uit Zuid- en Noord-Amerika. Om de negatieve BBP trend in 2030 tegen te gaan, zijn al op korte termijn investeringen in binnenlandse biomassaconversie capaciteit noodzakelijk. In scenario's waarin ontwikkelingen in bio-economie wordt aangenomen, hetzij in conversietechnologieën of in biomassamarkten, hebben een positief effect op het BBP van maximaal 1 miljard €/jaar vanaf 2030. De high-tech ontwikkelingsscenario's leveren de grootste opbrengst voor de Nederlandse economieën. In de eerste plaats creëren ze een toegevoegde waarde voor de binnenlandse agrarische sector. Ten tweede kunnen opkomende sectoren zoals biochemicaliën ongeveer 250 M€/jaar aan toegevoegde waarde genereren, wat vergelijkbaar is met de toegevoegde waarde van bio-elektriciteit. Echter wordt deze laatste ondersteund door beleid, terwijl geen soortgelijke stimulans bestaat voor biochemicaliën. In de derde plaats, terwijl de totale werkgelegenheid in de bio-economie sectoren (landbouw, bio-energie en biochemicaliën) daalt in 2030, wordt dit effect deels gedempt door investeringen in de bio-economie vergeleken met een counterfactual scenario. Tot slot, beperken ontwikkelingen van de bio-economie een verslechtering van de handelsbalans door de export van tweede generatie biobrandstoffen en biochemicaliën.

IV. Hoe kunnen onderzoeksinstrumenten en methoden worden verbeterd voor het verkennen van de ontwikkelingen in de bio-economie op de huidige en middellange termijn ?

In dit proefschrift zijn erkende onderzoeksinstrumenten en methoden gebruikt in context van specifieke producten en regio's ter verbetering van de weergave van relevante aspecten van de bio-economie. Hoofdstuk 2 tot en met 6 geven een uitgebreid overzicht van de verbeteringen en beperkingen van dit onderzoek, dat hieronder worden besproken als antwoord op deze onderzoeksvraag.

Product-specifieke verkenning
Verbeteringen: productsystemen van bio-energie en biochemicaliën moeten worden opgesplitst per productiestap, in tegenstelling tot het aannemen van inventarisatie data als achtergrond processen. Dit maakt het mogelijk om belangrijke inputs - zoals de energievoorzieningsmix en energie-intensieve processen - op subsysteemniveau te modelleren ten opzichte van de aanpak van het multifunctionaliteit probleem (systeemuitbreiding of allocatie). Het opsplitsten van de subsystemen wordt noodzakelijk geacht voor de productie in de regionale context en is vooral van belang wanneer systeemexpansie wordt toegepast om dubbeltelling of ter voorkoming van misleidende resultaten door het verschuiven van lasten tussen product-systemen. Het presenteren van resultaten op basis van alle benaderingen voor het toerekenen van co-producten, voorkomt dat bijvoorbeeld inefficiënte productiesystemen worden overgewaardeerd.

Volledigheid van inventarisaties en betrouwbare en controleerbare proces informatie (bijv.

gebaseerd op industriële productie data), zijn van even groot belang. De verhoogde nauwkeurigheid van de resultaten als gevolg van het opnemen van bottom-up gegevens in LCA studies, zoals toegepast in dit proefschrift, leidt tot meer conservatieve schattingen van milieuprestaties van de biochemische toepassingen. Tegelijkertijd leidt het ook tot beter begrip van de afwegingen tussen impactcategorieën en belicht het mogelijke verbeteringen.

*Beperkingen***:** de consequentiële LCA methode die is toegepast in dit proefschrift heeft beperkingen. Lange termijn veranderingen in de energievoorziening door de additionele inzet van bio-energie of biochemicaliën worden niet meegenomen. Dergelijke veranderingen kunnen bijvoorbeeld worden veroorzaakt door een sterk klimaatbeleid dat tot een groter aandeel hernieuwbare energie kan leiden, of door lage fossiele brandstofprijzen waardoor het aanbod van fossiele grondstoffen dominant zal zijn.

