YIELDING A FRUITFUL HARVEST

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PhD dissertation

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EEN VRUCHTBARE OOGST

Geavanceerde methoden en analyse van regionale potenties
voor duurzame biomassa ketens gekoppeld aan milieu- en
landgebruik-effecten van landbouwintensivering
(met een samenvatting in het Nederlands)

Proefschrift

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ABBREVIATIONS

AD	Anaerobic digestion
BTX	Benzene, toluene, and xylenes
BC	Biochemical
BFM	Bone-free meat
CBP	Consolidated Bioprocessing
CC	Combined cycle
CFB	Circulating fluidized bed
CHP	Combined heat and power
CI	Conventional intensification
CW	Carcass weight equivalent
DG	Directorate General (European Commission)
DDGS	Dried distillers grain soluble
DLUC	Direct Land Use Change
ECH	Epichlorohydrin;
ETE	Ethanol-to-ethylene
EU	European Union
FAME	Fatty acid methyl ester
FAO	UN Food and Agricultural Organisation
FMD	Foot-and-Mouth Disease
FPCM	Fat and protein corrected milk equivalent
FT	Fischer-Tropsch
GAEZ	Global Agro-Ecological Zones
GATT	General Agreement on Tariffs and Trade
GHG	Greenhouse Gas
GLOBIOM	Global Biosphere Management Model
GT	Gas turbine
GWP	Global warming potential
HHV	Higher Heating Value
HTU	Hydrothermal upgrading
ICE	Internal combustion engine
IGCC	Integrated gasification combined cycle
IGFC	Integrated gasification fuel cell
II	Intermediate sustainable intensification
ILUC	Indirect Land Use Change
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
LHV	Lower Heating Value
LR	Learning Rate
LUC	Land Use Change
LU	Livestock Unit
LULUC	Land use and land use change
LW	Live weight
MIRAGE-BioF	Modeling International Relationships in Applied General Equilibrium for Biofuel
MSW	Munical solid waste
MTBE	Methyl tertiary butyl ether
MTO	Methanol-to-olefins
NG	Natural gas
NGCC	Natural gas combined cycle
NGGT	Natural gas-gas turbine
NOP	Natural oil polyol
NREAP	National Renewable Energy Action Plan
NUE	Nutrient use efficiency
O&M	Operation and maintenance
ORC	Organic Rankine Cycle
PA	Polyamide
PBR	Photobioreactor
PBT	Polybutylene terephthalate
PC	Pulverized coal
PDO	1,3-propanediol
PE	Polyethylene
PET	Polyethylene terephtalate
PFP	Partial factor productivity
PHA	Polyhydroxyalkanoates
PLA	Polylactide
PP	Polypropylene
PS	Polystyrene
PTT	Polytrimethylene terephtalate
PUR	Polyurethanes
PVC	Polyvinylchloride
RED	Renewable Energy Directive (EU)
R&D	Research & Development
SC	Steam cycle
SHF	Separate Hydrolysis and Fermentation
SI	Sustainable intensification
SNG	Substitute/Synthetic Natural Gas
SOC	Soil organic carbon
SRC	Short rotation crops
SSCF	Simultaneous Saccharification and Co-Fermentation
SSF	Simultaneous Saccharification and Fermentation
TC	Thermochemical
TOP	Torrefied and pelletized biomass
UAA	Utilized Agricultural Area
WTW	Well-to-wheel

CHAPTER 1

Introduction

1.1 CURRENT AND FUTURE RESOURCES FOR ENERGY AND MATERIALS

To fulfill the demand for energy, food and materials, society depends on the resources available on earth. These resources include, for example, fossil fuels, land, and water. However, the amount of resources available is limited and we need to use them efficiently. Fossil fuels (oil, coal and gas) account for a large share of the current global primary energy supply. Although estimates about the availability and recoverability of fossil resources are often highly uncertain and frequently contested, fossil fuels reserves will be exhausted eventually. In addition, fossil fuel consumption is an important source of greenhouse gas (GHG) emissions. In 2010, CO_2 emissions from fossil fuel and industrial processes contributed to 62-65% of the global annual anthropogenic GHG emissions [1]. Thereby, fossil fuels are a major cause of climate change [1]. For these reasons, alternative resources for energy and materials need to be found. There are several renewable energy sources, like wind and solar, that can provide alternatives, but most of these can only provide electricity and/or heat. In contrast, biomass applications are more versatile and their applications also include the production of liquid biofuels, biochemicals and biomaterials. Therefore, biomass is vital for substituting fossil fuels in the transportation sector, especially in aviation, ocean shipping and long distance trucking, as well as in the chemical sector.

1.2 BIOMASS DEMAND AND POTENTIALS

Biomass covers a broad range of resources, including agricultural and forestry residues, agricultural crops (sugar, starch and oil crops) and dedicated bioenergy crops (e.g. short rotation crops and perennials), manure, other organic wastes and algae [2,3]. In 2010, global bioenergy use accounted for a primary biomass supply of approximately 62 exajoule (EJ_p) or 12% of the global primary energy use [4]. Modern uses of biomass for heat, electricity and biofuels amounted to about 22 EJ_p, while traditional bioenergy contributed for the remaining 40 EJ_p [4]. The current biomass demand for bio-based materials is equal to about 60% of the biomass demand for energy [5], but this demand particularly covers the traditional use for wood, paper and textile products (e.g. cotton). The biomass demand for novel synthetic organic materials at the moment is minor (0.6 EJ) [6].

To attain a low carbon energy supply, the required biomass contribution in 2050 is projected to be in the range of 80 to 180 EJ_p [7-12]. In addition, Saygin et al. [6] estimate an economic potential of biomass use of almost 20 EJ_p yr^{-1} for substitution of synthetic organic material in the chemical industry in 2050. Hence, a total biomass supply of approximately 100-200 EJ_p yr^{-1} would be required to meet the projected demand for both

bioenergy and biomaterials in 2050. By the same year, the IPCC estimates the global technical biomass supply potential to range from 100 to 300 EJ_p [2,3].
A comparison of the biomass requirements and supply potentials shows that, to mitigate climate change and to avoid competition between biomass feedstocks for food, feed, energy and materials, it is important to mobilize large amounts of biomass and to adopt biomass value chains that are most promising for producing heat, power, fuels and materials in terms of their technological, economic and environmental performance. But although biomass is widely considered an important option for reducing GHG emissions, the recent rapid increase in global biomass production for modern bioenergy and biomaterial purposes has also raised concerns about the role biomass can actually play in mitigating climate change. One of the main topics of concern is unwanted land use change (LUC), and especially indirect lands use change (ILUC). Here, ILUC is defined as a change in land use that takes place if biofuel feedstock production displaces agricultural production of food, feed and fibres and this displacement results in food, feed and fibres being produced elsewhere 1) to continue to meet the demand, or 2) because of increased food prices triggering additional production [13-15]. When ILUC entails a conversion of high carbon stock lands, e.g. forests or grasslands, this can lead to increased GHG emissions which reduces or even cancels out the GHG benefits of biofuels compared to fossil fuels [15]. Since the first publication on the negative effects of ILUC by Searchinger et al. [15], multiple studies have attempted to model and quantify the extent of (I)LUC and the level of related GHG emissions caused by biofuel production [16-20]. However, the modeling of LUC and (I)LUC-related GHG emissions is characterized by major limitations, uncertainties and challenges [21-23]. Although recent model improvements have often allowed a downward revision of (I)LUC-related GHG emissions compared to the original estimates by Searchinger et al. [15], the results still vary significantly between studies and outcomes are expected to remain uncertain [13,22,24]. Therefore, to mobilize large amounts of biomass while reducing GHG emissions and mitigating climate change, it is important to investigate how ILUC can be mitigated or prevented [13].

ILUC of biofuels can only be prevented when the direct LUC (DLUC) of the displaced activity is addressed as well. Therefore, it is necessary to take an integrated perspective on all land use, whether for food, feed, fibre and fuels. Previous research has identified the several measures to mitigate ILUC [13,25-27]. One of these measures is agricultural intensification, i.e. increased food production from existing agricultural land. The importance of agricultural intensification is also reflected in the estimates of biomass supply potentials. According to these estimates, a significant contribution to the total potential can come from energy crops grown on surplus agricultural land that is not required for food, feed and fibre production [3]. However, estimates of future surplus agricultural land available for biomass production deviate significantly between studies and the consensus on the potentials decreases for higher projections [4]. This disagreement can largely be explained by uncertainties about the possibilities of agricultural intensification and different

assumptions with respect to potential pathways for intensification and modernization of agriculture [28-30].
Agricultural intensification may not only contribute to increased biomass potentials, ILUC mitigation and improved GHG balances of biomass value chains, it may also improve the environmental performance of the agricultural sector itself. For example, Tilman et al. [31] show that there is a significant potential in agriculture to reduce global land clearing, GHG emissions and nitrogen use through improved technology and adoption and transfer of high yielding technologies to underyielding regions. Also, Havlik et al. [32] show that the transition of livestock production towards more efficient systems would significantly decrease livestock-induced GHG emissions. However, the GHG and other environmental impacts of agricultural intensification depend on how this intensification is organized (see e.g. Valin et al. [33]).

Many studies have investigated the effects of agricultural intensification on the availability of surplus agricultural land and biomass potentials, the GHG and other environmental impacts of agricultural intensification and biomass production, and/or the environmental and economic performance of biomass value chains. However, these studies often insufficiently address the regional possibilities for agricultural intensification, the GHG and other environmental impacts of different pathways for agricultural intensification, and the comparative performance of biomass value chains for bioenergy and materials. The next section provides an overview of the existing literature, its findings, and its limitations with respect to these aspects.

1.3 STATE OF THE ART AND KNOWLEDGE GAPS

REGIONAL DIFFERENCES IN THE POSSIBILITIES FOR AGRICULTURAL INTENSIFICATION AND ITS IMPACTS

Several studies have investigated the effects of agricultural intensification on the global availability of surplus agricultural land, technical biomass potentials and the environment. For example, van Vuuren et al. [34] assessed the impact of food crop yield changes on the global woody biomass potential in 2050. They found that an additional yield improvement of 12.5% compared to the baseline scenario resulted in an increase of the biomass potential from 150 to 230 EJ. Erb et al. [35] found that the biomass potential in 2050 would be 79 EJ yr^{-1} in the case of intermediate agricultural intensification and humane livestock rearing and 105 EJ yr^{-1} in the case of greater intensification of crop and livestock production[1].
Dornburg et al. [29] estimated that improvements in agricultural management could

1 For both intensification scenarios, it is assumed that current trends in diet and crop land area expansion are continued.

account for 140 EJ yr^{-1} of the total biomass supply potential of 500 EJ yr^{-1} in 2050. Slade et al. [28] derived from a review study that more than 1 Gha of high yielding agricultural land, equal to about 20% of the global agricultural land area in 2010, could be made available for bioenergy crops in 2050 if food crop yields increase at a higher rate than food demand and if the consumption of livestock products is limited.

While the global potential for agricultural intensification is significant, large differences exist between regions due to significant differences in the current yield gap [36]. Also, the GHG impacts of agricultural intensification are very region specific as these depend on, for example, the degree of intensification possible [33]. Regional analyses of potentials for agricultural intensification and biomass production allow considering the specific characteristics of a region such as, for example, biophysical conditions and agricultural practices. Various case studies have investigated regional potentials for agricultural intensification and the impact on biomass potentials, but more regions need to be investigated. Also, the case studies often only include improvements in crop production. Intensification potentials in the livestock sector have received much less attention in literature than agricultural crops – despite the fact that livestock production accounts for 70% of the total agricultural land and livestock intensification is assumed to play a key role in making land available for biomass production [4,37]. Also, Valin et al. [33] find that, on a global level, improvements in livestock production have a larger GHG mitigation potential than the intensification of crop production.

Studies that assess agricultural intensification potentials and/or global land availability for biomass generally base their crop yield projections on historical developments. Many of these studies also account for (a limited number of) endogenous drivers of future yields. These factors are related to, for example, climate change [38], crop or land prices [20,39-41] or management changes like the increased use of fertilizer and other production factors [40,42,43]. This diversity of factors reflects that in reality yield developments depend on numerous factors of various origins (e.g. economic, technological, ecological). The role of these different driving factors and their impacts will vary between regions and over time, but both aspects are not yet well understood. It is therefore important to assess the drivers for yield improvements at local scales. For example, de Wit et al. [44] find that agricultural yield developments in Europe in the past five decades are clearly correlated to agricultural policy. However, driving factors have not been investigated yet in other regions that are of critical importance in future biomass supply such as Latin America and Sub-Saharan Africa [45,46].
Although several studies have assessed the environmental impacts of bioenergy expansion while mitigating ILUC through agricultural intensification [47-52], often only the impacts of bioenergy production are considered. De Wit et al. [48] and Melillo et al. [47] assess

the net GHG impacts of bioenergy production and agricultural intensification, but do this on a European and global scale, respectively. Their GHG balances are not very detailed as they only account for nitrogen emissions, net soil organic carbon (SOC) fluxes and abated fossil emissions [47,48]. Several regional case studies investigated the GHG balance of bioenergy production in more detail and often also assessed other environmental impacts [49-52]. But, in most of these studies, the impacts of agricultural intensification, in terms of e.g. additional GHG emissions due to increased fertilizer use, are not or only partly taken into account [49,51] or assumed to be negligible because intensification only takes place through improved management [50]. Thus, regional case studies are needed that assess the GHG and other environmental impacts of both bioenergy expansion and agricultural intensification in detail.

PATHWAYS FOR AGRICULTURAL INTENSIFICATION

For agricultural intensification different pathways are possible. For example, with regard to crop yield improvements, a distinction can be made between conventional intensification and sustainable intensification. In the case of conventional intensification, yield increases are attained by applying more fertilizers, pesticides, irrigation and mechanization [33]. Sustainable intensification implies that yield improvements are achieved without increased input, but mainly through practices which improve the resource use efficiency, reduce environmental impacts and maintain or strengthen the capacity to continue producing food in the future [33,53,54]. With regard to livestock production, a distinction between intensification pathways can be made based on the existence of different production systems. To study global livestock production, production practices are generally categorized into three well-contrasted systems, i.e. pasture-based, mixed and industrial systems [55,56]. This means that intensification can take place within one system (incremental change) and through system transitions to more efficient and productive systems (transformational change, e.g. from pasture-based to mixed).

Different intensification pathways will have different impacts on, for example, GHG emissions and other environmental aspects. For instance, Valin et al. [33] assess the effects of different crop yield and livestock feed conversion efficiency scenarios on the GHG balance of agricultural production in developing countries. They find that when above-baseline gains in crop yield are attained by intensive fertilizer application, the global GHG savings compared to the baseline are about 450 $MtCO_2$-eq in 2050. In the case of sustainable intensification, i.e. through practices that improve crop yields without additional synthetic fertilizer, the emission savings are one third higher [33]. Also, with regard to livestock production, it is expected that intensification within one system and through system transitions will have different impacts on the GHG balance and on land occupation. But, although a large number of studies has investigated the

GHG performance of dairy and beef production systems, and to a lesser extent also the potential of GHG mitigation options [32,57], it is unclear yet how the impacts of the two development pathways compare to each other and how the results of such a comparison differ between regions.

Of the studies that assess the net GHG balance of bioenergy expansion and agricultural production, de Wit et al. [48] and Melillo et al. [47] only include one pathway for agricultural intensification. Van der Hilst et al. [52] consider two intensification pathways. The sustainable intensification pathway includes a few GHG mitigation measures like reduced tillage, but both pathways assume balanced fertilization of crops. All three studies exclude the GHG impacts of livestock intensification. It is thus not yet well known how different intensification pathways would affect the overall GHG balance of bioenergy expansion and agricultural production. In addition, the three studies exclude other environmental impacts of agricultural intensification (e.g. on biodiversity) [47,48,52]. However, agricultural intensification can have other environmental impacts that can be both positive and negative. These impacts are also highly dependent on the region in which intensification takes place. Thus, improved insight is needed in the regional GHG balance and other environmental impacts of different intensification pathways in both the crop and livestock sector.

ECONOMIC AND ENVIRONMENTAL PERFORMANCE OF BIOMASS VALUE CHAINS

Biomass use for energy and materials is projected to expand. Given limited biomass resources, more insight is needed into which biomass value chains are most promising for producing heat, power, fuels and materials in terms of their technological, economic and environmental performance. This requires: i) a detailed overview of the status and prospects of potential value chains; and ii) assessment and comparison of their economic and environmental performance on the short and long term. Assessment of the performance over time is important because biomass value chains are in different stages of development and have different potentials for improvement. For example, on the short term, new technologies may be more expensive than established technologies. But, as capacity deployment increases, with technological learning, they could become cheaper in the long term.

Although the economic and environmental performance of biomass value chains have been assessed widely in literature, earlier (review) work mainly considers bioenergy, and especially biofuels [58-61]. This literature generally considers either environmental or economic aspects [58,60-63]. In addition, most studies that consider biomaterials focus on environmental impacts (see e.g. [64-66]), while the number of economic assessments is limited [67,68]. Comparative work between bioenergy and biomaterials only includes

environmental aspects [69] or biomass use in the manufacturing sector [70]. However, as energy and material applications in different sectors are competing for biomass feedstocks, only an assessment that includes both their economic and environmental impact can generate better insights into the overall performance of the various biomass value chains.

1.4 RESEARCH QUESTIONS AND THESIS OUTLINE

Based on the knowledge gaps identified in the previous sections, the main aim of this thesis is to assess the regional potentials, the environmental impacts and economic performance of different pathways for agricultural intensification, biomass production and biomass use. To this end, the following research questions have been formulated:

1. What are regional possibilities for agricultural intensification and what are their impacts on the demand for agricultural land and biomass potentials?
2. What is the effect of different agricultural intensification pathways and biomass production on GHG emissions and the environment?
3. Which biomass value chains are preferred today as well as in the future with regard to their economic performance and GHG balance?

The research questions are addressed in Chapters 2 to 6. Table 1-1 presents an overview of the chapters and the research questions addressed. Chapter 2 analyzes the pace and direction of historical yield developments (1961-2010) for various crops and livestock products in seven countries in different world regions, and examines the technological, economic and institutional driving factors behind these developments. In addition, this chapter discusses how yield projections are defined in models that assess biomass potentials and impacts of biomass production, and how these projections can be improved based on the findings of the historical analysis. Chapter 3 investigates two pathways for intensification of cattle production systems and compares the impacts of these two pathways on farm gate GHG emissions, land occupation and LUC-related emissions (kg CO_2-eq per kg of milk or beef) in nine world regions. Chapter 4 examines the potentials for agricultural intensification and four other measures to mitigate ILUC in the province of Lublin, Poland. In addition, it quantifies how much miscanthus-based ethanol can be produced in this region with a low risk of causing ILUC. Chapter 5 builds on Chapter 4 and assesses how the implementation of agricultural intensification and other ILUC mitigation measures influences the GHG balance and other environmental impacts of agricultural and biofuel production in Lublin province. It specifically investigates the effects of three pathways for agricultural intensification in terms of sustainability (e.g. with regard to nutrient use efficiency and tillage practices). Chapter 6 reviews the status and prospects of biomass value chains for heat, power, fuels and materials, assesses their

current and long-term levelized production costs and avoided emissions, and compares their greenhouse gas abatement costs.

TABLE 1-1 | Overview of thesis chapters and the research question(s) addressed in them.

Chapter	Region(s)	Research questions		
		1	2	3
Chapter 2: Assessment of driving factors for yield and productivity developments in crop and cattle production as key to increasing sustainable biomass potentials	Australia, Brazil, China, India, USA, Zambia, Zimbabwe	x		
Chapter 3: Intensification pathways for beef and dairy cattle production systems: Impacts on GHG emissions, land occupation and land use change	Europe, Brazil, North America (detailed bottom-up data) 9 world regions (disaggregated global data and model results)	x	x	
Chapter 4: Bioethanol potential from miscanthus with low ILUC risk in the province of Lublin, Poland.	Lublin (Poland)	x		
Chapter 5: GHG emissions and other environmental impacts of ILUC mitigation	Lublin (Poland)		x	x
Chapter 6: Competing uses of biomass: Assessment and comparison of the performance of bio-based heat, power, fuels and materials	Global		x	x

The remainder of this PhD thesis is structured as follows:

Chapter 2 addresses research question 1 by assessing historical yield developments (1961-2010) and their driving factors for five major crops, beef and cow milk in Australia, Brazil, China, India, USA, Zambia and Zimbabwe. The driving factors are identified by comparing (temporal shifts in) historical yield trends to technological, economic and institutional developments in the country. In addition, this chapter discusses how yield projections are defined in models that assess biomass potentials and impacts of biomass production, and how these projections can be improved based on the findings of the historical analysis.

Chapter 3 addresses research questions 1 and 2 by assessing two intensification pathways in beef and dairy cattle production. It compares the impact of intensification within one system and of system transitions on GHG emissions without LUC, land occupation and LUC-related emissions. First, a review is conducted of bottom-up studies on farm gate emissions from dairy production in Europe and beef production in North America and Brazil. Then, a global data set on GHG emissions from cattle production is used to extrapolate the findings from this review to other regions. Finally, the Global Biosphere Management Model (GLOBIOM) is applied to perform a global assessment of land occupation and LUC-related emissions in nine world regions.

Chapter 4 addresses research question 1 by assessing how much biofuel can be produced with a low risk of causing ILUC by implementing agricultural intensification and four other ILUC mitigation measures in the Polish province of Lublin in 2020. It combines a top-down approach to define a reference scenario for agricultural production and land use in Lublin in 2020 and a bottom-up approach to calculate how the ILUC mitigation strategies can contribute to the availability of surplus agricultural land on which biomass can be produced without causing unwanted LUC. The case study focuses on bioethanol production from miscanthus.

Chapter 5 addresses research questions 2 and 3 by assessing how the implementation of ILUC mitigation measures and miscanthus-based ethanol production influences the net GHG balance of agricultural and bioethanol production in the province of Lublin in 2020. It specifically investigates three different agricultural intensification pathways in terms of sustainability and quantifies the effect of each pathway on the GHG balance and qualitatively assesses the influence on other environmental impacts.

Chapter 6 addresses research questions 2 and 3 by reviewing the status and prospects of biomass value chains for heat, power, fuels and materials and comparing the economic performance, GHG balance and GHG abatement costs of different biomass value chains and their fossil reference. The analysis of the economic performance and GHG balance includes literature data for both the present level of technology and projections for 2030.

Chapter 7 summarizes the findings from Chapters 2 to 6, provides answers to the research questions and gives recommendations for further research and policy.

REFERENCES

[1] IPCC. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change: Geneva, Switzerland; 2014.

[2] Smith P, Bustamante M, Ahammad H, et al. Agriculture, Forestry and Other Land Use (AFOLU). In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change . Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA.; 2014, p. 811-922.

[3] Chum H, Faaij A, Moreira J, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 209-332.

[4] Woods J, Lynd LR, Laser M, et al. Chapter 9. Land and Bioenergy. In: Souza GM, Victoria RL, Joly CA, Verdade LM, editors. Bioenergy & Sustainability: bridging the gaps. Scientific Committee on Problems of the Environment (SCOPE): Paris, France; 2015, p. 258-300.

[5] Piotrowski S, Carus M and Essel R. Global bioeconomy in the conflict between biomass supply and demand. nova-Institute GmbH: Hürth, Germany; 2015. nova paper #7 on bio-based economy 2015-09.

[6] Saygın D, Gielen DJ, Draeck M, et al. Assessment of the technical and economic potentials of biomass use for the production of steam, chemicals and polymers. Renewable and Sustainable Energy Reviews 2014;40:1153-1167.

[7] Fischedick M, Schaeffer R, Adedoyin A, et al. Mitigation potential and costs. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 791-864.

[8] IEA. Energy Technology Perspectives 2012: Pathways to a Clean Energy System. International Energy Agency: Paris, France; 2012.

[9] GEA. Global Energy Assessment - Toward a Sustainable Future. Cambridge University Press and the International Institute for Applied Systems Analysis: Cambridge, UK and New York, NY, USA and Laxenburg, Austria; 2012.

[10] WWF. The Energy Report: 100% Renewable Energy by 2050. WWF – World Wide Fund For Nature: Gland, Switzerland; 2011.

[11] Greenpeace. Energy [R]evolution: A Sustainable World Energy Outlook. Greenpeace International, The Global Wind Energy Council (GWEC) and European Renewable Energy Council (EREC): Amsterdam, The Netherlands and Brussels, Belgium; 2012.

[12] Dale BE, Anderson JE, Brown RC, et al. Take a Closer Look: Biofuels Can Support Environmental, Economic and Social Goals. Environmental science & technology 2014;48(13):7200-7203.

[13] Wicke B, Verweij P, van Meijl H, et al. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 2012;3(1):87-100.

[14] Plevin RJ, O'Hare M, Jones AD, et al. Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. Environmental science & technology 2010;44(21):8015-8021.

[15] Searchinger T, Heimlich R, Houghton RA, et al. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. Science 2008;319(5867):1238-1240.

[16] Tyner WE, Taheripour F, Zhuang Q, et al. Land use changes and consequent CO2 emissions due to US corn ethanol production: a comprehensive analysis. Center for Global Trade Analysis, Purdue University: West Lafayette, IN, USA; 2010.

[17] Al-Riffai P, Dimaranan B and Laborde D. Global trade and environmental impact study of the EU biofuels mandate. International Food Policy Research Institute: Washington, DC, USA; 2010.

[18] Hertel TW, Golub AA, Jones AD, et al. Effects of US Maize ethanol on global land use and greenhouse gas emissions: Estimating market-mediated responses. Bioscience 2010;60(3):223-231.

[19] Laborde D. Assessing the land use change consequences of European biofuels policies. International Food Policy Research Institute: Washington, DC, USA; 2011. Available from: http://trade.ec.europa.eu/doclib/docs/2011/october/tradoc_148289.pdf

[20] EPA. Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis. Environmental Protection Agency: Washington, DC, USA; 2010. EPA-420-R-10-006.

[21] Warner E, Zhang Y, Inman D, et al. Challenges in the estimation of greenhouse gas emissions from biofuel-induced global land-use change. Biofuels, Bioproducts and Biorefining 2014;8(1):114-125.

[22] Plevin RJ, Beckman J, Golub AA, et al. Carbon Accounting and Economic Model Uncertainty of Emissions from Biofuels-Induced Land Use Change. Environmental science & technology 2015;49(5):2656-2664.

[23] Verstegen JA, van der Hilst F, Woltjer G, et al. What can and can't we say about indirect land-use change in Brazil using an integrated economic - land-use change model? GCB Bioenergy 2015.

[24] Macedo IC, Nassar AM, Cowie AL, et al. Chapter 17. Greenhouse Gas Emissions from Bioenergy. In: Souza GM, Victoria RL, Joly CA, Verdade LM, editors. Bioenergy & Sustainability: bridging the gaps. Scientific Committee on Problems of the Environment (SCOPE): Paris, France; 2015, p. 582-617.

[25] Brinkman M, Wicke B, Gerssen-Gondelach S, et al. Methodology for assessing ILUC prevention. Copernicus Institute of Sustainable Development, Utrecht University: Utrecht, The Netherlands; 2014.

[26] van de Staaij J, Peters D, Dehue B, et al. Low Indirect Impact Biofuel (LIIB) Methodology - version Zero. 2012.

[27] Witcover J, Yeh S, Sperling D. Policy options to address global land use change from biofuels. Energy Policy 2013;56:63-74.

[28] Slade R, Saunders R, Gross R, et al. Energy from biomass: the size of the global resource. Imperial College Centre for Energy Policy and Technology and UK Energy Research Centre: Londen, UK; 2011.

[29] Dornburg V, van Vuuren D, van de Ven G, et al. Bioenergy revisited: Key factors in global potentials of bioenergy. Energy & Environmental Science 2010;3(3):258-267.

[30] Batidzirai B, Smeets EMW, Faaij APC. Harmonising bioenergy resource potentials—Methodological lessons from review of state of the art bioenergy potential assessments. Renewable and Sustainable Energy Reviews 2012;16(9):6598-6630.

[31] Tilman D, Balzer C, Hill J, et al. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences 2011;108(50):20260-20264.

[32] Havlík P, Valin H, Herrero M, et al. Climate change mitigation through livestock system transitions. Proceedings of the National Academy of Sciences 2014;111(10):3709-3714.

[33] Valin H, Havlik P, Mosnier A, et al. Agricultural productivity and greenhouse gas emissions: trade-offs or synergies between mitigation and food security? Environmental Research Letters 2013;8(3):9.

[34] van Vuuren DP, van Vliet J, Stehfest E. Future bio-energy potential under various natural constraints. Energy Policy 2009;37(11):4220-4230.

[35] Erb K, Haberl H, Krausmann F, et al. Eating the Planet: Feeding and fuelling the world sustainably, fairly and humanely– a scoping study. Commissioned by Compassion in World Farming and Friends of the Earth UK. Institute of Social Ecology: Vienna, Austria; 2009. Social Ecology Working Paper No. 116.

[36] Neumann K, Verburg PH, Stehfest E, et al. The yield gap of global grain production: A spatial analysis. Agricultural Systems 2010;103(5):316-326.

[37] Steinfeld H, Gerber P, Wassenaar T, et al. Livestock's long shadow: environmental issues and options. FAO: Rome, Italy; 2006.

[38] Jaggard KW, Qi A, Ober ES. Possible changes to arable crop yields by 2050. Philosophical Transactions of the Royal Society B: Biological Sciences 2010;365(1554):2835-2851.

[39] Rosegrant MW, Ringler C, Msangi S, et al. International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT): Model Description. International Food Policy Research Institute: Washington, D.C.; 2008. Available from: http://www.ifpri.org/sites/default/files/publications/impactwater.pdf

[40] Eickhout B, van Meijl JCM, Tabeau AA, et al. The Impact of Environmental and Climate Constraints on global food supply. Netherlands Environmental Assessment Agency (MNP) / Agricultural Economics Research Institute (LEI): Bilthoven / The Hague, The Netherlands; 2008.

[41] Khanna M, Crago CL, Black M. Can biofuels be a solution to climate change? The implications of land use change-related emissions for policy. Interface Focus 2011;1(2):233-247.

[42] Beach RH, Adams D, Alig R, et al. Model Documentation for the Forest and Agricultural Sector Optimization Model with Greenhouse Gases (FASOMGHG). RTI International: 2010. Available from: http://www.cof.orst.edu/cof/fr/research/tamm/FASOMGHG_Model_Documentation_Aug2010.pdf

[43] Mosnier A, Havlík P, Valin H, et al. The Net Global Effects of Alternative U.S. Biofuel Mandates: Fossil Fuel Displacement, Indirect Land Use Change, and the Role of Agricultural Productivity Growth — Nicholas Institute. Nicholas Institute for Environmental Policy Solutions: Durham, NC, USA; 2012. NI R 12-01. Available from: http://nicholasinstitute.duke.edu/climate/policydesign/net-global-effects-of-alternative-u.s.-biofuel-mandates

[44] de Wit M, Londo M, Faaij A. Productivity developments in European agriculture: Relations to and opportunities for biomass production. Renewable and Sustainable Energy Reviews 2011;15(5):2397-2412.

[45] Smeets EMW, Faaij APC, Lewandowski IM, et al. A bottom-up assessment and review of global bio-energy potentials to 2050. Progress in Energy and Combustion Science 2007;33(1):56-106.

[46] Hoogwijk M, Faaij A, Eickhout B, et al. Potential of biomass energy out to 2100, for four IPCC SRES land-use scenarios. Biomass and Bioenergy 2005;29(4):225-257.

[47] Melillo JM, Reilly JM, Kicklighter DW, et al. Indirect Emissions from Biofuels: How Important? Science 2009;326(5958):1397-1399.

[48] de Wit MP, Lesschen JP, Londo MHM, et al. Greenhouse gas mitigation effects of integrating biomass production into European agriculture. Biofuels, Bioproducts and Biorefining 2014;8(3):374-390.

[49] van Dam J, Faaij APC, Hilbert J, et al. Large-scale bioenergy production from soybeans and switchgrass in Argentina: Part B. Environmental and socio-economic impacts on a regional level. Renewable and Sustainable Energy Reviews 2009;13(8):1679-1709.

[50] van der Hilst F, Lesschen JP, van Dam JMC, et al. Spatial variation of environmental impacts of regional biomass chains. Renewable and Sustainable Energy Reviews 2012;16(4):2053-2069.

[51] Smeets EMW, Lewandowski IM, Faaij APC. The economical and environmental performance of miscanthus and switchgrass production and supply chains in a European setting. Renewable and Sustainable Energy Reviews 2009;13(6-7):1230-1245.

[52] van der Hilst F, Verstegen JA, Zheliezna T, et al. Integrated spatiotemporal modelling of bioenergy production potentials, agricultural land use, and related GHG balances; demonstrated for Ukraine. Biofuels, Bioproducts and Biorefining 2014;8(3):391-411.

[53] Garnett T, Appleby MC, Balmford A, et al. Sustainable Intensification in Agriculture: Premises and Policies. Science 2013;341(6141):33-34.

[54] Hochman Z, Carberry PS, Robertson MJ, et al. Prospects for ecological intensification of Australian agriculture. European Journal of Agronomy 2013;44:109-123.

[55] Robinson T, Thornton PK, Franceschini G, et al. Global livestock production systems. Food and Agriculture Organization of the United Nations (FAO) and International Livestock Research Institute (ILRI): Rome; 2011.

[56] Seré C, Steinfeld H and Groenewold J. World livestock production systems: Current status, issues and trends. Rome; 1996.

[57] Schader C, Jud K, Meier MS, et al. Quantification of the effectiveness of greenhouse gas mitigation measures in Swiss organic milk production using a life cycle assessment approach. Journal of Cleaner Production 2014;73:227-235.

[58] Gnansounou E, Dauriat A, Villegas J, et al. Life cycle assessment of biofuels: Energy and greenhouse gas balances. Bioresource technology 2009;100(21):4919-4930.

[59] Bauen A, Berndes G, Junginger M, et al. Bioenergy - A Sustainable and Reliable Energy Source: A Review of Status and Prospects. IEA Bioenergy: 2009. IEA Bioenergy ExCo:2009:06.

[60] Bain RL. World Biofuels Assessment, Worldwide Biomass Potential: Technology Characterizations. National Renewable Energy Laboratory: Golden, CO, USA; 2007. NREL/MP-510-42467.

[61] Obernberger I, Thek G and Reiter D. Economic Evaluation of Decentralised CHP Applications Based on Biomass Combustion and Biomass Gasification. BIOS Bioenergiesysteme GmbH: Graz, Austria; 2008.

[62] Larson ED. A review of life-cycle analysis studies on liquid biofuel systems for the transport sector. Energy for Sustainable Development 2006;10(2):109-126.

[63] von Blottnitz H and Curran MA. A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. Journal of Cleaner Production 2007;15(7):607-619.

[64] Hermann BG, Blok K, Patel MK. Producing Bio-Based Bulk Chemicals Using Industrial Biotechnology Saves Energy and Combats Climate Change. Environmental science & technology 2007;41(22):7915-7921.

[65] Weiss M, Haufe J, Carus M, et al. A Review of the Environmental Impacts of Biobased Materials. Journal of Industrial Ecology 2012;16:S169-S181.

[66] Ren T and Patel MK. Basic petrochemicals from natural gas, coal and biomass: Energy use and CO2 emissions. Resources, Conservation and Recycling 2009;53(9):513-528.

[67] Ren T, Daniëls B, Patel MK, et al. Petrochemicals from oil, natural gas, coal and biomass: Production costs in 2030–2050. Resources, Conservation and Recycling 2009;53(12):653-663.

[68] Hermann B and Patel M. Today's and tomorrow's bio-based bulk chemicals from white biotechnology. Applied Biochemistry and Biotechnology 2007;136(3):361-388.

[69] Dornburg V, Lewandowski I, Patel M. Comparing the land requirements, energy savings, and greenhouse gas emissions reduction of biobased polymers and bioenergy: An analysis and system extension of life-cycle assessment studies. Journal of Industrial Ecology 2004;7(3-4):93-116.

[70] Saygın D and Patel MK. Renewables for industry: An overview of the opportunities for biomass use. Utrecht University, Group of Science, Technology and Society / Copernicus Institute: Utrecht, Netherlands; 2010.

CHAPTER 2

Assessment of driving factors for yield and productivity developments in crop and cattle production as key to increasing sustainable biomass potentials

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ABSTRACT

The sustainable production potential of biomass for energy and material purposes largely depends on the future availability of surplus agricultural lands made available through yield improvements in crop and livestock production. However, the rates at which yields may develop, and the influence of technological, economic and institutional factors on these growth rates are key uncertainties in assessing the potentials and impacts of biomass production. This study analyzes the pace and direction of historical yield developments (1961-2010) of five major crops, beef and cow milk in Australia, Brazil, China, India, USA, Zambia and Zimbabwe, and examines the driving factors behind these developments. In addition, it explores how future yields are modeled and how modeling efforts may be improved. Average yield growth rates over the investigated period ranged in most cases between 0.7-1.6% yr^{-1} for crops, 1.0-1.5% yr^{-1} for milk and 0.4-0.8% yr^{-1} for beef (relative to 2010). The role of different drivers is region specific. Yet, supporting agricultural policies have played an important role in increasing yields in all countries, especially for crops. In cattle production, a key factor was the importance of commercial beef and milk production for the national or export market. Based on regional differences in drivers and yield developments, models that assess biomass potentials and impacts should take into account regional drivers, yield gaps and potential policy pathways.

2.1 INTRODUCTION

The use of biomass for energy, chemicals and materials is considered an important alternative to fossil resources [1,2]. For biomass to deliver a sizeable contribution, the availability of sufficient sustainable and affordable biomass feedstocks is crucial. Assessment studies that have evaluated the current and future global availability of biomass resources show that the largest future potential contribution can come from energy crops grown on various types of land [3,4]. The most important land type is surplus agricultural land, which can be released through increased production efficiency of food, animal feed or pasture. There is, however, disagreement about the availability of surplus agricultural land. Key uncertainties in predicting this area of land are technological progress in agricultural production systems and the related increases in crop and livestock yields [5-7]. Several studies have investigated the effects of yields on the availability of surplus agricultural land and biomass potentials and impacts. For example, van Vuuren et al. [8] assessed the impact of food crop yield changes on the global woody biomass potential in 2050. They found that an additional yield improvement of 12.5% compared to the baseline scenario resulted in an increase of the biomass potential from 150 to 230 EJ. Erb et al. [9] found that the biomass potential in 2050 would be 79 EJ yr^{-1} in the case of intermediate agricultural intensification and humane livestock rearing and 105 EJ yr^{-1} in the case of greater intensification of crop and livestock production[1]. Dornburg et al. [6] estimated that improvements in agricultural management could account for 140 EJ yr^{-1} of the total biomass supply potential of 500 EJ yr^{-1} in 2050. Slade et al. [5] derived from a review study that more than 1 Gha of high yielding agricultural land, equal to about 20% of the global agricultural land area in 2010, could be made available for bioenergy crops in 2050 if food crop yields increase at a higher rate than food demand and if the consumption of livestock products is limited.

The degree of yield improvements also affects the environmental performance of biomass production. Without sufficient improvements in yields, there is a large risk of direct or indirect land use change (DLUC and ILUC, respectively), which can result in high greenhouse gas (GHG) emissions [10,11]. In addition, advances in agricultural production systems may also improve the performance of the agricultural sector as a whole. For example, Tilman et al. [12] show that there is a significant potential in agriculture to reduce global land clearing, GHG emissions and nitrogen use through improved technology and adaption and transfer of high yielding technologies to underyielding regions. Also, Havlik et al. [13] show that the transition of livestock production toward more efficient systems would significantly decrease livestock-induced GHG emissions. These emission savings are mainly a result of a reduction in land use change [13].

Models that assess land availability, land use change induced by biomass demand and other impacts of biomass production, such as those used in the studies mentioned

1 Assuming continuation of current trends in diet and crop land area expansion

above, generally base their crop yield projections on historical developments. Many of these studies also account for (a limited number of) endogenous drivers of future yields. These factors are related to, for example, climate change [14], crop or land prices [15-18] or management changes like the increased use of fertilizer and other production factors [16,19,20]. This diversity of factors reflects that in reality yield developments depend on numerous factors of various origins (e.g. economic, technological, ecological). The question arises what role these different driving factors play, how they relate to each other and if their impact varies between regions. Moreover, productivity developments in the livestock sector have received much less attention in literature and modeling efforts than agricultural crops – despite the fact that livestock production accounts for 70% of the total agricultural land and one third of the arable land area is used for feed crop production [21]. For this sector, the lack of insight into the possibilities to increase yields, the rate at which this can be established and the role of different driving factors is even larger.