Energiesystemen-modellen kunnen dergelijke scenario's verkennen, zoals aangetoond in hoofdstuk 4 en hoofdstuk 5. Verder is de consequentiële LCA methode te statisch om directe en indirecte effecten van suikerriet expansie te verkennen wanneer de vraag naar biobrandstoffen of biochemicaliën stijgt. Verbeterde feedback mechanismen en iteraties met andere modellen kunnen helpen om deze beperking aan te pakken en de precisie van product-specifieke analyse te verbeteren. CGE-modellen kunnen deze functies verschaffen omdat ze de gevolgen van de toegenomen vraag op het globale landgebruiksysteem kunnen modeleren, net als het MAGNET model dat wordt gebruikt in hoofdstuk 6.

Verkenning van het energiesysteem
Verbeteringen: Vanuit een energiesysteem perspectief, vereist de uitbreiding van een bestaand model (MARKAL-NL-UU) met biochemicaliën, een gedetailleerde weergave van de referentie-petrochemische industrie. Omdat deze industrie zeer goot en complex is, is een vereenvoudiging van processtromen in de petrochemische industrie noodzakelijk.

De belangrijkste processtappen die expliciet moeten worden weergegeven in de structuur van het model zijn grondstoffen, chemische basisproducten, chemische tussenproducten en eindproducten (hoofdstuk 4). Het focussen op een nationale economie maakt het mogelijk om andere parameters in de structuur van het model op te nemen en te verkennen (de bestaande kapitaalvoorraad, de kosten en de technische prestaties in de tijd, en parametrering om de effecten van variabele fossiele brandstofprijzen te kunnen beoordelen). Bij middellange termijn verkenningen, beïnvloeden de kapitaalvoorraad en de gerelateerde resterende technische levensduur het tijdstip van de inzet biochemicaliën. Ten tweede moeten meerdere bioraffinage outputs gekoppeld worden aan eindgebruiksectoren, in tegenstelling tot bijvoorbeeld productiekosten bepalen in een gereduceerde vorm (bijvoorbeeld door toewijzing van credits).

Dit is vooral van belang voor de conversie van biomassa naar geavanceerde biobrandstoffen en biochemicaliën om sectoroverschrijdende synergieën te modeleren en om mogelijke concurrentie voor biomassa - die als gevolg van sectorspecifieke beleidsdoelstellingen kunnen ontstaan - te beperken. Ten derde kan uitsplitsing van het aanbod van biomassa voor verschillende grondstoffen en aanbod regio's nuttig zijn om gedetailleerde biomassa cost-supply curves te ontwikkelen en te koppelen aan specifieke conversiepaden van biomassa. Hierdoor wordt, het mogelijk om complete productieketens te verkennen.

Ten slotte zijn technologische ontwikkelingen en technologische leren een bepalende factor van het concurrentievermogen van de bio-economie. Hoofdstuk 4 richt zich op technologisch leren op basis van verbeteringen in proces efficiëntie en opbrengst in de tijd, op basis van technische bottom-up studies en deskundigenadvies. Bovendien, schat hoofdstuk 4 de toekomstige kosten van technologieën in door schaaleffecten in aanmerking te nemen. Deze worden als input voor het model gebruikt en worden gevarieerd in twee verschillende technologieontwikkeling scenario's. Hierdoor kan scenario-analyse op de technologische ontwikkeling worden gebruikt om een belangrijke factor van systeemprestaties te verkennen.

Beperkingen: het MARKAL-NL-UU model is nuttig om middellange termijn bio-economie ontwikkelingen in het Nederlandse energiesysteem te verkennen. Toch kunnen verschillende verbeteringen in het model worden opgenomen om het begrip van andere parameters te verhogen, die relevant zijn voor weergave van de bio-economie in modellen. Ten eerste worden de bestaande olieraffinaderijen niet expliciet gemodelleerd. Als gevolg daarvan kan ook de co-afhankelijkheid van brandstoffen (diesel, benzine, kerosine) met nafta productie niet worden gemodelleerd. Dit kan nuttig zijn om op te nemen om na te gaan of de co-afhankelijkheid een beperkende factor kan zijn voor de ontwikkeling van bio-energie en biochemicaliën. In de tweede plaats zijn end-of-life behandelingsopties en cascadering slechts gedeeltelijk uitgewerkt. Verbranding van organisch afval, al dan niet met terugwinning van energie, is niet verbonden met de output van biochemicaliën. Recycling is ook niet opgenomen. Deze opties kunnen de efficiëntie van het gebruik van biomassa in de economie vergroten. Ten derde, als gevolg van de grote bijdrage van biomassa warmte in hernieuwbare energie, moeten efficiëntie maatregelen in de industrie en de residentiële en commerciële sector ook expliciet opgenomen worden. Dit betekent ook een verbeterde weergave van de warmte sector. Andere aspecten die zeer relevant zijn voor de dynamiek van het systeem zijn globale concurrentie van de petrochemische industrie, het gebruik van andere fossiele grondstoffen (bijv. Schaliegas) en gerelateerde impact op de prijsdynamiek, en de productie en exportvraag van Nederland. Deze vragen kunnen deels worden opgelost door middel van samenwerking tussen modellen (model samenwerking).

Macro-economische verkenning

Verbeteringen: CGE-modellen zijn in staat om complexe en dynamische economische relaties van grootschalige inzet van biomassa te adresseren. De hoge sectorale en technologische aggregatie en de beperkte vertegenwoordiging van toekomstige structurele veranderingen (bijvoorbeeld diegene veroorzaakt door bio-economie ontwikkelingen die nog niet beschikbaar zijn) zijn echter belangrijke tekortkomingen van CGE-modellen. In dit proefschrift worden deze beperkingen aangepakt door samenwerking van het CGE-model MAGNET (hoofdstuk 6) en het energiesystemen-model MARKAL-NL-UU (hoofdstuk 4 en 5). Een belangrijke verbetering van MAGNET is de splitsing van de chemische sector in drie sub-sectoren: namelijk een conventionele fossiele chemische sector, een gemengde bio-based en fossiele chemische sector (voorziening van zowel fossiele en bio-based grondstoffen voor in de chemische industrie), en een biochemicaliën sector (volledige vervanging van chemische producten door de inzet van biomassaconversie technologieën). Om deze splitsing te bewerkstelligen zijn concessies vereist bij de selectie van de meest representatieve productieroutes, voor het vereiste aggregatieniveau van MAGNET. Op basis van MARKAL-NL-UU worden de kost-structuren van geselecteerde technologieën aangeleverd die als meest kosten efficiënt gelden op de middellange termijn. Deze technologieën zijn ook het meest representatief op het gebied van paden die voor biochemische implementatiestrategieën worden beschouwd. De sectoren elektriciteit en brandstof voor wegvervoer zijn in MAGNET verder opgesplitst om rekening te houden met het meestoken van biomassa (naast andere hernieuwbare energie alternatieven) en de productie van eerste en tweede generatie biobrandstoffen. Op deze manier zijn bestaande en geavanceerde conversiepaden voor biomassa geïntegreerd in de structuur van het macro-economische model. Aan de basis van deze structuur, ligt de productie en levering van de eerste generatie en lignocellulose grondstoffen op regionaal niveau, die kan worden gebruikt door alle sectoren van de bio-economie. Een belangrijke verbetering in de representatie van het aanbod van biomassa in MAGNET is de toevoeging van de pellet sector. Dit maakt, naast de handel in vloeibare biobrandstoffen, ook de verkenning van handel in vaste biomassa mogelijk. Technologie verandering wordt verkend aan de hand van scenario's, vergelijkbaar met de aannames in MARKAL-NL-UU. Ten slotte is de parametrisering en integratie van nationale en regionale beleidsambities voor hernieuwbare energie essentieel omdat deze van invloed zijn op de inzet volumes van biomassa en andere hernieuwbare energiebronnen op korte termijn.

Beperkingen: Een belangrijke stap op weg naar een beter begrip van de macro-economische effecten op basis van MAGNET uitkomsten ligt in de vertaling van de belangrijkste outputs in monetaire eenheden naar fysische eenheden (bv. PJ, Mt). Dit geldt voor de biomassavoorziening, handel, en de conversie van biomassa naar bio-elektriciteit, biobrandstoffen, en biochemicaliën.