Recently, de Wit et al. [22] discussed what growth rates and maximum (sustainable) yields could be achieved in European agriculture. They assessed agricultural yield developments in the past five decades and compared these to policy developments, structural changes and trends in the use of production factors (inputs). De Wit et al. found that yield developments were clearly correlated to agricultural policy, but yield growth did not always coincide with more efficient use of inputs [22]. De Wit et al. focused on Europe and did not investigate other regions that are of critical importance in future biomass supply such as Latin America and Sub-Saharan Africa [3,4]. Given the importance of yield projections in determining biomass supply and impacts, the aim of this study is to assess for seven countries in different world regions (i.e., Australia, Brazil, China, India, USA, Zambia, and Zimbabwe):

i. what the historical agricultural developments and their drivers are,
ii. to what extent and at what growth rate crop and livestock product yields can improve in the future, and
iii. how different settings and drivers can influence future yield developments.

These insights contribute to several aspects identified as a key to improving the assessment of biomass potentials and impacts, such as 1) the use of bottom-up analyses to enhance the understanding of current (agricultural) systems, options for improvement, the degree to which yields can be increased, drivers, and regional differences, and 2) a more explicit discussion of assumptions (including yields) [7,23].

The remainder of this chapter is organized as follows. Section 2 presents the selection of agricultural products and countries for this study and describes how historical agricultural developments and future yield projections are assessed. Section 3 starts by discussing historical yield developments and their driving factors. This is followed by an assessment of yield projections in models and how these projections can be improved based on the

knowledge gained from the historical analysis. Section 4 offers a discussion of our work, and conclusions are drawn in section 5.

2.2 METHODS

2.2.1 Selection of agricultural products and producing countries

The potential area of surplus agricultural land is expected to be largely influenced by efficiency developments in the production of major agricultural products. Therefore, the agricultural developments are assessed for five crops that are most dominant in terms of global production and cultivated area: wheat, corn, rice, sugarcane and soybean [24]. In addition, for livestock we only take into account beef and cow milk production since cattle uses most of the agricultural area for grazing. Pig and chicken production are often landless, but land is required for producing feed crops [25]. The area of cropland needed largely depends on the crop yields, which are already taken into account in this study. Therefore, pig and chicken production are not included in the assessment.

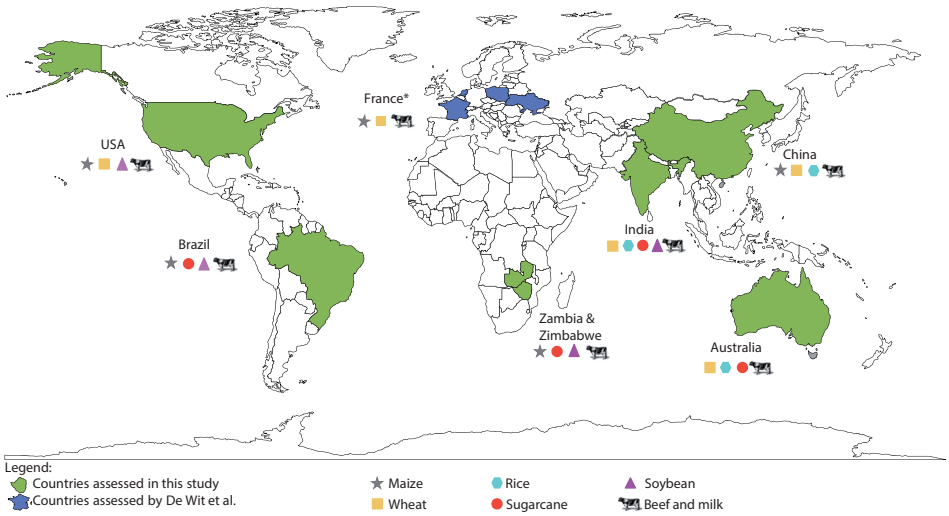

FIGURE 2-1 | Selected countries and agricultural products

* France was earlier assessed by de Wit et al. [22], but for comparison, data for France are also presented in this study.

Agricultural developments are assessed in seven countries: Australia, Brazil, China, India, USA, Zambia, and Zimbabwe (Figure 2-1). There are two reasons for this selection. First, Brazil, China, India, and the USA are major producers of the selected crops and cattle products [24]. Second, Australia, USA, Zambia, and Zimbabwe can potentially release a large area of agricultural land for biomass feedstock production [3,4]. De Wit et al. [22] assessed agricultural developments in France, The Netherlands, Poland and Ukraine. For

comparison, we have included data about France in the results section. In addition, we compare the general findings from de Wit et al. with our own results.

2.2.2 Historical developments in driving factors

The analysis starts with a description of the current status of agriculture in the selected countries and developments in driving factors that have taken place since 1961 (this part of the study is presented in appendices A2.1 to A2.8). Based on literature, the drivers of yield developments are classified into three types [22,26-30]: technological/management, economic, and institutional. Economic drivers are, for example, market developments and agricultural R&D investments. Institutional drivers include agricultural policies and governance systems. The discussion of economic and institutional drivers is based on literature review. For technological/management drivers the following are assessed: labor intensity and level of mechanization, irrigation, nutrient and pesticide use. These indicators are derived from time series data (1961-2010) collected from the UN Food and Agricultural Organization statistical division (FAOSTAT [24]). These statistics are aggregated on a country level, e.g. annual national consumption of fertilizers (tonne yr^{-1}). To enable comparison of the management levels between countries, the factors are expressed in average intensity per hectare of agricultural land, e.g. the national number of tractors used divided by the total area of agricultural land. For cattle, another indicator for the management level or production intensity is the proportion of ruminants to the area of meadows and pastures (hereafter the ruminant density). This is also derived from FAO statistics. It is not chosen to evaluate the cattle density, because this neglects the importance of other ruminants in the occupation of meadows and pastures for grazing and hay. As a result of the feed requirements varying between ruminant species, the number of ruminants is expressed in livestock units (LU), where one unit represents the energy requirements for maintenance and production of a typical cow in North America. The livestock unit coefficients are obtained from the FAO [31]. The ruminant density is then calculated as the number of livestock units per hectare of meadows and pastures. The ruminants included are: buffaloes, camel, cattle, goats and sheep. The area of meadows and pastures consists of the total land area available for both for grazing and for the production of conserved forages. This approach thus also accounts for systems that combine grazing and confinement. In this study, we only consider a limited number of drivers that can be actively steered. Literature shows that more factors can influence yields, these are discussed in section 2.4.

2.2.3 Historical yield and productivity developments

To assess historical yield trends and yield growth rates for the selected products and countries, time series data (1961-2010) are collected from FAOSTAT [24]. The crop yield is defined as the annual production quantity per hectare of area harvested (tonne ha^{-1}

yr^{-1}); the beef yield is given in terms of carcass weight (kg $animal^{-1}$); the milk yield is the annual milk production per cow (kg $animal^{-1}$ yr^{-1}). All numbers are national averages. The average beef and milk production per animal, however, are not the best indicators to study developments in livestock product yields. Beef and milk production can take place in different production systems ranging from pastoral to landless. But intensification does not always lead to higher beef or milk production per animal (e.g. because faster weight gain leads to shorter lifespans). Therefore, a better parameter for milk and beef yields would be the feed conversion efficiency (FCE, kg animal product per kg feed intake). The use of the FCE, however, has also limitations. These are discussed in section 2.4.

Average annual yield growth rates are obtained by applying linear regression to the historical yield data and are presented per product, per country, per decade and for the entire period. Growth rates are both expressed in absolute growth per year (e.g. t ha^{-1} yr^{-2}) and in percentage per year (simple annual growth rate, relative to the initial year). Temporal shifts are identified for each product on country level and differences between products within a country are described. Explanations are sought by comparing the observed changes with the technological/management, economic, and institutional developments. In addition, developments in productivity of the total agricultural sector and of the total livestock sector are assessed and discussed. This productivity is defined as the proportion of aggregated outputs to aggregated inputs (output-input ratio). To derive the aggregated inputs and outputs, the trend of all inputs and outputs in physical units is calculated as an index (base year 1961). From these indices, an (unweighted) average of all inputs and all outputs is calculated for each year. For the agricultural sector the included inputs are: agricultural land, fertilizer and tractors; the outputs are: crops, meat, milk and eggs. For the livestock sector, the inputs are: feed crops, meadow and pasture land; the outputs are: meat, milk and eggs.

2.2.4 Future yield projections and the role of driving factors

Yield growth rates from projections in literature and models are compared to linear extrapolation of historical trends. It is discussed how yield projections are defined and how they can be improved based on the findings on historical driving factors. For each country, key factors are identified that may stimulate or limit future yield developments. To better understand what possible pathways could be defined for future yield developments, also the magnitude of yield gaps is taken into account. The yield gaps are derived from current yield levels and data on maximum attainable rain-fed or irrigated yields in 2020 as derived from the Global Agro-Ecological Zones database for the IPCC SRES B1 Scenario from the Australian Commonwealth Scientific and Research Organization (CSIRO) Mark 2 Mode [32].

2.3 RESULTS

2.3.1 Yield and productivity developments

For each country, historical developments in agricultural inputs and yields are assessed. For each country, the status of agriculture and the developments in the different driving factors are discussed in detail in appendices A2.1-A2.8. Here a synthesis of historical yield developments and driving factors is presented for the case of Zimbabwe. Syntheses for the other countries can be found in appendices A2.2 to A2.7. The key findings for each country are presented and compared in section 2.3.1.2.

2.3.1.1 Example: developments and driving factors in Zimbabwe

Zimbabwe's first green revolution by commercial farmers [33,34] is clearly represented by the increase of irrigated area and fertilizer use in the 1960s and early 1970s (Figure 2-2). The yields of corn and soybeans only started to improve from the second half of the 1960s (Figure 2-3), which coincides with the shift from tobacco production to other crops because of export sanctions imposed in 1965 [34,35] (also see appendix A2.8). During the 1970s, corn yields declined again, while soy yields only fell down in 1979. In addition, irrigation levels stagnated and fertilizer use dropped in the 1970s. Thus, management conditions seem to be affected by the civil war [36] and corn production suffered more from this than soybean cultivation. Sugarcane yields appear to be even less affected by economic and political changes in the 1960s and 1970s. Apart from significant fluctuations, sugarcane yields have increased from 1960 until 1986. The improvement rate of 0.9% yr^{-1} in these years, however, is significantly lower compared to a growth rate of 3.1% yr^{-1} for corn and 16.9% yr^{-1} for soybeans between 1960 and 1980 (all relative to 1961), also see Table 2-1.

The introduction of smallholder support in the early 1980s led to a second green revolution [34], but this is not clearly reflected in the statistics. A major factor is the severe drought in 1983, resulting in a significant drop of corn and soybean yields. Good climate conditions in 1985 led to high yields [34]. But due to the reduction of smallholder support, and maybe the decline of agricultural R&D as well [34], fertilizer use and crop yields declined in the late 1980s and the 1990s. Remarkably, the area under irrigation increased in the same period. Due to a severe drought, crop yields plummeted in 1992 [34]. With the fast-track land reform in 2000 [37], yields and input levels dropped further and continued declining in the following years. Overall, crop yields and also agricultural productivity (Figure 2-4) have fluctuated considerably throughout the period 1961-2010.

Considering cattle production, beef yields were stable in the 1960s and early 1970s but declined during the civil war in the late 1970s (see Figure 2-14 in appendix A2.9). In the 1980s and 1990s, the yields improved at an average rate of 1.8% yr^{-1} (relative to 1981), but dropped temporarily during the drought of 1992.

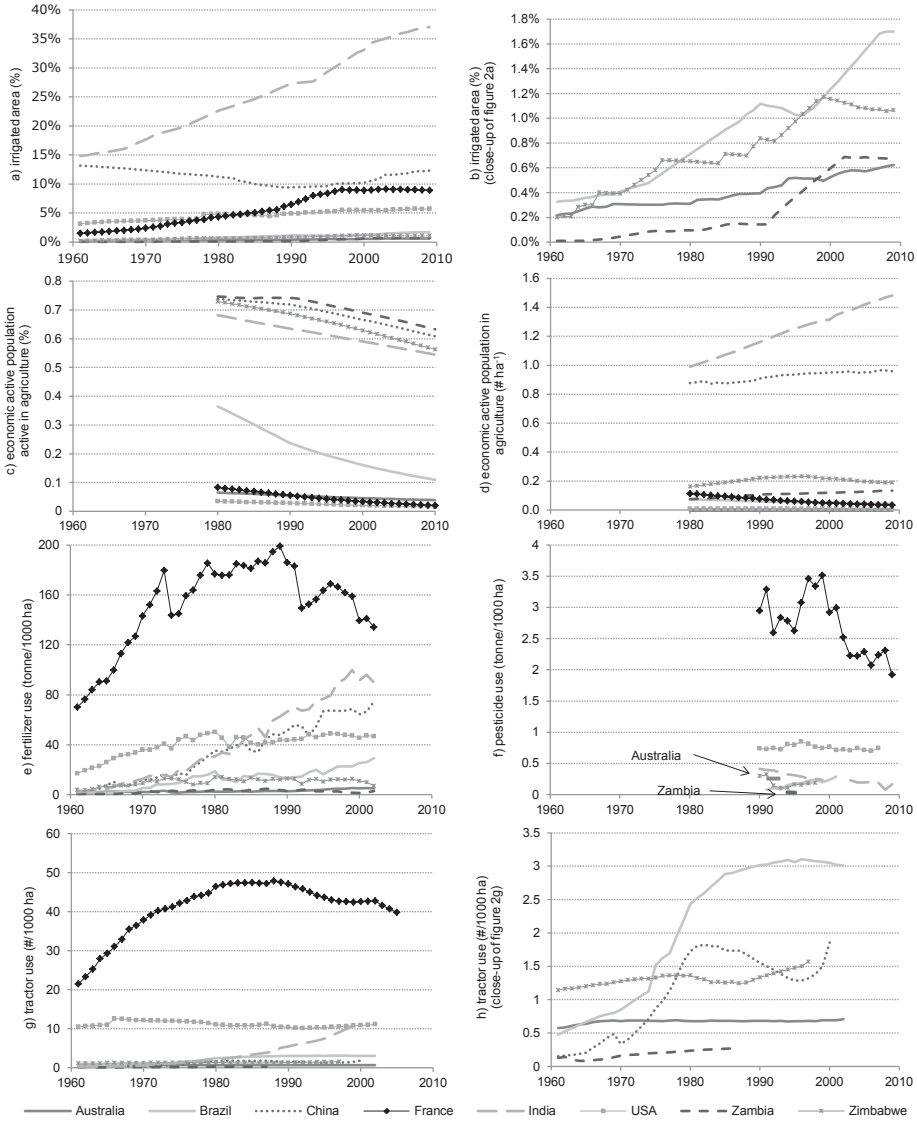

FIGURE 2-2 | Development in agricultural inputs.

All parameters are calculated from FAOSTAT data [24] according to the following definitions:
a) agricultural area equipped for irrigation = total area equipped for irrigation /agricultural area
b) close-up of panel a, presenting a selection of the data to reveal differences between the countries at the lowest irrigation levels.
c) labor share = total economically active population in agriculture / total economically active population
d) labor intensity= total economically active population in agriculture / agricultural area
e) fertilizer = total fertilizers / agricultural area
f) pesticides = (insecticides total + herbicides total + fungicides & bactericides total) / agricultural area
g) tractors = agricultural tractors / agricultural area
h) close-up of panel g, presenting a selection of the data to reveal differences between the countries at the lowest levels of tractor use.
note: when no data is shown for a certain country and/or year, no data is available for this country or year
note: for tractors, the capacity or size of machinery is not taken into account

According to the FAO statistics, beef yields have stagnated since the hyper-inflation and outbreak of foot-and-mouth disease [38,39]. Between 1960 and 1990, milk yields increased at a very low rate of 0.2% yr^{-1} (relative to 1961). Thus, it seems that technological improvements were limited while economic and political changes did not significantly affect milk yields either. After a small drop in 1992, yields peaked in 1993 and have stabilized since the late 1990s.

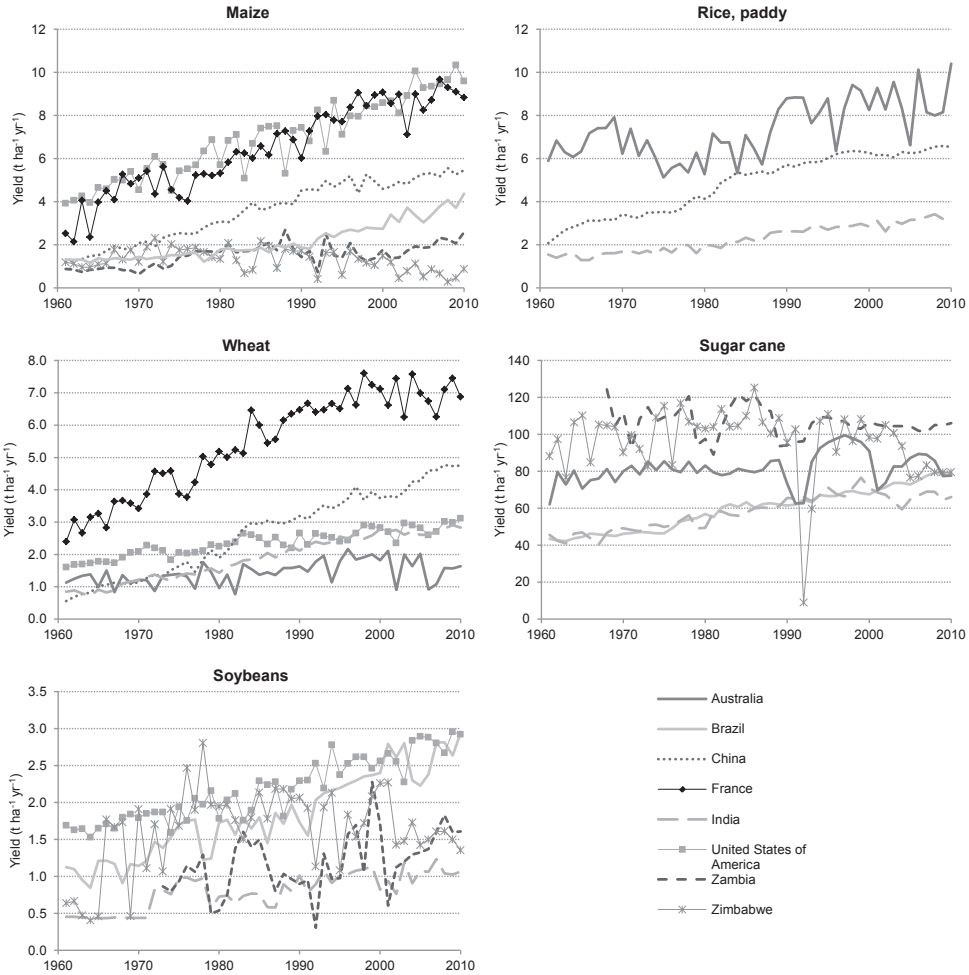

FIGURE 2-3 | Historical yield developments (1961-2010) for the crops corn, paddy rice, wheat, sugarcane and soybeans (FAOSTAT [24]).

TABLE 2-1 | Absolute and relative growth in crop, beef and milk yields for the period 1961-2010 and per decade (based on FAO statistics).

Product	Country	Production Mt (kt)	Yield t ha^{-1} yr^{-1} / kg animal^{-1} yr^{-1}		Average annual yield change kg ha^{-1} yr^{-2} / kg animal^{-1} yr^{-2} % yr^{-1}						
					Per decade[c,d]					Period	
		2010	1961	2010	1961-1970	1971-1980	1981-1990	1991-2000	2001-2010	1961-2010	1961-2010[e]
Corn	Brazil	56.1	1.3	4.4	12	21	23	85	101	55	
					0.9%	1.5%	1.3%	4.0%	3.2%	7.0%	1.6%
	China	177.5	1.2	5.5	95	111	116	24	79	93	
					8.0%	5.4%	3.6%	0.5%	1.6%	7.3%	1.6%
	France	14.0	2.5	8.8	325	33	83	179	101	131	
					13.4%	0.7%	1.4%	2.4%	1.2%	4.1%	1.4%
	USA	316.2	3.9	9.6	132	84	63	159	156	117	
					3.3%	1.6%	1.0%	2.2%	1.8%	2.9%	1.2%
	Zambia	2.8	0.9	2.6	-11	102	4	-4	117	25	
					-1.2%	11.2%	0.2%	-0.3%	8.3%	2.8%	1.2%
	Zimbabwe	1.2	1.2	0.9	54	-57	33	16	-39	-15	
					5.4%	-2.9%	2.4%	1.4%	-4.3%	-0.9%	-1.6%
Rice, paddy	Australia	0.2	5.9	10.4	127	-157	172	24	27	62	
					2.1%	-2.4%	2.8%	0.3%	0.3%	1.1%	0.7%
	China	197.2	2.1	6.5	127	105	108	83	56	93	
					5.5%	3.3%	2.3%	1.5%	0.9%	3.6%	1.3%
	India	120.6	1.5	3.3	18	30	80	30	48	43	
					1.2%	1.8%	4.2%	1.1%	1.6%	3.4%	1.3%
Soybeans	Brazil	68.8	1.1	2.9	8	18	15	70	20	37	
					0.8%	1.3%	0.9%	3.8%	0.8%	4.1%	1.4%
	India	12.7	0.5	1.1	-2	13	22	16	19	14	
					-0.4%	1.8%	3.4%	1.8%	2.0%	2.9%	1.2%
	USA	90.6	1.7	2.9	22	19	25	22	42	27	
					1.4%	1.1%	1.3%	0.9%	1.7%	1.8%	1.0%
	Zambia	0.0	0.9[a]	1.6	0	-33	-40	134	100	17	
		(41.0)			0.0%	-4.0%	-3.0%	19.8%	11.0%	2.0%	1.2%
	Zimbabwe	0.1	0.6	1.4	130	114	43	46	-45	12	
		(57.3)			30.0%	8.5%	2.5%	2.9%	-2.5%	0.9%	0.6%
Sugarcane	Australia	31.5	62.2	77.7	947	-3	211	3,475	811	221	
					1.3%	0.0%	0.3%	4.8%	1.0%	0.3%	0.3%
	Brazil	716.2	43.4	79.2	399	1,239	439	671	1,081	775	
					0.9%	2.8%	0.7%	1.1%	1.5%	1.9%	1.0%
	India	277.8	45.6	66.1	530	329	738	930	94	593	
					1.2%	0.7%	1.3%	1.4%	0.1%	1.4%	0.8%
	Zambia	4.1	124.4[b]	106.1	-6,198	-27	-344	747	4	-152	
					-5.2%	0.0%	-0.3%	0.7%	0.0%	-0.1%	-0.1%
	Zimbabwe	2.8	88.3	79.5	1,071	1,385	-649	4,972	-2,897	-270	
					1.2%	1.5%	-0.6%	7.5%	-2.9%	-0.3%	-0.3%

TABLE 2-1 | Continued

Product	Country	Production Mt (kt)	Yield t ha^{-1} yr^{-1} / kg animal^{-1} yr^{-1}		Average annual yield change kg ha^{-1} yr^{-2} / kg animal^{-1} yr^{-2} % yr^{-1}						
					Per decade[c,d]					Period	
		2010	1961	2010	1961-1970	1971-1980	1981-1990	1991-2000	2001-2010	1961-2010	1961-2010[e]
Wheat	Australia	22.1	1.1	1.6	-7	16	42	42	-27	12	
					-0.6%	1.4%	3.3%	2.6%	-1.6%	1.0%	0.7%
	China	115.2	0.6	4.7	65	82	90	79	126	88	
					10.0%	6.5%	3.7%	2.4%	3.0%	17.0%	1.8%
	France	38.2	2.4	6.4	114	90	139	97	-9	98	
					4.3%	2.2%	2.7%	1.5%	-0.1%	3.2%	1.3%
	India	80.7	0.9	2.8	43	26	58	45	19	47	
					5.8%	2.0%	3.5%	2.0%	0.7%	6.2%	1.5%
	USA	60.1	1.6	3.1	49	6	-3	51	46	25	
					3.1%	0.3%	-0.1%	2.1%	1.8%	1.5%	0.8%
Beef	Australia	2.1	150	254	1.7	-0.9	4.0	1.5	2.5	2.1	
					1.1%	-0.5%	2.3%	0.7%	1.1%	1.4%	0.8%
	Brazil	7.0	192	238	0.1	-2.4	-0.1	3.2	4.1	0.8	
					0.0%	-1.2%	0.0%	1.6%	2.1%	0.4%	0.4%
	China	6.2	97	141	0.1	0.2	5.4	-2.4	1.3	1.2	
					0.1%	0.2%	5.8%	-1.6%	1.0%	1.3%	0.8%
	France	1.5	186	296	1.1	2.7	3.9	0.1	1.7	2.7	
					0.6%	1.3%	1.7%	0.0%	0.6%	1.5%	0.9%
	India	1.1	80	103	0.0	0.9	1.1	0.1	0.0	0.6	
					0.0%	1.1%	1.3%	0.1%	0.0%	0.8%	0.6%
	USA	12.0	215	341	4.2	0.3	3.4	2.0	2.1	2.7	
					2.0%	0.1%	1.3%	0.7%	0.6%	1.3%	0.8%
	Zambia	0.1	190	160	-0.6	-2.7	0.0	0.0	0.0	-0.7	
		(60.8)			-0.3%	-2.0%	0.0%	0.0%	0.0%	-0.4%	-0.4%
	Zimbabwe	0.1	167	225	0.0	-2.4	6.2	3.8	0.0	1.6	
		(99.6)			0.0%	-1.4%	4.1%	2.1%	0.0%	1.1%	0.7%
Cow milk	Australia	9.0	1,985	5,810	93	7	113	80	91	75	
					4.7%	0.3%	3.8%	1.9%	1.7%	4.0%	1.4%
	Brazil	30.7	707	1,340	9	-7	8	51	19	12	
					1.2%	-0.9%	1.2%	6.9%	1.6%	2.0%	1.0%
	France	23.3	2,671	6,278	73	47	114	93	30	84	
					2.8%	1.5%	3.0%	1.8%	0.5%	3.5%	1.3%
	China	36.0	1,208	2,882	6	55	-43	6	80	29	
					0.5%	4.6%	-2.3%	0.4%	3.5%	3.0%	1.2%
	India	50.0	424	1,284	2	7	19	27	36	17	
					0.4%	1.4%	3.4%	3.7%	3.8%	6.0%	1.5%
	USA	87.5	3,307	9,595	125	95	136	147	147	128	
					3.8%	2.1%	2.5%	2.1%	1.8%	4.1%	1.4%
	Zambia	0.1	300	300	0	0	0	0	0	0	
		(88.5)			0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Zimbabwe	0.4	406	430	-1	3	0	1	0	1	
					-0.3%	0.7%	-0.1%	0.1%	0.0%	0.2%	0.2%

Negative growth in grey; [a] yield in 1973; [b] yield in 1968; [c] the average annual yield change in terms of percentage is given relative to the first year of the selected period, i.e. the average growth rate is expressed as a percentage of the estimated yield in the first year of selected period (e.g., from 1971-1980, the average annual growth of corn yield in Brazil was 1.5% of the yield level in 1971 as derived from linear regression); [d] note that the yield growth rates per decade are calculated for a limited time period of 10 years. This means that the choice for a certain timeframe and outliers in the data can have significant influence on the result of the linear regression. The use of longer timeframes may show a very different trend in yield development; [e] relative to 2010.

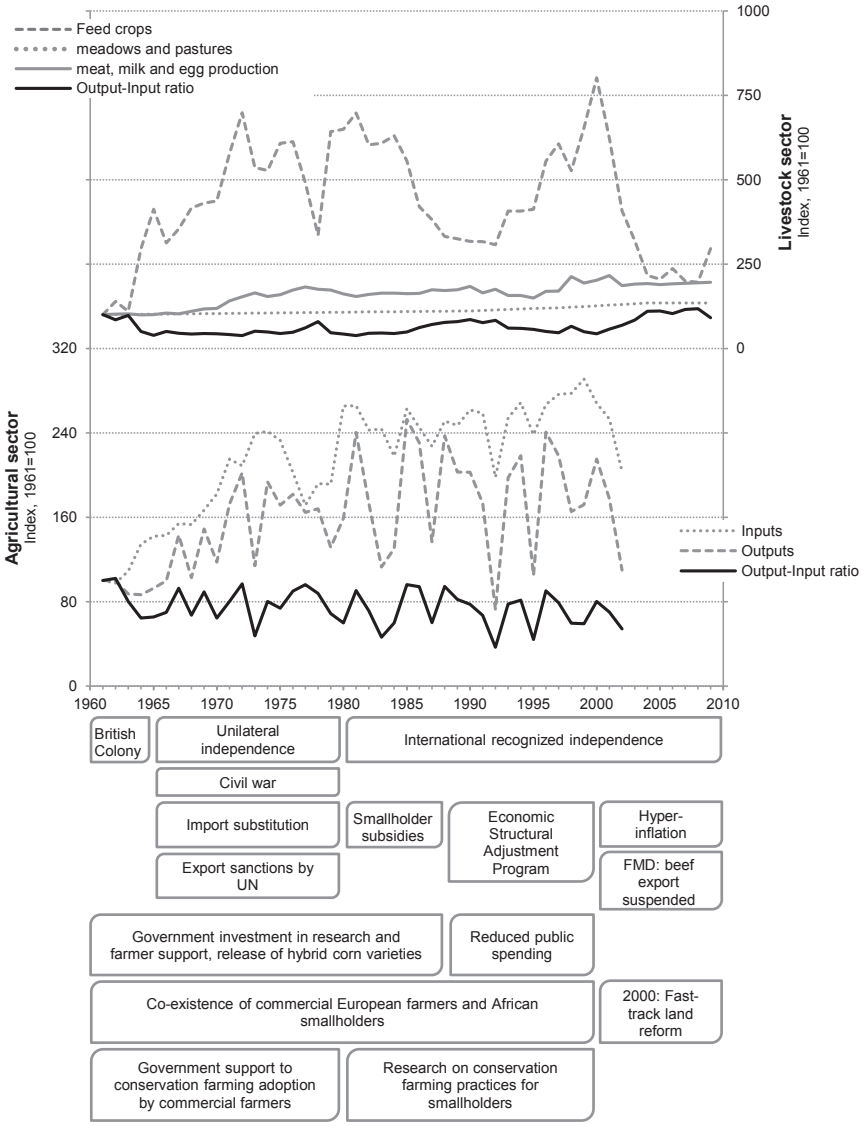

FIGURE 2-4 | Productivity developments in the Zimbabwean agricultural and livestock sector and institutional, economic and technological/management developments. Because of limited data, agricultural tractors are not included in the inputs and in the output-input ratio for the agricultural sector. (FMD: foot-and-mouth disease).

2.3.1.2 Summary and comparison between countries

Over the past five decades, most crop yields showed an upward trend (Table 2-1). The yield growth rates, however, varied significantly between regions. Average yield growth rates over the period investigated (1961-2010) ranged in most cases between 0.7-1.6% yr^{-1} for crops, 1.0-1.5% yr^{-1} for milk and 0.4-0.8% yr^{-1} for beef (all relative to 2010). Highest rates were found for wheat in China (1.8% yr^{-1}), milk in India (1.5% yr^{-1}) and beef in France (0.9% yr^{-1}). The lowest rates for a crop are found for sugarcane (-0.3 to 1.0% yr^{-1}), for any one country the rates are lowest for Zimbabwe (-1.6 to 0.7% yr^{-1}). For comparison, in the European countries studied by de Wit et al. [22], average growth rates of wheat are 1.0% yr^{-1} for Poland to 1.3% yr^{-1} for France (relative to 2010). This is lower compared to wheat growth rates in China and India, but higher compared the figures for wheat in Australia and the USA. Absolute wheat yield growth in France and the Netherlands (approximately 100 kg ha^{-1} yr^{-2} [22]), however, was higher compared to the four countries producing wheat in the present study. For beef, the absolute growth in France and Poland is comparable to the USA, but average growth rates in these European countries are higher than in all non-European countries assessed in this study (0.9% yr^{-1} for France and 1.2% yr^{-1} for Poland). Absolute and relative yield growth figures for beef in the Netherlands are comparable to Brazil.

In this study, the most observed trend over five decades for crop yield growth is linear. This is in accordance with other studies [40-42]. Yet, in each case, the analysis revealed periods during which yields improved at a higher rate compared to the long-term average as well as periods during which yield growth rates were lower than this average. In each case, technological as well as economic and institutional factors have played a role and these drivers often influenced each other. Yet, the importance and the effect of a driving factor varied from case to case. Table 2-2 gives an overview of the most important factors behind yield and productivity developments in each country (for more details, see appendices A2.2 to A2.8; for France see de Wit et al. [22]).

Improvements in agricultural technology and management have often led to considerable yield growth (Figure 2-2 and Figure 2-3). Especially in China and India, large scale adoption of new technologies (including high yielding crops) resulted in high average yield growth rates (appendices A2.4 and A2.5). In France and the USA, improved technologies resulted in considerable absolute yield improvements (appendix A2.6). The technological improvements, however, included a significant increase in the use of inputs like fertilizer and often caused a decline in agricultural productivity (Figure 2-5). In the cattle sector, yield improvements were often achieved through the increased use of feed crops. In Australia and Zimbabwe, however, the consumption of feed crops grew faster than the production of meat and milk, which led to a reduction in the productivity of the livestock sector. Other countries, like India and the USA were able to compensate for the higher input levels by increasing the production output levels of the livestock sector at a similar or even higher rate.

Economic factors often play a vital role in the improvement of agricultural technology. Investments in R&D enabled the development of new technologies. In most countries, these were mainly public investments. In the USA, also private investments were very important (appendix A2.6). These investments had already started in the period of industrial and agricultural protectionism. Only in Australia, industrial protectionism indirectly biased the agricultural sector and hindered improvements in production practices and crop yields (appendix A2.2). The introduction of economic liberalization provided an incentive for many farmers to (further) improve yields and increase or stabilize agricultural productivity. In Australia, yields and yield improvement rates improved quickly after liberalization started. In Brazil and the USA, yield improvement rates reduced in the first instance but increased again after about ten years (appendices A2.3 and A2.6). Agricultural production in China diversified after the reforms and yield improvements of predominant crops slowed down (appendix A2.4). Commercial farmers in Zambia profited from liberalization as they were able to improve soybean yields at high rates, while corn yields of smallholders decreased (appendix A2.7). For the cattle sector, the importance of commercial beef and milk production for the domestic or export market was found to be a key factor for yield improvements. In Australia and the USA, such markets already existed during the period of protectionism, while the beef market in Brazil has especially grown after economic liberalization (appendices A2.2, A2.3 and A2.6). In France (and the EU as a whole), dairy markets got saturated and a quota on milk production was introduced. The number of dairy cows reduced significantly, but milk yields continued to increase through improved management [43]. This, however, led to a reduction in livestock productivity in terms of the output-input ratio.

In accordance with the findings of de Wit et al. [22] for European countries, policies are found to be an important instrument in steering changes in the agricultural sector in the seven countries investigated in the present study. New technologies could be adopted by farmers because of farmer support programs, e.g. in India (appendix A2.5). Market liberalization policies created new markets for agricultural products, e.g. in Australia (appendix A2.2). In some cases, yield improvements have been attained by policies that were focused on a specific commodity: for example in Zambia, the focus of policies on corn production during a long period resulted in significantly higher yield growth rates for corn compared to other crops (appendix A2.7). In Brazil, the ProÁlcool program positively affected sugarcane yields (appendix A2.3). The import substitution policy for edible oils in India especially stimulated the increase of soybean yields. In contrast to the successful implementation of policies in the above examples, Zimbabwe also shows the impact of a lack of good governance and stimulating policies. A civil war in the 1970s and economic reforms around the 1990s disrupted agricultural production and the economy. Due to the long-term unstable situation, crop yields and agricultural productivity have fluctuated heavily.

TABLE 2-2 | Key driving factors behind historical yield and productivity developments[a].

	Driving factors	Effect on other driving factors	Effect on crops	Effect on cattle
Australia	•Market reforms, trade liberalization, opening of export markets		+ Rice and wheat yields + Agricultural productivity	+/- Beef and milk yields
	•Cattle: growth of export markets	+ Cattle management and technology		+ Milk and beef yields - Livestock productivity
	•Introduction of agri-environmental policies	+ Agricultural production, fertilizer use, irrigation	+/- Agricultural productivity	+ Decline in livestock productivity slowed down
Brazil	•Industrial protection and fertilizer subsidies	+ Fertilizer use	+ Corn and soybean yields - Agricultural productivity	+/- Beef and milk yields
	•Economic reforms	- & +/- Fertilizer use	- Corn and soybean yield growth rates +/- Agricultural productivity	+/- Beef and milk yields
	•Opening of agricultural export markets	+ Fertilizer use	+ Yield growth rates corn and soybean	+ Beef and milk yield growth rate + Livestock productivity increase
	•ProÁlcool program	+ Tractor use, irrigation	+ Sugarcane yields	
China	•Agricultural reforms, public investments in infrastructure and R&D	+ irrigation, fertilizer use, mechanization	+ Yields	
	•Economic reforms	+ Agricultural diversification	- Yield growth rates	+ Beef and milk yields
	•Increased consumption of milk and dairy products			+ Milk yields
France	•Protection of agricultural markets, stimulation of mechanization and fertilizer use	+ Inputs	+ Yields - Agricultural productivity	+ Milk and beef yields
	•Stimulation of modernization and scaling-up, land reforms	+/- Fertilizer use	+ Yields + Agricultural productivity	+ Milk and beef yields
	•Shift to high-yielding crops		+ Yields + Agricultural productivity	
	•Agri-environmental policy, reform of farmer support programs and stimulation of organic farming	- Inputs	- Yield growth rates	- Yield growth rates
	•Quotation milk production			- Livestock productivity

TABLE 2-2 | Continued

	Driving factors	Effect on other driving factors	Effect on crops	Effect on cattle
India	•Public investments & subsidies	+ Inputs	+ Yields - Agricultural productivity	
	•Increased milk consumption			+ Milk yields
USA	•Investment in R&D, biotechnology	+ Livestock technology and management	+ Yields	+ Beef and milk yields
	•Trade liberalization and reform of farmer support policies		- Yield growth rates (during reforms) + Yield growth rates (after reforms)	
	•Growing milk market			+ Milk yields
	•Agri-environmental programs	- Fertilizer use	+ Agricultural productivity	
Zambia	•Fertilizer subsidies	+ Fertilizer use	+ Corn yields	
	•Economic liberalization, elimination of fertilizer subsidies	- Fertilizer use + Irrigation	- Corn yields + Soybean yields	
	•Conservation farming technologies		+ Agricultural productivity	
	•Fertilizer Support Program	+ Fertilizer use +/- Irrigation	- Agricultural productivity + Corn yields	
Zimbabwe	•R&D (commercial farmers)	+ Irrigation + Fertilizer use		
	•Civil war	- Irrigation - Fertilizer use	- Yields	- Beef yields
	•Economic reforms, reduction of smallholder support and of agricultural R&D	- Fertilizer use	- Yields	
	•Economic crisis and fast track land reform	- Inputs	- Yields	
	•FMD and ban on beef export			- Beef yields

[a] Effect: +, increase; - decrease; +/- stabilization
FMD, Foot-and-Mouth Disease

In addition to agricultural policies aimed at economics and production, most countries have also introduced agri-environmental policies which aimed at, for example, enhanced quality of degraded agricultural lands (Australia, China, Zambia, Zimbabwe), balanced use of inputs (China, France) and controlled use and management of natural resources (Australia, India, USA). In the USA, China and Zambia, this led to improved agricultural productivity (appendices A2.4, A2.6 and A2.7). In Australia the productivity did not improve considerably compared to previous years (appendix A2.2). In India, the productivity continued to decline due to weak enforcement of agri-environmental policies (appendix A2.5).

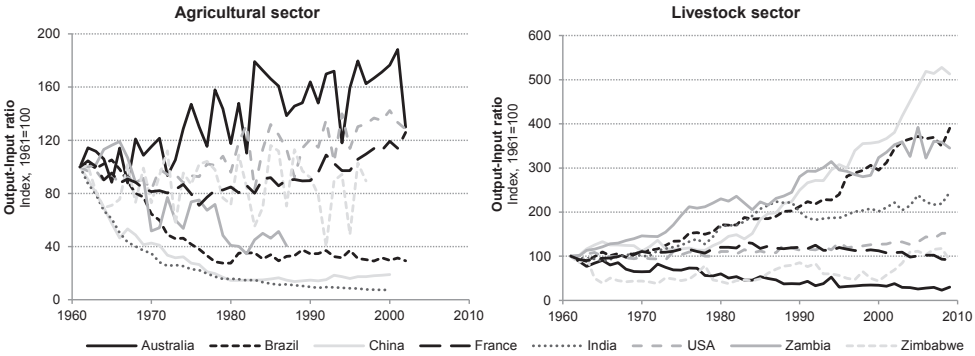

FIGURE 2-5 | Comparison of developments in productivity (output-input ratio) of the agricultural and livestock sector in the seven selected countries. The input-output ratio is indexed to 100 for the year 1961. This means that when the ratio is higher than 100 in one year, the productivity has improved compared to 1961; when the ratio is lower than 100, the productivity has declined. For the agricultural sector the included inputs are: agricultural land, fertilizer and tractors; the outputs are: crops, meat, milk and eggs. For the livestock sector, the inputs are: feed crops, meadow and pasture land; the outputs are: meat, milk and eggs.

2.3.2 Yield projections

2.3.2.1 Crops

Models that assess biomass potentials and/or impacts of biomass production apply either only exogenous yield projections (determined by factors outside the model) or a combination of exogenously and endogenously (determined by the model based on internal factors) defined yield projections. The exogenous yield projections are based on historical trends. As mentioned in section 2.3.1.2, the analysis of historical yield developments shows that the most observed trend for crop yield growth is linear. Yet, longer historical time series show that in, for example, the USA crop yield growth has not always followed the current linear trend (see Figure 2-16 in appendix A2.9). Also, over shorter time frames, variability in the trend is found with periods of decline, stagnation and/or strong growth. Therefore, yield projections based on historical trends depend on the historical time frame taken into account. For example, Fischer et al. [42] find global yield growth rates of 1.0% yr^{-1} for wheat, rice and soybean and 1.5% yr^{-1} for corn based on

the linear trend for 1991-2010. For the period 1961-2010, the present study finds global growth rates that are slightly higher for wheat, rice and soybean (1.1-1.3% yr^{-1}) and lower for corn (1.3 % yr^{-1}, all relative to 2010), see Table 2-4 in appendix A2.9. Although these differences seem small, they may have considerable impact on future biomass potentials and impacts. This is illustrated by Fischer et al. [42] who state that, in order to meet the projected food demand in 2050 with limited increase of real prices of crops, the minimum global yield growth rate for staple crops between 2010 and 2050 is 1.1% yr^{-1} relative to 2010 [42]. Higher growth rates of e.g. 1.3% yr^{-1} are preferred to account for factors that may influence supply and demand of crops, including increasing biofuel demand [42]. In model assessments, it is therefore important to make explicit what historical time frame is considered to define exogenous yield projections.

Although exogenous yield projections are based on historical trends, some models assume extrapolation of this trend (e.g. in [14]), while most models assume the overall future yield growth to slow down compared to the historical trend (e.g. in [10,44]). Van Dijk and Meijerink [45] give several reasons for assuming decreasing yield growth. First, the opportunities for increasing yields and exploiting existing yield gaps are more and more exhausted [46]. Also, investments in agricultural R&D have declined and considerable socio-economic constraints in many developing countries are considered to remain a limiting factor for yield growth [47-49]. Although these motivations are reasonable, these are mainly expectations about how different factors are likely to develop. There may, however, also be other possible pathways. Indeed, the present study shows that in the past 50 years, significant developments in technology, e.g. mechanization, fertilizer use and crop breeding, have driven yield improvements. But although these technologies are wide-spread now, there still exist considerable differences in technology level and yield gaps between regions (see section 2.3.1 and Table 2-5 in appendix A2.9). The present study and the study by de Wit et al. [22] show that in regions where historical yield growth was high, the development and adoption of new technologies was primarily driven by policies (e.g. subsidies for farmers, trade liberalization and public investments in R&D). In regions where there is still room for considerable yield improvements, stimulating policies could thus play a vital role in materializing this potential. Similarly, other factors could also have a significant effect on future yield developments. The presence of different potential pathways shows that, in the assessment of future biomass potentials and the impacts of biomass production, it is important to investigate different scenarios and to include various endogenous driving factors.

To determine the endogenous yield projections, models relate yield change to driving factors like land or crop prices, climate change or management. According to Dietrich et al. [50], technological change is considered to be the key driver for yield change. There is, however, little consensus about the drivers of technological change and the influence of these drivers on yield change [50,51]. Therefore, the number of endogenous factors is generally limited in models, often to only one or two factors. As a result, the different

technical, economic and institutional driving factors are not covered well. Furthermore, the findings from the historical analysis (Section 2.3.1) make clear that future yield developments largely depend on how the different driving factors develop in each region. As the number of endogenous factors is generally limited, models do not properly distinguish different driving factors between regions. Important examples of how different drivers affect yield growth potentials are the following:

- In the case of corn in the USA, current yields have attained almost 80% of the *maximum attainable yield*[2] (see Table 2-5 on yield gaps, appendix A2.9). Thus, it is likely that the current technological limits will constrain and slow down yield growth in the near future. Continuation of the historical yield improvement trend would require significant *technological progress*, e.g. increase of the potential agro-climatic yield through biotechnology.
- On the other side of the spectrum, yields of rice and soy in India and corn and soy in Zambia and Zimbabwe are less than 40% of the maximum attainable yield. In various other cases, the yield gap is smaller but still leaves room for considerable yield improvements as well. In such situations, the historical analysis shows that accelerated yield growth compared to the longer term trend is possible under favorable circumstances with regard to e.g. governance.
- Zambia and Zimbabwe are examples of cases where *stable agricultural and trade policies* are needed to improve and support the agricultural sector and market and the economy in general. Under such conditions, farmers may be able to adopt improved technologies and management practices. As more advanced technologies and practices already exist, farmers could realize a significant acceleration in yield growth compared to the average trend over the past five decades.
- In Southern India, rice yields could be increased significantly if *management conditions* and *market access* would be improved (see appendix A2.5 on agricultural characteristics). In Northern India, rice (and also wheat) is mainly produced on irrigated lands and yields are higher compared to Southern India. Ground water depletion, however, poses a risk to future yield improvements.
- An important measure to attain yield improvements in an environmentally sound way is *sustainable agricultural intensification*. In Australia, for example, management levels have been low relative to most other countries. The same is true for yield growth rates. Although intensification could have significant

2 Maximum attainable yield: the yield resulting from combining i) the constraint-free potential agro-climatic yield with regard to temperature, radiation and soil moisture conditions prevailing in the specific region and ii) reduction factors related to climate (e.g. pests and diseases), soil and terrain conditions and assumptions regarding the management level [32]. Generally, it is assumed that there is a minimum yield gap where the actual yield level is equal to the economically attainable yield. Fischer et al. [42] consider this economically attainable yield to be about 23% below the maximum attainable yield. Larger yield gaps are assumed to be 'economically exploitable yield gaps'. These caps could (largely) be closed with existing technologies [42].

environmental impact, it is found that this can also be realized while improving the productivity, i.e. increasing the crop production (output) per unit resource use (input), and reducing negative effects like emissions and water pollution (see also [42,52,53]).

Thus, for future modeling work, detailed regional assessment of the most important driving factors for yield development and the implementation of more drivers are needed. For this purpose, Table 2-3 identifies some key drivers that may either limit or stimulate future yield improvements in each country assessed in this study.

We highlighted the importance of stimulating policies especially for Zambia and Zimbabwe earlier. The historical analysis showed that in all countries agricultural and trade policies play an important role in steering yield developments. Van Dijk and Meijerink [45] show that in economic models, policies and institutions are included, but it seems that their effect on yields is not considered yet. Given our findings, it is important to include policies as a driving factor for yield changes in model assessments. For example, Dietrich et al. [50] have attempted to implement endogenous yield change related to investments in R&D. Linking yield developments to R&D and other policy-related drivers could be used to define scenarios for different policy pathways and to evaluate their impact on yield changes. In addition, yield gap figures are a good indicator both for the degree of technological progress that can still be attained, and for the potential yield growth rates. It is useful to apply yield gaps in the models to define the yield development projections. For that, more research is needed on, for example, the (crop and region specific) correlation between yield gaps and yield growth rates. In the historical analysis, it was also found that yields have often fluctuated in the past and this affected the average yield growth rate. In current assessment models, projections are based on historical trends and the influence of fluctuations on the long term trend is neglected. The regional identification of key drivers for yield developments could include an assessment of risk factors for yield fluctuations (e.g. the occurrence of extreme climate conditions), which can be taken into account in the yield projections.

A comparison of yield projections from global outlook studies in van Dijk and Meijerink [45] shows that the projected yield growth varies significantly and depends on the underlying assumptions made. The influence of underlying assumptions on yield projections is also illustrated by a comparison of yield projections from the integrated assessment model IMAGE (which is used for the assessment of, e.g. biomass potentials and impacts; see e.g. van Vuuren et al. [8]) and the economic model MIRAGE models (which is used for analyses on, e.g. land use change induced by biofuel targets; see e.g. Laborde [10]). In Table 2-4 (Appendix A2.9), growth rates derived from the yield projections in IMAGE and MIRAGE are compared to the extrapolation of historical trends. Both models combine exogenous yield projections with endogenously determined yield changes. The exogenous yield projections used in IMAGE and MIRAGE assume that, on the global level, yield growth

rates will slow down compared to the historical linear trend [10,44]. Nevertheless, the two models do not always agree on whether yields in a certain region or even globally will improve at a pace higher or lower compared to the linear growth trend. Large differences in projections between the models are found for corn and soybean in Brazil and rice in China. Also, the projected global growth rates from MIRAGE for corn, rice and soybean are higher compared to the projections based on linear extrapolation of the historical trends from 1961-2010. In IMAGE, the global projections for wheat, corn, rice and soybean result in lower yield growth rates compared to linear extrapolation. It is most likely that these contrasting results can be explained by differences in how endogenous yield changes are modeled. The insights from this and other studies would help to make the underlying assumptions for endogenous yield projections more explicit and detailed, and help to assess how yield projections are influenced by different assumptions.

TABLE 2-3 | Key threats and opportunities for future yield improvements.

	Threats to future yield improvements	Opportunities for future yield improvements
Australia	Climate and climate change	Sustainable intensification of crop and livestock sector (see e.g. [52])
Brazil	Weak enforcement or mitigation of land conservation policies	Intensification of cattle production
China	Decreasing water availability Loss of fertile land and land degradation	Expansion of region-specific policies Continuation or expansion of policies to improve productivity Increase of mechanization
India	Land degradation Decreasing water availability	Improvement of market access, management and production of smallholders in Southern India Enforcement of agri-environmental policies Improvement of productivity
USA	Lack of new advances in biotechnology: no significant improvements in the maximum attainable yields	Significant funding for and advances in biotechnology: shift of maximum attainable yields
Zambia	Soil erosion Climate variability	Stimulating policy; e.g. improvement of market access of smallholders (investment in infrastructure) Increase in the adoption of conservation farming
Zimbabwe	Continuation of unstable economic situation Climate variability	Re-establishment of (beef) export markets: improvement of knowledge and production of smallholders

The factors are identified based on the historical assessment in section 2.3.1.

In the introduction, several studies were mentioned that assess the influence of increased yields on the biomass potential. Van Vuuren et al. [8], Erb et al. [9] and Dornburg et al. [6] used yield projections from the FAO (Bruinsma [54], the presented yield projections from the IMAGE model are in line with these projections [44]). Van Vuuren et al. [8] and Dornburg et al. [6] take this FAO scenario as baseline and assess the extra biomass potential from additional yield increases. Erb et al., [9] however, assume that the FAO scenario represents a high intensification scenario and baseline yield improvements are lower. This

shows that the perception of the baseline varies. This again underlines the importance to make assumptions more explicit. In addition, it is important to discuss each scenario and address under what conditions the projected yields and the resulting biomass potentials and impacts can be attained; e.g. the degree to which investments have to be increased, or required changes in policy. This is considered to be highly valuable for decision making.

2.3.2.2 Beef and milk

As opposed to historical yield growth trends for crops often being linear, for beef and milk production we only found linear trends for Australia, India and the USA; these are countries where we found that yields had significantly improved over a longer timeframe because of the existence of a commercial market. The absence of a yield trend in the past makes it more difficult to define yield growth scenarios for the future. Also, compared to crops, less information can be found about the projections used in the models. Several studies, e.g. IMAGE [16,55], apply yield projections from the FAO, which are presented for aggregated world regions level in Wirsenius et al. [56]. A comparison of these projections with historical growth figures indicates that in developed regions, the average annual yield growth rate is projected to be significantly lower than in the last five decades. For example, in North America and Oceania the increase in beef yield is projected to decline from 1.0% yr^{-1} in the period 1961-2005 to 0.2% yr^{-1} from 1997/99-2030. Also on the global level, yield growth of beef and milk will be slower compared to the historical trend. Acceleration of yield improvements is projected to especially take place in Sub-Saharan Africa. In this region the yield growth rates of milk are projected to increase from -0.4% yr^{-1} (1961-2005) to 0.8% yr^{-1} (1997/99-2030). In addition to the FAO projections, Wirsenius et al. [56] defined an improved livestock production (ILP) scenario, assuming faster intensification of livestock production in low- and medium income regions as a result of increased competition for land and stricter policies related to land use and livestock production. In this scenario, more regions (e.g. Asia) will realize accelerated yield increases compared to the past. Also on the global level, yield growth will increase from 0.9% yr^{-1} to 1.5% yr^{-1} for beef and from 0.5% yr^{-1} to 2.2% yr^{-1} for milk [56]. The scenarios from FAO and Wirsenius et al. again illustrate regional differences, which should be taken into account in the models.

To define yield projections, it is again helpful to consider the potential role of different driving factors per region. In the historical assessment it was found that the role of commercial livestock production for the domestic or export market is an important factor for explaining yield improvements. For modelling purposes, several scenarios could be defined for market development based on assumptions regarding the size and location of beef and milk consumption and production. In addition, the speed of yield improvements may be based on other driving factors like the possibilities for technological developments and the introduction of agri-environmental policies. Similar to crops, the technological improvement potential could be assessed through yield gap analysis. For

livestock, however, no standardized methods exist to assess the yield gap [57]. One approach is similar to the conceptual framework for crops and is based on three groups of production factors; production defining (climate and animal genetic characteristics), production limiting (water and feed intake) and growth reducing (diseases, pollutants) [58]. This method is still new; the first calculations of potential beef production were recently conducted by van der Linden et al. [59].

2.4 DISCUSSION

2.4.1 FAO data

This study analyses a large amount of statistical data which is obtained from the FAOSTAT database. The quality of FAO data, however, can vary significantly. When available, the FAO presents *official data*, which means that the data is collected directly from the states. Yet, the data collection capacities and practices vary between countries and affect the reliability of the data. In addition, the concepts, definitions, coverage and classifications used by the countries are not uniform and require harmonization to enable international comparison. When no official data is available, the FAO gives figures from secondary semi-official or unofficial datasets or own estimations [24]. With regard to crop yields used in this study, the amount of underlying data that is non-official data is limited and mainly restricted to Zambia and Zimbabwe. In the case of milk and beef yields, secondary and estimated data are more common and also presented on a regular basis for Brazil, China and India [24]. Sometimes these data seem to be artificial as yields remain constant over one to five decades (see for example yields from beef and milk production in Zambia and Zimbabwe, Figure 2-14).

To get an impression of the reliability of FAO yield statistics, their consistency with USDA data was analyzed (Figure 2-17, Appendix A2.9). In some cases, FAO and USDA diverge significantly. For example, in Zambia, corn yield development between 1961 and 1982 is highly uncertain; figures from the FAO show an increasing yield trend, while the USDA data present a downward trend. This has significant impact on the interpretation of historical developments. The FAO data suggests that corn yield improvements started in the early 1970s with the introduction of fertilizer subsidies for smallholder farmers, while the USDA data implies that the yields were negatively affected by the new pricing and subsidy policies and only started to increase after the introduction of the first new corn varieties in the late 1970s. Differences between the two statistical sources were also found for soybean in Zimbabwe, milk in Brazil and China and beef in China and India. Not all data sets could be compared as the USDA database had no statistics on milk and beef production in Zambia and Zimbabwe and on sugarcane yields. The varying quality and reliability must be taken into account when interpreting the results. Still, the FAO database is the most complete source for yield figures currently available.

2.4.2 Yield indicators for cattle

With regard to cattle production, it is preferred to assess yield developments in terms of changes in feed conversion efficiency (FCE, kg animal product per kg feed intake) instead of beef or milk production per animal. The main problem of using the production level per animal is that this figure does not always reflect technological advancements. For example, an improved beef cattle production system may achieve faster weight gain and be able to reduce the cattle lifetime. As a result, the beef production per animal may remain constant or even reduce, while the total production can be increased. Also, de Wit et al. [22] showed that beef yields in the Netherland decreased in the 1990s and 2000s because of the large share of dairy cows that are optimized for milk and not for meat production. In the present study, the historical data show that the average yield per animal has continued to increase in the main cattle producing countries. The rates at which these yields have increased, however, are likely to differ from improvement rates in feed conversion efficiency.

Another reason why the use of the feed conversion efficiency is preferred is the underlying idea of this study that yield improvements have an important role in making land available for biomass production without increasing overall land use. While crop yields are directly related to land use, figures of beef or milk production per animal give no indication of the related land use. As feed consumption can be linked more directly to land use, the feed conversion efficiency would give a better insight in how developments in the cattle sector would influence land requirements.

Ideally, the FCE is measured over the lifetime of an animal because its value is not constant over time. To analyze historical developments in average FCE, a more simple but less accurate way is to calculate the feed conversion efficiency by dividing the produced amount of beef or milk by the gross feed intake. This feed intake is based on estimated energy requirements and the amount of energy supplied by feed inputs, factors that are highly depending on the production system. Therefore, the examination of developments in feed conversion efficiencies over time would require the allocation of animal populations and production quantities to the different production systems for at least several points in time, e.g. building on previous work by Seré and Steinfeld [25] and Bouwman et al. [55]. This was not feasible for the present study. As the carcass weight and annual milk production per animal are the best available data over a longer historical time period, these figures were used here to assess beef and milk yield developments. The same data was used by de Wit et al. [22] and Wirsenius et al. [56] to study livestock yield growth rates.

2.4.3 Yield projections and assessment of biomass potentials

This study investigated historical developments in yields and their drivers and provides suggestions for how potential studies can better account for yield developments and driving factors. The assessment focused on three types of driving factors: technological/ management, economic, and institutional. Other factors, however, may also influence yield developments. Climate change, for example, may either have a positive or negative effect on yield growth depending on the location (see e.g. Jaggard et al. [14]). Also, several studies indicate that yield improvement rates of crops are related to the GDP level of a country [41,60]. This correlation, however, does not necessarily mean that GDP itself is a driver of yield development. It is more likely that GDP is an indicator of other driving factors, such as market conditions and technology levels, which are included in the present study.

As shown in section 2.3.2.1, historical yield growth rates depend on the time frame considered. This is also seen when comparing the yield growth rates for France as calculated in this study and in de Wit et al. [22]. For wheat, for example, the present study found a growth rate of 4.3 % yr^{-1} (114 kg ha^{-1} yr^{-2}) for the period 1961 to 1970 while de Wit et al found a growth rate of 5.2 % yr^{-1} (136 kg ha^{-1} yr^{-2}) for the period 1961 to 1969. Thus, the yield growth rates are highly sensitive to the timeframe applied, especially in the case of short time frames. The growth rates should thus be considered with great care and only be used as an indicator of the extent of yield growth or decline. Nevertheless, both studies show that the yield growth rates are very useful to assess the impact of driving factors on yield developments.

Although the historical assessment gives important insights in how different factors may influence future yield developments, it is not possible to predict future yield growth rates. The insights can thus only be used to assess how yields may develop under certain conditions. Particularly the application of endogenous factors and scenarios is useful to assess how yield developments change under different assumptions and how this affects biomass potentials. As mentioned in section 2.3.2.1, it is important to translate each scenario to conditions for meeting the projected yield developments. This can help identify (regional) strategies for increasing yields.

Finally, in addition to yields, there are also other factors that may affect biomass production potentials. For example, market developments and incentives could influence the balance between crop and livestock production on the one hand and biomass production on the other hand. Also, sustainability criteria could affect the area of land that is excluded from biomass production. An overview of more key factors is provided by Dornburg et al. [6]. Similar to the drivers for yield developments, it is important to make the assumptions regarding these factors explicit. Also, the application of scenarios could be useful.

2.5 CONCLUSIONS

Global, sustainable biomass production potentials of energy crops largely depend on the future availability of surplus agricultural lands made available through yield improvements in crop and livestock production. This study analyzed the pace and direction of historical yield developments between 1961 and 2010 in Australia, Brazil, China, India, USA, Zambia and Zimbabwe. Furthermore, it assessed the technological, economic and institutional driving forces behind these developments and explored how the insights gained can help to improve the modeling of future yields.

This study showed that historical yield growth (especially of crops) has often followed a linear trend. Mainly, the average yield improvement rates for crops and milk were between 0.7% and 1.6% yr^{-1}. For beef, the rates were lower (maximum of 0.8% yr^{-1} in Australia; all relative to 2010). In all cases, yields and yield growth rates have fluctuated to various degrees. Large fluctuations were especially found for crops when driving factors changed strongly (e.g. extreme climate conditions in Australia). Also, in each case, the analysis revealed periods during which yields improved at a higher rate compared to the long-term average as well as periods during which yield growth rates were lower than this average. The periods of high yield growth, e.g. 8.5% yr^{-1} for soybean in Zimbabwe in the 1970s, show that relatively fast improvements can be attained in cases where the yield gap is still large. Such significant improvements can especially be realized under favorable conditions with regard to economics and governance that stimulate improvements in agricultural technology and management. The future development of yields depends on how driving factors will change in each region.

The historical assessment shows that all three types of driving forces have influenced yield changes. The importance and the effect of each factor, however, is country- and even regional- specific. Overall, supporting agricultural policies have played an important role in increasing yields. Examples of successful policies are subsidies to stimulate adoption of new technologies, trade liberalization (resulting in increased demand for agricultural products which stimulated investments and innovations in the agricultural sector) and public investments in R&D. In some periods and countries, such policies were absent or eliminated (e.g. Australia in the 1960s and Zambia in the 1990s). As a result, yields stagnated or declined. Although agricultural policies led to yield increases in many cases, they failed to improve output-input ratios (i.e. unsuccessful to realize more efficient use of resources like fertilizers). Some countries like the USA and (to a lesser extent) China were able to increase this productivity by implementing specific agri-environmental policies. Other countries adopted such policies as well, but the result largely depended on the success to enforce these policies (e.g. productivity stabilized in Australia, but no effect was seen in India). The importance of policies in steering yields was especially high for crops. With regard to yield improvements in cattle production, a key factor was the importance of commercial beef and milk production for the national or export market. But

policy and market can be closely related: in many cases, trade liberalization created new markets, which stimulated investments and resulted in improved yields as demonstrated in, for example, Brazil.

Current models that assess biomass potentials and impacts only take into account one or a limited number of endogenous factors influencing yields. Also, an explicit discussion of the assumptions behind yield projections is lacking, which hampers a comparison of yield projections between the models. Several suggestions are made to improve the models and thereby our understanding of potential future pathways for agricultural yield developments and for sustainable biomass production. First, scenarios based on regional assessment of key factors for yield development, as conducted in this study, could help to gain more insight in potential pathways and regionally differentiated effects. Second, to define such scenarios, yield gap figures are an important indicator of possible technological progress and the potential rate of yield improvement. Also, different policy strategies should be included and tested in the scenarios. Finally, the assessment of important factors for yield development could help to make the underlying assumptions of yield projections more explicit. The implementation of these suggestions will help to identify policy options and preconditions for specific development pathways.

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REFERENCES

[1] Harvey M and Pilgrim S. The new competition for land: Food, energy, and climate change. Food Policy 2011;36:S40-S51.

[2] Chum H, Faaij A, Moreira J, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 209-332.

[3] Smeets EMW, Faaij APC, Lewandowski IM, et al. A bottom-up assessment and review of global bio-energy potentials to 2050. Progress in Energy and Combustion Science 2007;33(1):56-106.

[4] Hoogwijk M, Faaij A, Eickhout B, et al. Potential of biomass energy out to 2100, for four IPCC SRES land-use scenarios. Biomass and Bioenergy 2005;29(4):225-257.

[5] Slade R, Saunders R, Gross R, et al. Energy from biomass: the size of the global resource. Imperial College Centre for Energy Policy and Technology and UK Energy Research Centre: Londen, UK; 2011.

[6] Dornburg V, van Vuuren D, van de Ven G, et al. Bioenergy revisited: Key factors in global potentials of bioenergy. Energy & Environmental Science 2010;3(3):258-267.

[7] Batidzirai B, Smeets EMW, Faaij APC. Harmonising bioenergy resource potentials—Methodological lessons from review of state of the art bioenergy potential assessments. Renewable and Sustainable Energy Reviews 2012;16(9):6598-6630.

[8] van Vuuren DP, van Vliet J, Stehfest E. Future bio-energy potential under various natural constraints. Energy Policy 2009;37(11):4220-4230.

[9] Erb K, Haberl H, Krausmann F, et al. Eating the Planet: Feeding and fuelling the world sustainably, fairly and humanely– a scoping study. Commissioned by Compassion in World Farming and Friends of the Earth UK. Institute of Social Ecology: Vienna, Austria; 2009. Social Ecology Working Paper No. 116.

[10] Laborde D. Assessing the land use change consequences of European biofuels policies. International Food Policy Research Institute: Washington, DC, USA; 2011. Available from: http://trade.ec.europa.eu/doclib/docs/2011/october/tradoc_148289.pdf

[11] Searchinger T, Heimlich R, Houghton RA, et al. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. Science 2008;319(5867):1238-1240.

[12] Tilman D, Balzer C, Hill J, et al. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences 2011;108(50):20260-20264.

[13] Havlík P, Valin H, Herrero M, et al. Climate change mitigation through livestock system transitions. Proceedings of the National Academy of Sciences 2014;111(10):3709-3714.

[14] Jaggard KW, Qi A, Ober ES. Possible changes to arable crop yields by 2050. Philosophical Transactions of the Royal Society B: Biological Sciences 2010;365(1554):2835-2851.

[15] Rosegrant MW, Ringler C, Msangi S, et al. International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT): Model Description. International Food Policy Research Institute: Washington, D.C.; 2008. Available from: http://www.ifpri.org/sites/default/files/publications/impactwater.pdf

[16] Eickhout B, van Meijl JCM, Tabeau AA, et al. The Impact of Environmental and Climate Constraints on global food supply. Netherlands Environmental Assessment Agency (MNP) / Agricultural Economics Research Institute (LEI): Bilthoven / The Hague, The Netherlands; 2008.

[17] Khanna M, Crago CL, Black M. Can biofuels be a solution to climate change? The implications of land use change-related emissions for policy. Interface Focus 2011;1(2):233-247.

[18] EPA. Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis. Environmental Protection Agency: Washington, DC, USA; 2010. EPA-420-R-10-006.

[19] Beach RH, Adams D, Alig R, et al. Model Documentation for the Forest and Agricultural Sector Optimization Model with Greenhouse Gases (FASOMGHG). RTI International: 2010. Available from: http://www.cof.orst.edu/cof/fr/research/tamm/FASOMGHG_Model_Documentation_Aug2010.pdf

[20] Mosnier A, Havlík P, Valin H, et al. The Net Global Effects of Alternative U.S. Biofuel Mandates: Fossil Fuel Displacement, Indirect Land Use Change, and the Role of Agricultural Productivity Growth — Nicholas Institute. Nicholas Institute for Environmental Policy Solutions: Durham, NC, USA; 2012. NI R 12-01. Available from: http://nicholasinstitute.duke.edu/climate/policydesign/net-global-effects-of-alternative-u.s.-biofuel-mandates

[21] Steinfeld H, Gerber P, Wassenaar T, et al. Livestock's long shadow: environmental issues and options. FAO: Rome, Italy; 2006.

[22] de Wit M, Londo M, Faaij A. Productivity developments in European agriculture: Relations to and opportunities for biomass production. Renewable and Sustainable Energy Reviews 2011;15(5):2397-2412.

[23] Wicke B, van der Hilst F, Daioglou V, et al. Model collaboration for the improved assessment of biomass supply, demand, and impacts. GCB Bioenergy 2015;7(3):422-437.

[24] FAO. FAOSTAT [Internet: updated 2014, accessed 2012-2014]. Available from: http://faostat.fao.org

[25] Seré C and Steinfeld H. World Livestock Production Systems: Current status, issues and trends. Food and Agriculture Organization of the United Nations: Rome; 1996. FAO Animal Production And Health Paper 127. Available from: http://www.fao.org/docrep/004/W0027E/W0027E00.HTM

[26] Hengsdijk H and Langeveld JWA. Yield trends and yield gap analysis of major crops in the world. Statutory Research Tasks Unit for Nature & the Environment (WOT Natuur & Milieu), Wageningen UR: Wageningen, The Netherlands; 2010. WOt-werkdocument 170.

[27] Neumann K, Verburg PH, Stehfest E, et al. The yield gap of global grain production: A spatial analysis. Agricultural Systems 2010;103(5):316-326.

[28] Piesse J and Thirtle C. Agricultural R&D, technology and productivity. Philosophical Transactions of the Royal Society B: Biological Sciences 2010;365(1554):3035-3047.

[29] Smith P, Gregory PJ, van Vuuren D, et al. Competition for land. Philosophical Transactions of the Royal Society B: Biological Sciences 2010;365(1554):2941-2957.

[30] Anderson K. Globalization's effects on world agricultural trade, 1960–2050. Philosophical Transactions of the Royal Society B: Biological Sciences 2010;365(1554):3007-3021.

[31] FAO. Guidelines for the preparation of livestock sector reviews. Animal Production and Health Guidelines. No. 5. FAO: Rome; 2011.

[32] FAO and IIASA. GAEZ Global Agri-Ecological Zones [Internet: updated 2014, accessed 19 February 2014]. Available from: http://gaez.fao.org/Main.html#

[33] Langyintuo AS and Setimela P. Assessing the effectiveness of a technical assistance program: The case of maize seed relief to vulnerable households in Zimbabwe. Food Policy 2009;34(4):377-387.

[34] Eicher CK. Zimbabwe's maize-based Green Revolution: Preconditions for replication. World Development 1995;23(5):805-818.

[35] Whitlow R. Soil erosion and conservation policy in Zimbabwe: Past, present and future. Land Use Policy 1988;5(4):419-433.

[36] Whitlow R. Conflicts in land use in Zimbabwe: Political, economic and environmental perspectives. Land Use Policy 1985;2(4):309-322.

[37] Matondi PB. 1. Understanding Fast Track Land Reforms in Zimbabwe. In: Matondi PB, editor. Zimbabwe's Fast-Track Land Reform. 1st ed., Zed Books: Londen; 2012, p. 1-17.

[38] Matondi PB. 5. Complexities in understanding agricultural production outcomes. In: Matondi PB, editor. Zimbabwe's Fast-Track Land Reform. 1st ed., Zed Books: Londen; 2012, p. 130-160.

[39] Marquette CM. Current poverty, structural adjustment, and drought in Zimbabwe. World Development 1997;25(7):1141-1149.

[40] Ray DK, Mueller ND, West PC, et al. Yield Trends Are Insufficient to Double Global Crop Production by 2050. PLoS ONE 2013;8(6):e66428.

[41] Hafner S. Trends in maize, rice, and wheat yields for 188 nations over the past 40 years: a prevalence of linear growth. Agriculture, Ecosystems & Environment 2003;97(1–3):275-283.

[42] Fischer RA, Byerlee D, Edmeades GO. Crop yields and global food security: will yield increase continue to feed the world? Australian Centre for International Agricultural Research: Canberra, Australia; 2014.

[43] Huyghe C. Country Pasture/Forage Resources Profiles. France [Internet: updated 2012, accessed 5 February 2015]. Available from: http://www.fao.org/ag/agp/AGPC/doc/Counprof/France/france.htm#4

[44] OECD. OECD Environmental Outlook to 2050. OECD publishing: 2012. Available from: http://dx.doi.org/10.1787/9789264122246-en

[45] van Dijk M and Meijerink GW. A review of global food security scenario and assessment studies: Results, gaps and research priorities. Global Food Security 2014;3(3–4):227-238.

[46] Searchinger T, Hanson C, Ranganathan J, et al. Creating a Sustainable Food Future. A menu of solutions to sustainably feed more than 9 billion people by 2050, World Resources Report 2013-14: Interim Findings. World Resources Institute (WRI): Washington DC, USA; 2013.

[47] Alexandratos N and Bruinsma J. World agriculture towards 2030/2050: the 2012 revision. Food and Agricultural Organisation of the United Nations: 2012. ESA working paper no. 12-03.

[48] McIntyre BD, Herren HR, Wakhungu J, et al. Agriculture at Crossroads. International Assessment of Agricultural Knowledge, Science and Technology for Development - Global Report. IAASTD: Washington DC, USA; 2009.

[49] Alston JM, Beddow JM, Pardey PG. Agricultural Research, Productivity, and Food Prices in the Long Run. Science 2009;325(5945):1209-1210.

[50] Dietrich JP, Schmitz C, Lotze-Campen H, et al. Forecasting technological change in agriculture—An endogenous implementation in a global land use model. Technological Forecasting and Social Change 2014;81:236-249.

[51] Robinson S, van Meijl H, Willenbockel D, et al. Comparing supply-side specifications in models of global agriculture and the food system. Agricultural Economics 2014;45(1):21-35.

[52] Hochman Z, Carberry PS, Robertson MJ, et al. Prospects for ecological intensification of Australian agriculture. European Journal of Agronomy 2013;44:109-123.

[53] de Wit MP, Lesschen JP, Londo MHM, et al. Greenhouse gas mitigation effects of integrating biomass production into European agriculture. Biofuels, Bioproducts and Biorefining 2014;8(3):374-390.

[54] Bruinsma J. World agriculture: towards 2015/2030. An FAO perspective. Earthscan Publications Ltd: London, UK; 2003.

[55] Bouwman AF, van der Hoek KW, Eickhout B, et al. Exploring changes in world ruminant production systems. Agricultural Systems 2005;84(2):121-153.

[56] Wirsenius S, Azar C, Berndes G. How much land is needed for global food production under scenarios of dietary changes and livestock productivity increases in 2030? Agricultural Systems 2010;103(9):621-638.

[57] ILRI. Closing livestock yield gaps in the developing world: imperatives for people and the planet [Internet: updated 2014, accessed 26 June 2014]. Available from: http://www.slideshare.net/ILRI/smith-gfsc-apr2014

[58] van de Ven GWJ, de Ridder N, van Keulen H, et al. Concepts in production ecology for analysis and design of animal and plant–animal production systems. Agricultural Systems 2003;76(2):507-525.

[59] van der Linden A, van de Ven GWJ, Oosting SJ, et al. Can we extend yield gap analysis to livestock production? [poster presentation]. First International Conference on Global Food Security; 29 Sept-2 Oct 2013; Noordwijkerhout, The Netherlands. 2013.

[60] Powell JP and Rutten M. Convergence of European wheat yields. Renewable and Sustainable Energy Reviews 2013;28:53-70.

[61] Hawkes C, Friel S, Lobstein T, et al. Linking agricultural policies with obesity and noncommunicable diseases: A new perspective for a globalising world. Food Policy 2012;37(3):343-353.

[62] Tauger MB. Agriculture in World History. 1st ed., Routledge: New York, NY, USA; 2011.

[63] Hawkes C and Murphy S. 2. An Overview of Global Food Trade. In: Hawkes C, Blouin C, Henson S, Drager N, Dubé L, editors. Trade, Food, Diet and Health: Perspectives and Policy Options. Wiley Blackwell: Oxford; 2010, p. 16-32.

[64] Chambers WB. World trade and concerns for the human environment. In: Brouwer F, Ervin DE, editors. Public concerns, environmental standards and agricultural trade. CAB International: Wallingford, Oxfordshire, UK; 2002, p. 39-55.

[65] Pink B. 2012 Year book Australia. Australian Bureau of Statistics (ABS): Canberra, Australia; 2012.

[66] Stringer R and Anderson K. Australia. In: Brouwer F, Ervin DE, editors. Public Concerns, Environmental Standards and Agricultural Trade. CAB International: Wallingford, Oxfordshire, UK; 2002, p. 181-214.

[67] Doyle PT and Stockdale CR. Dairy Farm Management Systems | Seasonal, Pasture-Based, Dairy Cow Breeds. In: Fuquay JW, editor. Encyclopedia of Dairy Sciences. 2nd ed., Academic Press: San Diego; 2011, p. 29-37.

[68] Anderson K, Lattimore R, Lloyd PJ, et al. 5 Australia and New Zealand. In: Anderson K, editor. Distortions to Agricultural Incentives: A Global Perspective, 1955-2007. The International Bank for Reconstruction and Development / The World Bank: Washington DC, USA; 2009, p. 289-322.

[69] Trewing D. Chapter 14 - Agriculture. In: Trewing D, editor. 2004 Year book Australia. Volume 86, Australian Bureau of Statistics (ABS): Canberra, Australia; 2004, p. 417-456.

[70] Anderson K and Valdés A. 7 Latin America and the Caribbean. In: Anderson K, editor. Distortions to Agricultural Incentives: A Global Perspective, 1955-2007. The International Bank for Reconstruction and Development / The World Bank: Washington DC, USA; 2009, p. 289-322.

[71] Dixon J, Gulliver A, Gibbob D. Farming Systems and Poverty: Improving farmers' livelihoods in a changing world. FAO and World Bank: Rome and Washington D.C.; 2001.

[72] Wint GRW and Robinson TP. Gridded livestock of the world 2007. FAO: Rome, Italy; 2007. Available from: http://www.fao.org/docrep/010/a1259e/a1259e00.htm

[73] Cederberg C, Meyer D and Flysjö A. Life cycle inventory of greenhouse gas emissions and use of land and energy in Brazilian beef production. Swedish Institute for Food and Biotechnology (SIK): Göteborg, Sweden; 2009. 792.

[74] Ferreira J, Pardini R, Metzger JP, et al. Towards environmentally sustainable agriculture in Brazil: challenges and opportunities for applied ecological research. Journal of Applied Ecology 2012;49(3):535-541.

[75] Martinelli LA, Naylor R, Vitousek PM, et al. Agriculture in Brazil: impacts, costs, and opportunities for a sustainable future. Current Opinion in Environmental Sustainability 2010;2(5–6):431-438.

[76] Baer W. Import Substitution and Industrialization in Latin Amercia: Experiences and Interpretations. Latin American Research Review 1972;7(1):95-122.

[77] Carvalho J. Agriculture, industrialization and the macroeconomic environment in Brazil. Food Policy 1991;16(1):48-57.

[78] Carmo Oliveira Jd. Trade policy, market 'distortions', and agriculture in the process of economic development Brazil, 1950–1974. Journal of Development Economics 1986;24(1):91-109.

[79] Stattman SL, Hospes O, Mol APJ. Governing biofuels in Brazil: A comparison of ethanol and biodiesel policies. Energy Policy 2013;61:22-30.

[80] Lamers P, Hamelinck CN, Junginger M, et al. International bioenergy trade—A review of past developments in the liquid biofuel market. Renewable and Sustainable Energy Reviews 2011;15(6):2655-2676.

[81] Banerjee O, Macpherson AJ, Alavalapati J. Toward a Policy of Sustainable Forest Management in Brazil: A Historical Analysis. The Journal of Environment & Development 2009;18(2):130-153.

[82] Bowman MS, Soares-Filho BS, Merry FD, et al. Persistence of cattle ranching in the Brazilian Amazon: A spatial analysis of the rationale for beef production. Land Use Policy 2012;29(3):558-568.

[83] Barros S. Brazil Biofuels Annual: Annual Report 2013. USDA Foreign Agricultural Service: Sao Paulo; 2013. BR13005. Available from: http://gain.fas.usda.gov/Recent%20GAIN%20Publications/Biofuels%20Annual_Sao%20Paulo%20ATO_Brazil_9-12-2013.pdf

[84] Veeck G, Pannell CW, Smith CJ, et al. China's Geography : Globalization and the Dynamics of Political, Economic, and Social Change. 2nd ed., Rowman & Littlefield Publishers: Plymouth, UK; 2011.

[85] Karplus VJ and Deng XW. Transformation in China's Agriculture in the Twentieth Century. In: Karplus VJ, Deng XW, editors. Agricultural biotechnology in China: origins and prospects. Springer: New York, USA; 2008, p. 27-44.

[86] Anderson K and Martin W. 9 China and Southeast Asia. In: Anderson K, editor. Distortions to agricultural incentives: A global perspective, 1955-2007. The International Bank for Reconstruction and Development / The World Bank: Washington DC, USA; 2009, p. 359-387.

[87] Karplus VJ and Deng XW. Agricultural Biotechnology Takes Root in China. In: Karplus VJ, Deng XW, editors. Agricultural biotechnology in China: origins and prospects. Springer: New York, USA; 2008, p. 55-77.

[88] Bao J. Dairy Production in Diverse Regions | China. In: Fuquay JW, editor. Encyclopedia of Dairy Sciences. 2nd ed., Academic Press: San Diego; 2011, p. 83-87.