Bovenstaande wordt momenteel beschouwd als een beperking, aangezien de resultaten uit MAGNET niet direct kunnen worden vergeleken met de biomassa kosten, het verbruik van biomassa, bio-energie en biochemicaliën voorziening uit MARKAL-NL-UU. Een consistente methode om de monetaire stromen uit MAGNET om te zetten naar fysieke stromen zal ook de mogelijkheden tot soft-linking tussen de modellen verbeteren. Een voorbeeld is de mogelijkheid om effecten van verbeteringen in landbouw of finale energievraag aan MARKAL-NL-UU te leveren. Bovendien kunnen consequentiële LCA studies zulke outputs gebruiken om de gevolgen van door de markt gemedieerde effecten te verkennen. Een andere beperking van MAGNET is dat CCS en BECCS technologieën in de elektriciteitssectoren en vloeibare brandstoffen niet expliciet in het model zijn opgenomen. Als gevolg daarvan kunnen de macro-economische effecten van ambitieuze klimaatmitigatiescenario's niet worden verkend.

Aanbevelingen voor stakeholders in de bio-economie

Sectorspecifiek (bv. energie, landbouw, bosbouw) en sector overkoepelend beleid (bv. over beperking van klimaatverandering, milieu) kunnen de schaal van de inzet en de impact van biomassa voor energie en chemicaliën beïnvloeden. Stakeholders in de bio-economie (bv. beleidsmakers, industrie, investeerders, andere marktspelers) moeten voldoende informatie krijgen over de gevolgen en impact van hun beslissingen. De gebruikte tools en conclusies in dit proefschrift kunnen van nut zijn om die informatie te verschaffen.

Beleidsmakers
Onder het huidige beleid zal Nederland wordt het moeilijk om de nodige vermindering van broeikasgasemissie in 2030 te bereiken (hoofdstuk 4 en hoofdstuk 6). Strengere doelstellingen en een versterkt stimulerend beleid rond bio-energie en andere hernieuwbare energiebronnen moet worden overwogen, vooral gezien de lage fossiele brandstofprijzen (bv. een hoger aandeel hernieuwbaar energie, strengere sectorspecifieke verplichtingen).

Als de bio-economie sector groeit, zullen beleidsmakers steeds vaker geconfronteerd worden met de behoefte om duurzaamheid van biomassa in de economie te garanderen en om verbeteringen in de waardeketen aan te sporen. Duurzame bio-energie conversieketens moeten worden geïdentificeerd en duurzaamheidscriteria moeten worden vastgesteld (bv. minimum vermindering van broeikasgasemissie) voor de conversie van vaste biomassa in bio-energie (bv. bio-elektriciteit, biomassa warmte (EC, 2014c)) en voor de conversie van biomassa naar bio-chemicaliën. LCA benaderingen, zoals naar verwezen wordt in de EU RED (EC, 2009b), kunnen in die richting van nut zijn aangezien de productie van bio-chemicaliën worden gekenmerkt door vergelijkbare duurzaamheidsproblemen als biobrandstoffen (bv. variatie van milieu-impact varieert met grondstof, productielocatie; hoofdstuk 3). Deze zijn echter wel complexer (bv. langere waardeketens, tijdelijke biogenische koolstof-opslag afhankelijk van de levensduur van het product, andere technische prestatie dan de petrochemische versie). Met de verwachte grootschalige import van biomassa naar conversiegebieden als Nederland (hoofdstuk 4 tot 6), moeten de duurzame inkoop en ontwikkeling van biomassamarkten worden gegarandeerd (bv. door certificeringsregelingen), aangezien de criteria rond minimum vermindering van broeikasgasemissies de milieu-impact in productiegebieden niet voldoende veilig kunnentellen. De politieke gevolgen van duurzaamheidscriteria op bio-chemicaliën via de productieketen kunnen worden geadresseerd met behulp van de consequentiële LCA methode.

Het energiesysteem-model dat in hoofdstuk 5 wordt gebruikt voor projecties van onder meer technologie neutrale scenario's (MARKAL-NL-UU) kan dienen om de meest veelbelovende ontwikkelingspaden te benadrukken. De scenarioresultaten tonen een grote productie van biomassawarmte in de industrie. Voor de warmtesector is echter nog niet voldoende aandacht, met uitzondering van energie-efficiëntie doelstellingen. Bovendien tonen hoofdstuk 4 tot en met 6 aan dat de inzet van geavanceerde technologieën een voorwaarde is voor efficiënt gebruik van biomassa in de bio-economie en om op een kostenefficiënte manier een sterke vermindering van broeikasgasemissie te bereiken. Vooruitgang in nieuwe technologieën voor biomassa-conversie (bv. vergassing, pyrolyse) en, dankzij bioraffinaderijen, sectoroverschrijdende synergieën tussen de transport- en de elektriciteitssector, en tussen de transport en chemische sector, zijn twee gebieden waar onderzoek, ontwikkeling en inzet (RD&D) subsidies nog steeds nodig zijn. Dit zal zorgen voor vroege verspreiding en snelle groei van geavanceerde technologieën in de loop van tijd (hoofdstuk 5). In dit proefschrift wordt een CO_2-prijsmechanisme gebruikt als enige instrument om emissiebeperking te stimuleren, wat voor een ongelijk speelveld voor bio-chemicaliën kan zorgen. Beleidsmakers moeten een krachtig beleid voeren (buiten RD&D) om te zorgen dat alle methodes voor biomassa-conversie kunnen concurreren op een gelijk speelveld en om sectoren te koppelen met behulp van bioraffinaderijen. Zo kan bijvoorbeeld een CO_2-belasting op fossiele grondstoffen in plaats van op CO_2-emissies worden toegepast.

Ten slotte toont hoofdstuk 5 ook pad afhankelijkheden aan die veroorzaakt kunnen worden door beleidsbeslissingen buiten de bio-economie. Bijvoorbeeld, de sluiting van steenkoolcentrales kan de behoefte aan de inzet van CCS in de stroomsector op middellange termijn beperken, net zoals de bijkomende uitdagingen (bv. proof of technology op commerciële schaal, publieke opinie). Tegelijk vereist dat aanzienlijke inspanningen om het onshore en offshore vermogen van windturbines te verbeteren en bijkomende problemen op te lossen (bv. discontinuïteit, transportnet-afhankelijkheid, connectiviteit, congestie, opslag).

Industrie en andere marktspelers
Aangezien merkeigenaars momenteel de grootste consumenten zijn van toepassingen van bulk bio-chemicaliën (bv. bioplastic), is het aanbevolen om chains of custody in te voeren die op korte termijn voor goede bedrijfsvoering zorgen in de productieketen. Daarbij is nauwlettend toezicht nodig gericht op de typen agrochemicaliën, oogstpraktijken, afvoer en behandeling van bedrijfsafval, en energieproductie van conversiesystemen uit koolstofarme bronnen (bv. door certificeringsregelingen, controles ter plaatse). Korte termijn verbeteringen in biomassaproductie (bv. landbouwmanagement) en industriële praktijken (bv. gebruik van beschikbare hernieuwbare bronnen voor energieproductie, efficiëntieverbeteringen van biomassa-conversie) zijn vereist in lijn met best practices en met betrekking tot de regionale context.

Vanuit het systeem-perspectief zal sectoroverschrijdende samenwerking in de waardeketen de groeikansen van de industrie maximaliseren en markttoegang bieden aan producenten. Zoals dit proefschrift aantoont, ontstaan er sectoroverschrijdende synergieën aan de vraagkant bij multi-output processen (bv. bioraffinaderijen), wat aangeeft dat de industrie nieuwe kansen moet verkennen op het gebied van samenwerking of nieuwe markten moet integreren door investering, demonstratie en opwaardering van nieuwe technologieën. Tegelijk kunnen die inspanningen alleen operationeel worden als hoge kwaliteit biomassaproductie verzekerd is tegen lage kosten. Samenwerking met upstream marktspelers in de waardeketen is nodig om te zorgen voor biomassaproductie tegen lage kosten (bv. investeringen in de landbouwsector rond gewasontwikkelingen, oogstverbeteringen, ophalen van restafval), productiekwaliteit (bv. biomassa voorverwerkingsfaciliteiten) en continuïteit en veiligheid van grondstoffen (bv. rond infrastructuur, opslag en logistiek), en om vraagmarkten een inzetbare en verhandelbare grondstof te bieden.