[89] Shah A. Agriculture and environment in India Policy implications in the context of North–South trade. In: Mukherjee S, Chakraborty D, editors. Environmental scenario in India: successes and predicaments. Routledge: Abingdon, Oxon, UK; 2012, p. 219-242.

[90] Gulati A and Pursell G. 10 India and other South Asian Countries. In: Anderson K, editor. Distortions to Agricultural Incentives: A Global Perspective, 1955-2007. The International Bank for Reconstruction and Development / The World Bank: Washington DC, USA; 2009, p. 389-415.

[91] Mukherjee S and Chakraborty D. Editors' introduction: the Indian growth story: towards a sustainable development? In: Mukherjee S, Chakraborty D, editors. Environmental scenario in India: successes and predicaments. Routledge: Abingdon, Oxon, UK; 2012, p. 1-18.

[92] Alston JM, James JS, Andersen MA, et al. A Brief History of U.S. Agriculture. In: Alston JM, James JS, Andersen MA, Pardey PG, editors. Persistence Pays. Springer: New York, NY, USA; 2010, p. 9-21.

[93] Carpentier CL and Ervin DE. USA. In: Brouwer F, Ervin DE, editors. Public concerns, environmental standards and agricultural trade. CAB International: Wallingford, Oxfordshire, UK; 2002, p. 95-139.

[94] McCormick ME. Dairy Farm Management Systems | Non-Seasonal, Pasture Optimized, Dairy Cow Breeds in the United States. In: Fuquay JW, editor. Encyclopedia of Dairy Sciences. 2nd ed., Academic Press: San Diego; 2011, p. 38-43.

[95] Pelletier N, Pirog R, Rasmussen R. Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. Agricultural Systems 2010;103(6):380-389.

[96] Silvis H and van Rijswick C. Agricultural policies and trade liberalisation. In: Brouwer F, Ervin DE, editors. Public concerns, environmental standards and agricultural trade. CAB International: Wallingford, Oxfordshire, UK; 2002, p. 11-37.

[97] Blandford D and Boisvert RN. 16 Policy for agricultural adjustment in the United States. In: Blandford D, Hill B, editors. Policy reform and adjustment in the agricultural sectors of developed countries. CAB International: Wallingford, Oxfordshire, UK; 2006, p. 237-253.

[98] Alston JM, James JS, Andersen MA, et al. Research Funding and Performance. In: Alston JM, James JS, Andersen MA, Pardey PG, editors. Persistence Pays. Springer: New York, NY, USA; 2010, p. 137-185.

[99] Alston JM, James JS, Andersen MA, et al. The Federal Role. In: Alston JM, James JS, Andersen MA, Pardey PG, editors. Persistence Pays. Springer: New York, NY, USA; 2010, p. 187-236.

[100] ASTI Agricultural Science and Technology Indicators. ASTI Data Tool version 1.1 [Internet: accessed 23 August 2013]. Available from: http://www.asti.cgiar.org/data/

[101] Bonaglia F. Zambia: Sustaining Agricultural Diversification. OECD Journal: General Papers 2009;2009(2):103-131.

[102] Howard JA and Mungoma C. Zambia's stop-and-go revolution: the impact of policies and organizations on the development and spread of maize technologies. International Development Working Paper No. 61. MSU Agricultural Economics: Michigan, USA; 1996.

[103] Pletcher J. The Politics of Liberalizing Zambia's Maize Markets. World Development 2000;28(1):129-142.

[104] Jayne TS, Govereh J, Wanzala M, et al. Fertilizer market development: a comparative analysis of Ethiopia, Kenya, and Zambia. Food Policy 2003;28(4):293-316.

[105] Djurfeldt G, Aryeetey E, Isinika AC. African Smallholders: Food Crops, Markets and Policy. CAB International: Wallingford, Oxfordshire, UK; 2011.

[106] Haggblade S and Tembo G. Conservation farming in Zambia. International Food Policy Research Institute: Washington, DC, USA; 2003. EPTD Dicussion Paper No. 108. Available from: http://www.ifpri.org/sites/default/files/publications/eptdp108.pdf

[107] Matondi. Zimbabwe's Fast-Track Land Reform. Zed Books: London, UK; New York, USA; 2012.

[108] Marongwe LS, Nyagumbo I, Kwazira K, Kassam A, Friedrich T. Conservation Agriculture and Sustainable Crop Intensification: A Zimbabwe Case Study. Integrated Crop Management 2012;171-29.

[109] USDA National Agricultural Statistics Service. Quick Stats [Internet: accessed September 2013]. Available from: http://quickstats.nass.usda.gov/?source_desc=CENSUS

[110] USDA Foreign Agricultural Service. PSD Online [Internet: accessed September 2013]. Available from: http://www.fas.usda.gov/psdonline/psdQuery.aspx

APPENDICES

A2.1 HISTORICAL DEVELOPMENTS IN GLOBAL AGRICULTURAL SECTOR

In the first half of the twentieth century, global agriculture had been affected by weather crises (droughts, floods, famines, plant and animal diseases), the great depression in 1930, and two world wars [61,62]. Later, after a decolonization process during the 1950s-1960s, the agricultural sector in former colonies was also under-performing [61]. At the same time, agriculture had to fulfill an important role in supporting the industrialization process by providing cheap food to urban work force [61,63]. As a result, during the 1940s-1970s, governments aimed to increase their production and continue the modernization and mechanization of agricultural systems which had started in the 19th century [62]. To support and protect agricultural production and prices, states adopted agricultural protection policies. Since the 1950s, many developed countries imposed strong state controls on agriculture and trade through instruments like import tariffs, export subsidies and producer support [22,29,63]. These policy instruments guaranteed a minimum return to farmers and compensated for price differences between the internal and global market [22,63]. Developing and industrializing countries heavily taxed agricultural exports and protected producers from import competition [61]. Rapid economic growth facilitated investments in agricultural R&D, which led to breeding of high-yielding crop varieties, increasing application of fertilizers and irrigation, and mechanical innovations [22,28,62]. This is referred to as the Green Revolution. By the 1970s, the Green revolution and state intervention resulted in overproduction in developed countries. Some less developed countries like China and Brazil were also able to increase agricultural production at a higher rate than population growth. But other developing countries (especially in Africa) lagged behind and suffered from under-production [61].

In the last 25 years, both developed and developing countries have begun to reform their agricultural policies, which has resulted in growing international trade of agricultural products. Trade liberalization started with the first General Agreement on Tariffs and Trade (GATT) in 1947. Although the trade of agricultural products was subject of international negotiations, it was typically excluded from multilateral trade agreements until 1990 [61,63]. In 1995, the Uruguay Round Agreement on Agriculture (AoA) came into effect and imposed measures on signatory countries to open their agricultural markets [61,63]. Another issue that remained unresolved in negotiations was the question how policies for sustainable development and environmental protection could be aligned with and integrated in trade regulations. Although environmental stewardship is a global issue, the interests in multilateral negotiations are diverse. While developed countries emphasize the need for an environmental reform of trade regulations, developing

countries are mainly concerned with questions related to market access, dumping and agricultural subsidies [64]. As will be shown in the next sections, this diversity in interests is also reflected in the varying degree of adoption and enforcement of agri-environmental policies in the selected countries.

A2.2 AUSTRALIA

Agricultural characteristics

Australia is a dry continent. Its climatic zones range from a tropical Northern region, through an arid interior, to a temperate Southern region [65]. The wet northern summer conditions allow beef cattle grazing and sugarcane production (east coast). The drier southern summer conditions favor wheat production, and grazing of sheep and dairy and beef cattle [65]. Rice is mainly grown in the South East of Australia [65]. About 10% of the agricultural area is cultivated (cropping and sown pastures and grasses) [24,65]. The remaining area consists of permanent pastures and meadows for livestock grazing [24]. The majority of the farms are engaged in either livestock farming or grain growing [65].
The management levels are low by OECD standards. Until the early 1980's, about 3.5% of the cultivated land (arable land) was equipped for irrigation. From the late 1980's, this share increased to about 5.5% in recent years [24]. Irrigation is mainly applied for vegetables and fruits, rice, and also sugarcane [65,66]. Because low rainfall limits the returns on fertilizer expenditure, fertilizer use is relatively low [66], see Figure 2-2. The highest rates of fertilizer and pesticide are applied in horticulture (fruits and vegetables) [66]. The average ruminant density on pastures is very low (see Figure 2-15 in appendix A2.9). Beef cattle are mainly held in Northern Australia, where production is extensive and the technology level low. In the South, production is more intensive. This is illustrated by higher stocking rates per hectare, improved pastures, and the use of fodder crops and animal health products [65]. Dairy production mainly takes place in the South-Eastern high-rainfall coastal areas and is based on year round pasture grazing. Feedlot-based dairying is expanding, but is still uncommon [65]. Between 1980 and 2000, farmers switched to another dairy cattle breed [67].

Economic and institutional developments

Compared to other OECD countries, Australia has been more protective to industry for most of the 20th century. The country did not participate in the General Agreement on Tariffs and Trade (GATT) between 1947-1979. In the 1950s and 1960s, Australia's policies were focused on industrial protectionism, characterized by price support, and trade protection (e.g. import restrictions on manufacturing products). These policies isolated farmers from national and international market signals [66] and resulted in indirect

disincentives for agriculture [68]. The subsidies and protection provided to the agricultural sector were limited and could not offset these disincentives [68].

From the 1970s, Australia has been reforming its trade policies [68]. The past two decades have been a period of especially rapid total factor productivity (TFP) growth [68]. One important factor explaining the increase is the openness of the Australian economy to trade and investment [42]. In 2002, Stringer [66] mentions that "four-fifths of the Australian agricultural production is exported". The main export markets are the USA and Asia [66].

Until the 1990's, agricultural policies were led by socio-economic objectives. Since the 1990's, more emphasis has been put on sustainable agricultural development [66]. Current agricultural policies aim to improve market responsiveness, and encourage sustainable agricultural practice (i.e. approaches that combine economic, environmental and social aspects). Projects also focus on food quality. Agri-environmental policies concentrate on water, soil erosion, salinity, and biodiversity loss [66]. Measures to achieve this include research, education, voluntary adoption of best practice, and development of guidelines in collaborations between governments, NGOs, industries, and communities. Over time, regulatory approaches have been complemented, or even substituted, with market oriented mechanisms like for example the polluter pays principle. The basis of these measures is that prices reflect social costs and benefits, i.e. positive and negative externalities are taken into account [66].

Yield developments

In Australia, a small average change in wheat and rice yields between 1961 and 1980 was followed by a significant increase in the 1980s (Figure 2-3 and Table 2-1). As this coincides with the participation in the General Agreement on Tariffs and Trade (GATT) after 1979, it is likely that the improvements are a result of the trade policy reforms which opened the international market [68]. Notably, the input use levels remained stable. Thus, it seems that the reforms motivated farmers to use their resources more efficiently, which resulted in higher production levels and improved productivity in the early 1980s (Figure 2-6). Agri-environmental policies were first implemented in the 1990s and aimed to improve market responsiveness and encourage sustainable agricultural practice [66]. Since their introduction, agricultural production has further increased. But, fertilizer use and irrigation levels increased as well and the productivity of the agricultural sector did not improve compared to the 1980s (Figure 2-6). Also, wheat and rice yields stagnated in the 1990s. Sugarcane yields have been relatively steady and did not significantly improve in the 1980s. A considerable drop in yield in the period 1990-92 was followed by a peak in the second half of the 1990s and another plunge in 2001-02 due to drought. Then, sugarcane yields seem to have stabilized again around previous levels.

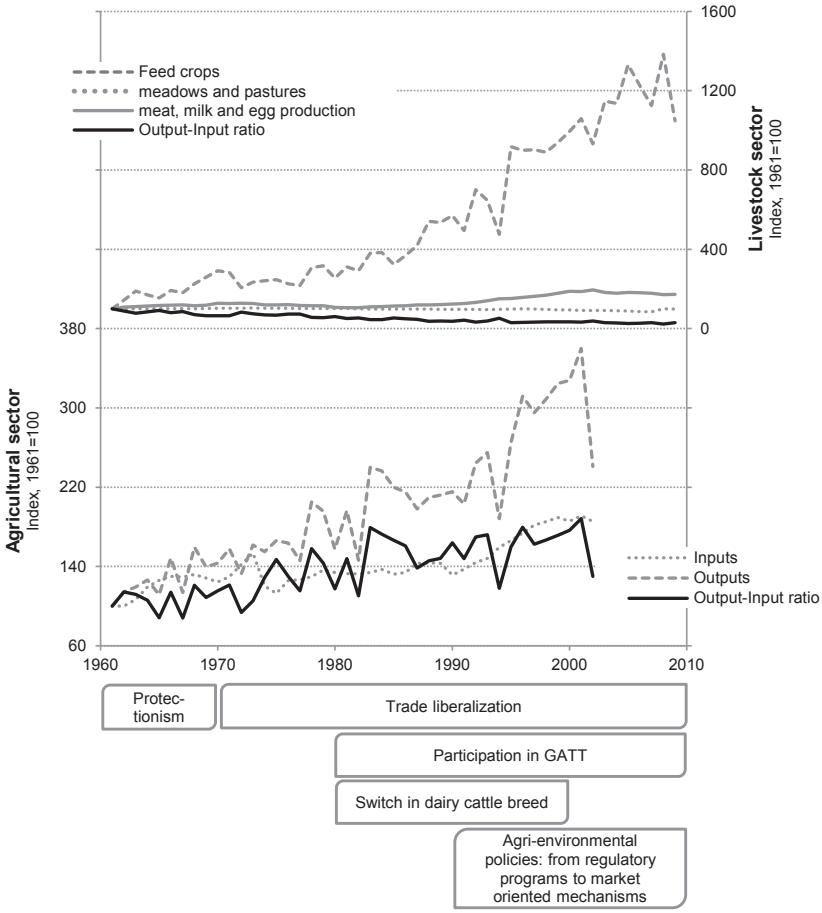

FIGURE 2-6 | Australian agricultural and livestock productivity developments and institutional, economic and technological/management developments. (GATT, General Agreement on Tariffs and Trade).

Beef and milk yields have almost continuously improved, except for a period of stagnation in the 1970s and early 1980s (Figure 2-14 and Table 2-1). In this period of market reforms, also the production and export of beef and milk temporarily declined [24]. Between 1980 and 2000, beef and milk yields improved while their export markets grew, feed crop consumption increased enormously (Figure 2-6), and a shift in dairy cattle breed from British breeds to Holstein-Friesian animals was made [67,69]. Although the majority of cattle production is still extensive, the rise in feed crop consumption is likely to be related to the intensification of beef production in South Australia and the expansion of feedlot-based dairying [65]. The above findings suggest that the development of export markets has been an important driver for changes in the production systems of beef and milk, and all these factors together have contributed to yield improvements. Due to the massive increase in feed crop use, however, the output-input ratio of the livestock sector has

decreased significantly. The decline has slowed down since the mid-1990s, which may be related to the introduction of agri-environmental policies.

A2.3 BRAZIL

Agricultural characteristics

Agriculture in Brazil is characterized by concentrated land ownership; medium- and large-scale commercial farms contribute the bulk of agricultural output [70]. Crop and livestock are combined in mixed farming systems [71]. The majority of cattle is held in *extensive mixed* (rain-fed) production systems [72]. These are found in the wooded and open savannah areas (the Cerrados) in the Central-West of Brazil, and also in the South-East [71,73]. Extensive ranching is the primary activity, but cultivation of soy and corn is increasing. In addition to this farming type, i*ntensive mixed farming* takes place in Eastern and Central Brazil. This system produces most of the sugarcane, which is mainly cultivated in the Central-South of Brazil. D*ryland mixed farming* (which is mainly semi-subsistance farming) is the major system in North-Eastern Brazil [71].
A significant expansion of the agricultural sector in the last decades was accompanied by a considerable increase in deforestation, replacement of native vegetation and biodiversity loss [74,75]. Also, the use of fertilizers and other inputs, and the ruminant density on meadows and pastures have risen significantly since the 1970s (Figure 2-2 and Figure 2-15). The ruminant density in Brazil is high compared to the other selected countries, except to India. The largest share of the cattle population is being held for beef production; about 10% are dairy cows [24,73]. Beef and milk production are mainly based on extensive systems, in which cattle grazes on pastures all year round [73]. In the emerging semi-extensive systems, herds also receive supplemental feed from crops and various concentrates. Feedlot-based, intensive systems are still rather uncommon [73].

Economic and institutional developments

Prior to 1950, the Brazilian market was concentrated around the export of food and raw materials, and the import of industrial products [76]. Between 1950 and the mid-1970's, the focus shifted to national industrialization, and policies aimed at replacing foreign imports with domestic production [77]. This is called Import Substitution Industrialization (ISI) [78]. To protect the industry, wage rates were kept low by restrained food prices [77]. In order to realize low food prices, Brazilian agriculture was heavily and increasingly taxed [78]. Levies consisted partly of direct export taxes, but were dominated by indirect taxation resulting from industrial protection policies [70]. Due to the industrial protection, also input prices increased. Therefore, the government provided credit and fertilizer subsidy to promote the use of fertilizer and other inputs [77].

Trade liberalization started in the 1980's, and continued to the mid-1990's [70]. Reforms included the removal of import and export restrictions, and the redistribution of resources from import-competing to export-oriented sectors [70]. This transformation took also place in the agricultural sector [70]. Today, major exports of agricultural products like soybean, sugar, beef and ethanol contribute to Brazil's positive balance of trade. Thus, the agricultural sector plays an important role in the economic development of Brazil [74,75]. During the last years of industrial protectionism, the oil crisis in 1973 prompted the Brazilian government to phase out fossil fuels. The ProÁlcool program was launched to promote the sugarcane industry and bio-ethanol production [79]. Blending ethanol to fossil fuel was already introduced in 1931, but the ProÁlcool program brought about a major increase in ethanol consumption and production [79]. Today, most of the ethanol produced is still intended for the domestic market. In 2009, almost 14% of the production was exported [80]. To improve Brazil's energy diversity and independence, a second biofuel program was implemented in 2004: the National Program of Production and Use of Biodiesel (PNPB). Due to the abundance of soy and the search for new soy markets, biodiesel production is largely based on the conversion of soybean oil [79].
The need to control deforestation was already recognized in the 1920s. The first Forestry Code, which dates from 1934, regulated the conservation of forests on private land [81]. In 1965, a second code expanded the land dedicated to preservation from forests to other sensitive areas. Also, it created conservation areas outside the private rural properties [81]. Due to economic priorities, however, enforcement of the codes was weak. Between 1974 and 1987, when the focus shifted from protectionism to trade liberalization, the government promoted livestock production, forestry, and mining in the Brazilian Amazon [81]. The markets for Brazilian beef have been growing since the 1970s and led to considerable expansion of extensive cattle ranching on cleared forest land in this region [82]. In addition, the more recent expansion of soy production on previous pastures causes further expansion of cattle ranching into the Amazon [82]. Policies and other initiatives that aim to intensify cattle production are in early stages yet [82].

Yield developments

In Brazil, the developments in the yield of corn and soybeans are comparable. From the 1960s to 1980s, yield growth rates were moderate and highest in the 1970s (Figure 2-3 and Table 2-1). In the 1990s, high improvement rates of more than 3.5% yr^{-1} were attained (relative to 1991). Fertilizer use increased substantially in the late 1960s and the 1970s, declined and stagnated in the 1980s and increased again in the 1990s (Figure 2-2). Thus, it appears that fertilizer subsidies provided during industrial protection (1950s-70s) [77] led to an increase of fertilizer use and of yields, especially in the 1970s. The economic reforms in the 1980s [70] temporarily hindered agricultural development, but the opening of agricultural export markets stimulated further improvements in the 1990s. Notably,

after agricultural productivity declined in the 1960s and 1970s, the output-input ratio has remained fairly constant from the 1980s until 2002 (Figure 2-7). Despite the introduction of the biodiesel program in 2004 [79], soybean yields stagnated in this decennium. Yet, the share of soybeans used for biodiesel production has been rather small in the first years (0.5% in 2006 compared to 12% in 2010) [24,83]. The development of sugarcane yields is similar to the trend for corn, but the first period of major growth is found between 1975 and 1985. This is clearly related to the introduction of the PróAlcool program in the early 1970s [79]. Tractor use and irrigation also grew significantly after 1973, thus are probably related to the rise of sugarcane production.

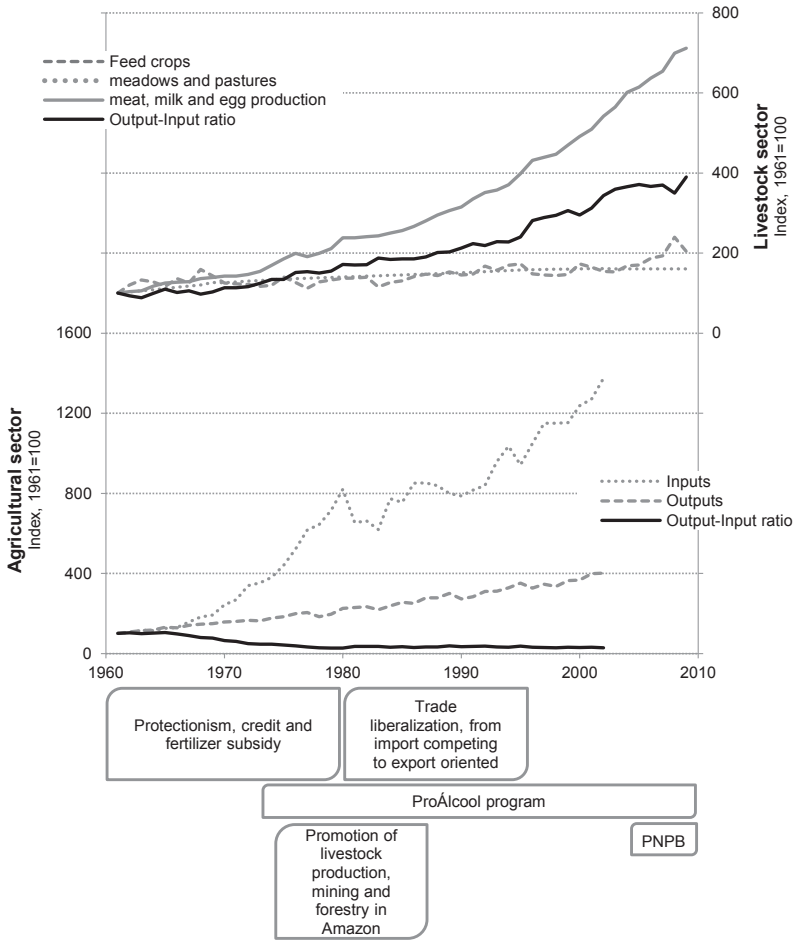

FIGURE 2-7 | Brazilian agricultural and livestock productivity developments and institutional, economic and technological/management developments. (PNPB, national program of production and use of biodiesel).

Considering beef and cow milk production, no significant yield improvements were attained from 1961 until the 1980s (Figure 2-14 and Table 2-1). This period of relatively stable yields was interrupted by a few years of decline in the mid-1970s. Major yield increases were only attained in the early 1990s. This suggests that the promotion and expansion of cattle ranching during the period of liberalization [81,82] did not directly stimulate yield improvements. Only when the export markets were fully opened in the early 1990s, yields significantly improved. For milk, this growth continued in the late 1990s and 2000s, but at a lower rate. The initial increase in beef yields was first followed by a decline in the late 1990s, before yields increased again in the 2000s. As the production of beef and milk has grown significantly faster than the use of feed crops and the area of pasture land, the output-input ratio of the livestock sector has continuously increased over the past five decades, especially since the mid-1990s (Figure 2-7).

A2.4 CHINA

Agricultural characteristics

Agriculture in China is characterized by large environmental diversity, and large diversity in agricultural products. The Qinling mountains divide China into water-deficit (North, West) and water-surplus regions (South, Northeast) [84]. There are four major farming systems [71]. *Lowland rice production* is found in humid and moist subhumid areas in South and Central East China. Rice production is rain-fed, with supplementary irrigation where available. Important livelihoods are, besides rice, subsidiary crops like corn and soybeans, livestock, and off-farm work. *Upland Intensive Mixed Farming* is found in upland and hill areas with humid and subhumid climate (South East and North China). A significant area, mainly rice, is irrigated. Livestock contributes draught power, meat, income and savings. Also off-farm work is an important income source. *Temperate mixed farming* is found in moist and dry subhumid areas in Central Northern China. The major crops are wheat and corn. Livestock is also an important livelihood.

Pastoral farming is located in semiarid and arid temperate climates in Western China. Pastoralism is based on extensive grazing of mixed herds (camels, cattle, sheep and goats) on native pasture. In local suitable areas, farmers apply irrigated crop production (e.g. cotton, barley, wheat). Characteristic for the agricultural sector of China are the majority of small-scale farms and the application of multiple cropping systems, i.e. the production of more than one crop per year on the same land [84].

China aims to be largely self-sufficient in grain production. Because of its large population, land availability for agricultural production is an important issue [84]. Economic growth has caused a major increase in population and demand for housing, transport and industry in the Eastern coastal area. Much land in this region, however, is fertile and highly productive agricultural land which is lost due to the urbanization process [84,85].

Due to this pressure, even marginal lands (e.g. with very limited precipitation or extreme slope) are cultivated [84].
Without irrigation, the dry areas are of marginal use for intensive agriculture. Water shortages hamper the improvement of agricultural production in these regions. Therefore, irrigation has expanded at a high rate (Figure 2-2). This, however, is causing severe water shortages as water consumption outpaces replacement through precipitation; there are major concerns that groundwater reserves are being depleted, especially in arid areas in Eastern and Western China [84,85]. Fertilizer use has increased dramatically in the past five decades (Figure 2-2). Fertilizer use is especially high in lowland rice and in temperate mixed farming systems [71]. The overuse of fertilizers is associated with land degradation, air pollution, and eutrophication of water sources [84].

Economic and institutional developments

The period from 1949-1976 in China is called the Maoist era. During this era, the Chinese Communist Party (CCP) was in power. Governance was characterized by a strong inward orientation and self-imposed isolation [84]. Also, policies were relatively homogenous for the nation as a whole, and the use of capital and resources was regulated centrally [84]. In this era, farming took place in large farming communes [62]. These communes were difficult to manage. In 1959, this caused a collapse of farm production and a huge famine. In order to solve the problems, agricultural reforms were introduced which included the reparation and construction of irrigation systems and the distribution of high-yield seeds [62,85]. Programs to develop improved crop varieties had already started in the early 1950s. The communes were broken up when the market reforms started in 1978. Since then, individual farmers have been leasing land from the local authorities. The resulting diversification of crop production and farmers' activities, income and education level became visible in the second half of the 1990s [84].
Since the end of the Maoist era, the CCP has still been in charge. The central government continues to play an important role in planning and guiding the direction of development (e.g. economic decision making), but the role for local governments is increasing [84]. Also, from 1978, the focus of the market shifted from import substitution industrialization toward export-oriented development strategies [86]. This resulted in an export-led industrial growth, and also a restructuring of the economy away from agriculture and heavy industry toward light manufacturing and service activities [84,86]. The taxation of agricultural exports has been reduced, but the protection of import-competing agriculture, especially of rice, has been increased [86]. Because of the importance of the agricultural sector, the government increased investments in agricultural R&D and started to fund research in biotechnology. This support was continued in the following decades [87]
Environmental protection laws were first introduced in the late 1980's. These laws aimed to prevent the loss of high-productivity cropland caused by the expansion of urban and

industrial areas in Eastern China [84]. Also, the increased awareness about environmental problems and the need for more efficient agriculture led to the implementation of the Comprehensive Agricultural Development (CAD) program in 1988 [84]. The CAD program was introduced because of the low productivity of a large share of arable land and increasing grain imports. The program aimed to enhance the quality of agricultural land through better land management, including improved fertility and drainage, balanced use of inorganic fertilizer, irrigation and water storage and conservation. The CAD still exists [84].

Yield developments

Since the agricultural reforms in the early 1960s [62,85], irrigation, fertilizer use and mechanization have been increasing almost continuously (Figure 2-2). In addition, fast yield growth is found for all crops (corn, rice and wheat), Table 2-1. On average, the average yield growth rate was highest for wheat (1.8% yr^{-1} relative to 2010), but the highest absolute improvement was achieved by corn and rice (93 kg ha^{-1} yr^{-2}). These considerable gains can mainly be attributed to the introduction of new technology, which was realized by significant public investments in infrastructure and research [85]. In the 1990s, however, yield improvements dropped. This may reflect the diversification of farmers' crop production after the economic reforms [84]. In the following decade, the growth rates of corn and wheat yields increased again, while rice yield improvements continued to slow down.

The yield growth rates for beef and milk have been lower than for crops. This may be explained by the major importance of cattle for delivering draught power, as agricultural production is still labor-intensive in China (Figure 2-2). The market reforms may have resulted in some improvements in cattle production; the beef and milk yield shortly increased around the 1980s. Afterwards, beef yields stabilized again. Milk yields also stagnated for some years, but have been increasing at a rate of 3.5% yr^{-1} since the late 1990s (relative to 2001, Figure 2-14 and Table 2-1). In the same time, the consumption of milk and dairy products by urban residents soared, which was caused by China's growing prosperity [88].

After the output-input ratio of the agricultural sector had declined rapidly in the 1960s and 1970s, it stabilized in the 1980s and improved gradually in the 1990s (Figure 2-8). It is likely that both market liberalization and agri-environmental policies, which aim to improve agricultural land quality [84], have contributed to this reversal of the downward trend in productivity.

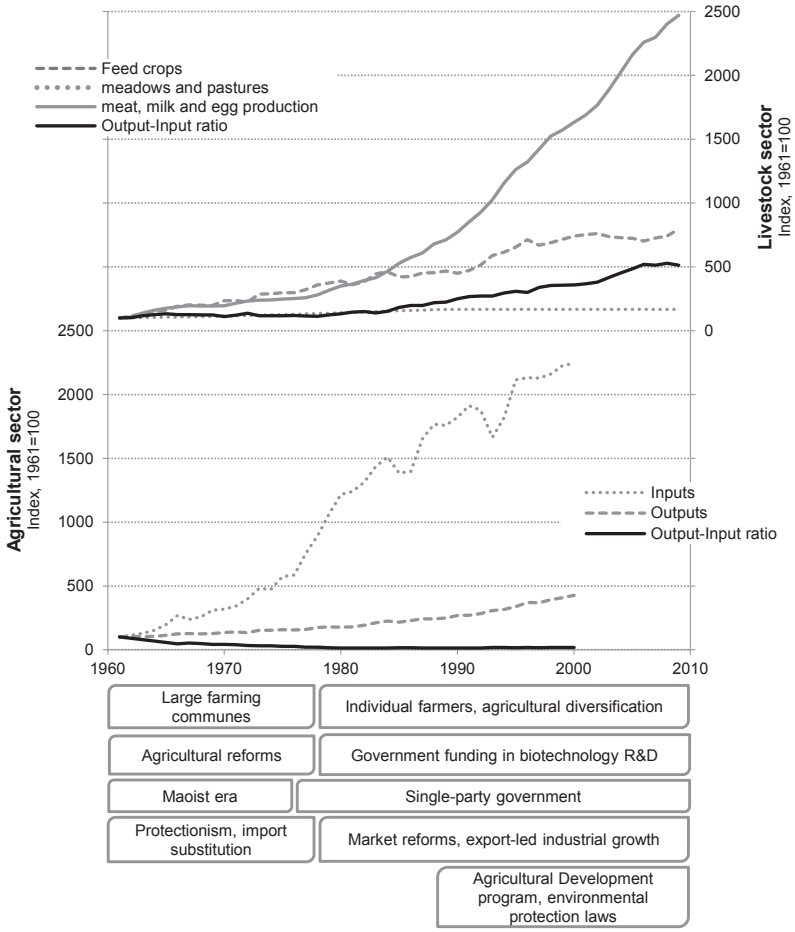

FIGURE 2-8 | Productivity developments in the Chinese agricultural and livestock sector and institutional, economic and technological/management developments.

A2.5 INDIA

Agricultural characteristics

India has two major farming systems: rice-wheat and rainfed mixed. The rice-wheat system in Northern India is characterized by wetland rice production in summer (monsoon season), and irrigated wheat production in winter (cool, dry season). A significant amount of livestock is held in this system, where bovines produce draft power, milk and manure for composting [71].

The rainfed mixed system occupies the largest area in India (Central and Southern India). It is mainly rain dependent, but according to Dixon [71], about 16 percent of the area cultivated under this system was equipped with simple, small-scale irrigation techniques

around 2000. Infrastructure and market access are poor, and agricultural activities are oriented toward subsistence. The main livelihoods are cereals, legumes, fodder crops, livestock and off-farm activities [71].

The input-intensity of the agricultural sector has increased substantially since 1961 (Figure 2-2). Yet, as the increase in input use outpaced the growth in total production, the output-input ratio has declined seriously (Figure 2-9). This has caused major environmental issues. Large-scale irrigation in Northern India has inflicted soil salinization and groundwater depletion [89]. Also, groundwater is polluted due to intensive use of fertilizers and rudimentary processing of livestock wastes. In addition, large livestock populations cause soil degradation through the conversion of natural vegetation [89]. In Southern India, soil erosion is the main problem. The vegetative cover and organic matter content of soils are low. Yet, farmers continue to cultivate crops on marginal lands to meet their basic needs [89].

Economic and institutional developments

India is a former colony of the UK and gained independence in 1947. In order to prevent famines, and to ensure affordable prices for basic foods, the Indian government has been intervening in the food market since its independence in 1947. In the public distribution system, which was established in 1958 and is still present, basic foods are sold at subsidized prices [90].

In response to droughts and famines in 1965-66, policies aimed at food grain self-sufficiency and agricultural imports began to be replaced by domestic production [62,90]. Green revolution technologies played an important role, as the government implemented many programs to modernize agriculture at a high speed. This included the development and planting of high-yielding wheat and rice varieties and large subsidies for electricity and fertilizers [62,90]. According to Dixon [71], however, agricultural development during India's Green Revolution did mainly take place in 10 percent of India's districts which had adequate local infrastructure for water management, transport and electricity (for tubewells). In the 1970's and early 1980's, the import of edible oils expanded significantly. This led to policies which aimed to decline these imports and substitute them with domestically produced oils [90]. Import substitution was abandoned in the 1990s and the focus shifted to an export oriented economy. Trade policies were reformed through the structural adjustment program (SAP), which was introduced 1991 [91]. A significant number of environmental policies exist that aim to control the use and management of natural resources. Enforcement of these regulations, however, is weak [89].

Yield developments

In India, crop yields have almost continuously grown in the last five decades, but at different rates. The highest rates are found for wheat and rice in the period 1961-1990 (Table 2-1). Explanations for these achievements can be found in the major investments by the government in the modernization of agriculture (in Northern India), the development and adoption of high-yielding rice and wheat varieties and agricultural subsidies [62,71,90]. After 1990, the absolute and relative yield growth of these crops has decreased. Soybean yields started to increase in 1972. This coincides with the introduction of the import substitution policy for edible oils [90]. Since 1961, sugarcane yields have increased at a moderate rate until 1999. After a decline in 2000-2004, yields returned to the level of the mid-1990s. For all crops, there is no clear relation between market reforms and yield developments in the 1990s. Although the linear regression data suggest that yield growth of rice, wheat and soybeans slowed down in this decade, the graphs in Figure 2-3 do not show a significant deviation from an earlier trend which can be linked to the liberalization process.

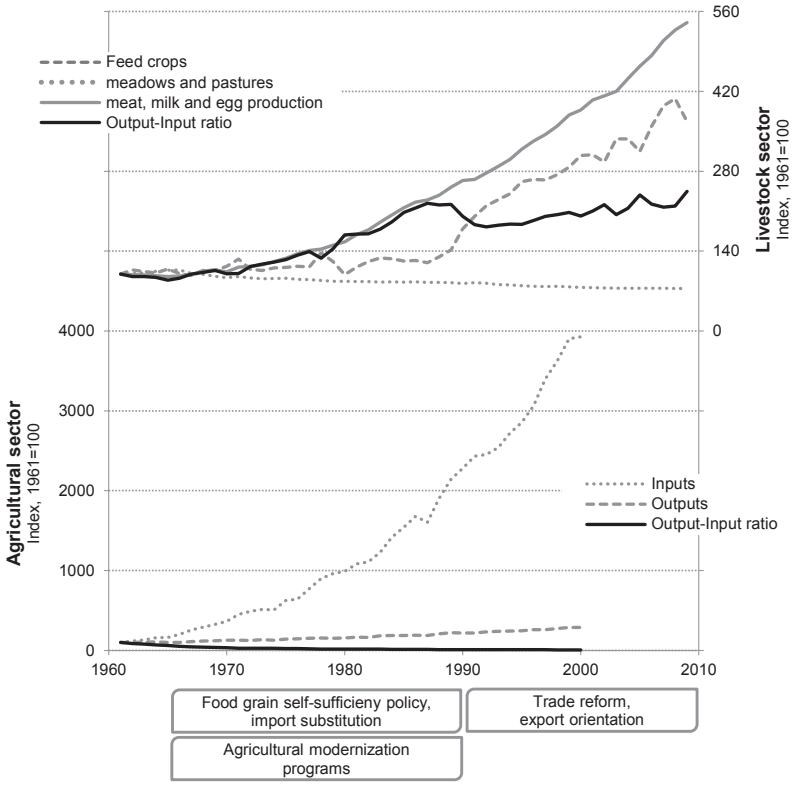

FIGURE 2-9 | Productivity developments in the Indian agricultural and livestock sector and institutional, economic and technological/management developments.

Regarding cattle product yields, there is a significant difference in developments between milk and beef. Cow milk yields have increased all five decades, and growth rates increased as well. Similar to China, this is likely to be related to increased milk consumption [24]. Beef yields are low compared to the other six countries. This is likely to be related to the protected status of cows in Hinduism, the major religion in India. Beef yields have been rather constant and only increased in the 1970s and 1980s. This temporary improvement may be explained by reduced need for draft power due to the mechanization process (Figure 2-2). Mechanization, however, has continued in 1990s, but this is not reflected in further improvement of beef yields.
Due to the enormous increase in inputs, the agricultural productivity has continuously decreased between 1961 and 2000 (Figure 2-9). Although the decline has slowed down, the lack of productivity improvements confirms the weak enforcement of agri-environmental policies in India [89].

A2.6 USA

Agricultural characteristics

In the USA, agricultural production is mainly concentrated in the Pacific and Central (Midwestern) regions and the Southern plains [92]. Over the past decades, the total number of farms in the USA has declined but the number of large scale farms has increased. Still, large-scale farms are a minority of all US farms, but they produce more than two-third of agricultural output [70,92]. The production practices depend on the farm size and the natural resource base (e.g. soil moisture and fertility). For example, the major practice in the corn belt (Midwestern USA) is dryland farming. In the Central valley of California (Eastern USA), irrigation is applied [93].

The agricultural production of the last five decades can be characterized by intensive management, especially with regard to mechanization (Figure 2-2). It seems, however, that the intensity of input uses has stabilized since the 1970s and 1980s. Also, the ruminant density on pastures has declined (Figure 2-15). In the livestock sector, dairy production is mainly confinement based [94]. Pasture use, however, has grown since the early 1990s [94]. In pasture based production systems, dairy cows may be at pasture during parts of the year. At the same time, and in winter months, they receive stored forage along with varied levels of supplemental concentrates throughout the year [94]. Beef production is mainly characterized by cow-calf herds on pasture and (winter) hay [95]. Beef cattle are finished in feedlots where they receive a mixed, high concentrate feed ration. Less than 1% of beef cattle are finished in pastures [95]. On pastures, no housing is provided for cow-calf herds. Hormone implants are employed in the feedlot stage. Calves can also be sent to feedlots directly. Pelletier [95] mentions that this is common practice in the US Upper Midwest.

The high intensity of agricultural management in the USA has led to a wide range of environmental issues. The most important problems are soil erosion (i.e. loss of the fertility and water-holding capacity of the soil) and contamination of water sources by agricultural chemicals and livestock manure [93].

Economic and institutional developments

From their introduction in the 1930's, agricultural policies in the USA have been differing significantly in composition from the EU and other OECD countries. The focus of agricultural policies has been on providing food aid and nutrition assistance. Assistance to farmers in the form of commodity support programs is placed second [96]. These commodity programs consisted of price support and direct income payments [96,97]. To limit payments by the government, crop programs, such as corn and wheat, placed limits on production. For other commodities, like milk, import restrictions were applied [97]. Market liberalization and multilateral trade agreements have changed the programs for farm support. Since 1985, income support has shifted to payments that are decoupled from prices and production. Also, production limitations have been replaced by more planting flexibility, enabling farmers to make market-based decisions [97].
The USA have been dominant in agricultural R&D expenditures [98]. For example, Figure 2-10 shows that American public spending in 2000 was twice as high compared to the investments made by China. In addition, agricultural R&D in the USA has been funded extensively by the private sector. The public and private sector contribute both about half of the total investments [98]. In other countries, especially the developing countries, the share of private spending has been much smaller (Figure 2-11). In addition, innovations in agricultural technology have been stimulated by intellectual property rights. Until the 1970s, however, this protection excluded inventions related to living organisms like plants and animals [99]. This changed in the 1980s and the legalization of patents on life forms cleared the way for biotechnology to rapidly expand [99].

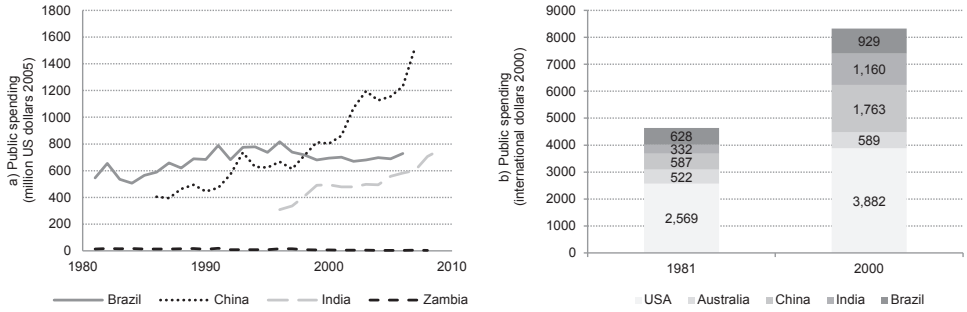

FIGURE 2-10 | Public agricultural R&D spending: a) historical progress in spending in developing countries [100]; b) spending in OECD and developing countries [98]

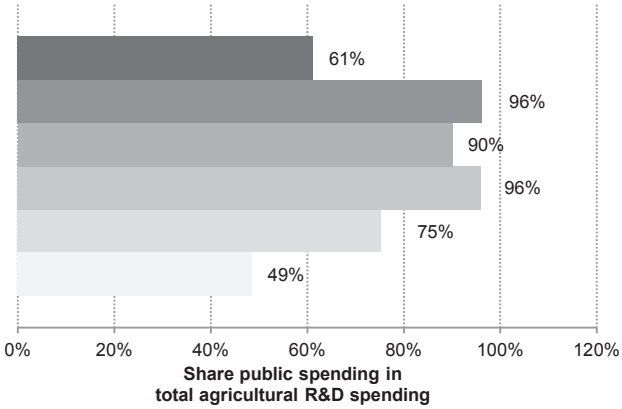

FIGURE 2-11 | Public share in total agricultural R&D spending in 2000 [98].

Agri-environmental policies in the US have been implemented from about 1970 [93]. Traditionally, broad programs were implemented in each state. More recently, individual state and local programs have emerged [93]. The majority of agri-environmental policies have been voluntary-payment programs [93]. The application of regulatory approaches which define input or performance standards for agriculture was limited by two factors. First, agriculture in the US is characterized by a high variety in production practices and local circumstances (ecosystems, quality and sensitivity of resources). Regulation of this diversity in operations is technically difficult and expensive. Second, a strong agricultural lobby exists in the US, and political influence on environmental protection has been modest [93]. As a result, regulatory programs and environmental standards for agriculture have only been introduced since the second half of the 1990's. These regulatory programs focus on the quantity and quality of production inputs (especially water quality in livestock production, and pesticide use) [93]. The most important environmental policies in the US are the Conservation Reserve Program (CRP, farm bill 1985) and the Environmental Quality Incentives Program (EQIP, farm bill 1996). The CRP is a voluntary program that provides payments to farmers who apply conservation practices on environmentally sensitive lands [96]. The EQIP provides financial and technical assistance to farmers to improve and protect the environmental quality of their properties (e.g. soil and water) [96].

Yield developments

In the USA, the crop yield growth trends have been positive for most of the period 1961-2010 (Figure 2-3). It is very likely that the substantial investments in agricultural R&D [98] have played an important role in achieving these improvements. From the mid-1980s, growth of corn and wheat yields slowed down for about a decade. Probably, this deceleration is related to the reforms of trade and farmer support policies in the same

period [97]. Afterwards, however, absolute growth reached a record high in the 1990s (Table 2-1). For soybeans, growth accelerated in the 2000s. In addition to the effect of trade liberalization, these significant increases in yield growth may be attributed to the rise of biotechnology since the 1980s [99]. Improvements in technology and management have also driven yield growth in beef and cow milk production. This technological progress is likely to be stimulated by investments in R&D and growing domestic milk consumption [24,98]. Although absolute and relative yield growth of beef was highest in the 1960s, yields have also been increasing considerably since the mid-1980s after a period of stagnation in the 1970s. Cow milk yields have almost continuously increased and while the relative growth rate has been fairly constant, the absolute growth accelerated in the 1980s and 1990s.

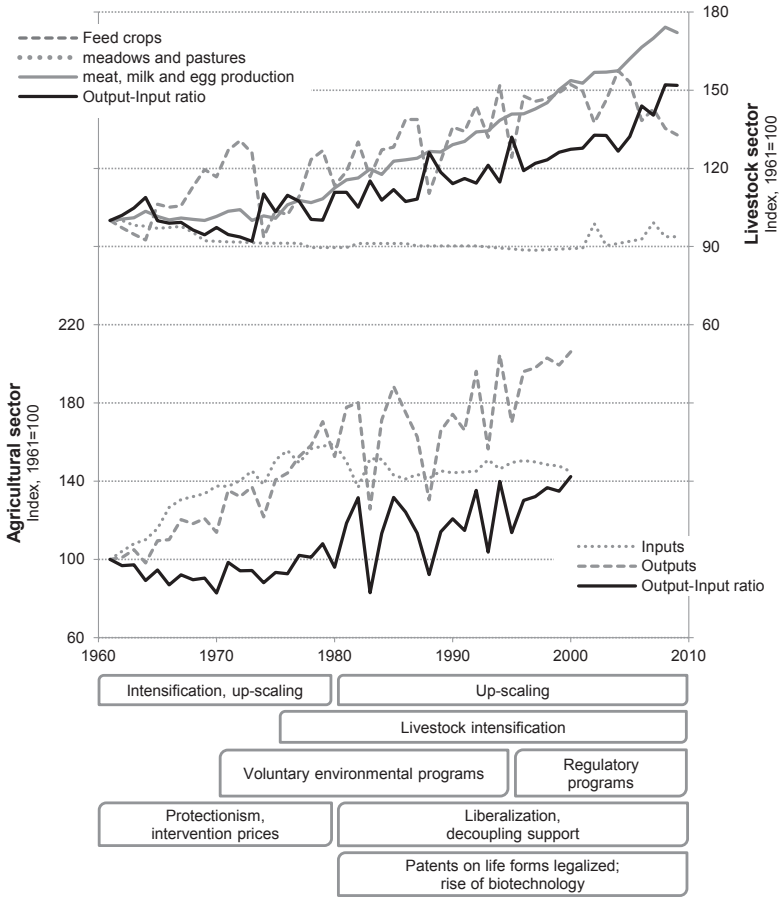

FIGURE 2-12 | Productivity developments in the American agricultural and livestock sector and institutional, economic and technological/management developments.

Regarding agricultural management, tractor use peaked in 1966 and fertilizer use reached the highest level in 1980 (Figure 2-2). Also, the output-input ratios of the agricultural and livestock have been improving since the 1970s (Figure 2-12). As agri-environmental programs were introduced in the same decade, the developments in input use and agricultural productivity may well be related to these policies.

A2.7 ZAMBIA

Agricultural characteristics

The agricultural sector in Zambia exists of a small number of large-scale commercial farmers who have good access to input and output markets, a few medium-scale commercial farmers for whom market access is difficult, and a majority of smallholders who are often engaged in subsistence farming [101,102]. The large-scale farmers produce and sell wheat, soybean, coffee, milk and other livestock products. Corn, however, dominates the agricultural sector and is mainly produced by the smallholders and medium-scale commercial farmers [102].

The major farming systems in Zambia are maize mixed (Central and East Zambia) and cereal-root crop mixed (West Zambia) [71]. Maize mixed systems are found in plateau and highland areas with a dry subhumid to moist subhumid climate. Besides corn, principal livelihoods are tobacco, cotton, cattle, goats, poultry, and off-farm work. Cattle are kept for ploughing, milk, manure, but also for savings [71]. Cereal-root crop mixed systems are situated in regions of lower altitude and higher temperatures. The number of livestock per household is higher compared to the maize mixed system. The major sources of income are corn, sorghum, millet, cassava, yams, legumes, and cattle [71].

In Zambia and other South African countries, the major environmental problem related to agriculture is declining soil fertility [71]. Soil degradation is caused by inappropriate management practices such as continuous cropping and overgrazing [71]. Average agricultural input levels in Zambia are low (Figure 2-2).

Economic and institutional developments

Zambia is a former colony of Britain (the region was named Northern Rhodesia) and gained independence in 1964. After independence, the economy heavily depended on copper exports and many people lived in the urban mining areas. To ensure food supply to these areas, the new government aimed to increase national corn production. In the colonial period, however, commercial corn production had mainly relied on large-scale European farmers. The new objective was to enhance the participation of smallholders in the commercial corn market [102]. The agricultural intervention system of price controls and subsidies, which also dated from the colonial period, was maintained. But, new

pricing policies favored smallholders in remote areas over commercial farmers with good market access [102,103]. In the early 1970s, agricultural policies were expanded with fertilizer subsidies, which mainly benefitted the corn sector [102,103]. Both corn seed and fertilizer were made accessible to smallholder farmers in remote areas through a network of cooperative depots. In addition, farmers could sell their corn to these depots [102]. In the meantime, the Zambian government had invested in a corn breeding program which resulted in the release of twelve new varieties between 1977 and 1994. The program was started in the early 1960s to reduce the import of crop varieties from Zimbabwe, on which European farmers had relied during the colonial period [102].

Between 1973 and 1991, Zambia had been governed by single party rule. This period coincided with an economic crisis in the late 1970s and 1980s due to a collapse of the copper price in 1975 and poorly managed governmental interventions in the market [103]. Although attempts were made to reform (agricultural) policies in the 1980s, economic liberalization only started when a new government came to power in 1991 [103,104]. Through liberalization, the corn market was fully privatized. But, intervention in the input markets for fertilizer and credit remained [103]. Fertilizer price subsidies had been eliminated in 1988, which resulted in high input costs for corn. Therefore, smallholders reduced their use of fertilizer and hybrid corn varieties and returned to the cultivation of traditional corn varieties and subsistence crops like sorghum [71,102]. The government then decided to continue fertilizer distribution on loan, but this undermined the ability of the private market to distribute fertilizer commercially [104]. Also, underinvestment in infrastructure and other public goods had made the purchase of fertilizer unprofitable to many farmers [104]. In response to the reduced fertilizer use (Figure 2-2) and corn production in the 1990s, a new policy for fertilizer distribution and subsidy (the Fertilizer Support Program) was implemented in 2002 [105].

Efforts to control soil erosion started in the mid-1980s, driven by the spreading problem of land degradation and the economic reforms in late 1980s and early 1990s. At first, commercial farmers adopted conservation farming technologies to improve the profitability of mechanized corn production [106]. In 1995, appropriate technologies for smallholders were introduced as well. The development and promotion of the technologies was collectively conducted by farmer organizations, private companies, NGOs and the government [106].

Yield developments

After corn yields declined and fertilizer use increased slowly in the 1960s, the introduction of fertilizer subsidies [102,103] caused these levels to increase significantly in the 1970s (Figure 2-2, Table 2-1). The fact that agricultural policies were mainly focused on corn production [102,103] is clearly reflected in the high yield improvement rate of 4.5% yr^{-1} for corn compared to 1.2% yr^{-1} for soybeans and 0.1% yr^{-1} for sugarcane between 1971

and 1990 (relative to 1971, 1973 for soy). After the elimination of fertilizer subsidy in 1988 due to economic liberalization [102,103], fertilizer use and corn yields declined in the 1990s. It appears that commercial farmers benefitted from the economic reforms, as irrigation levels increased considerably and soybean yields improved at a very high rate of almost 20% yr^{-1} in the 1990s[3] (relative to 1991). Sugarcane yields increased at 0.7% yr^{-1} in the same decennium. In addition, Figure 2-13 shows that the output-input ratio of Zambia's agricultural sector improved in the late 1980s and 1990s.

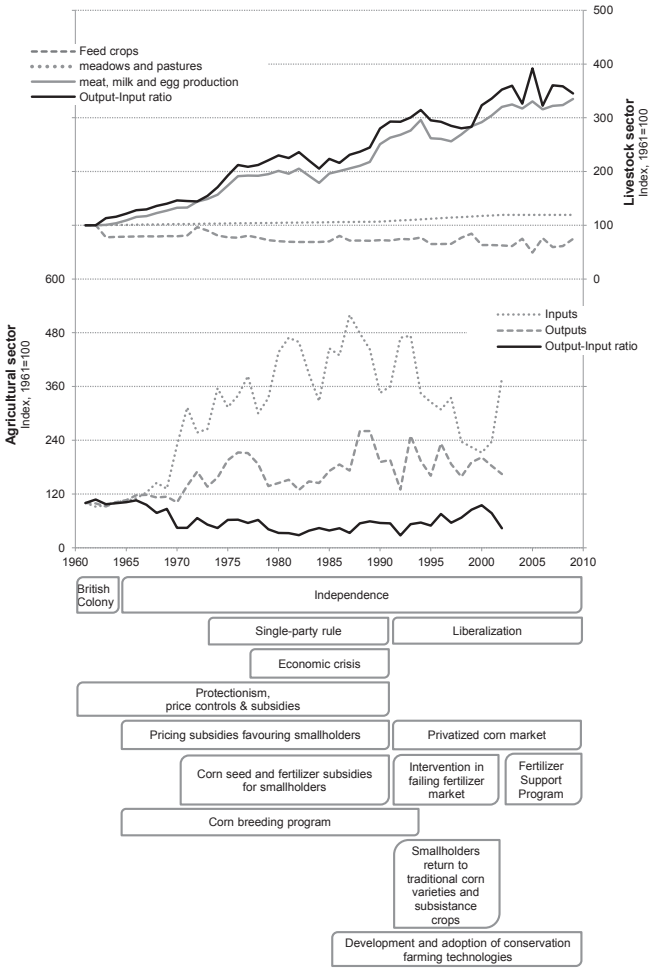

FIGURE 2-13 | Productivity developments in the Zambian agricultural and livestock sector and institutional, economic and technological/management developments. Because of limited data, agricultural tractors are not included in the inputs and in the output-input ratio for the agricultural sector.

3 The high growth rate for soybean yields is also related to the drought induced yield drop in 1992. Without this outlier in the dataset, however, the improvement rate is still 12.2 % yr^{-1}. Sugarcane yields were not considerably affected by the drought.

Besides reduced fertilizer use, these advances in productivity may also be the result of the introduction of conservation farming technologies [106]. After the adoption of the Fertilizer Support Program in 2002 [105], however, fertilizer use increased again, the area equipped for irrigation stabilized (Figure 2-2) and overall agricultural productivity dropped. Corn yields rose again and sugarcane yields stabilized. The effect on soybean yields is unclear; after a steep decline in 2001 due to drought, yields recovered and returned to levels comparable to the 1990s.

The FAO data show constant milk yields from 1961 until 2010. Beef yields have also been relatively stable, except for a decline of 1.8% yr^{-1} between 1968 and 1980 (relative to 1970). An explanation may be that the shift in focus of agricultural policies towards smallholders in this period has affected commercial farmers. This theory can, however, not be confirmed by statistics.

A2.8 ZIMBABWE

Agricultural characteristics

Until 2000, the agricultural sector of Zimbabwe consisted of two major farming systems, which both occupied half of the arable land [34]. The commercial farming system was dominated by a relatively small group of European farmers. These large scale farms were located in the higher rainfall areas in North-Eastern Zimbabwe [36,107]. Production was mainly focused on crops and input intensive [36]. The smallholder farming system involved a large number of African farmers. These small-scale farms were located in the drier and more remote areas with poor market access. Farming included crop and livestock production and was mainly subsistence driven [36].

From 2000, the Zimbabwean government acquired land from the European commercial farmers on a large scale. The land was divided into smallholder farms and commercial farms of varying scales, and redistributed to black farmers [37]. Because of the limited knowledge and skills of the new farmers and poor access to inputs and new technologies, the national level of irrigation and fertilizer use declined after 2000, see Figure 2-2 [38]. Also, a loss of knowledge about livestock management led to more disease related deaths [38]. As a result, the number of ruminants and the ruminant density on pastures declined (Figure 2-15).

At the start of the 21st century, the major farming system was maize mixed [71], see the description for Zambia. As more than half of the land is not suitable for crop production without irrigation, cattle production plays an important role in Zimbabwe. The relatively high human and livestock populations and densities on marginal suitable lands, however, has resulted in large-scale soil degradation in smallholder farming areas [35,36].

Economic and institutional developments

Together with Zambia, Zimbabwe is a former colony of Britain (the region was named Southern Rhodesia). But, while Zambia was directly administered by the British during its colonization period, Zimbabwe was a self-governing colony. In 1965, The Zimbabwean government declared independence unilaterally, which was only recognized internationally in 1980. During the period of unilateral independence, the United Nations (UN) imposed sanctions on exports [34]. To face these embargos, the government adopted a policy of import substitution [39]. Policies aimed at agricultural diversification and commercial production of export oriented tobacco was replaced by cultivation of previously imported crops like corn, wheat and soybeans [34,35]. The period between 1965 and 1980 was also accompanied by a civil war, which was partly concerned with the uneven distribution of land between commercial farmers and smallholders [36]. Intensification of this guerilla in the late 1970s led to the abandonment of commercial farms in more remote areas and occupation by peasants [35].

After independence in 1980, the government aimed to support the development of smallholders. A new land reform policy allowed the sale of commercial farmland on a 'willing buyer, willing seller' basis [34]. In addition, smallholders were enabled to obtain credit to purchase seed and fertilizer and to make use of subsidized marketing services. This led to a rapid adoption of hybrid corn varieties [34]. According to Eicher [34], this successful smallholder green revolution in the first half of the 1980s could be realized because of good political, institutional, technological and economic conditions. An important factor to success was the investment in research, education and farmer support in previous decennia, which had already led to a green revolution by white commercial farmers in the 1960s [33,34]. Government financed research on high yielding crops in the 1970s and 1980s led to the release of more than 30 new hybrid corn varieties by 1990 [33,62]. The success of the smallholder support system, however, resulted in high expenses for subsidies. In the late 1980s and early 1990s, the government lowered subsidies and encouraged farmers to diversify crop production [34]. These reductions in public spending were part of an economic structural adjustment program (ESAP), which also included other measures to liberalize the economy [39]. In addition, Zimbabwe's public R&D system slowly started to deteriorate; many European agricultural experts left Zimbabwe in the years after independence, while the shifted focus of agricultural research programs from commercial farmers to smallholders required experienced researchers [34].

A series of events in the 1990s led to hyper-inflation and a collapse of the economy in the 2000s [38]. The ESAP had seriously affected Zimbabwe's economy and the fast-track land reform program in 2000 disrupted commercial agricultural production. Also, beef exports to the EU were suspended because of foot-and-mouth disease [38,39]. Due to this hyper-inflation, farmers' incomes dropped dramatically and inputs became unaffordable to many farmers [33]. While Zimbabwe was once called the bread-basket of South-Africa, now international support programs are needed to improve food security among

impoverished rural households [33]. The program initiated by the British government in 2003 also aimed at promoting conservation farming practices [33,108]. The first agri-environmental policies, however, were already introduced in the early 20th century. The government provided significant financial support to apply conservation farming practices on commercial farms, which was very successful in the 1960s and 1970s [35]. Policies to address soil degradation in peasant farming had some success in the 1960s, but ceased in 1970s due to the political situation, increasing human and livestock populations, and a lack of (financial) support [35]. In the 1980s and 1990s, several research activities on conservation agriculture were initiated, but did not lead to significant uptake of conservation practices by smallholders [108].

A2.9 ADDITIONAL FIGURES AND TABLES

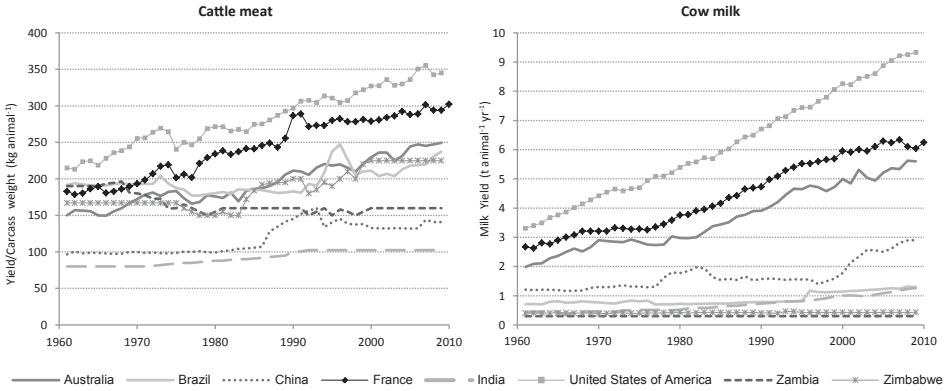

FIGURE 2-14 | Historical yield developments (1961-2010) for the production of cattle meat and cow milk (FAOSTAT [24]).

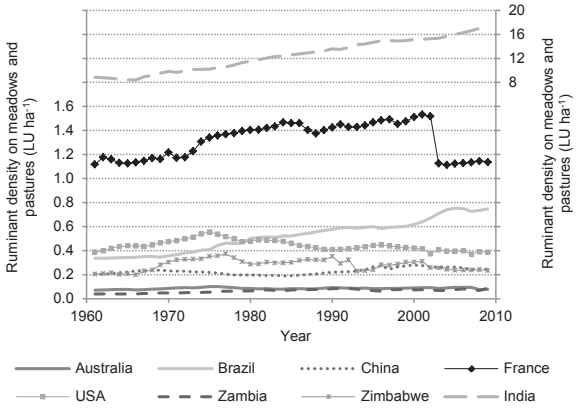

FIGURE 2-15 | Development in livestock production intensity: ruminant density on pastures and meadows in livestock units per hectare (LU ha^{-1}), derived from FAOSTAT data [24]. Ruminants included: buffaloes, camel, cattle, goats, and sheep. Note the different scale for India compared to the other countries.

TABLE 2-4 | Comparison of average annual crop yield growth rates (% yr^{-1}) derived from a) extrapolation of linear regression (2010-2050); b) IMAGE projections (2010-2050); c) MIRAGE projections (2008-2020); d) projections from Jaggard et al. (2007-2050).

			Australia	Brazil	China	India	Zambia	Zimbabwe	USA	World
Wheat	Linear extrapolation[a]		0.7%		1.8%	1.5%			0.8%	1.3%
	IMAGE projection[a,d]		1.2%[f]		2.0%	1.4%			0.5%	1.2%
	MIRAGE projection[b]	bau			1.4%				0.8%	1.0%
	Yield projections in Jaggard et al.[c]	bau	1.1%		2.4%	2.0%			1.2%	
		min	0.8%		1.7%	1.4%			0.9%	
		max	1.9%		4.6%	3.9%			2.2%	
Corn	Linear extrapolation[a]			1.6%	1.6%		1.2%	-1.6%	1.2%	1.3%
	IMAGE projection[a]			2.4%	1.8%		1.6%[g]		1.2%	0.9%
	MIRAGE projection[b]	bau		4.9%	1.8%		1.6%[h]		1.2%	1.5%
	Yield projections in Jaggard et al.[c]	bau		1.7%	1.9%				1.3%	
		min		1.2%	1.3%				0.8%	
		max		3.5%	3.8%				2.7%	
Rice	Linear extrapolation[a]		0.7%		1.3%	1.3%			1.0%	1.2%
	IMAGE projection[a]		0.5%[f]		0.4%	1.8%			1.1%	1.1%
	MIRAGE projection[b]	bau			1.6%				0.9%	2.2%
	Yield projections in Jaggard et al.[c]	bau	0.9%		1.5%	1.4%			1.2%	
		min	0.6%		1.0%	0.9%			0.9%	
		max	1.7%		3.1%	3.0%			2.4%	
Soybean	Linear extrapolation[a]			1.4%		1.2%	1.2%	0.6%	1.0%	1.1%
	IMAGE projection[a,e]			1.0%		1.5%	1.7%[g]		1.4%	1.0%
	MIRAGE projection[b]	bau		3.1%			2.0%[h]		1.1%	2.0%
	Yield projections in Jaggard et al.[c]	bau		1.6%		1.4%			1.1%	
		min		1.1%		1.0%			0.8%	
		max		3.2%		2.9%			2.2%	
Sugarcane	Linear extrapolation[a]		0.3%	1.0%		0.8%	-0.1%	-0.3%		0.6%
	IMAGE projection[a]									
	MIRAGE projection[b]	bau								
	Yield projections in Jaggard et al.[c]	bau	0.4%	1.2%		1.1%				
		min	0.3%	0.9%		0.8%				
		max	0.7%	2.4%		2.1%				

[a] Relative to 2010 yields; [b] Relative to 2008 yields; [c] Relative to 2007 yields; [d] In IMAGE, wheat is aggregated into the product group temperate cereals; [e] In IMAGE, soybean is aggregated into the product group oil crops; [f] In IMAGE, Australia is aggregated into the region Oceania; [g] In IMAGE, Zambia and Zimbabwe are aggregated into the region Southern Africa; [h] In MIRAGE, Zambia and Zimbabwe are aggregated into the region Sub-Saharan Africa.

The IMAGE model adopts yield projections from the FAO and combines these with endogenous assumptions on yield changes [44,54]. The MIRAGE model uses a baseline scenario from Aglink-Cosimo, which is also complemented by endogenous assumptions on yield developments [10]. Jaggard et al. [14] assume a continuation of current yield trends, but also take into account relative changes owing to increasing carbon dioxide (CO_2) and ozone (O_3) concentrations, climate change and technological developments.

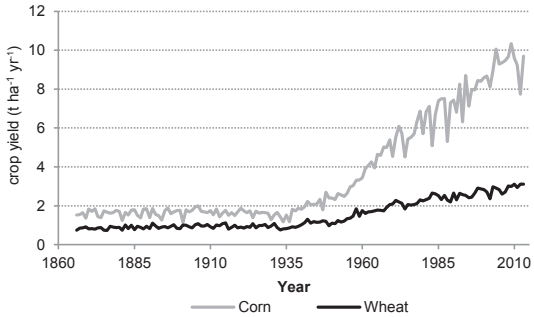

FIGURE 2-16 | Long term historical yield trends for corn and wheat in the USA [109]**.**

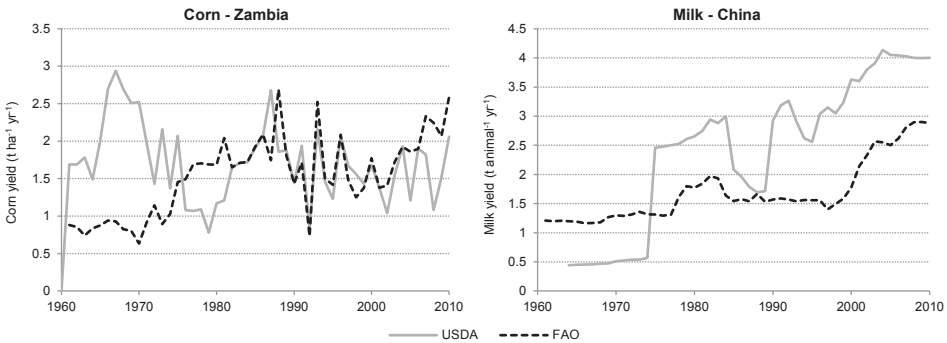

FIGURE 2-17 | Examples of diverging statistical yield data between FAO [24] and USDA [110].

TABLE 2-5 | Recent yields [24], maximum attainable yields and yield gaps [32] (t ha^{-1} yr^{-1})

			AU	BR	CN	FR	IN	ZM	ZW	US
Wheat	Average yield 2008-2010	t ha^{-1}	1.6		4.7	7.1	2.8			3.0
	Max attainable yield[a,c]	t ha^{-1}	3.8		6.4	8.2	3.9			6.1
	Yield gap	t ha^{-1}	2.2		1.7	1.1	1.1			3.1
	Current yield as % of max	%	42		74	87	72			50
Corn	Average yield 2008-2010	t ha^{-1}		4.1	5.4	9.1		2.3	0.5	9.9
	Max attainable yield[a,d]	t ha^{-1}		6.6	8.4	8.8		10.9	9.8	12.6
	Yield gap	t ha^{-1}		2.5	2.9	-0.3		8.6	9.2	2.7
	Current yield as % of max	%		62	65	103		21	6	78
Rice	Average yield 2008-2010	t ha^{-1}	8.9		6.6		3.3			
	Max attainable yield[b,e,f]	t ha^{-1}	10.7		9.5		9.2			
	Yield gap	t ha^{-1}	1.9		3.0		5.9			
	Current yield as % of max	%	82		69		36			
Soybean	Average yield 2008-2010	t ha^{-1}		2.8			1.0	1.7	1.5	2.9
	Max attainable yield[a,g]	t ha^{-1}		3.4			3.1	4.4	4.1	3.7
	Yield gap	t ha^{-1}		0.6			2.0	2.8	2.6	0.8
	Current yield as % of max	%		82			34	38	36	78
Sugarcane	Average yield 2008-2010	t ha^{-1}	80.3	79.1			66.5	105.4	79.5	
	Max attainable yield[b,e]	t ha^{-1}	135.3	106.8			123.1	144.1	142.1	
	Yield gap	t ha^{-1}	55.1	27.6			56.6	38.8	62.6	
	Current yield as % of max	%	59	74			54	73	56	

AU, Australia; BR, Brazil; CN, China; FR, France; IN, India; ZM, Zambia, ZW, Zimbabwe; US, United States

[a] Maximum attainable yield in 2020 as calculated for the IPCC SRES B1 Scenario from the Australian Commonwealth Scientific and Research Organization (CSIRO) Mark 2 Model [32]; [b] Maximum attainable yield based on the average climatic conditions for the period 1961-1990, applied in case no projection for 2020 was available [32]; [c] Australia, France and USA: high input level, rain fed conditions; China and India: high input level, irrigated conditions; [d] all countries except France and USA: high input level, rain fed conditions; France and USA: high input level, irrigated conditions; [e] High input level, irrigated conditions; [f] Maximum agri-ecological attainable yield for Indica wetland rice (150 days); [g] High input level, rain fed conditions

High input level: "Under a high level of input (advanced management assumption), the farming system is mainly market oriented. Commercial production is a management objective. Production is based on improved or high yielding varieties, is fully mechanized with low labor intensity and uses optimum applications of nutrients and chemical pest, disease and weed control." [32]

CHAPTER 3

Intensification pathways for beef and dairy cattle production systems: impacts on GHG emissions, land occupation and land use change

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ABSTRACT

Cattle production is characterized by high greenhouse gas (GHG) emissions and land requirements, and can have a significant impact on land use change (LUC) and associated emissions. Intensification of production systems is considered an important strategy for mitigating anthropogenic GHG emissions. When categorizing production practices into three systems, i.e. pasture-based, mixed and industrial systems, intensification can either take place within one system or through the transition to another more productive system. This study investigates the impacts of these two pathways on farm gate emissions and LUC-related emissions (kg CO_2-eq per kg of milk or beef) in nine world regions. First, a review is conducted of bottom-up studies on farm gate emissions from dairy production in Europe and beef production in North America and Brazil. Then, a global data set on GHG emissions from cattle production is used to extrapolate the findings from this review to other regions. Finally, the GLOBIOM model is applied to perform a global assessment of land occupation and LUC-related emissions. For dairy in Europe, farm gate emission reductions of 1%-14% are found for intensification within one system and 2%-26% for system transitions. In Europe as well as other developed regions, the comparative influence of both pathways on the GHG balance largely depends on the specific design of the initial and final production systems. In developing countries, there is especially a great potential for emission reductions through intensification within the pasture-based system. The additional reduction potential of moving from pasture-based to mixed and industrial production is limited. Also, emission reductions of intensification within the mixed system are smaller compared to the pasture-based system. For beef production in Brazil, intensification within pasture-based systems can attain significant farm gate emission reductions (>50%). The same is true for pasture-based systems in other developing regions and also some developed regions. Furthermore, the additional GHG reduction potentials of moving from pasture-based to mixed systems, and of intensification within mixed systems are considered larger than for dairy. Although both the dairy and beef sector can often attain significant farm gate emission reductions through intensification within pasture-based systems, the transition to mixed systems is important to reduce land occupation and LUC-related emissions. LUC mitigation is considered to be the most important GHG mitigation strategy for cattle production in Sub-Saharan Africa and Latin America. Important strategies to reduce both farm gate and LUC-related emissions include increasing the productivity of grassland and cropland and increasing the animal productivity through improved feed quality.

3.1 INTRODUCTION

The livestock sector is an important user of natural resources and has significant influence on local landscapes and ecosystems [1-3]. The livestock sector is responsible for approximately 15% of the global greenhouse gas (GHG) emissions and is therefore one of the main contributors to climate change [4,5]. In addition, the land required for livestock production, both direct for grazing and indirect for feed crop cultivation, accounts for 70% of the global agricultural land area and covers up to 30% of the ice-free terrestrial surface of the planet [6]. The impact on GHG emissions and land occupation is especially large for cattle production, which accounts for an estimated 65% [5] or even 77% [7] of the total livestock-related GHG emissions. Also, GHG emissions caused by land use change (LUC) can have a significant impact on the GHG balance of cattle production [8].

There is a significant potential to reduce the GHG impacts from dairy and beef production. Gerber et al. [5] estimate that the livestock sector emissions can be reduced by approximately 30%. About 65% of these reductions can be attained in the cattle sector. To reduce emissions, numerous GHG mitigation options are suggested [9-11]. Such mitigation strategies are often related to intensification of cattle production. Intensification can be realized by, for example, fertilizing pastures to enhance the pasture productivity, reducing the grazing period and adding more concentrated (less fibrous) feed to the diet [9-11]. As a result of improved feed quality (feed digestibility), methane emissions from enteric fermentation decline [7]. The higher feed quality also increases the animal productivity (quantity of milk or beef produced per animal), which leads to a further decline of the non-CO_2 emission intensity. However, the housing of animals, production of feed crops and use of fertilizers may increase the emissions from manure management, feed production and energy use, and partially counteract the direct cattle emission reductions from intensification.

To study global livestock production, production practices are generally categorized into three well-contrasted systems, i.e. pasture-based, mixed and industrial systems [12,13]. When using this system classification in the context of intensification in the cattle sector, a distinction can be made between intensification within one system and system transitions to more efficient and productive systems (i.e. from pasture-based to mixed and from mixed to industrial). Due to the clear distinction between the systems, these two pathways imply different natures of change. While intensification within one system is characterized by incremental change, a system transition involves transformational change. Therefore, it is expected that these two development pathways will have different impacts on the GHG balance and on land occupation. Although a large number of studies has investigated the GHG performance of dairy and beef production systems, and to a lesser extent also

the potential of GHG mitigation options [8,14], it is unclear yet how the impacts of the two development pathways compare to each other and how the results of such a comparison differ between regions. Better insight in these aspects is considered very valuable for designing strategies and policies for future sustainable development of the cattle sector. Therefore, the aim of this study is to compare the impact of intensification within one system and of system transitions on GHG emissions without LUC, land occupation and LUC-related emissions. The assessment considers both dairy and beef production in nine world regions, based on results from studies in the literature and on data and simulations from the Global Biosphere Management Model (GLOBIOM).

The remainder of this chapter is structured as follows: Section 2 describes our approach, the production systems considered, and the impact categories selected to assess the effects of intensification. Section 3 discusses the respective impacts of each development pathway on GHG emissions without LUC, land occupation and LUC-related emissions, and compares the impacts of the two pathways in each region. Section 4 offers a discussion, and conclusions are drawn in section 5.

3.2 MATERIALS AND METHODS

3.2.1 Literature review and data

For the assessment, literature was collected that conducts studies on the GHG impacts and land requirements of dairy or beef production systems in specific regions, based on bottom-up data. Each studies was selected based on the use of similar system boundaries and emission sources, the availability of the total milk or beef production, and the ability to convert the results to the functional unit used in the present study (see section 3.2.3). In total, 69 studies on dairy production (from 30 publications) and 47 studies on beef production (from 17 publications) were found. The majority of the studies was published in 2009 or later and assessed production systems that represent typical systems in the considered region. Therefore, the studies are considered to provide a good representation of current production practices in the regions covered. A detailed overview of all studies, including their main specifications and results, is provided in Appendices A3.1 to A3.4.

Table 3-1 gives an overview of the number of studies per region and shows that the different world regions are not equally covered. Therefore, the dataset from Herrero et al. [7] is used to extrapolate the results from the studies to other regions. This dataset provides a consistent picture of, for example, feed use, feed conversion efficiency and non-CO_2 GHG emissions for cattle production in 30 regions (see Table 3-12 in Appendix A3.6) in the year 2000. However, the trade-off compared to the studies from the literature is that the data is reconstructed from different global datasets instead of from bottom-up data [7].

TABLE 3-1 | Number of studies on GHG emissions from dairy and beef cattle by region and production system. An overview of the studies included in this review and their main characteristics is provided in Appendices A3.1 and A3.2.

Production system	Europe		Asia[a]		Africa[b]		North America		Latin America and the Caribbean[c]		Oceania		Total	
	Dairy	Beef	Dairy	Beef	Dairy	Beef	Dairy	Beef	Dairy	Beef	Dairy	Beef	Dairy	Beef
Pasture-based	17	2						1	1	10	2	4	20	17
Mixed	26	17					4	7	1	0	3	4	34	28
Industrial	6			1			9	1					15	2
Total	49	19	0	1	0	0	13	9	2	10	5	8	69	47

[a] When considering nine world regions, Asia is divided into three world regions [7].
[b] When considering nine world regions, Africa is divided into two world regions [7].
[c] In the rest of this chapter, this region will be referred to as Latin America

The studies in the literature do not always include an analysis of land occupation and LUC-related emissions. Therefore, the impacts of intensification are first discussed based on GHG emissions without LUC. Thereafter, the dataset from Herrero et al. [7] is used in the Global Biosphere Management Model (GLOBIOM) to perform a systematic assessment of land occupation and LUC-related emissions in nine world regions [8]. GLOBIOM is a global partial equilibrium model integrating the agricultural and forestry sectors in a bottom-up setting based on detailed grid cell information. The model is used to analyse the competition for land between agriculture, forestry, and bioenergy, and has been applied in recent key studies about the effect of livestock productivity developments on climate change mitigation [8,15].

3.2.2 Cattle production systems

All studies from the literature are classified into three production systems: pasture-based, mixed and industrial systems. This is based on the systems classification from Seré and Steinfeld [13] and Robinson et al. [12], see Table 3-11 in Appendix A3.6. The three production systems are defined as following:

1. Pasture-based: production system in which cattle are grazing year-round or for a large part of the year. The diet includes pasture forage, indoor grass feeding, grass silage and hay [16]. For beef, pastures may also include grains or legumes like wheat or clover [17,18]. A small share of the diet can consist of imported, low quality concentrates [16]. Milk production, beef production and stocking rate per hectare are relatively low [19-21].
2. Mixed: production system in which grazing is still important, but complemented with more concentrated, higher quality feed like soybean meal [12,22]. In beef production, calves are first sent to pasture before they are finished on feedlots [23]. Feed crops are imported and/or partially produced on the farm [12,22]. Milk

production, beef production and stocking rate per hectare are higher than from pasture-based systems.

3. Industrial: production system in which cattle are confined to a stall barn or feedlot. The diet mainly consists of a concentrated, high-controlled mix of grains, residues and soy complemented with a combination of feed supplements, vitamins and medicine [5,24]. Milk production, beef production and stocking rates per hectare are highest for this production system.

Herrero et al. [7] and GLOBIOM also use the livestock classification system from Robinson et al. [12] and include pasture-based and mixed systems but exclude industrial systems, see Table 3-11 in Appendix A3.6. They also include *urban* and *other* production systems, which cannot be categorized as pasture, mixed or industrial. These systems are excluded from the assessment.

3.2.3 Functional unit and data standardization

In the literature, GHG emissions and land occupation are expressed in various functional units. The present study uses the *fat and protein corrected milk equivalent (FPCM)* for dairy cattle and *carcass weight equivalent (CW)* for beef cattle as functional units. Both are widely used in literature and in key modelling studies [5,25]. The FPCM is a standard used to compare milk with different fat and protein contents to evaluate milk production of different dairy cattle on a common basis. FPCM is calculated from the amount of raw milk and the fat and protein content, equation 1 [25]. If the fat or protein percentage is unknown, milk is converted to FPCM with 4.0% fat and 3.3% protein [5].