Het komende decennium kan een beslissende periode worden waar zulke kansen gevaloriseerd kunnen worden en bij kunnen dragen aan maatschappelijke uitdagingen. Zoals hoofdstuk 5 toont, zullen er op korte termijn investeringen plaatsvinden in de petrochemische industrie in Nederland, het concurrentievermogen van biochemische productietechnologieën zal sterk afnemen, vooral in het licht van lage fossiele brandstofprijzen. Een dergelijke ontwikkeling kan de productie in de petrochemische industrie blokkeren en de ontwikkeling van de bio-economie belemmeren. In een onzeker investeringsomgeving moet de industrie zich actief betrekken in strategische beslissingen die op korte termijn misschien geen winst op zullen leveren maar dat wel zullen doen op de langere termijn.

Aanbevelingen voor verder onderzoek

De inzet van verbeterde methodes en tools voor de verkenning van de bio-economie, gepresenteerd in dit proefschrift, brengt ook grote tekortkomingen aan het licht die met toekomstig onderzoek kunnen worden opgelost. De tijdspanne (nu tot 2030) en het geografische bereik (Nederland) toegepast in dit proefschrift had als hoofddoel de milieu-impact van gevestigde bio-energie en biochemische routes te verkennen en de complexiteiten rond opkomende bio-economie sectoren in de modellen MARKAL-NL-UU en MAGNET te verwerken. Hoewel dit is bereikt, zijn energiesystemen en macro-economische studies met grotere tijdshorizon en regionaal bereik vereist om de rol en bijdrage van de bio-economie bij sterke emissiebeperking te evalueren.

Aanbevelingen over product-specifieke LCA studies

Product-specifieke en productieketen-specifieke studies gebaseerd op attributionele LCA zijn waardevol om de milieu-impact van gevestigde productieketens binnen specifieke systeemgrenzen te evalueren. Maar de resultaten zijn afhankelijk van de kwaliteit van de gebruikte data. Kwaliteit en geldigheid van data kwaliteit en geldigheid geldt als een overkoepelende aanbeveling (bv. volledigheid van inventarissen en betrouwbare en controleerbare procesinformatie gebaseerd op industriële productiedata), maar is in het bijzonder relevant voor LCA's bedoeld om de impact van nieuwe biochemische productieketens nauwkeurig te bepalen. Alle attributionele LCA’s moeten gepaard gaan met verbetering in data-transparantie en gevoeligheidsanalyse. Dit geldt met name voor algemene achtergrond inputs zoals pesticiden die de menselijke gezondheid en kwaliteit van het ecosysteem beïnvloeden. Voor het reduceren van onzekerheid die gepaard gaat met methodologieën om de interactie van de technosfeer met het natuurlijk systeem te bepalen, wordt het aanbevolen om inzichten uit milieu-impact assessments te gebruiken die gebaseerd zijn op de lokale omgeving en die ook lokale niveaus van schadelijke stoffen meten. Ten slotte is modellering en opsplitsing van product-systemen aanbevolen in zoverre mogelijk, aangezien het een onderliggende reden kan zijn voor verschil in resultaten. Voor de consequentiële LCA methode is model samenwerking vereist tussen bottom-up LCA en top-down CGE modellen, wat hieronder verder besproken wordt.

Aanbevelingen voor energiesystemen en economische modellen

Modellen die gebruikt worden voor de bio-economie moeten de petrochemische industrie en haar grondstofstromen bevatten, evenals biochemische productietechnologieën op diverse niveaus (bv. grondstof, basischemicaliën, tussenproducten, eindproducten). Bovendien moet met de expliciete modellering van multi-output systemen (bv. bioraffinaderijen) sectoren worden gekoppeld. Gezien de emissiebeperking moeten andere relevante industriesectoren voor ontwikkeling van de bio-economie geïntegreerd worden (bv. papier en pulp, staal, bouw). Een langere tijdspanne vereist ook een uitbreiding van de technologieportfolio van bio-chemicaliën (bv. naar specialty chemicaliën, chemicaliën op basis van algen, chemicaliën op basis van lignine). Specialty chemicaliën worden in lage volumes geproduceerd en er wordt niet verwacht dat ze een significante hoeveelheid fossiele brandstoffen zullen vervangen of direct zullen bijdragen aan vermindering van broeikasgasemissies. Maar ze hebben hoog gefunctionaliseerde toepassingen en hebben typisch een hoge marktprijs. Daarom kunnen ze de bijdrage van biomassa in de economie nog meer stimuleren dan aangegeven in hoofdstuk 6. Door de hoge relevantie van de niet-energetische sector en warmtesector, moeten aan het technologisch bereik van verbeterde verkenningen ook vraagtechnologieën en maatregelen (bv. energie-efficiëntie) worden toegevoegd en gecombineerd worden met verkenningen van energiesystemen. Ten slotte, door de aanwijzingen voor groei in bio-chemicaliën productie (hoofdstuk 4 tot en met 6), wordt het relevant om de gevolgen van beleidsinstrumenten te evalueren die biomassa voor bio-energie ondersteunen en of die zorgen voor een niet-competitieve omgeving voor biochemicaliën. Een multi-sectorale CO_2 emissie belastingregeling, of belasting op fossiele koolstof houdende grondstoffen, en niet op CO_2-emissies, kan potentieel toegepast worden.