Equation 1:

$$FPCM\ (kg) = raw\ milk\ (kg) * (0.337 + 0.116 * fat\ content\ (\%) + 0.06 * protein\ content(\%))$$

The CW can be calculated from the live weight (LW) or bone-free meat (BFM) by using a dressing percentage, equations 2 and 3. The dressing percentage varies per country because the breed, gender, diet, season of slaughter and other factors affect the dressing rate [26]. However, data on dressing percentages is scarce. Therefore, the dressing percentage for all countries and regions has been assumed to be 60% for live weight (LW) and 150% for bone-free meat (BFM) [27].

Equation 2:

$$CW\ (kg) = LW\ (kg) * dressing\ percentage\ (\%)$$

Equation 3:

$$CW\ (kg) = BFM\ (kg) * dressing\ percentage\ (\%)$$

3.2.4 GHG emissions without LUC

The present study investigates the cradle to farm gate GHG emissions per kg of milk or beef produced. This includes emissions from all upstream processes in cattle production up to the point where the animals or products leave the farm. The main emission sources are enteric fermentation, manure management and feed production [7], see Table 3-2. The studies from the literature also take into account indirect emissions from the production of farm inputs, emissions from energy consumption, and sometimes emissions from land use (Table 3-2). GHG emissions from the production and use of pesticides, medicines or detergents are excluded in the present study, as their share in the total GHG balance is very small and most studies do not cover these sources [5]. A majority of the studies calculates the GHG emissions based on life-cycle assessment (LCA). 13 publications use another approach or model to assess the GHG emissions [24,28], see Appendices A3.1 and A3.2. In Herrero et al. [7], cattle non-CO_2 emissions are calculated using a digestion and metabolism model for ruminants.

TABLE 3-2 | Summary of the main GHG emission sources and the related processes included in the studies from literature and in GLOBIOM.

Main source of GHG emissions	Characteristics	Produced greenhouse gas	Studies from literature	GLOBIOM
Enteric fermentation	Enteric fermentation	CH_4	X	X
Manure management	Manure storage, processing and deposit	Direct and indirect N_2O and CH_4	X	X
Feed production	Manure applied to feed crops and pasture or directly deposited on pastures by animals	Direct N_2O	X	X
	Synthetic fertilizer applied to feed crops and pastures from decomposition of crop residues	Direct and indirect N_2O	X	X
	Volatilisation and leaching	Indirect N_2O	X	X
	Production, processing and transport of feed and fertilizer	CO_2	X	-
Energy consumption	On-farm energy use	CO_2	X	-
Land use	SOC change during cultivation	CO_2	X[b]	X[c]
LUC	Pasture expansion or decline	CO_2	X[b]	-
	Cropland expansion or decline	CO_2	X[b]	-
	Conversion from forest land to pasture or cropland (deforestation)	CO_2	-	X
	Conversion from natural land[a] to pasture or cropland	CO_2	-	X

[a] Natural land is all land other than forest, cropland, land being grazed by livestock or other agricultural land, wetlands, bareland and urban areas [29]
[b] Only included in some studies from the literature
[c] Only accounted for in Europe [30]

The results from the literature for different regions and from different references are difficult to compare because of the influence of local conditions like climate and the differences in assumptions made with regard to, for example, the functional unit, allocation methods and characterization of the production processes [17,18]. Therefore, the GHG emission impacts of intensification within one system and of system transitions are first discussed based on studies from the same reference and for the same region or country. This comparison concentrates on one or two larger regions for which all production systems are covered by the literature. For dairy, Europe is considered and for beef the analysis includes the USA, Canada and Brazil. Based on the findings from this assessment, the data from Herrero et al. [7] is then used to extrapolate the results to other regions.

3.2.5 Land occupation

Land occupation is defined as the area of land needed to produce one kg of milk or beef. This includes grassland and cropland, as well as on-farm land use and land required to produce imported feed. When land occupation is not directly available from a study, but sufficient data is available about total land use and total beef or milk production, land occupation is derived by applying Equation 4.

Equation 4:

$$LO(m^2/kg\ FPCM\ or\ CW) = \frac{land\ requirement\ (ha)*10000\ (m^2/ha)}{total\ milk\ or\ beef\ production\ (kg\ FCPM\ or\ CW)}$$

Comparable to the GHG emissions, the impact of the two development pathways on land occupation is first discussed for Europe and North America based on results from the studies in literature. Then, the results from GLOBIOM are used to discuss how the development pathways may influence land occupation in other regions.

3.2.6 LUC-related emissions

LUC-related emissions are defined as GHG emissions caused by a change from one land use to another, e.g. the change from forests to cropland in order to allow the expansion of crop production. As the number of studies that assess LUC-related emissions is small [16,31-34], only results for dairy production in Europe are discussed. The results from GLOBIOM are used as the main input for the discussion on the share and size of LUC-related emissions in the GHG balance of cattle production in each region [8]. In GLOBIOM, land use change results from choosing land use and processing activities with the aim to maximize social welfare (while subject to resource, technological, and policy constraints) [35]. The relevant land use change processes in GLOBIOM are the conversion of forest

or natural land to pasture or cropland (Table 3-2). CO_2 emissions or sequestration are calculated as the difference in carbon content in above- and below-ground living biomass between the initial and new land use [29].

3.3 RESULTS

3.3.1 Impact of intensification on GHG emissions without LUC

3.3.1.1 Intensification within one system

When considering the GHG balance without LUC-related emissions, the review of studies on dairy production in Europe shows that intensification within both the pasture-based and mixed production system often has a positive effect on the GHG emissions. For example, in pasture-based systems, Hörtenhuber et al. [16] find that emissions are about 0.07 kg CO_2-eq/kg FPCM or 6%-7% lower in upland pasture systems compared to alpine pasture systems that have lower grass productivity and cattle stocking density. Also, O'Brien et al. [36] find that a higher stocking rate reduces emissions by 0.01 to 0.03 kg CO_2-eq/kg FPCM (1-3%). Regarding mixed systems, Hörtenhuber et al. [16] and Bell et al. [37] find that the emissions are 0.10-0.13 kg CO_2-eq/kg FPCM or 8-14% lower in systems that attain higher milk yields. However, not all studies find the same trend and this depends on the precise strategy used for intensification. For example, according to O'Brien et al. [36], emissions in both pasture-based and mixed systems are 0.05 to 0.14 kg CO_2-eq/kg FPCM (5-16%) higher for cows solely selected for high milk production compared to cows selected for both increased productivity and fertility. The difference is largest for pasture-based systems [36]. Also, Haas et al. [38] find that the emissions of an intensive mixed system are 0.30 kg CO_2-eq/kg FPCM (30%) higher compared to an extensive mixed system. In the intensive mixed system, benefits from improved animal productivity are reduced by increased fossil energy and fertilizer consumption [38]. When comparing conventional and organic production, Hörtenhuber et al. [16] find for both pasture-based and mixed systems that emissions from organic dairy production are up to 0.08 kg CO_2-eq/kg FPCM lower compared to conventional production because of, for example, not using synthetic fertilizers in organic production (see Table 3-3). The difference between conventional and organic production is largest for pasture-based systems [16]. But also here, results differ across studies. Kristensen et al. [39] and Olesen et al. [40] find for mixed dairy systems that the emissions from organic production are respectively 0.07 and 0.14 kg CO_2-eq/kg FPCM higher compared to conventional production, because of e.g. higher feed quality and animal productivity in conventional production.

For industrial dairy production, fewer studies are available. Bell et al. [37] find that an improved system with cows selected for increased milk fat and protein production attains a higher milk yield per cow and slightly reduces emissions per kg of milk compared to a

system with average milk fat and protein production. The difference in emissions between the two industrial systems is 0.09 kg CO_2-eq/kg FPCM (8%), which is a bit smaller than for mixed systems [37].

TABLE 3-3 | Key factors identified as reasons for differences in emissions between dairy production system.

Production system(s) compared	Production system types compared	Main reasons for differences in emissions	Main emission sources influenced
Pasture-based	Conventional	Grass productivity [16], cattle stocking density [16,36], cow genotype [36]	Feed production
	Organic vs. conventional	Synthetic fertilizer use, SOC sequestration in organic and SOC losses in conventional [16]	Feed production
Mixed	Conventional	Energy and fertilizer use [38], animal productivity[a] [16,37], cow genotype, genetic selection for increased milk production and fertility [36], genetic selection of cows for increased milk fat and protein production [37]	Enteric fermentation, feed production
	Similar systems (organic or conventional)	Feed conversion efficiency, land use intensity, share of imported feed [39]	Enteric fermentation, feed production
	Organic vs. conventional	Feed quality [16,40], synthetic fertilizer use [16], cattle stocking density, grazing period, animal productivity[a] [39]	Enteric fermentation, feed production
Industrial	Conventional	Genetic selection of cows for increased milk fat and protein production, animal productivity[a] [37]	All
Pasture-based vs. mixed	Conventional	Climatic conditions [14], pasture productivity [28], feed quality [36], animal productivity[a] [14,16,28,36]	Enteric fermentation, feed production
	Organic pasture-based vs. conventional mixed	Cattle stocking density [41], feed quality [42], animal productivity[a], fossil energy and fertilizer consumption [38]	Enteric fermentation, manure management
Pasture-based or mixed vs. industrial	Conventional	Manure management, energy consumption [24], feed quality [42], animal productivity[a] [37]	All

[a] Animal productivity here refers to milk production per animal.

Intensification in pasture-based beef production systems in Brazil has a significant potential for reducing GHG emissions. For example, Dick et al. [18] compare an extensive grazing system to an improved system characterized by improved pasture, the introduction of other forage species and rotational grazing. They find that the emissions of the improved system are less than half of the extensive system (15.4 vs. 36.0 kg CO_2-eq/kg CW). Ruvario et al. [17] compare various pasture-based systems with different grasses and crops

(natural and improved natural grass, ryegrass, sorghum) and management levels. The variation in emissions between the systems is large (ranging from 32.0 to 68.2 kg CO_2-eq/kg CW). Overall, the lowest emissions are attained in systems with the shortest animal lifetime and highest cattle density. In most cases, these are also the systems with the highest feed quality (improved pastures). One exception is the system based on natural grass and a protein-energy mineralized salt supplement. This system has a lower feed quality and cattle density compared to the system based on natural grass and ryegrass, but the salt supplement improves the feed conversion rate, shortens the lifetime of the animals and thereby reduces the total emissions (37.4 kg CO_2-eq/kg CW) compared to the natural grass and ryegrass-based system (47.4 kg CO_2-eq/kg CW) [17].
For intensification of mixed beef systems in Canada, GHG reduction potentials are lower. Basarab et al. [43] investigate four systems which differ in animal lifetime (age at which animals are started on their finishing diet) and use of hormone implants. For animals implanted with hormones, the total emissions are 1.2-1.3 kg CO_2-eq/kg CW or 6% lower compared to systems not implanted with hormones. Basarab et al. [43] also find that emissions are 1.3-1.4 kg CO_2-eq/kg CW (7%) higher for systems with longer animal lifetimes. For beef, no studies were found that compared different industrial production systems.

3.3.1.2 System transitions

When comparing pasture-based and mixed dairy systems in Europe, Hörtenhuber et al. [16], Thomassen et al. [41], Haas et al. [38], Williams et al. [42], O'Brien et al. [36] and Schader et al. [14] find that emissions for mixed systems are generally lower compared to pasture-based systems (reductions of 0.02-0.30 kg CO_2-eq/kg FPCM, or approximately 2%-26%). Key reasons for emission reductions are increased feed quality, animal stocking density and animal productivity, see Table 3-3. In Thomassen et al. [41], Haas et al. [38] and Williams et al. [42], the pasture-based system is an organic system, while the mixed system is based on conventional production. When the emissions from a conventional pasture-based system are higher compared to an organic pasture-based system, as found by Hörtenhuber et al. [16], the emissions of mixed dairy production will also be lower compared to conventional pasture-based production. In Lovett et al. [28], the emissions from the mixed system are 0.09 kg CO_2-eq/kg FPCM (10%) higher compared to pasture-based production. In this study, the pasture-based system attains a higher pasture productivity compared to the mixed system. The additional concentrates in the mixed system do not compensate for this. As a result, animal productivity is also higher in the pasture-based system [28].

A transition from pasture-based or mixed to industrial dairy production in Europe may decrease as well as increase the GHG balance depending on, for example, the increase

in feed quality and in emissions related to manure management [24,37,42], see Table 3-3. Williams et al. [42] find that emissions in an industrial system are reduced by 0.25 kg CO_2-eq/kg FPCM (20%) compared to pasture-based production and by 0.08 kg CO_2-eq/kg FPCM (8%) compared to mixed production. Bell et al. [37] also find a reduction in emissions compared to mixed dairy production (difference of 0.16-0.18 kg CO_2-eq/kg FPCM or 14%). In contrast, O'Brien et al. [24] find an increase of 0.05 kg CO_2-eq/kg FPCM (6%) when moving from a pasture-based to an industrial system.

For beef production systems in the USA, Pelletier et al. [23] compared pasture-based, mixed and industrial systems. A transition from pasture-based to mixed (backgrounding and feedlot finishing) and from mixed to industrial (finishing on feedlots only) production both result in decreased emissions (for key factors behind these differences see Table 3-4). Similar to the results for dairy production, the benefit is larger for the step from a pasture-based to a mixed system (from 30.7 to 25.9 kg CO_2-eq/kg CW or 16% reduction) than from a mixed to an industrial system (from 25.9 to 23.7 kg CO_2-eq/kg CW, 9% reduction) (Pelletier et al. 2010).

TABLE 3-4 | Key factors identified as reasons for differences in emission between beef production systems.

Production system(s) compared	Production system types compared	Main reasons for differences in emissions	Main emission sources influenced
Pasture	Conventional	Pasture productivity, feed quality, weight gain, animal productivitya [18], lifetime, cattle density [17]	Enteric fermentation, manure management, feed production
Mixed	Conventional	Hormone implants, carcass weight, lifetime [43]	Enteric fermentation, manure management, feed production, energy use
Pasture vs. mixed and mixed vs. industrial	Conventional	Feed quality, animal growth rate, lifetime [23]	Enteric fermentation, manure management, feed production

[a]Animal productivity: beef production per animal

3.3.1.3 Comparing and extrapolating results for intensification within one system and system transitions

Most studies on dairy production in Europe show that intensification results in decreased GHG emissions per kg of milk produced. Reduction potentials found for intensification within one system are 1%-7% for pasture-based systems, 8-14% for mixed systems and about 8% for industrial systems. GHG mitigation potentials of system transitions are 2%-26% from pasture-based to mixed production and 8%-14% from mixed to industrial systems. However, these figures are based on a limited number of studies and should only be considered as an indication of attainable reduction potentials. In addition, for

both development pathways a few exceptions are found for which intensification did not result in GHG emission reductions. First, organic or extensive dairy production sometimes reduces emissions compared to more intensive conventional production because of not using any synthetic fertilizers, lower fossil fuel consumption and the occurrence of soil organic carbon sequestration [16,38]. Second, moving to an industrial production system may either decrease emissions because of increased feed quality and animal productivity or increase emissions because of higher emissions from manure management and energy consumption [24,37,42].

A comparison of the results from the studies in literature and the data from Herrero et al. [7] (Figure 3-1A) shows that the emission values of all production systems in Europe are relatively close to one another compared to other regions. Therefore, whether the influence on the GHG balance is larger for intensification within one system or for a system transition will depend on the specific design of the initial and final production systems. When considering more regions (Figure 3-1A), the same seems to apply for other developed regions, i.e. North America and Oceania. In developing regions, however, the differences within and between pasture-based and mixed systems are more significant than in developed regions. Also, average non-CO_2 emissions in these regions, especially from pasture-based production, are higher compared to developed regions. Although practices in developed countries cannot be adopted one to one in developing countries, these differences suggest that there is a great potential to mitigate GHG emissions in developing regions through intensification within pasture-based systems. This potential may often be as high as the emission reduction that can be attained by a transition from pasture-based to mixed production. Probably, the most important limitation for emission reductions in the pasture-based system is the climate. As shown in GLOBIOM results, emissions are generally significantly higher in arid areas than in humid and temperate regions. Nevertheless, the GLOBIOM results show that pasture-based dairy production in arid regions in South Africa attains higher milk yields and lower emissions compared to other Sub-Saharan arid regions. Management practices in South Africa may therefore provide good options for improvements in the other regions. Thus, for all developing regions it is interesting to compare current management practices with best practices available in the own region and in other regions and to investigate options to improve the production system. With regard to mixed systems, the differences in emissions between regions are smaller than for pasture-based systems. However, in some regions and especially Sub-Saharan Africa, there may still be significant potentials to reduce emissions through intensification within the mixed system. In addition, the studies from literature indicate that a smaller additional emission reduction may be attained by a transition to the industrial system. In this case it is important to adopt measures that minimize GHG emissions from manure management and energy consumption.

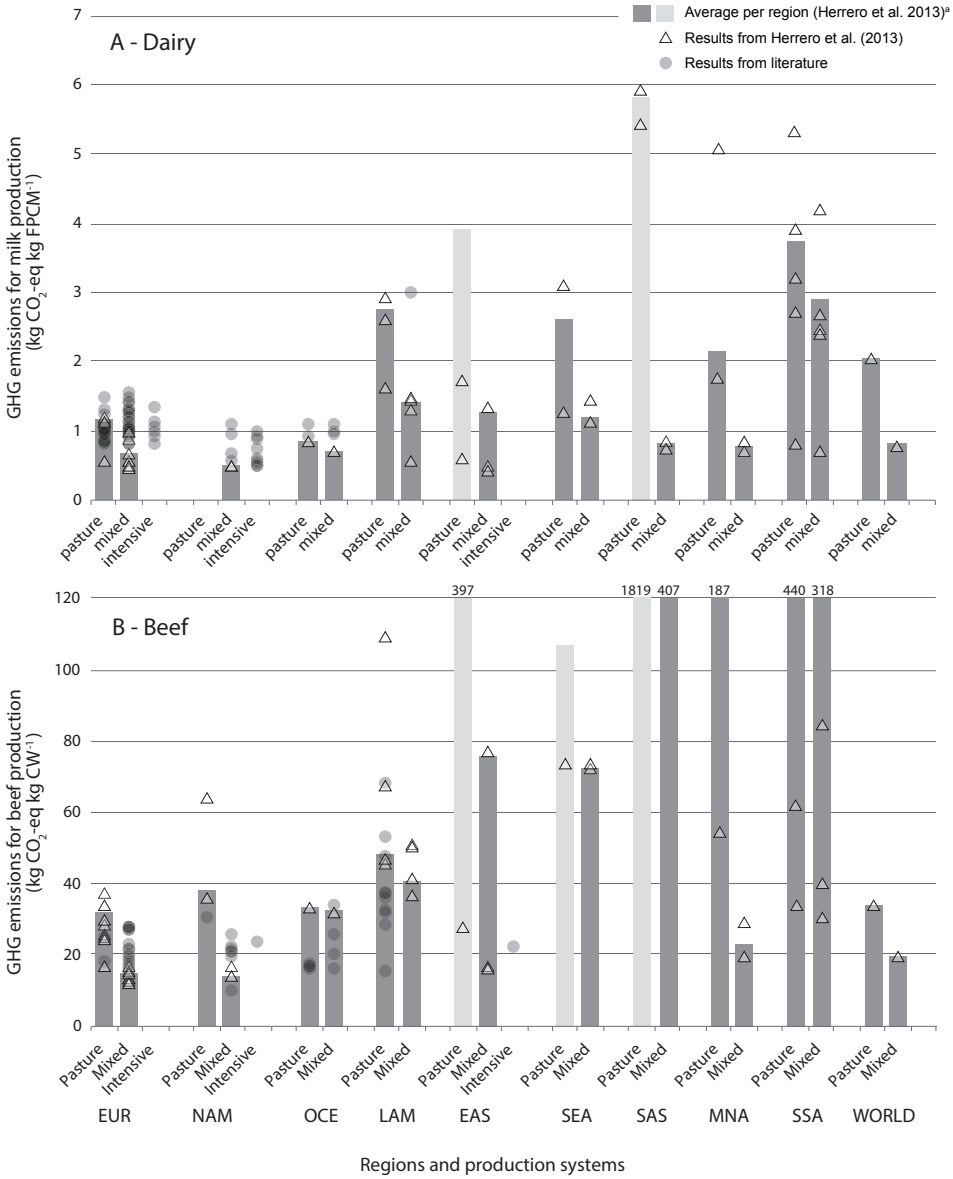

FIGURE 3-1 | GHG emissions without LUC emissions for dairy (A) and beef (B) systems per region and production system. The triangles represent the results per sub-region from Herrero et al. [7] and the dots represent the results from the studies in literature. Average per region is the weighted average of all sub-regions from Herrero et al. [7].

EUR, Europe; NAM, North America; OCE, Oceania; LAM, Latin America and the Caribbean; EAS, East Asia; SEA, South-East Asia; SAS, South Asia; MNA, Middle-East and Northern Africa; SSA, Sub-Saharan Africa.

[a] In panel A, the bars for the weighted average emissions of pasture-based dairy production in East Asia and South Asia are given a different color than the other regions, because the number of livestock units in these pasture-based systems are very small compared to the total number of LU in these regions. Similarly, in panel B, the bars for the weighted average emissions of pasture-based beef production in East Asia, South East Asia and South Asia are given a

Based on the studies on beef production, GHG emission reductions in pasture-based systems in Brazil could exceed 50% when changing from extensive, natural grass-based pastures to improved pastures. However, when the initial and/or final design of the production system are in between these two ends, the mitigation potential will be lower. Intensification of mixed systems in Canada is found to result in an emission reduction of 6%-7%. Also, for a transition from pasture-based to mixed production in the USA, emissions are found to be reduced by 16%. Transitioning from a mixed to industrial system results in a decrease of 9% in GHG emissions. When considering the GLOBIOM results for all regions, several observations are made. First, it is seen that for both pasture-based and mixed beef systems the variation in non-CO_2 emissions between the developed regions (Europe, Oceania and North America) is larger compared to dairy systems (Figure 3-1B). Second, also the difference in average emissions between pasture-based and mixed beef systems in Europe is more significant than for dairy. Third, with regard to mixed systems, the emissions vary more between all regions compared to milk production. An important explanation for these observations is considered to be the larger variation in production systems, for example with regard to pasture types and animal slaughter age. These are aspects that have a significant impact on the feed quality, feed conversion efficiency and beef yield, and thus on the resulting GHG emissions. This implies that intensification within pasture-based production systems (through adoption of best practises available) cannot only attain considerable GHG reductions in developing regions but also in some developed regions, especially Canada. In addition, GHG mitigation potentials from intensification within mixed systems, e.g. in Oceania, parts of Asia and Sub-Saharan Africa, may be larger compared to dairy production. Fourth, the additional GHG reduction potential of moving from pasture-based to mixed systems compared to intensification within pasture-based systems is considered larger than for dairy.

different color than the other regions, because the number of livestock units in these pasture-based systems are very small compared the total number of LU in these regions.
Note 1: In Herrero et al. [7], there are no results for dairy pasture-based systems in North America and EU regions because pure grass-based milk production is insignificant in these regions and all dairy cattle production systems are classified as mixed. The results presented for pasture-based dairy production in Europe are for the following regions: former USSR, rest of Central Eastern Europe and rest of Western Europe.
Note 2: In panel A and B, the emissions of all systems in the Pacific Islands are very high compared to the average emissions in Oceania, while the number of livestock units (LU) in this sub-region is very low compared to the total LU in Oceania [7]. Therefore, the individual results for the Pacific Islands are not included in the figure.
Note 3: In panel B, the results from Herrero et al. [7] for the following production systems and regions are outside the range of the y-axis: pasture-based systems in South Asia (incl. India), Turkey, Eastern, Western and Southern Africa and parts of South East Asia, and mixed systems in South Asia (incl. India), Eastern and Southern Africa.

The finding that the potential emission reductions from intensification within the pasture-based system can often be as significant as from transitioning to a mixed system is confirmed by Gerber et al. [5]. They state that global GHG emissions from beef and dairy production can be reduced by 17% to 32% when farmers apply the best practices available for their production system in the same region and climate zone. Also, as significant emission reductions can be attained within pasture-based systems, Gerber et al. [5] find that only an additional 3% to 5% emission reduction can be attained when farmers are allowed to make the transition from pasture-based to mixed production systems. In line with the findings from the present chapter, the additional reductions are highest for beef production [5].

3.3.2 Impact of intensification on land use change and associated emissions

Despite the more limited additional GHG benefits from a system transition compared to intensification within a pasture-based system, such a transition may still be required for other reasons, such as land scarcity, land use change and associated emissions [8], specifically if LUC occurs in e.g. forest frontier areas. A system transition is important because mixed and industrial systems are generally characterized by lower land occupation than pasture-based systems (see Table 3-5). For example, for beef production in the USA, Pelletier et al. (2010) find that land occupation is 114.7 m^2 kg_{CW}^{-1} for pasture-based, 91.2 m^2 kg_{CW}^{-1} for mixed and 74.5 m^2 kg_{CW}^{-1} for industrial production. According to Williams et al. (2006), the land requirement for beef in the UK is 38.5-42.1 m^2 kg_{CW}^{-1} in pasture-based systems compared to 22.8-24.1 m^2 kg_{CW}^{-1} in mixed production. For dairy production in Europe, Hürtenhuber et al. [16] find that land occupation is 1.5-2.4 m^2 kg_{FPCM}^{-1} for pasture-based and 1.2-1.7 kg_{FPCM}^{-1} for mixed production. In addition, based on Bell et al. [37], land occupation is 1.1-1.2 m^2 kg_{FPCM}^{-1} for mixed and 0.6-0.7 m^2 kg_{FPCM}^{-1} for industrial production. However, according to Williams et al. [42] and O'Brien et al. [24], the land requirement in an industrial system is respectively equal to mixed production or 0.2 m^2 kg_{FPCM}^{-1} higher compared to a pasture-based system due to a higher annual feed intake per cow in the industrial system. According to Bell et al. [37] and Williams et al. [42], the difference in land occupation between two or three systems in the same production system is smaller than the difference between production systems.

In the previous sections, it was found that organic or extensive dairy production can reduce emissions compared to more intensive conventional production. However, land occupation in an organic or extensive system is higher compared to conventional or more intensive production [16,39], which may then be associated with increased LUC emissions depending on the local circumstances. Thus, from a land use perspective, conventional production and intensification may be preferred compared to organic or extensive production. However, the studies by Hörtenhuber et al. [16] and Haas et al. [38] highlight

the importance to use fertilizers and energy as efficiently as possible, as is demonstrated in other studies as well [5,28,40]. This is especially true for mixed and industrial systems, for which the share of emissions from feed production and energy use increase compared to pasture-based systems.

TABLE 3-5 | Land occupation for dairy and beef production per region in grassland and cropland, as calculated in GLOBIOM.

Region		Dairy			Beef		
		Grassland (m^2/ kg FPCM)	Cropland (m^2/ kg FPCM)	Total Land (m^2/ kg FPCM)	Grassland (m^2/kg CW)	Cropland (m^2/kg CW)	Total Land (m^2/kg CW)
EUR	PB[a]	21.8	-	22	78	1.3	79
	MI	3.5	0.4	4	27	2.2	29
OCE	PB	36.3	-	36	623	-	623
	MI	4.7	0.1	5	115	-	115
NAM	PB[a]	-	-	-	222	3.4	226
	MI	2.5	0.6	3	25	3.6	29
LAM	PB	46.5	0.1	47	416	-	416
	MI	10.1	0.2	10	156	0.1	156
EAS	PB	8.1	0.2	8	429	2.1	431
	MI	0.7	0.4	1	47	4.9	52
SEA	PB	45.4	-	45	733	-	733
	MI	2.2	0.4	2.5	51	-	51
SAS	PB	64.7	-	65	11,909	-	11,909
	MI	1.8	0.6	2	377	-	377
MNA	PB	78.3	-	78	1,627	1.4	1,629
	MI	3.0	1.0	4	153	6.5	160
SSA	PB	166.8	-	167	16,897	-	16,897
	MI	68.7	0.2	69	9,249	0.4	9,249
WORLD	PB	49.2	-	49	639	0.7	639
	MI	3.8	0.4	4	199	1.5	201

The regions are defined as in Figure 3-1. Production systems: PB, pasture-based; MI, mixed.
[a]There are no GLOBIOM results for dairy pasture-based systems in North America and EU regions, see note 1 for Figure 3-1.

When translating land occupation to land use change emissions, studies from the literature estimate that the effect of the conversion of natural lands and forests to agricultural land was limited to a maximum of 11% of the total emissions from dairy production in Europe [16,24,44]. Also, the share of LUC-related emissions is larger for mixed and industrial systems compared to pasture-based production, because of increased feed imports from South America [16,24]. However, when the GHG balance without LUC-related emissions was lower for mixed systems compared to pasture-based production, Hörtenhuber et al. [16] show that this remains true for the GHG balance including LUC. In accordance with the studies from literature, the GLOBIOM results show that the share of LUC-related emissions is small for dairy and beef production in Europe and North America and for

beef production in Oceania (Figure 3-2). However, LUC-related emissions in both dairy and beef production account for 20% up to more than 50% of the total emissions in Latin America, South East Asia and Sub-Saharan Africa. Also, in developing regions, the amount and share of emissions related to LUC are generally higher for pasture-based systems compared to mixed systems (see also Table 3-10 in Appendix A3.5). In these regions, high numbers of livestock units graze in low productive areas. The low animal productivity and high land occupation then cause significant natural land conversion, degradation and/or deforestation.

Because of the large contribution of LUC, its mitigation is a key strategy for reducing GHG emissions from dairy production in Latin America and from dairy and beef production in Sub-Saharan Africa and parts of Asia. For example, Cohn et al. [45] show that policy-driven intensification within pasture-based cattle production systems in Brazil could reduce the pasture area by 16-21 million hectares (Mha). As a result, 15-17 Mha of forest could spared from deforestation and emissions associated with deforestation could drop by 75%-80% [45]. In Latin America, the share of LUC-related emissions in the total GHG balance is lower for beef (up to 20%) than for dairy (up to 40%). Therefore, the GHG reduction potential of LUC mitigation (in terms of percentage) is also lower for beef. However, as the number of livestock units in the beef sector is significantly higher than in the dairy sector, the absolute GHG reduction potential may be larger for the beef sector.

Several strategies can be identified to reduce LUC-related emissions. First, when new farms are established, these should be based on mixed production systems instead of pastures-based systems. In addition, as illustrated by the study of Cohn et al. [45], land use change can be mitigated through a reduction in the land occupation of existing cattle production. This can either be attained by moving from a pasture-based to mixed production system, or by reducing the land occupation in the current system. The literature provides several options to realize reduced land occupation: improving the productivity of grassland and cropland, increasing the animal productivity through improved feed quality and introducing or increasing the amount of crops in the feed ration [16,18,37,45,46]. As these options are also identified as strategies to mitigate emissions in general, the total GHG reduction potential of these measures is even larger. In addition, the studies illustrate the importance of good soil management to reduce soil organic carbon losses or even stimulate soil carbon accumulation. For example, Belflower et al. [33] account for changes in soil organic carbon due to the conversion of cropland to perennial grassland in mixed dairy production in the USA. The resulting SOC sequestration reduces total emissions by 0.07 kg CO_2-eq kg_{FPCM}^{-1} or 15%. Also, in the study by Basarab et al. [43] on mixed beef production in Canada, land has been under rotation between grassland and cropland (cereals and oilseed crops). This rotation results in a net SOC sequestration and an emission reduction of 2.2 to 3.6 kg CO_2-eq kg_{CW}^{-1} (11-16%) [43].

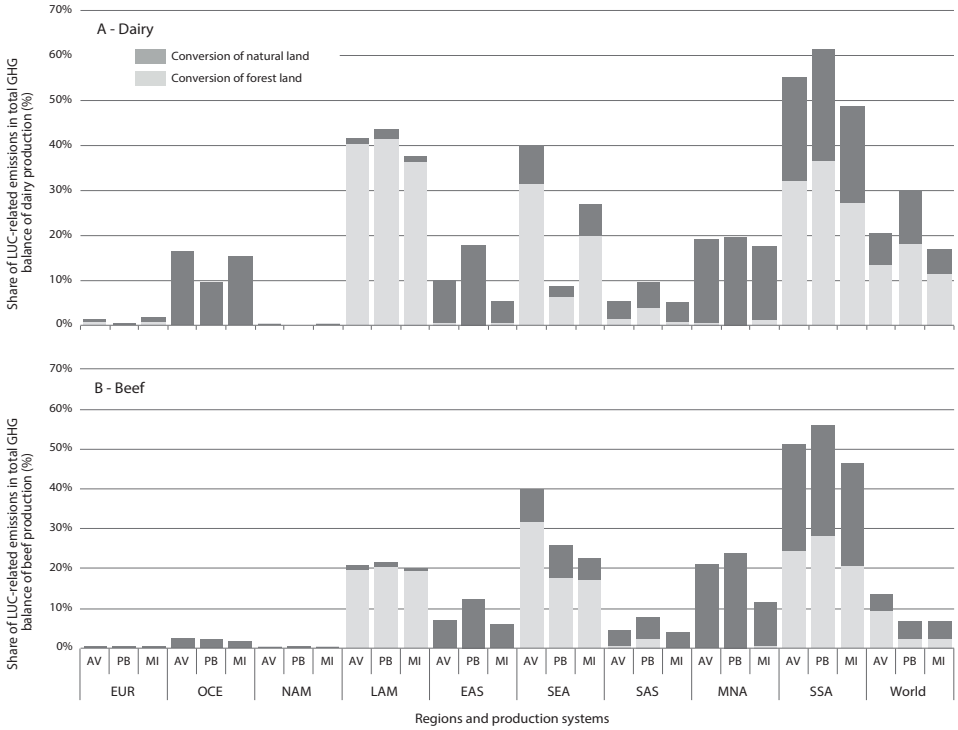

FIGURE 3-2 | LUC emissions (related to natural land conversion and deforestation) in terms of percentage of the total GHG emissions for dairy production (A) and for beef production (B). Estimates are obtained from GLOBIOM simulations in Havlik et al. [8]. Regions are defined as in Figure 3-1. Production systems: AV, weighted average of all production systems (also including *urban* and *other*); PB, pasture-based; MI, mixed. For absolute values of LUC-related emissions, see Appendices A3.3 to A3.5.

Note 1: there are no GLOBIOM results for dairy pasture-based systems in North America, see note 1 for Figure 3-1.
Note 2: in South-East Asia, the weighted average LUC-related emissions of all production systems are higher than the emissions from both pasture-based and mixed systems, because the system categories *other* and *urban* are also included in this average.

3.4 DISCUSSION

3.4.1 Data cover and data quality

The majority of the studies in literature evaluate pasture-based and mixed systems. Fewer studies have assessed industrial systems, especially in the case of beef production. As a result, the environmental performance of industrial systems in terms of GHG emissions and land occupation cannot be investigated well. The same is true for the GHG reduction potential of the transition from mixed to industrial systems. The few cases available indicate that the intensification step from a mixed to an industrial production can either have a positive or a negative impact on the GHG balance while the effect on land occupation is generally positive. The influence on LUC and LUC-related emissions,

however, is largely unknown yet. It is likely that the assessment of industrial systems is lacking because mixed and pasture-based systems are currently dominant in milk and beef production. For example, Havlík et al. [8] state that not enough data was available at the global level to differentiate between more than two aggregate production systems. However, to improve the insights in GHG emissions and mitigation potentials for different systems, it is desirable to include industrial systems as a separate category in models and pay more attention to these systems in bottom-up studies. Better understanding of the options for cattle production and GHG mitigation could, for example, help to develop strategies to improve future beef and milk production. An approach to assess industrial systems while data is limited or lacking could be to use a conceptual design of industrial production systems, i.e. not based on existing but on potential production practices. Possibly, different designs and/or scenarios could be included to compare their influence on future GHG emissions and LUC.

In addition to the unequal coverage of production systems, also regions are covered unequally in studies from the literature. Almost all studies investigate dairy and beef production in Europe, North America and Oceania and beef production in Brazil. While studies on cattle production in developing countries are lacking, GLOBIOM shows that there are significant differences in results between and within developing regions. These should be investigated in more detail. Also, increasing population, growing demand for livestock products and rapid urbanization will have a significant impact on livestock production in these regions [47]. It is likely that this will also influence the GHG emission performance of cattle production. Finally, the results show that the contribution of developing countries to land use change is considerable. For these reasons, it is highly recommended to dedicate more studies on developing countries and collect more data for these regions.

Based on the description of the management system in the bottom-up studies, the classification to one of the three production systems was not always clear, especially for dairy. Many studies use production system classifications which are very different from the classification applied in the present study, e.g. average dairy system or improved natural grass system. Other studies classify the system as pasture-based or confinement system, while the present study categorizes it as mixed based on the detailed system description. For dairy, the impact on the results is expected to be limited as the differences in GHG emissions within and between pasture-based and mixed systems in developed countries are relatively small. However, for beef production, especially in Latin America, the results are more likely to change when the system classification would be altered. Because of the large variation in production practices, the classification system applied in the present study is appropriate for models, but less suitable to apply to case specific studies.

Although this cannot be solved easily, a first step for improvement could be to further develop the definitions of the different production systems. These definitions could, for example, include specifications about the time spent at pasture and the amount of grass, crops and concentrates in the diet.

3.4.2 LUC

Although the importance of LUC is widely recognized among scientists, LUC has not been included in most bottom-up studies because of conceptual and methodological limitations [48]. One of the limitations is insufficient or unavailable information related to LUC [17]. For example, the origin of feedstuff and the affected ecosystem are often unknown [49]. Nevertheless, including LUC in bottom-up studies is very valuable because it allows assessing region specific impacts and trade-offs of different practices such as organic and conventional production. It is especially interesting to include LUC in LCA studies, because these do not only include GHG emissions, but also consider effects on other impact categories like water consumption. The very few studies that include LUC consider different aspects, e.g. on-farm LUC due to rotation between crop- and grassland and/or LUC related to imported feeds [16,31-34]. Nguyen et al. [34] only present the combined emissions of land use and land use change (LULUC). Also, studies apply different methods to calculate LUC-related emissions and results can vary significantly depending on the selected approach [50]. Attempts are made to develop more uniform methods that can be widely applied, but there is no consensus yet on how to account for LUC-related emissions in bottom-up (LCA) studies [49,50].

Using global models for the assessment of LUC is the only way to apply a uniform method for assessing global LUC and comparing regional results. The advantage is that it provides a consistent framework where agricultural production and land use are correctly balanced across all regions, through direct occupation of land, or through indirect effects as a result of feed consumption and international trade. Also, effects of varying important parameters or policy assumptions can easily be tested. A disadvantage is the level of uncertainty on some data inputs such as global land cover or grassland productivity. In addition, some limitations for assessing LUC in bottom-up studies also apply to models. First, the models apply different approaches, e.g. with regard to what causes and types of LUC are included [5,51]. Second, as illustrated in the example of indirect land use change studies for biofuels [52], the variety of approaches and parameterisation across models lead to significant ranges of uncertainty for LUC estimates. As both bottom-up studies and global models have advantages and disadvantages to investigate LUC, it is recommended that both approaches are used complementary to each other. In all cases, the methods and underlying assumptions should be explained clearly [50].

3.5 CONCLUSIONS

This study aimed to review the GHG emissions and land occupation of dairy and beef production in different world regions and especially to compare the impact of intensification within one production system (pasture, mixed or industrial) and of transitions to another system. To this end, the results of different bottom-up studies for the same region and from the same reference were discussed for dairy production in Europe and beef production in North America and Brazil. The findings from this assessment were then compared to results from GLOBIOM and extrapolated to nine world regions.

Based on the studies for dairy in Europe, GHG emissions excluding land use change are reduced by approximately 1% to 14% through intensification in the pasture-based, mixed or industrial system and by approximately 2-26% by moving from pasture-based to mixed production. Moving from mixed to industrial production can either decrease emissions by 8%-14% or increase the GHG balance depending on, for example, the reduction in emissions due to improved feed quality and the increase in emissions related to manure management. Overall, whether the influence on the GHG balance is larger for intensification within one system or for system transitions depends on the specific design of the initial and final production systems. This also applies to other developed regions. In developing countries, the differences within and between pasture-based and mixed systems are more significant and there is a great potential to reduce emissions by intensification within the pasture-based system. The additional GHG reduction potential of a transition from pasture-based to mixed and industrial production is limited. Also, emission reductions of intensification within the mixed system are smaller compared to the pasture-based system.

For beef production in Brazil, the emission reduction potential for intensification within the pasture-based system could exceed 50% when changing from extensive, natural grass-based pastures to improved pastures. Intensification of mixed systems in Canada reduces emissions by 6%-7%. Moving from pasture-based to mixed production in the USA decreases emissions by 16% and moving from mixed to industrial production results in an emission reduction of 9%. When considering all global regions, it is found that in every region the difference in emissions within and between production systems is larger compared to dairy production. This implies that intensification within pasture-based beef production systems (through adoption of best practises available) cannot only attain considerable GHG reductions in developing regions but also in some developed regions. Also, the additional GHG reduction potentials of transitions from pasture-based to mixed systems compared to intensification within pasture-based systems, as well as the GHG mitigation potentials of intensification within mixed systems are considered larger for

beef than for dairy. One study suggests that moving from mixed to industrial systems in the USA has an additional reduction potential as well, but more assessments are needed to further investigate this for the USA and other regions.

Although significant GHG emission reductions can often be attained by intensification within pasture-based system, moving to the mixed system is an important strategy to significantly reduce land occupation and mitigate land use change and associated emissions. In developing regions, especially Sub-Saharan Africa and Latin America, land use change mitigation is often the most important strategy to reduce GHG emissions from dairy and beef production.

While the largest challenges and potentials to mitigate GHG emissions are found in developing regions, studies in the literature focus on cattle production in developed countries. Therefore, more studies should be dedicated to developing countries and more data should be collected for these regions. In addition, industrial production systems are currently not included in GLOBIOM and only assessed in a limited number of studies. However, because of the increasing importance of reducing GHG emissions and LUC caused by the cattle sector, it should be investigated what role industrial production could play in the future. Finally, it is recommended to investigate LUC and associated emissions more intensively. To this aim, bottom-up studies and global models may be used complementary to each other. Following these directions would greatly help improving the understanding of the GHG emission performance of cattle production systems. The insights gained can be used to design strategies for future sustainable developments in cattle production.

REFERENCES

[1] Herrero M, Gerber P, Vellinga T, et al. Livestock and greenhouse gas emissions: The importance of getting the numbers right. Animal Feed Science and Technology 2011;166-167:779-782.

[2] Phillips SJ, Anderson RP, Schapire RE. Maximum entropy modeling of species geographic distributions. Ecological Modelling 2006;190(3–4):231-259.

[3] McMichael AJ, Powles JW, Butler CD, et al. Food, livestock production, energy, climate change, and health. The Lancet 2007;370(9594):1253-1263.

[4] Bellarby J, Tirado R, Leip A, et al. Livestock greenhouse gas emissions and mitigation potential in Europe. Global Change Biology 2013;19(1):3-18.

[5] Gerber P, Steinfeld H, Henderson B, et al. Tackling Climate Change Through Livestock: A Global Assessment of Emissions and Mitigation Opportunities. Food and Agriculture Organization of the United Nations (FAO): Rome; 2013.

[6] Steinfeld H, Gerber P, Wassenaar T, et al. Livestock's long shadow: environmental issues and options. FAO: Rome, Italy; 2006.

[7] Herrero M, Havlík P, Valin H, et al. Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. Proceedings of the National Academy of Sciences of the United States of America 2013;110(52):20888-20893.

[8] Havlík P, Valin H, Herrero M, et al. Climate change mitigation through livestock system transitions. Proceedings of the National Academy of Sciences 2014;111(10):3709-3714.

[9] Smith P, Martino D, Cai Z, et al. Greenhouse gas mitigation in agriculture. Philosophical transactions of the Royal Society of London.Biological Sciences 2008;363(1492):789-813.

[10] Eckard RJ, Grainger C, de Klein CAM. Options for the abatement of methane and nitrous oxide from ruminant production: A review. Livestock Science 2010;130(1-3):47-56.

[11] Hristov AN, Oh J, Lee C, et al. Mitigation of greenhouse gas emissions from livestock production: A review of technical options for non-CO2 emissions. Food and Agriculture Organization of the United Nations (FAO): Rome; 2013.

[12] Robinson T, Thornton PK, Franceschini G, et al. Global livestock production systems. Food and Agriculture Organization of the United Nations (FAO) and International Livestock Research Institute (ILRI): Rome; 2011.

[13] Seré C, Steinfeld H and Groenewold J. World livestock production systems: Current status, issues and trends. Rome; 1996.

[14] Schader C, Jud K, Meier MS, et al. Quantification of the effectiveness of greenhouse gas mitigation measures in Swiss organic milk production using a life cycle assessment approach. Journal of Cleaner Production 2014;73:227-235.

[15] Valin H, Havlik P, Mosnier A, et al. Agricultural productivity and greenhouse gas emissions: trade-offs or synergies between mitigation and food security? Environmental Research Letters 2013;8(3):9.

[16] Hörtenhuber S, Lindenthal T, Amon B, et al. Greenhouse gas emissions from selected Austrian dairy production systems-model calculations considering the effects of land use change. Renewable Agriculture and Food Systems 2010;25(04):316-329.

[17] Ruviaro CF, de Léis CM, Lampert VDN, et al. Carbon footprint in different beef production systems on a southern Brazilian farm: a case study. Journal of Cleaner Production 2015;96:435-443.

[18] Dick M, Abreu dS, Dewes H. Life cycle assessment of beef cattle production in two typical grassland systems of southern Brazil. Journal of Cleaner Production 2015;96:426-434.

[19] Smeets EMW, Faaij APC, Lewandowski IM, et al. A bottom-up assessment and review of global bio-energy potentials to 2050. Progress in Energy and Combustion Science 2007;33(1):56-106.

[20] Flysjö A, Henriksson M, Cederberg C, et al. The impact of various parameters on the carbon footprint of milk production in New Zealand and Sweden. Agricultural Systems 2011;104(6):459-469.

[21] Cederberg C, Meyer D and Flysjö A. Life cycle inventory of greenhouse gas emissions and use of land and energy in Brazilian beef production. Swedish Institute for Food and Biotechnology (SIK): Göteborg, Sweden; 2009. 792.

[22] Herrero M, Thornton PK, Gerber P, et al. Livestock, livelihoods and the environment: understanding the trade-offs. Environmental Sustainability 2009;1(2):111-120.

[23] Pelletier N, Pirog R, Rasmussen R. Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. Agricultural Systems 2010;103(6):380-389.

[24] O'Brien D, Shalloo L, Patton J, et al. A life cycle assessment of seasonal grass-based and confinement dairy farms. Agricultural Systems 2012;107:33-46.

[25] Gerber P, Vellinga T, Opio C, et al. Greenhouse Gas Emissions from the Dairy Sector. Rome; 2010.

[26] McKiernan B, Gaden B and Sundstrom B. Dressing percentages for cattle. NSW Department of Primary Industries: New South Wales, Australia; 2007. Primefact 340.

[27] Opio C, Gerber P, Mottet A, et al. Greenhouse gas emissions from ruminant supply chains- A global life cycle assessment. Food and Agriculture Organization of the United Nations (FAO): Rome; 2013.

[28] Lovett DK, Shalloo L, Dillon P, et al. Greenhouse gas emissions from pastoral based dairying systems: The effect of uncertainty and management change under two contrasting production systems. Livestock Science 2008;116(1-3):260-274.

[29] Mosnier A, Havlík P, Valin H, et al. Alternative U.S. biofuel mandates and global GHG emissions: The role of land use change, crop management and yield growth. Energy Policy 2013;57:602-614.

[30] Frank S, Schmid E, Havlík P, et al. The dynamic soil organic carbon mitigation potential of European cropland. Global Environmental Change 2015.

[31] Bartl K, Gómez CA, Nemecek T. Life cycle assessment of milk produced in two smallholder dairy systems in the highlands and the coast of Peru. Journal of Cleaner Production 2011;19(13):1494-1505.

[32] Bonesmo H, Beauchemin KA, Harstad OM, et al. Greenhouse gas emission intensities of grass silage based dairy and beef production: A systems analysis of Norwegian farms. Livestock Science 2013;152(2-3):239-252.

[33] Belflower JB, Bernard JK, Gattie DK, et al. A case study of the potential environmental impacts of different dairy production systems in Georgia. Agricultural Systems 2012;108(1):84-93.

[34] Nguyen TLT, van dW, Eugène M, et al. Effects of type of ration and allocation methods on the environmental impacts of beef-production systems. Livestock Science 2012;145(1-3):239-251.

[35] Havlík P, Schneider UA, Schmid E, et al. Global land-use implications of first and second generation biofuel targets. Energy Policy 2011;39(10):5690-5702.

[36] O'Brien D, Shalloo L, Grainger C, et al. The influence of strain of Holstein-Friesian cow and feeding system on greenhouse gas emissions from pastoral dairy farms. Journal of dairy science 2010;93(7):3390-3402.

[37] Bell MJ, Wall E, Russell G, et al. The effect of improving cow productivity, fertility, and longevity on the global warming potential of dairy systems. Journal of dairy science 2011;94(7):3662-3678.

[38] Haas G, Wetterich F, Köpke U. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. Agriculture, Ecosystems & Environment 2001;83(1-2):43-53.

[39] Kristensen T, Mogensen L, Knudsen MT, et al. Effect of production system and farming strategy on greenhouse gas emissions from commercial dairy farms in a life cycle approach. Livestock Science 2011;140(1-3):136-148.

[40] Olesen JE, Schelde K, Weiske A, et al. Modelling greenhouse gas emissions from European conventional and organic dairy farms. Agriculture, Ecosystems & Environment 2006;112(2-3):207-220.

[41] Thomassen MA, van Calker KJ, Smits MCJ, et al. Life cycle assessment of conventional and organic milk production in the Netherlands. Agricultural Systems 2008;96(1-3):95-107.

[42] Williams AG, Audsley E and Sandars DL. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Natural Resource Management Institute, Cranfield University, Silsoe Research Institute: Bedford, UK; 2006.

[43] Basarab J, Baron V, López-Campos Ó, et al. Greenhouse gas emissions from calf- and yearling-fed beef production systems, with and without the use of growth promotants. Animals 2012;2(4):195-220.

[44] O'Brien D, Capper JL, Garnsworthy PC, et al. A case study of the carbon footprint of milk from high-performing confinement and grass-based dairy farms. Journal of dairy science 2014;97(3):1835-1851.

[45] Cohn AS, Mosnier A, Havlík P, et al. Cattle ranching intensification in Brazil can reduce global greenhouse gas emissions by sparing land from deforestation. Proceedings of the National Academy of Sciences 2014;111(20):7236-7241.

[46] Ridoutt BG, Page G, Opie K, et al. Carbon, water and land use footprints of beef cattle production systems in southern Australia. Journal of Cleaner Production 2014;73:24-30.

[47] Thornton PK, van de Steeg J, Notenbaert A, et al. The impacts of climate change on livestock and livestock systems in developing countries: A review of what we know and what we need to know. Agricultural Systems 2009;101(3):113-127.

[48] Dalgaard R, Schmidt J, Halberg N, et al. Case Study LCA of Soybean Meal. International Journal of LCA 2008;13(3):240-254.

[49] Hörtenhuber S, Piringer G, Zollitsch W, et al. Land use and land use change in agricultural life cycle assessments and carbon footprints - the case for regionally specific land use change versus other methods. Journal of Cleaner Production 2014;73:31-39.

[50] Flysjö A, Cederberg C, Henriksson M, et al. The interaction between milk and beef production and emissions from land use change - critical considerations in life cycle assessment and carbon footprint studies of milk. Journal of Cleaner Production 2012;28:134-142.

[51] Valin H, Havlík P, Forsell N, et al. Description of the GLOBIOM (IIASA) model and comparison with the MIRAGE-BioF (IFPRI) model. Laxenburg; 2013.

[52] Wicke B, Verweij P, van Meijl H, et al. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 2012;3(1):87-100.

[53] Basset-Mens C, Ledgard S, Boyes M. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecological Economics 2009;68(6):1615-1625.

[54] Beukes PC, Gregorini P, Romera AJ, et al. Improving production efficiency as a strategy to mitigate greenhouse gas emissions on pastoral dairy farms in New Zealand. Agriculture, Ecosystems & Environment 2010;136(3-4):358-365.

[55] Cederberg C and Stadig M. LCA Case Studies System Expansion and Allocation in Life Cycle Assessment of Milk and Beef Production. International Journal of LCA 2003;8(6):350-356.

[56] Henriksson M, Flysjö A, Cederberg C, et al. Variation in carbon footprint of milk due to management differences between Swedish dairy farms. Animal : an international journal of animal bioscience 2011;5(9):1474-84.

[57] Casey JW and Holden NM. Analysis of greenhouse gas emissions from the average Irish milk production system. Agricultural Systems 2005;86(1):97-114.

[58] Shortall OK and Barnes AP. Greenhouse gas emissions and the technical efficiency of dairy farmers. Ecological Indicators 2013;29:478-488.

[59] Castanheira ÉG, Dias aC, Arroja L, et al. The environmental performance of milk production on a typical Portuguese dairy farm. Agricultural Systems 2010;103(7):498-507.

[60] Arsenault N, Tyedmers P, Fredeen A. Comparing the environmental impacts of pasture-based and confinement-based dairy systems in Nova Scotia (Canada) using life cycle assessment. International Journal of Agricultural Sustainability 2009;7(1):19-41.

[61] Phetteplace HW, Johnson DE, Seidl AF. Greenhouse gas emissions from simulated beef and dairy livestock systems in the United States. Nutrient Cycling in Agroecosystems 2001;60:99-102.

[62] Rotz CA, Montes F, Chianese DS. The carbon footprint of dairy production systems through partial life cycle assessment. Journal of dairy science 2010;93(3):1266-1282.

[63] Gollnow S, Lundie S, Moore AD, et al. Carbon footprint of milk production from dairy cows in Australia. International Dairy Journal 2014;37(1):31-38.

[64] Christie KM, Rawnsley RP, Eckard RJ. A whole farm systems analysis of greenhouse gas emissions of 60 Tasmanian dairy farms. Animal Feed Science and Technology 2011;166-167(1):653-662.

[65] Thomassen MA, Dolman MA, van Calker KJ, et al. Relating life cycle assessment indicators to gross value added for Dutch dairy farms. Ecological Economics 2009;68(8-9):2278-2284.

[66] McGeough EJ, Little SM, Janzen HH, et al. Life-cycle assessment of greenhouse gas emissions from dairy production in Eastern Canada: a case study. Journal of dairy science 2012;95(9):5164-5175.

[67] Crosson P, Shalloo L, O'Brien D, et al. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. Animal Feed Science and Technology 2011;166-167:29-45.

[68] Beauchemin KA, Henry Janzen H, Little SM, et al. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. Agricultural Systems 2010;103(6):371-379.

[69] Casey JW and Holden NM. Quantification of GHG emissions from sucker-beef production in Ireland. Agricultural Systems 2006;90(1-3):79-98.

[70] Foley PA, Crosson P, Lovett DK, et al. Whole-farm systems modelling of greenhouse gas emissions from pastoral suckler beef cow production systems. Agriculture, Ecosystems & Environment 2011;142(3-4):222-230.

[71] Nguyen TLT, Hermansen JE, Mogensen L. Environmental consequences of different beef production systems in the EU. Journal of Cleaner Production 2010;18(8):756-766.

[72] White TA, Snow VO, King WM. Intensification of New Zealand beef farming systems. Agricultural Systems 2010;103(1):21-35.

[73] Ogino A, Kaku K, Osada T, et al. Environmental impacts of the Japanese beef-fattening system with different feeding lengths as evaluated by a life-cycle assessment method. Journal of animal science 2004;82(7):2115-2122.

APPENDICES

A3.1 Literature review of dairy production systems

TABLE 3-6 | Case studies on dairy production systems, their characteristics, total GHG emissions and land occupation.

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Haas et al. [38]	Germany	Organic dairy system	Pasture-based	LCA approach	Insufficient information about the system boundaries. EC and FP are included.	No	1.30	-	-
Hörtenhuber et al. [16]	Austria	Alpine, conventional	Pasture-based	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	1.10	1.17	2.2
Hörtenhuber et al. [16]	Austria	Alpine, organic	Pasture-based	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	1.02	1.02	2.4
Hörtenhuber et al. [16]	Austria	Upland, pasture-based, conventional	Pasture-based	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	1.03	1.03	1.5
Hörtenhuber et al. [16]	Austria	Upland, pasture-based, organic	Pasture-based	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	0.95	0.95	1.6
Thomassen et al. [41]	The Netherlands	Organic dairy system	Pasture-based	LCA approach	EF, MM, FP and EC	No	1.50	-	1.8
Wiliams et al. [42]	United Kingdom	Organic dairy system	Pasture-based	LCA approach	EF, MM, FP and EC	No	**1.23**	-	2.0
Lovett et al. [28]	Ireland	Moorepark	Pasture-based	PME (Pastoral Milk Emissions Model)	EF, MM, FP and EC	No	0.86	-	-
O'Brien et al. [24]	Ireland	Grass-based dairy system	Pasture-based	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.86	0.87	0.7

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
O'Brien et al. [44]	Ireland	High performance grass-based system	Pasture-based	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	Yes	**0.84**[1]	0.84	-
O'Brien et al. [36]	Ireland	Moorepark, New Zealand cows	Pasture-based	Moorepark Dairy System Model (MDSM)	EF, MM, FP and EC	No	0.98		
O'Brien et al. [36]	Ireland	Moorepark, High durability cows	Pasture-based	Moorepark Dairy System Model (MDSM)	EF, MM, FP and EC	No	1.02		
O'Brien et al. [36]	Ireland	Moorepark, High production cows	Pasture-based	Moorepark Dairy System Model (MDSM)	EF, MM, FP and EC	No	1.10		
O'Brien et al. [36]	Ireland	High stocking rate, New Zealand cows	Pasture-based	Moorepark Dairy System Model (MDSM)	EF, MM, FP and EC	No	0.95		
O'Brien et al. [36]	Ireland	High stocking rate, High durability cows	Pasture-based	Moorepark Dairy System Model (MDSM)	EF, MM, FP and EC	No	1.01		
O'Brien et al. [36]	Ireland	High stocking rate, High production cows	Pasture-based	Moorepark Dairy System Model (MDSM)	EF, MM, FP and EC	No	1.09		
Schader et al. [14]	Switzerland	Highland organic system	Pasture-based	Single-farm model based on a LCA approach	EF, MM, FP and EC	No	1.10	-	-
Bartl et al. [31]	Peru	Highland pasture-based system	Pasture-based	LCA approach	EF, MM, FP and EC	Yes	13.68	13.68	23.1
Basset-Mens et al. [53]	New Zealand	Average New Zealand dairy system	Pasture-based	LCA approach	EF, MM, FP and EC	No	1.09	-	1.3
Beukes et al. [54]	New Zealand	Pasture-based dairy system	Pasture-based	DairyBase database	EF, MM, FP and EC	No	0.91*	-	-
GLOBIOM	Europe	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.16**	1.16	21.8
Herrero et al. [7][2]	Former USSR	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.21**		
Herrero et al. [7][2]	RCEU	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.14**		

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Herrero et al. [7][2]	ROWE	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**0.56**		
GLOBIOM	Oceania	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**0.85**	0.94	36.4
Herrero et al. [7][2]	ANZ	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**0.85**		
Herrero et al. [7][2]	Pacific Islands	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**6.32**		
GLOBIOM	North America	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**-**	-	-
GLOBIOM	Latin America and the Caribbean	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.77**	4.90	46.6
Herrero et al. [7][2]	Brazil	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.61**		
Herrero et al. [7][2]	RCAM	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.62**		
Herrero et al. [7][2]	RSAM	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.93**		
GLOBIOM	East Asia	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**3.91**	4.76	8.3
Herrero et al. [7][2]	China	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.73**		
Herrero et al. [7][2]	Japan	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**0.62**		
GLOBIOM	South East Asia	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.63**	2.88	45.4
Herrero et al. [7][2]	RSEA OPA	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.26**		
Herrero et al. [7][2]	RSEA PAC	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**3.11**		
GLOBIOM	South Asia	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**5.82**	6.43	64.7
Herrero et al. [7][2]	India	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**5.43**		
Herrero et al. [7][2]	RSAS	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**5.93**		
GLOBIOM	Middle-East and North Africa	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.14**	2.66	78.3
Herrero et al. [7][2]	MENA	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**1.76**		
Herrero et al. [7][2]	Turkey	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**5.09**		
GLOBIOM	Sub-Saharan Africa	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**3.73**	9.70	166.8

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Herrero et al. [7][2]	Congo Basin	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**3.21**		
Herrero et al. [7][2]	Eastern Africa	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**3.91**		
Herrero et al. [7][2]	Western Africa	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**5.33**		
Herrero et al. [7][2]	Southern Africa	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.71**		
Herrero et al. [7][2]	South Africa	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**0.82**		
GLOBIOM	World	-	Pasture-based	GLOBIOM model	EF, MM and FP	Yes	**2.05**	2.93	49.3
Haas et al. [38]	Germany	Extensive dairy system	Mixed	LCA approach	EF, MM, FP and EC	No	1.00	-	-
Haas et al. [38]	Germany	Intensive dairy system	Mixed	LCA approach	Insufficient information about the system boundaries. FP and EC are included.	No	1.30	-	-
Thomassen et al. [41]	Netherlands	Conventional dairy system	Mixed	LCA approach	EF, MM, FP and EC	No	1.40	-	1.3
Hörtenhuber et al. [16]	Austria	Upland, conventional	Mixed	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	0.95	1.03	1.6
Hörtenhuber et al. [16]	Austria	Upland, organic	Mixed	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	0.91	0.91	1.7
Hörtenhuber et al. [16]	Austria	Lowland, conventional	Mixed	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	0.82	0.90	1.2

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Hörtenhuber et al. [16]	Austria	Lowland, organic	Mixed	Model design per on-farm production system (alpine or upland or lowland)	EF, MM, FP and EC	Yes	0.82	0.82	1.3
Kristensen et al. [39]	Denmark	Organic dairy system	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.28	-	2.4
Kristensen et al. [39]	Denmark	Conventional dairy system	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.21	-	1.8
Bonesmo et al. [32][3]	Norway	Average Norway dairy system	Mixed	HolosNor model	EF, MM, FP and EC	Yes	**1.02**[1]	-	-
Cederberg and Stadig [55]	Sweden	Average Swedish dairy system	Mixed	LCA approach consistent with ISA 14041 standard	Direct on- farm, purchased inputs and indirect N_2O emissions (Crosson et al. 2011)	No	1.05	-	4.0
Flysjö et al. [20]	Sweden	Average Sweden dairy system	Mixed	LCA approach consistent with ISA 14040- 14044 standards	EF, MM, FP and EC	No	**1.14**	-	-
Henriksson et al. [56]	Sweden	Average Sweden dairy system	Mixed	LCA approach consistent with ISA 14040- 14044 standards	EF, MM, FP and EC	No	1.12	-	-
Casey and Holden [57]	Ireland	Pasture-based dairy system	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.47	-	1.2
Lovett et al. [28]	Ireland	Kilmaley	Mixed	PME (Pastoral Milk Emissions Model)	EF, MM, FP and EC	No	**0.95**	-	-
O'Brien et al. [36]	Ireland	High concentrate, New Zealand cows	Mixed	MDSM (Moorepark Dairy System Model)	EF, MM, FP and EC	No	0.98		
O'Brien et al. [36]	Ireland	High concentrate, High durability cows	Mixed	MDSM (Moorepark Dairy System Model)	EF, MM, FP and EC	No	0.99		
O'Brien et al. [36]	Ireland	High concentrate, High production cows	Mixed	MDSM (Moorepark Dairy System Model)	EF, MM, FP and EC	No	1.04		

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Williams et al. [42]	United Kingdom	Non-organic dairy system	Mixed	LCA approach	EF, MM, FP and EC	No	1.06	-	1.2
Bell et al. [37]	Scotland	High forage control	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.32	-	1.2
Bell et al. [37]	Scotland	High forage select	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.21	-	1.1
Shortall and Barnes [58]	Scotland	Average Scottish dairy system	Mixed	DEA (Data Envelopment Analysis) using farm data from surveys. Equations and emission factors are obtained from the IPCC	EF, MM, FP and EC	No	1.00	-	-
Schader et al. [14]	Switzerland	Lowland organic mixed dairy system	Mixed	Single-farm model based on a LCA approach	EF, MM, FP and EC	No	0.91	-	-
Castanheira et al. [59]	Portugal	Average Portuguese dairy system	Mixed	LCA approach consistent with ISA 14040- 14044 standards	EF, MM, FP and EC	No	1.02	-	-
Olesen et al. [40]	Europe	Organic dairy system	Mixed	FarmGHG model	EF, MM, FP and EC	No	1.57*	-	-
Olesen et al. [40]	Europe	Various dairy systems in Europe	Mixed	FarmGHG model	EF, MM, FP and EC	No	1.43*	-	-
Arsenault et al. [60]	Canada	Pasture-based dairy system	Mixed	LCA approach	EF, MM, FP and EC	No	0.97	-	2.8
Belflower et al. [33]	United States	Pasture-based dairy system	Mixed	IFSM (Integrated Farm System Model)	EF, MM, FP and EC	Yes	0.57	0.48[4]	-
Phetteplace et al. [61]	United States	Average US dairy system	Mixed	Whole-farm approach using equations and emissions factors from IPCC	EF, MM, FP and EC	No	1.09	-	-

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Rotz et al. [62]	United States	Grass-based system	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.67	-	-
Bartl et al. [31]	Peru	Coastal system	Mixed	LCA approach	EF, MM, FP and EC	Yes	3.00	3.16	1.0
Flysjö et al. [20]	New Zealand	Average New Zealand dairy system	Mixed	LCA approach consistent with ISA 14040- 14044 standards	EF, MM, FP and EC	No	**0.97**	-	-
Gollnow et al. [63]	Australia	Average Australian dairy system	Mixed	IDF (International Dairy Federation) Guidelines	EF, MM, FP and EC	Yes	1.10	1.11	-
Christie et al. [64]	Tasmania	Average Tasmanian dairy system	Mixed	Simapro LCA software	EF, MM, FP and EC	No	**1.00**[5]	-	-
GLOBIOM	Europe	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.68**	0.69	3.9
Herrero et al. [7]	EU Baltic	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.57**		
Herrero et al. [7]	EU CentralEast	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.58**		
Herrero et al. [7]	EU MidWest	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.48**		
Herrero et al. [7]	EU North	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.47**		
Herrero et al. [7]	EU South	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.67**		
Herrero et al. [7]	Former USSR	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.87**		
Herrero et al. [7]	RCEU	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.00**		
Herrero et al. [7]	ROWE	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.49**		
GLOBIOM	Oceania	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.70**	0.82	4.7
Herrero et al. [7]	ANZ	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.70**		
Herrero et al. [7]	Pacific Islands	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.82**		
GLOBIOM	North America	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.51**	0.51	3.0
Herrero et al. [7]	Canada	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.51**		
Herrero et al. [7]	USA	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.51**		

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
GLOBIOM	Latin America and the Caribbean	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.40**	2.25	10.3
Herrero et al. [7]	Brazil	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.46**		
Herrero et al. [7]	Mexico	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.58**		
Herrero et al. [7]	RCAM	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.31**		
Herrero et al. [7]	RSAM	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.47**		
GLOBIOM	East Asia	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.26**	1.33	1.2
Herrero et al. [7]	China	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.33**		
Herrero et al. [7]	Japan	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.50**		
Herrero et al. [7]	South Korea	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.44**		
GLOBIOM	South East Asia	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.21**	1.65	2.5
Herrero et al. [7]	RSEA OPA	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.12**		
Herrero et al. [7]	RSEA PAC	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**1.45**		
GLOBIOM	South Asia	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.83**	0.88	2.3
Herrero et al. [7]	India	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.86**		
Herrero et al. [7]	RSAS	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.76**		
GLOBIOM	Middle-East and North Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.77**	0.93	4.0
Herrero et al. [7]	MENA	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.73**		
Herrero et al. [7]	Turkey	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.84**		
GLOBIOM	Sub-Saharan Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**2.89**	5.65	68.9
Herrero et al. [7]	Congo Basin	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**2.46**		
Herrero et al. [7]	Eastern Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**2.67**		
Herrero et al. [7]	Western Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**4.22**		

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
Herrero et al. [7]	Southern Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**2.41**		
Herrero et al. [7]	South Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.70**		
GLOBIOM	World	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**0.80**	0.96	4.2
Bell et al. [37]	Scotland	Non- grazing low forage control	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.14	-	0.7
Bell et al. [37]	Scotland	Non- grazing low forage select	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	1.05	-	0.6
Thomassen et al. [65]	Netherlands	Conventional dairy system	Industrial	LCA approach	EF, MM, FP and EC	No	1.36	-	1.3[12]
O'Brien et al. [24]	Ireland	Confinement dairy system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.91	1.03	0.9[7]
O'Brien et al. [44]	United Kingdom	High performance confinement dairy system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	Yes	**0.83**[1]	0.88	-
Wiliams et al. [42]	United Kingdom	High maize-feed dairy system	Industrial	LCA approach	EF, MM, FP and EC	No	0.98	-	1.2
Belflower et al. [33]	United States	Confinement dairy system	Industrial	IFSM (Integrated Farm System Model)	EF, MM, FP and EC	Yes	0.53	0.53[4]	-
O'Brien et al. [44]	United States	Top performance confinement dairy system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	Yes	**0.90**[1]	0.90	-
Rotz et al. [62]	United States	Dry lot system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.50	-	-
Rotz et al. [62]	United States	Dry lot system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.51	-	-
Rotz et al. [62]	United States	Confinement dairy system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.74	-	-
Rotz et al. [62]	United States	Confinement dairy system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.57	-	-
Rotz et al. [62]	United States	Confinement dairy system	Industrial	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	0.61	-	-
Arsenault et al. [60]	Canada	Confinement system	Industrial	LCA approach	EF, MM, FP and EC	No	0.99	-	2.6

TABLE 3-6 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg FPCM)	GHG emissions with LUC (kg CO_2-eq / kg FPCM)	Land occupation (m^2 / kg FPCM)
McGeough et al. [66]	Canada	Confinement dairy system	Industrial	LCA approach using the HOLOS model	EF, MM, FP and EC	No	**0.92**	-	-
GLOBIOM	Europe	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**0.78**	0.79	6.7
GLOBIOM	Oceania	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**0.86**	1.04	15.6
GLOBIOM	North America	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**0.51**	0.51	2.6
GLOBIOM	Latin America and the Caribbean	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**1.85**	3.17	20.4
GLOBIOM	East Asia	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**1.17**	1.30	1.7
GLOBIOM	South East Asia	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**1.96**	3.26	18.9
GLOBIOM	South Asia	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**1.03**	1.09	4.3
GLOBIOM	Middle-East and North Africa	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**1.11**	1.37	24.4
GLOBIOM	Sub-Saharan Africa	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**3.07**	6.84	99.5
GLOBIOM	World	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**0.97**	1.22	6.2

*The final values could not be derived from the original publication and were derived from Crosson et al. [67].
When emission results are presented in bold, more detailed information about the distribution among emission sources is available (see Appendix A3.3, Table 3-8).
[a] EF, enteric fermentation; MM, manure management; FP, feed production; EC, energy consumption
[1] The GHG emissions include carbon sequestration.
[2] In Herrero et al. [7], pasture-based milk production is not important in the regions EU Baltic, EU CentralEast, EU MidWest, EU North, EU South, Canada, USA, Mexico and South Korea. Therefore, no results are provided for these regions.
[3] GHG emissions are based on an analysis of 30 different dairy farms in Norway [32]. Not enough information is provided about the variation in the dairy production systems, so based on the provided information mixed is chosen as the production system.
[4] The LUC-related emissions involve carbon sequestration [33].
[5] Christie et al [64] calculated the GHG emissions of 60 different Tasmanian dairy farms. In this table, the mean GHG emissions from these farms is presented.

A3.2 Literature review of beef production systems

TABLE 3-7 | Case studies on beef production systems, their characteristics, total GHG emissions and land occupation.

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Pelletier et al. [23]	United States	Pasture- based beef system	Pasture- based	ISO-compliant LCA approach using SimaPro 7.1 LCA software	EF, MM and FP[1]	No	30.7	-	-
Cederberg et al. [21]	Brazil	Typical Brazilian beef system	Pasture- based	LCA approach	EF, MM, FP and EC	No	28.2	-	175.0
Dick et al. [18]	Brazil	Extensive beef system	Pasture- based	LCA approach consistent with ISA 14040- 14044 standards	EF and MM. Only direct N_2O represents FP	Yes	36.0	41.5	234.8
Dick et al. [18]	Brazil	Improved beef system	Pasture- based	LCA approach consistent with ISA 14040- 14044 standards	EF and MM. Only direct N_2O represents FP	Yes	15.4	18.5	21.0
Ruviaro et al. [17]	Brazil	Natural grass system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	68.2	-	-
Ruviaro et al. [17]	Brazil	Improved natural grass system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	32.3	-	-
Ruviaro et al. [17]	Brazil	Cultivated ryegrass and sorghum system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	32.0	-	-
Ruviaro et al. [17]	Brazil	Improved natural grass/ sorghum system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	37.4	-	-
Ruviaro et al. [17]	Brazil	Natural grass supplemented with protein mineralized salt system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	53.3	-	-

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Ruviaro et al. [17]	Brazil	Natural grass supplemented with protein- energy mineralized salt system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	37.4	-	-
Ruviaro et al. [17]	Brazil	Natural grass/ ryegrass system	Pasture- based	LCA approach with default data provided by IPCC	EF, MM, FP and EC	No	47.4	-	-
Williams et al. [42]	United Kingdom	Organic beef system	Pasture- based	LCA approach	EF, MM, FP and EC	No	18.2	-	42.1
Williams et al. [42]	United Kingdom	100% suckler beef system	Pasture- based	LCA approach	EF, MM, FP and EC	No	25.3	-	38.5
Ridoutt et al. [46]	Australia	Japanese ox grass- fed steers system	Pasture- based	LCA approach (PAS 2050)	EF, MM, FP and EC	No	16.3	-	275.2
Ridoutt et al. [46]	Australia	EU cattle system	Pasture- based	LCA approach (PAS 2050)	EF, MM, FP and EC	No	17.3	-	160.0
Ridoutt et al. [46]	Australia	Yearling in Gundagai system	Pasture- based	LCA approach (PAS 2050)	EF, MM, FP and EC	No	16.6	-	137.6
Ridoutt et al. [46]	Australia	Yearling in Bathurst system	Pasture- based	LCA approach (PAS 2050)	EF, MM, FP and EC	No	17.0	-	140.8
GLOBIOM	Europe	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**31.9**	31.9	79.2
Herrero et al. [7][2]	EU Baltic	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**17.0**		
Herrero et al. [7][2]	EU CentralEast	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**28.6**		
Herrero et al. [7][2]	EU MidWest	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**37.2**		
Herrero et al. [7][2]	EU North	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**33.8**		
Herrero et al. [7][2]	EU South	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**25.6**		
Herrero et al. [7][2]	Former USSR	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**25.0**		
Herrero et al. [7][2]	RCEU	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**30.0**		
Herrero et al. [7][2]	ROWE	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**24.2**		
GLOBIOM	Oceania	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**33.3**	33.9	623.4
Herrero et al. [7][2]	ANZ	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**33.1**		

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Herrero et al. [7][2]	Pacific Islands	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**135.0**		
GLOBIOM	North America	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**37.8**	37.8	225.8
Herrero et al. [7][2]	Canada	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**64.3**		
Herrero et al. [7][2]	USA	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**36.1**		
GLOBIOM	Latin America and the Caribbean	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**48.6**	61.9	416.2
Herrero et al. [7][2]	Brazil	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**45.9**		
Herrero et al. [7][2]	Mexico	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**67.7**		
Herrero et al. [7][2]	RCAM	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**109.5**		
Herrero et al. [7][2]	RSAM	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**47.3**		
GLOBIOM	East Asia	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**296.6**	337.7	431.2
Herrero et al. [7][2]	China	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**297.4**		
Herrero et al. [7][2]	Japan	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**28.1**		
GLOBIOM	South East Asia	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**106.9**	144.4	733.0
Herrero et al. [7][2]	RSEA OPA	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**73.8**		
Herrero et al. [7][2]	RSEA PAC	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**132.9**		
GLOBIOM	South Asia	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**1819.4**	1975.7	11908.6
Herrero et al. [7][2]	India	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**1766.5**		
Herrero et al. [7][2]	RSAS	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**1849.0**		
GLOBIOM	Middle-East and North Africa	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**187.0**	245.2	1628.7
Herrero et al. [7][2]	MENA	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**54.5**		
Herrero et al. [7][2]	Turkey	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**1530.7**		
GLOBIOM	Sub-Saharan Africa	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**439.8**	998.8	16897.1

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Herrero et al. [7][2]	Congo Basin	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**62.0**		
Herrero et al. [7][2]	Eastern Africa	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**642.2**		
Herrero et al. [7][2]	Western Africa	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**212.3**		
Herrero et al. [7][2]	Southern Africa	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**561.8**		
Herrero et al. [7][2]	South Africa	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**33.7**		
GLOBIOM	World	-	Pasture- based	GLOBIOM model	EF, MM and FP	Yes	**33.8**	36.3	639.4
Basarab et al. [43]	Canada	Calf-fed production system (harvested after 11- 14 months)	Mixed	ISO-compliant LCA	EF, MM, FP and EC	Yes[3]	21.1	18.8[3]	-
Basarab et al. [43]	Canada	Calf- fed with growth implants (harvested after 11- 14 months)	Mixed	ISO-compliant LCA	EF, MM, FP and EC	Yes[3]	19.9	17.7[3]	-
Basarab et al. [43]	Canada	Year- fed production system (harvested after 19- 23 months)	Mixed	ISO-compliant LCA	EF, MM, FP and EC	Yes[3]	22.5	18.9[3]	-
Basarab et al. [43]	Canada	Year- fed with growth implants (harvested after 19- 23 months)	Mixed	ISO-compliant LCA	EF, MM, FP and EC	Yes[3]	21.2	17.9[3]	-
Beauchemin et al. [68]	Canada	Various beef production systems	Mixed	LCA approach	EF, MM, FP and EC	**No**	**21.7**	-	-
Phetteplace et al. [61]	United States	Cow- calf to feedlot	Mixed	Whole-farm approach using equations and emissions factors from IPCC	EF, MM, FP and EC	No	9.9	-	-
Pelletier et al. [23]	United States	Feedlot and back-ground beef system	Mixed	ISO-compliant LCA approach using SimaPro 7.1 LCA software	EF, MM and FP and EC	No	25.9	-	-

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Bonesmo et al. [32][4]	Norway	Culled cows and heifers beef system	Mixed	HolosNor model	EF, MM, FP and EC	Yes	**19.1**	21.7	-
Bonesmo et al. [32][4]	Norway	Finishing young bulls beef system	Mixed	HolosNor model	EF, MM, FP and EC	Yes	**14.6**	17.3	-
Cederberg and Stadig [55]	Sweden	Typical Swedish cow-calf finishing system	Mixed	LCA approach consistent with ISA 14041 standard	Direct on- farm, purchased inputs and indirect N_2O emissions.	No	16.7*	-	58.5
Casey and Holden [69]	Ireland	Typical Irish Suckle beef system	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM, FP and EC	No	18.0	-	-
Foley et al. [70]	Ireland	National Farm Survey Ireland	Mixed	BEEFGEM model	EF, MM, FP and EC	No	22.8	-	-
Foley et al. [70]	Ireland	Production system finishing males as bulls with moderate stocking rate	Mixed	BEEFGEM model	EF, MM, FP and EC	No	**19.4**	-	-
Foley et al. [70]	Ireland	Production system finishing males as bulls with intensive stocking rate	Mixed	BEEFGEM model	EF, MM, FP and EC	No	**21.7**	-	-
Foley et al. [70]	Ireland	Production system finishing males as steers with moderate stocking rate	Mixed	BEEFGEM model	EF, MM, FP and EC	No	**18.7**	-	-
Foley et al. [70]	Ireland	Production system finishing males as steers with intensive stocking rate	Mixed	BEEFGEM model	EF, MM, FP and EC	No	**20.1**	-	-
Williams et al. [42]	United Kingdom	Non- organic system	Mixed	LCA approach	EF, MM, FP and EC	No	15.8	-	42.1

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Williams et al. [42]	United Kingdom	Lowland beef system[2]	Mixed	LCA approach	EF, MM, FP and EC	No	15.6	-	38.5
Williams et al. [42]	United Kingdom	Hill and upland beef system[2]	Mixed	LCA approach	EF, MM, FP and EC	No	16.4	-	24.1
Nguyen et al. [34]	France	Standard suckle cow- calf herd with finishing heifers and bull- fattening herd using diet rich based on maize silage	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM and FP	Yes[5]	27.8	25.5[5]	-
Nguyen et al. [34]	France	Suckle cow- calf herd with finishing heifers enriched in omega-3 FA through pasture and grass silage and bull- fattening herd using diet rich based on maize silage and supplemented with linseeds	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM and FP	Yes[5]	27.7	25.5[5]	-
Nguyen et al. [34]	France	Standard suckle co- calf herd with finishing heifers and bull- fattening herd using a fibre- based concentrate diet	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM and FP	Yes[5]	27.9	25.3[5]	-
Nguyen et al. [34]	France	Suckle cow- calf herd with finishing heifers enriched in omega-3 FA through pasture and grass silage and bull- fattening herd using a starch-based concentrate supplemented with linseeds	Mixed	Partial LCA approach (cradle-to-gate)	EF, MM and FP	Yes[5]	27.0	24.4[5]	-