Aanbevelingen voor modelsamenwerking

Met deze verbeteringen kan een consistenter en harmonieus framework ontwikkeld worden waarin LCA, energiesystemen en CGE modellen geëngageerd worden naast sectorspecifieke modellen (bv. elektriciteitssector). Gezien de verwachte potentiële productie van bio-elektriciteit van bioraffinaderijen maar ook de grote productie van intermitente windenergie, moet hun operationele flexibiliteit (bv. op uurbasis) gerealiseerd worden met gebruik van een dispatch model. Dit is aangetoond in Brouwer et al. (2015). Een gelijkaardige soft-linking benadering is aanbevolen, die rekening houdt met bevindingen gebaseerd op dit proefschrift zoals de grote bijdrage van 2^{e} generatie biobrandstoffen tot hernieuwbare energie doelstellingen en de productie van bio-elektriciteit uit bioraffinaderijen. Bovendien kan de ontwikkeling en toepassing van een consistente methode om de monetaire eenheden van het MAGNET model om te zetten in fysieke eenheden en daarbij het soft-linking proces verbeteren tussen een macro-economisch- en een energie-systeemmodel. Dit is bijzonder relevant als er lange termijn verkenningen uitgevoerd worden waarin economische verbindingen in en tussen regio's kunnen leiden tot veranderingen in productiecapaciteit (bv. relevant voor de chemische industrie), wat tevens de regionale productievraag, energievraag en biomassaproductie beïnvloedt. Ten slotte, kunnen de product-gebaseerde en productieketen-specifieke verkenningen gebaseerd op de LCA methode significant baat hebben bij de outputs van energiesystemen en macro-economische CGE modellen. Consequentiële LCA's kunnen baat hebben bij de integratie van een middellange termijn en lange termijn energieproductie mix zoals geschat door energiesysteem modellen voor een specifieke regio. Op dezelfde manier kunnen CGE modellen inzicht geven in biomassaproductie, land gebruik en emissies door veranderingen in landgebruik. Dit kan de nauwkeurigheid en consistentie verhogen van consequentiële LCA resultaten, die momenteel zeer uiteen lopen. Zo kunnen comparatieve LCA studies uitgevoerd worden op basis van verschillende productieketens in lijn met effecten afkomstig van verbeterde systeemwijde en economiewijde modellen.

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CURRICULUM VITAE

Ioannis Tsiropoulos was born in Athens, Greece on the 29th of March, 1983. He graduated from the National and Kapodistrian University of Athens and received his B.Sc. degree in Physics in 2006. Over the course of his studies he specialised in Environmental Physics and Meteorology and worked as an intern at the Institute for Environmental Research and Sustainable Development in Athens. After completing his mandatory military service he moved to the Netherlands where he enrolled in the master's programme Sustainable Development – Energy and Resources at Utrecht University, graduating in 2010. Between 2009 and 2011 he worked as a sustainability researcher at the OnePlanet Catalogue and People 4 Earth foundations. Early in 2011, he began working as a junior researcher at Utrecht University. He worked on a Life Cycle Assessment project for Nestlé Waters, on a bioeconomy framework project (SAT-BBE) under the 7th Framework Programme of the European Commission and on an energy systems modelling project (MEV II) within the BE-Basic R&D Program.

PEER REVIEWED PUBLICATIONS

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