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Nguyen et al. [71]	Europe	Suckle cow- calf beef system	Mixed	LCA approach	EF, MM, FP and EC	Yes	27.3	27.4	42.9
Ridoutt et al. [46]	Australia	Inland weaners, grass fattened and feedlot finished system	Mixed	LCA approach (PAS 2050)	EF, MM, FP and EC	No	16.2	-	140.8
Ridoutt et al. [46]	Australia	North coast weaners, grass fattened and feedlot finished system	Mixed	LCA approach (PAS 2050)	EF, MM, FP and EC	No	20.3	-	204.8
White et al. [72]	New Zealand	Lowland pasture- based beef system[9]	Mixed	Farmax Pro tool from the model Stockpol	EF, MM, FP and EC	No	26.0	-	-
White et al. [72]	New Zealand	Upland pasture- based beef system[9]	Mixed	Farmax Pro tool from the model Stockpol	EF, MM, FP and EC	No	34.0	-	-
GLOBIOM	Europe	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**14.8**	14.9	28.9
Herrero et al. [7]	EU Baltic	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**13.7**		
Herrero et al. [7]	EU CentralEast	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**13.3**		
Herrero et al. [7]	EU MidWest	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**14.4**		
Herrero et al. [7]	EU North	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**12.9**		
Herrero et al. [7]	EU South	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**16.7**		
Herrero et al. [7]	Former USSR	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**15.7**		
Herrero et al. [7]	RCEU	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**15.0**		
Herrero et al. [7]	ROWE	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**12.4**		
GLOBIOM	Oceania	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**32.7**	33.3	115.8
Herrero et al. [7]	ANZ	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**31.9**		
Herrero et al. [7]	Pacific Islands	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**100.3**		
GLOBIOM	North America	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**14.4**	14.4	28.6
Herrero et al. [7]	Canada	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**17.1**		

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Herrero et al. [7]	USA	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**14.0**		
GLOBIOM	Latin America and the Caribbean	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**41.1**	51.5	156.2
Herrero et al. [7]	Brazil	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**41.3**		
Herrero et al. [7]	Mexico	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**50.8**		
Herrero et al. [7]	RCAM	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**50.4**		
Herrero et al. [7]	RSAM	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**36.8**		
GLOBIOM	East Asia	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**75.9**	80.5	52.1
Herrero et al. [7]	China	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**77.5**		
Herrero et al. [7]	Japan	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**17.2**		
Herrero et al. [7]	South Korea	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**16.2**		
GLOBIOM	South East Asia	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**72.7**	93.8	51.1
Herrero et al. [7]	RSEA OPA	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**72.2**		
Herrero et al. [7]	RSEA PAC	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**73.9**		
GLOBIOM	South Asia	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**407.2**	423.7	376.8
Herrero et al. [7]	India	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**438.7**		
Herrero et al. [7]	RSAS	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**286.1**		
GLOBIOM	Middle-East and North Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**22.7**	25.7	159.5
Herrero et al. [7]	MENA	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**19.5**		
Herrero et al. [7]	Turkey	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**29.3**		
GLOBIOM	Sub-Saharan Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**317.5**	594.4	9249.4
Herrero et al. [7]	Congo Basin	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**40.0**		

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
Herrero et al. [7]	Eastern Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**452.6**		
Herrero et al. [7]	Western Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**84.5**		
Herrero et al. [7]	Southern Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**444.4**		
Herrero et al. [7]	South Africa	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**30.8**		
GLOBIOM	World	-	Mixed	GLOBIOM model	EF, MM and FP	Yes	**19.7**	21.1	200.7
Pelletier et al. [23]	United States	Feedlot beef system	Industrial	ISO-compliant LCA approach using SimaPro 7.1 LCA software	EF, MM, FP and EC	No	23.7	-	-
Ogino et al. [73]	Japan	Feedlot (beef fattening) system	Industrial	LCA approach consistent with ISA 14040 standards	EF, MM, FP and EC	No	22.6*	-	-
GLOBIOM	Europe	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**23.9**	24.0	63.7
GLOBIOM	Oceania	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**35.5**	36.3	376.4
GLOBIOM	North America	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**25.8**	25.8	86.4
GLOBIOM	Latin America and the Caribbean	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**53.6**	67.5	255.7
GLOBIOM	East Asia	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**72.6**	77.9	59.4
GLOBIOM	South East Asia	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**87.9**	146.3	203.9
GLOBIOM	South Asia	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**378.7**	396.1	592.2

TABLE 3-7 | Continued

Reference	Region	Production system as defined in reference	Production system as used in present study	Method GHG emissions	Main GHG sources included[a]	LUC included	GHG emissions without LUC (kg CO_2-eq / kg CW)	GHG emissions with LUC (kg CO_2-eq / kg CW)	Land occupation (m^2 / kg CW)
GLOBIOM	Middle-East and North Africa	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**71.0**	90.0	590.9
GLOBIOM	Sub-Saharan Africa	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**335.7**	689.2	11470.6
GLOBIOM	World	-	Average per region	GLOBIOM model	EF, MM and FP	Yes	**53.8**	61.6	177.7

* The final values could not be derived from the original publication and were derived from Crosson et al. [67].
When emission results are presented in bold, more detailed information about the distribution among emission sources is available (see Appendix A3.4, Table 3-9).
[a] EF, enteric fermentation; MM, manure management; FP, feed production; EC, energy consumption.
[1] In Pelletier et al. [23], energy use in the pasture-based system is negligible as 94% percent of cumulative energy use is allocated to feed production.
[2] In Herrero et al. [7], pasture-based beef production is not important in South Korea. Therefore, no results are provided for this region.
[3] In Basarab et al. [43], the LUC-related emissions include SOC sequestration due to land rotation between cropland and grassland.
[4] Beef is produced from dairy herds [32]. The GHG emissions are based on an analysis of 18 different farms in Norway [32]. Not enough information is provided about the variation in the dairy production systems, so based on the provided information mixed is chosen as the production system.
[5] Nguyen et al. [34] present the combined GHG emissions from LULUC, which includes SOC sequestration in permanent grassland, land rotation between grass and crops (carbon sequestration) and conversion of Brazilian forest to soybean (carbon loss).

A3.3 Extended review of various literature reviews of dairy production systems

TABLE 3-8 | Dairy production in kg CO_2-eq per kg FPCM (fat and protein corrected milk) per year.

Literature review	Region	Dairy production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg FPCM)
Lovett et al. [28]	Ireland	Moorepark	Pasture-based	-	0.41	0.06	0.02	0.08[1]	0.10	0.01	0.15	0.03	0.86
O'Brien et al. [44]	Ireland	High performance grass-based dairy system	Pasture-based	0.001	0.43	0.04	0.03		0.28		-0.002	0.04	0.84
GLOBIOM	Europe	-	Pasture-based	0	0.82	0.15	0.06		0.14		-	-	1.16
Herrero et al. [7][2]	Former USSR	-	Pasture-based		0.86	0.16	0.06		0.14		-	-	1.21
Herrero et al. [7][2]	RCEU	-	Pasture-based		0.80	0.15	0.05		0.14		-	-	1.14
Herrero et al. [7][2]	ROWE	-	Pasture-based		0.35	0.05	0.03		0.11		-	-	0.56
GLOBIOM	Oceania	-	Pasture-based	0.09	0.57	0.03	0.07		0.19		-	-	0.94
Herrero et al. [7][2]	ANZ	-	Pasture-based		0.57	0.02	0.07		0.19		-	-	0.85
Herrero et al. [7][2]	Pacific Islands	-	Pasture-based		4.58	0.65	0.17		0.93		-	-	6.32
GLOBIOM	North America	-	Pasture-based	-	-	-	-		-		-	-	-
GLOBIOM	Latin America and the Caribbean	-	Pasture-based	2.13	1.99	0.07	0.19		0.52		-	-	4.90
Herrero et al. [7][2]	Brazil	-	Pasture-based		1.87	0.06	0.18		0.50		-	-	2.61
Herrero et al. [7][2]	RCAM	-	Pasture-based		1.18	0.03	0.11		0.29		-	-	1.62
Herrero et al. [7][2]	RSAM	-	Pasture-based		2.11	0.07	0.21		0.54		-	-	2.93

TABLE 3-8 | Continued

Literature review	Region	Dairy production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg FPCM)
GLOBIOM	East Asia	-	Pasture-based	0.85	3.00	0.06	0.16		0.68		-	-	4.76
Herrero et al. [7][2]	China	-	Pasture-based		1.24	0.02	0.13		0.33		-	-	1.73
Herrero et al. [7][2]	Japan	-	Pasture-based		0.38	0.01	0.08		0.15		-	-	0.62
GLOBIOM	South East Asia	-	Pasture-based	0.25	1.92	0.07	0.23		0.41		-	-	2.88
Herrero et al. [7][2]	RSEA OPA	-	Pasture-based		0.93	0.06	0.04		0.22		-	-	1.26
Herrero et al. [7][2]	RSEA PAC	-	Pasture-based		2.27	0.07	0.30		0.47		-	-	3.11
GLOBIOM	South Asia	-	Pasture-based	0.61	4.48	0.13	0.22		0.99		-	-	6.43
Herrero et al. [7][2]	India	-	Pasture-based		4.17	0.12	0.21		0.93		-	-	5.43
Herrero et al. [7][2]	RSAS	-	Pasture-based		4.57	0.13	0.22		1.01		-	-	5.93
GLOBIOM	Middle-East and North Africa	-	Pasture-based	0.52	1.60	0.04	0.14		0.36		-	-	2.66
Herrero et al. [7][2]	MENA	-	Pasture-based		1.30	0.03	0.12		0.31		-	-	1.76
Herrero et al. [7][2]	Turkey	-	Pasture-based		3.93	0.09	0.29		0.78		-	-	5.09
GLOBIOM	Sub-Saharan Africa	-	Pasture-based	5.97	2.83	0.08	0.24		0.58		-	-	9.70
Herrero et al. [7][2]	Congo Basin	-	Pasture-based		2.43	0.07	0.22		0.49		-	-	3.21
Herrero et al. [7][2]	Eastern Africa	-	Pasture-based		2.97	0.09	0.24		0.60		-	-	3.91
Herrero et al. [7][2]	Western Africa	-	Pasture-based		3.98	0.12	0.35		0.86		-	-	5.33
Herrero et al. [7][2]	Southern Africa	-	Pasture-based		2.08	0.06	0.16		0.40		-	-	2.71
Herrero et al. [7][2]	South Africa	-	Pasture-based		0.50	0.01	0.07		0.23		-	-	0.82

TABLE 3-8 | Continued

Literature review	Region	Dairy production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg FPCM)
GLOBIOM	World	-	Pasture-based	0.87	1.50	0.07	0.13		0.35		-	-	2.92
Flysjö et al. [20]	New Zealand	Average New Zealand dairy system	Mixed	-	0.62	0.01	0.01[3]	0.01[3]	0.22	0.03	0.05	0.02	0.97
Christie et al. [64]	Tasmania	Average Tasmanian dairy system	Mixed	-	0.55	0.01	0.07	-	0.03	0.11[4]	0.12	0.11	1.00
Lovett et al. [28]	Ireland	Kilmaley	Mixed	-	0.39	0.08	0.03	0.08[1]	0.08	0.02	0.23	0.04	0.95
Flysjö et al. [20]	Sweden	Average Sweden dairy system	Mixed	-	0.58	0.04	0.05[3]	0.02[3]	0.21	0.03	0.04	0.16[3]	1.14
Bonesmo et al. [32]	Norway	Average Norrish dairy system	Mixed	-	0.39	0.18[5]			0.21[5]		0.19[5]	0.05	1.02
GLOBIOM	Europe	-	Mixed	0.01	0.41	0.10	0.05		0.12		-	-	0.69
Herrero et al. [7]	EU Baltic	-	Mixed		0.39	0.05	0.03		0.10		-	-	0.57
Herrero et al. [7]	EU CentralEast	-	Mixed		0.39	0.04	0.04		0.10		-	-	0.58
Herrero et al. [7]	EU MidWest	-	Mixed		0.29	0.05	0.05		0.09		-	-	0.48
Herrero et al. [7]	EU North	-	Mixed		0.29	0.05	0.04		0.10		-	-	0.47
Herrero et al. [7]	EU South	-	Mixed		0.44	0.08	0.06		0.09		-	-	0.67
Herrero et al. [7]	Former USSR	-	Mixed		0.50	0.15	0.05		0.17		-	-	0.87
Herrero et al. [7]	RCEU	-	Mixed		0.59	0.21	0.05		0.15		-	-	1.00
Herrero et al. [7]	ROWE	-	Mixed		0.29	0.07	0.03		0.09		-	-	0.49
GLOBIOM	Oceania	-	Mixed	0.12	0.48	0.04	0.04		0.14		-	-	0.82

TABLE 3-8 | Continued

Literature review	Region	Dairy production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg FPCM)
Herrero et al. [7]	ANZ	-	Mixed		0.48	0.04	0.04		0.14		-	-	0.70
Herrero et al. [7]	Pacific Islands	-	Mixed		1.24	0.21	0.07		0.30		-	-	1.82
GLOBIOM	North America	-	Mixed	0	0.30	0.03	0.04		0.13		-	-	0.51
Herrero et al. [7]	Canada	-	Mixed		0.28	0.04	0.03		0.16		-	-	0.51
Herrero et al. [7]	USA	-	Mixed		0.31	0.03	0.04		0.13		-	-	0.51
GLOBIOM	Latin America and the Caribbean	-	Mixed	0.85	0.99	0.06	0.07		0.29		-	-	2.25
Herrero et al. [7]	Brazil	-	Mixed		1.03	0.06	0.07		0.30		-	-	1.46
Herrero et al. [7]	Mexico	-	Mixed		0.33	0.01	0.05		0.19		-	-	0.58
Herrero et al. [7]	RCAM	-	Mixed		0.91	0.06	0.06		0.28		-	-	1.31
Herrero et al. [7]	RSAM	-	Mixed		1.05	0.06	0.09		0.27		-	-	1.47
GLOBIOM	East Asia	-	Mixed	0.07	0.80	0.02	0.12		0.32		-	-	1.33
Herrero et al. [7]	China	-	Mixed		0.83	0.03	0.13		0.34		-	-	1.33
Herrero et al. [7]	Japan	-	Mixed		0.32	0.01	0.04		0.13		-	-	0.50
Herrero et al. [7]	South Korea	-	Mixed		0.27	0.01	0.04		0.11		-	-	0.44
GLOBIOM	South East Asia	-	Mixed	0.44	0.74	0.05	0.11		0.32		-	-	1.65
Herrero et al. [7]	RSEA OPA	-	Mixed		0.68	0.04	0.10		0.31		-	-	1.12
Herrero et al. [7]	RSEA PAC	-	Mixed		0.89	0.06	0.14		0.36		-	-	1.45

TABLE 3-8 | Continued

Literature review	Region	Dairy production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg FPCM)
GLOBIOM	South Asia	-	Mixed	0.05	0.60	0.03	0.07		0.14		-	-	0.88
Herrero et al. [7]	India	-	Mixed		0.60	0.03	0.07		0.16		-	-	0.86
Herrero et al. [7]	RSAS	-	Mixed		0.58	0.03	0.05		0.09		-	-	0.76
GLOBIOM	Middle-East and North Africa	-	Mixed	0.16	0.45	0.02	0.08		0.22		-	-	0.93
Herrero et al. [7]	MENA	-	Mixed		0.40	0.01	0.11		0.21		-	-	0.73
Herrero et al. [7]	Turkey	-	Mixed		0.53	0.04	0.04		0.24		-	-	0.84
GLOBIOM	Sub-Saharan Africa	-	Mixed	2.76	2.10	0.10	0.31		0.38		-	-	5.65
Herrero et al. [7]	Congo Basin	-	Mixed		1.78	0.08	0.25		0.35		-	-	2.46
Herrero et al. [7]	Eastern Africa	-	Mixed		1.98	0.11	0.26		0.33		-	-	2.67
Herrero et al. [7]	Western Africa	-	Mixed		3.02	0.11	0.51		0.57		-	-	4.22
Herrero et al. [7]	Southern Africa	-	Mixed		1.65	0.07	0.31		0.39		-	-	2.41
Herrero et al. [7]	South Africa	-	Mixed		0.38	0.01	0.09		0.21		-	-	0.70
GLOBIOM	World	-	Mixed	0.16	0.53	0.05	0.06		0.16		-	-	0.96
O'Brien et al. [44]	United Kingdom	High performance confinement dairy system	Industrial	0.06	0.38	0.12	0.08		0.08		0.08	0.09	0.88
O'Brien et al. [44]	United States	Top performance confinement dairy system	Industrial	0	0.37	0.12	0.15		0.08		0.07	0.10	0.90

TABLE 3-8 | Continued

Literature review	Region	Dairy production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg FPCM)
McGeough et al. [66]	Canada	Confinement dairy system	Industrial	-	0.44	0.08	0.07		0.24	0.06	0.02	0.02	0.92
GLOBIOM	Europe	-	Average per region	0.01	0.49	0.12	0.05		0.12		-	-	0.79
GLOBIOM	Oceania	-	Average per region	0.17	0.54	0.04	0.06		0.22		-	-	1.04
GLOBIOM	North America	-	Average per region	0	0.31	0.03	0.04		0.13		-	-	0.51
GLOBIOM	Latin America and the Caribbean	-	Average per region	1.32	1.32	0.07	0.10		0.36		-	-	3.17
GLOBIOM	East Asia	-	Average per region	0.13	0.78	0.02	0.12		0.25		-	-	1.30
GLOBIOM	South East Asia	-	Average per region	1.30	1.37	0.06	0.19		0.34		-	-	3.26
GLOBIOM	South Asia	-	Average per region	0.06	0.75	0.03	0.07		0.18		-	-	1.09
GLOBIOM	Middle-East and North Africa	-	Average per region	0.26	0.74	0.03	0.09		0.25		-	-	1.37
GLOBIOM	Sub-Saharan Africa	-	Average per region	3.78	2.27	0.09	0.27		0.43		-	-	6.84
GLOBIOM	World	-	Average per region	0.25	0.66	0.06	0.07		0.18		-	-	1.22

LUC, land use change; EF, enteric fermentation; MM, manure management; FP, feed production; EC, energy consumption.

[1] Lovett et al. [28] present the combined results for indirect N_2O emissions from feed production and manure management. Therefore, the total indirect N_2O emissions are presented under manure management.

[2] In Herrero et al. [7], pasture-based milk production is not important in the regions EU Baltic, EU CentralEast, EU MidWest, EU North, EU South, Canada, USA, Mexico and South Korea. Therefore, no results are provided for these regions.

[3] Flysjö et al. [20] include the emission categories 'N_2O other' and 'CO_2 other energy'. In this table, the results for 'N_2O other' are included under manure management. The results for 'CO_2 other energy' are displayed as energy consumption.

[4] Christie et al. [64] present the combined results for indirect N_2O emissions from feed production and manure management. Therefore, the total indirect N_2O emissions are presented under indirect N_2O emissions from feed production.

[5] In Bonesmo et al. [32], the CO_2 emissions from food production include carbon sequestration. Bonesmo et al. [32] present the combined results for N_2O and CH_4 emissions from manure management and for direct and indirect N_2O from feed production.

A3.4 Extended review of various literature reviews of beef production systems

TABLE 3-9 | Beef production in kg CO_2-eq per kg CW (carcass weight) per year.

Literature review	Region	Beef production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg CW)
GLOBIOM	Europe	-	Pasture-based	0.05	22.78	2.45	1.65		5.03		-	-	31.96
Herrero et al. [7][1]	EU Baltic	-	Pasture-based		11.51	0.94	1.44		3.16		-	-	17.0
Herrero et al. [7][1]	EU CentralEast	-	Pasture-based		21.98	1.72	2.14		2.79		-	-	28.6
Herrero et al. [7][1]	EU MidWest	-	Pasture-based		27.35	2.08	2.11		5.68		-	-	37.2
Herrero et al. [7][1]	EU North	-	Pasture-based		23.52	3.48	1.74		5.09		-	-	33.8
Herrero et al. [7][1]	EU South	-	Pasture-based		18.42	1.17	1.11		4.87		-	-	25.6
Herrero et al. [7][1]	Former USSR	-	Pasture-based		18.37	1.24	1.09		4.32		-	-	25.0
Herrero et al. [7][1]	RCEU	-	Pasture-based		20.74	1.18	1.47		6.57		-	-	30.0
Herrero et al. [7][1]	ROWE	-	Pasture-based		16.18	1.24	1.48		5.28		-	-	24.2
GLOBIOM	Oceania	-	Pasture-based	0.66	21.72	0.82	1.84		8.91		-	-	33.94
Herrero et al. [7][1]	ANZ	-	Pasture-based		21.60	0.81	1.83		8.88		-	-	33.1
Herrero et al. [7][1]	Pacific Islands	-	Pasture-based		95.60	4.99	7.38		26.99		-	-	135.0
GLOBIOM	North America	-	Pasture-based	0.01	27.30	1.21	2.13		7.20		-	-	37.86
Herrero et al. [7][1]	Canada	-	Pasture-based		46.30	2.36	5.54		10.13		-	-	64.3
Herrero et al. [7][1]	USA	-	Pasture-based		26.06	1.14	1.91		7.01		-	-	36.1
GLOBIOM	Latin America and the Caribbean	-	Pasture-based	13.36	34.22	1.09	2.31		10.94		-	-	61.93
Herrero et al. [7][1]	Brazil	-	Pasture-based		32.91	1.01	2.05		9.92		-	-	45.9
Herrero et al. [7][1]	Mexico	-	Pasture-based		48.45	1.59	3.12		14.56		-	-	67.7
Herrero et al. [7][1]	RCAM	-	Pasture-based		77.95	2.56	5.07		23.93		-	-	109.5
Herrero et al. [7][1]	RSAM	-	Pasture-based		32.73	1.06	2.38		11.10		-	-	47.3
GLOBIOM	East Asia	-	Pasture-based	41.07	222.29	4.23	13.42		56.67		-	-	337.68
Herrero et al. [7][1]	China	-	Pasture-based		222.91	4.24	13.45		56.82		-	-	297.4
Herrero et al. [7][1]	Japan	-	Pasture-based		17.72	0.38	2.84		7.17		-	-	28.1
GLOBIOM	South East Asia	-	Pasture-based	37.42	77.13	2.36	6.76		20.71		-	-	144.39
Herrero et al. [7][1]	RSEA OPA	-	Pasture-based		53.79	1.59	3.41		15.04		-	-	73.8

TABLE 3-9 | Continued

Literature review	Region	Beef production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg CW)
Herrero et al. [7][1]	RSEA PAC	-	Pasture-based		95.37	2.96	9.38		25.15		-	-	132.9
GLOBIOM	South Asia	-	Pasture-based	156.30	1372.68	38.74	72.90		335.10		-	-	1975.72
Herrero et al. [7][1]	India	-	Pasture-based		1332.55	37.61	70.84		325.49		-	-	1766.5
Herrero et al. [7][1]	RSAS	-	Pasture-based		1395.10	39.38	74.05		340.47		-	-	1849.0
GLOBIOM	Middle-East and North Africa	-	Pasture-based	58.22	140.15	2.98	7.21		36.68		-	-	245.24
Herrero et al. [7][1]	MENA	-	Pasture-based		39.06	1.10	2.37		12.01		-	-	54.5
Herrero et al. [7][1]	Turkey	-	Pasture-based		1165.45	22.06	56.38		286.82		-	-	1530.7
GLOBIOM	Sub-Saharan Africa	-	Pasture-based	559.02	331.11	9.41	17.44		81.83		-	-	998.81
Herrero et al. [7][1]	Congo Basin	-	Pasture-based		44.48	1.39	3.61		12.54		-	-	62.0
Herrero et al. [7][1]	Eastern Africa	-	Pasture-based		484.87	13.68	25.04		118.64		-	-	642.2
Herrero et al. [7][1]	Western Africa	-	Pasture-based		156.49	4.72	9.12		41.94		-	-	212.3
Herrero et al. [7][1]	Southern Africa	-	Pasture-based		423.97	11.97	21.97		103.88		-	-	561.8
Herrero et al. [7][1]	South Africa	-	Pasture-based		24.48	0.71	1.66		6.84		-	-	33.7
GLOBIOM	World	-	Pasture-based	2.45	23.81	1.39	2.51		6.14		-	-	36.30
Beauchemin et al. [68]	Canada	Various beef production systems: Cow- calf herd, background- and feedlot system	Mixed	0	13.74	1.09	5.00[2]		0.83[2]		1.11		21.77
Foley et al. [70]	Ireland	National Farm Survey Ireland	Mixed	-	11.34	2.10[3]	-[3]		4.83	1.34[3]	2.50	0.72[3]	22.83
Foley et al. [70]	Ireland	Production system finishing males as bulls with moderate stocking rate	Mixed	-	10.17	1.85[3]	-[3]		3.94	1.10[3]	1.89	0.49[3]	19.44

TABLE 3-9 | Continued

Literature review	Region	Beef production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg CW)
Foley et al. [70]	Ireland	Production system finishing males as bulls with intensive stocking rate	Mixed	-	10.08	1.32[3]	-[3]		5.24	1.43[3]	3.28	0.40[3]	21.75
Foley et al. [70]	Ireland	Production system finishing males as steers with moderate stocking rate	Mixed	-	9.98	1.68[3]	-[3]		3.50	0.98[3]	2.04	0.47[3]	18.65
Foley et al. [70]	Ireland	Production system finishing males as steers with intensive stocking rate	Mixed	-	9.87	1.14[3]	-[3]		4.41	1.27[3]	3.06	0.37[3]	20.12
Bonesmo et al. [32]	Norway	Culled cows and heifers	Mixed	2.59	8.34	3.89[4]			4.37[4]		1.39	1.09	21.67
Bonesmo et al. [32]	Norway	Finishing young bulls	Mixed	2.63	6.84	2.98[4]			3.08[4]		0.97	0.75	17.25
GLOBIOM	Europe	-	Mixed	0.07	8.81	2.37	0.96		2.65		-	-	14.86
Herrero et al. [7]	EU Baltic	-	Mixed		8.22	1.69	1.05		2.71		-	-	13.7
Herrero et al. [7]	EU CentralEast	-	Mixed		8.21	1.79	1.25		2.07		-	-	13.3
Herrero et al. [7]	EU MidWest	-	Mixed		8.75	2.55	0.98		2.14		-	-	14.4
Herrero et al. [7]	EU North	-	Mixed		7.67	1.83	0.85		2.53		-	-	12.9
Herrero et al. [7]	EU South	-	Mixed		10.72	2.29	1.19		2.54		-	-	16.7
Herrero et al. [7]	Former USSR	-	Mixed		8.85	2.54	0.82		3.47		-	-	15.7
Herrero et al. [7]	RCEU	-	Mixed		8.44	2.42	0.76		3.35		-	-	15.0
Herrero et al. [7]	ROWE	-	Mixed		6.33	2.29	0.68		3.13		-	-	12.4
GLOBIOM	Oceania	-	Mixed	0.56	21.24	1.21	2.37		7.89		-	-	33.27
Herrero et al. [7]	ANZ	-	Mixed		20.67	1.12	2.36		7.75		-	-	31.9

TABLE 3-9 | Continued

Literature review	Region	Beef production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg CW)
Herrero et al. [7]	Pacific Islands	-	Mixed		68.62	9.56	2.98		19.10		-	-	100.3
GLOBIOM	North America	-	Mixed	0	8.44	0.42	1.69		3.81		-	-	14.37
Herrero et al. [7]	Canada	-	Mixed		11.43	0.56	1.72		3.40		-	-	17.1
Herrero et al. [7]	USA	-	Mixed		8.00	0.40	1.68		3.87		-	-	14.0
GLOBIOM	Latin America and the Caribbean	-	Mixed	10.38	27.95	0.96	2.51		9.70		-	-	51.49
Herrero et al. [7]	Brazil	-	Mixed		28.21	0.95	2.35		9.80		-	-	41.3
Herrero et al. [7]	Mexico	-	Mixed		32.17	1.12	4.74		12.81		-	-	50.8
Herrero et al. [7]	RCAM	-	Mixed		34.97	1.24	2.84		11.39		-	-	50.4
Herrero et al. [7]	RSAM	-	Mixed		25.13	0.88	2.28		8.49		-	-	36.8
GLOBIOM	East Asia	-	Mixed	4.55	47.01	1.82	6.24		20.86		-	-	80.49
Herrero et al. [7]	China	-	Mixed		47.93	1.85	6.37		21.29		-	-	77.5
Herrero et al. [7]	Japan	-	Mixed		10.99	0.57	1.15		4.48		-	-	17.2
Herrero et al. [7]	South Korea	-	Mixed		10.90	0.51	1.23		3.59		-	-	16.2
GLOBIOM	South East Asia	-	Mixed	21.17	44.28	3.71	3.77		20.91		-	-	93.83
Herrero et al. [7]	RSEA OPA	-	Mixed		43.96	3.63	3.79		20.78		-	-	72.2
Herrero et al. [7]	RSEA PAC	-	Mixed		45.06	3.90	3.71		21.22		-	-	73.9
GLOBIOM	South Asia	-	Mixed	16.52	243.80	9.90	55.49		98.03		-	-	423.74
Herrero et al. [7]	India	-	Mixed		256.51	8.72	66.05		107.40		-	-	438.7
Herrero et al. [7]	RSAS	-	Mixed		194.85	14.46	14.81		61.94		-	-	286.1
GLOBIOM	Middle-East and North Africa	-	Mixed	2.99	14.43	0.68	2.17		5.46		-	-	25.73
Herrero et al. [7]	MENA	-	Mixed		12.37	0.53	1.97		4.62		-	-	19.5
Herrero et al. [7]	Turkey	-	Mixed		18.56	0.99	2.56		7.15		-	-	29.3
GLOBIOM	Sub-Saharan Africa	-	Mixed	276.89	193.61	5.77	45.41		72.73		-	-	594.42
Herrero et al. [7]	Congo Basin	-	Mixed		26.03	0.92	4.85		8.19		-	-	40.0
Herrero et al. [7]	Eastern Africa	-	Mixed		256.89	7.58	72.44		115.72		-	-	452.6
Herrero et al. [7]	Western Africa-	-	Mixed		53.34	1.87	11.08		18.16		-	-	84.5

TABLE 3-9 | Continued

Literature review	Region	Beef production system as in reference	Production system as in present study	LUC (CO_2)	EF (CH_4)	MM (CH_4)	MM (direct N_2O)	MM (indirect N_2O)	FP (direct N_2O)	FP (indirect N_2O)	FP (CO_2)	EC (CO_2)	Total GHG emissions (kg CO_2-eq per kg CW)
Herrero et al. [7]	Southern Africa	-	Mixed		331.33	9.53	39.91		63.63		-	-	444.4
Herrero et al. [7]	South Africa	-	Mixed		17.60	0.76	4.39		7.99		-	-	30.8
GLOBIOM	World	-	Mixed	1.40	11.87	1.56	1.85		4.46		-	-	21.15
GLOBIOM	Europe	-	Average per region	0.07	15.61	3.15	1.43		3.76		-	-	24.03
GLOBIOM	Oceania	-	Average per region	0.79	23.68	1.10	2.05		8.70		-	-	36.33
GLOBIOM	North America	-	Average per region	0	17.54	0.78	2.15		5.33		-	-	25.80
GLOBIOM	Latin America and the Caribbean	-	Average per region	13.97	37.11	1.21	2.97		12.29		-	-	67.55
GLOBIOM	East Asia	-	Average per region	5.34	47.58	1.61	5.46		17.91		-	-	77.91
GLOBIOM	South East Asia	-	Average per region	58.34	58.05	3.94	5.34		20.64		-	-	146.30
GLOBIOM	South Asia	-	Average per region	17.45	235.69	9.40	46.17		87.41		-	-	396.12
GLOBIOM	Middle-East and North Africa	-	Average per region	19.04	51.25	1.60	3.58		14.58		-	-	90.04
GLOBIOM	Sub-Saharan Africa	-	Average per region	353.57	226.49	6.62	32.29		70.27		-	-	689.23
GLOBIOM	World	-	Average per region	8.38	35.12	1.84	4.46		11.76		-	-	61.56

LUC, land use change; EF, enteric fermentation; MM, manure management; FP, feed production; EC, energy consumption.

[1] In Herrero et al. [7], pasture-based milk production is not important in South Korea. Therefore, no results are provided for this region.

[2] Beauchemin et al. [68] only present the combined results for direct or indirect N_2O emissions from MM and FP.

[3] Foley et al. [70] only present the combined results for CH_4 and direct N_2O emissions from manure management and for indirect N_2O emissions from manure management and feed production. The total indirect N_2O emissions are shown under FP. The CO_2 emissions from electricity production and diesel production are presented under EC.

[4] In Bonesmo et al. [32], the CO_2 emissions from food production include carbon sequestration. Bonesmo et al. [32] present the combined results for N_2O and CH_4 emissions from manure management and for direct and indirect N_2O from feed production.

A3.5 Land use change emissions per region

TABLE 3-10 | Simulated Land use change emissions from dairy and beef production per region using the GLOBIOM model. LUC includes deforestation and natural land conversion.

		Dairy			Beef		
		Deforestation	**Natural land conversion**	**Total LUC**	**Deforestation**	**Natural land conversion**	**Total LUC**
		(kg CO_2-eq/ kg FPCM)	**(kg CO_2-eq/ kg FPCM)**	**(kg CO_2-eq/ kg FPCM)**	**(kg CO_2-eq/ kg FPCM)**	**(kg CO_2-eq/ kg FPCM)**	**(kg CO_2-eq/ kg FPCM)**
EUR	PB[a]	0.00	0.00	0.00	0.04	0.01	0.05
	MI	0.01	0.01	0.01	0.06	0.00	0.07
OCE	PB	0.00	0.09	0.09	0.00	0.66	0.66
	MI	0.00	0.12	0.13	0.00	0.56	0.56
NAM	PB[a]				0.01	0.00	0.01
	MI	0.00	0.00	0.00	0.00	0.00	0.00
LAM	PB	2.04	0.09	2.13	12.70	0.66	13.36
	MI	0.82	0.03	0.84	9.94	0.43	10.38
EAS	PB	0.00	0.85	0.86	0.02	41.05	41.07
	MI	0.00	0.07	0.07	0.05	4.50	4.55
SEA	PB	0.18	0.07	0.25	25.51	11.91	37.42
	MI	0.33	0.11	0.44	16.04	5.13	21.17
SAS	PB	0.24	0.37	0.61	43.09	113.21	156.30
	MI	0.01	0.04	0.05	1.20	15.32	16.52
MNA	PB	0.00	0.52	0.52	0.09	58.13	58.22
	MI	0.01	0.15	0.16	0.17	2.82	2.98
SSA	PB	3.57	2.40	5.96	281.92	277.10	559.02
	MI	1.54	1.22	2.75	124.60	152.29	276.89
WORLD	PB	0.53	0.34	0.88	0.83	1.62	2.45
	MI	0.11	0.05	0.16			

The regions are defined as in Figure 3-1. Production systems: PB, pasture-based; MI, mixed.
[a]There are no GLOBIOM results for dairy pasture-based systems in North America and EU regions, see note 1 for Figure 3-1.

A3.6 Livestock classification and countries and regions included in literature and GLOBIOM

Classification systems

TABLE 3-11 | Cattle production classification systems as adopted in the present chapter, defined by Robinson et al. [12] and applied in Herrero et al. [7] and in GLOBIOM [8].

Livestock classification system applied in the present chapter for studies from literature	Livestock classification system Robinson et al. [12]	Livestock classification system as applied in Herrero et al. [7] and GLOBIOM [8]
Pasture-based	Grassland-based[a]	Grazing[a]
Mixed	Mixed crop-livestock[b,c]	Mixed crop-livestock[c]
Industrial	Landless	Not included
	Urban & Other	Urban & Other
Average per country or region		ANY[d]

[a] Grassland-based/grazing is subdivided into the following systems based on climate: LGA, arid/semi-arid; LGH, humid/sub-humid; LGT, temperate/tropical highlands. In the present study, the results for pasture-based systems are presented as the weighted average of all grassland-based systems (LGA, LGH, LGT) per country or region, based on the number of livestock units in each system.
[b] Mixed is subdivided into rain-fed and irrigated
[c] Mixed is subdivided into the following systems based on climate: MRA, arid/semi-arid; MRH, humid/sub-humid; MRT, temperate/tropical highlands. In the present study, the results for mixed systems are presented as the weighted average of all mixed systems (MRA, MRH, MRT) per country or region, based on the number of livestock units in each system.
[c] In the present study, the average results per region (ANY) are presented as the weighted average of all production systems in this region, based on the number of livestock units in each system. This average also includes the systems Urban and Other.

In Herrero et al. [7] and GLOBIOM [8], the pasture-based and mixed production systems are split into climate zones (see notes Table 3-11). Production systems in temperate regions are considered as the most intensive system because these regions provide better conditions for high yielding cattle breeds and for feeding practices with higher quality feeds. Production systems in arid regions are the least intensive. The average emissions from pasture-based or mixed production in each country or region are calculated as the weighted average based on the number of livestock units (LU) in each sub-category, equation A1. For example, the average emission level from pasture-based systems is the weighted average of the emissions from the grassland-based categories LGA, LGH and LGT.

Equation A1

$$GHG\ Emissions_{region,average} = \frac{\sum_i (\#\ LU_{category\ i} * GHG\ Emissions_{region,category\ i})}{total\ of\ LU}$$

Regions

TABLE 3-12 | A list of countries and regions from GLOBIOM, Herrero et al. [7] and the case studies.

Regions in GLOBIOM	Sub regions in Herrero et al. [7]	Countries included in Herrero et al. [7] and GLOBIOM	Countries or regions covered by case studies
Europe (EUR)	EU Baltic	*Estonia, Latvia, Lithuania*	*Europe*
	EU Central East	*Bulgaria, Czech Republic, Hungary, Poland, Romania, Slovakia, Slovenia*	
	EU Mid-West	*Austria, Belgium, Germany, France, Luxembourg, Netherlands*	*Austria, Germany, France, Netherlands*
	EU North	*Denmark, Finland, Ireland, Sweden, United Kingdom*	*Denmark, Ireland, Sweden, United Kingdom (Scotland)*
	EU South	*Cyprus, Greece, Italy, Malta, Portugal, Spain*	*Portugal*
	Former USSR	*Armenia, Azerbaijan, Belarus, Georgia, Kazakhstan, Kyrgyzstan, Moldova, Russian Federation, Tajikistan, Turkmenistan, Ukraine, Uzbekistan*	
	RCEU	*Albania, Bosnia and Herzegovina, Croatia, Macedonia, Serbia-Montenegro*	
	ROWE	*Gibraltar, Iceland, Norway, Switzerland*	*Norway, Switzerland*
Oceania (OCE)	ANZ	*Australia, New Zealand*	*Australia (Tasmania), New Zealand*
	Pacific Islands	*Fiji Islands, Kiribati, Papua New Guinea, Samoa, Solomon Islands, Tonga, Vanuatu*	
North America (NAM)	Canada		Canada
	United States of America (USA)		*USA*
Latin America and the Caribbean (LAM)	Brazil		Brazil
	Mexico		
	RCAM	*Bahamas, Barbados, Belize, Bermuda, Costa Rica, Cuba, Dominica, Dominican Republic, El Salvador, Grenada, Guatemala, Haiti, Honduras, Jamaica, Nicaragua, Netherland Antilles, Panama, St Lucia, St Vincent, Trinidad and Tobago*	
	RSAM	*Argentina, Bolivia, Chile, Colombia, Ecuador, Guyana, Paraguay, Peru, Suriname, Uruguay, Venezuela*	*Peru*
East Asia (EAS)	China		
	Japan		Japan
	South Korea		
South East Asia (SEA)	RSEA OPA	*Brunei Daressalaam, Indonesia, Singapore, Malaysia, Myanmar, Philippines, Thailand*	
	RSEA PAC	*Cambodia, Korea DPR, Laos, Mongolia, Vietnam*	
South Asia (SAS)	India		
	RSAS	*Afghanistan, Bangladesh, Bhutan, Maldives, Nepal, Pakistan, Sri Lanka*	

TABLE 3-12 | Continued

Regions in GLOBIOM	Sub regions in Herrero et al. [7]	Countries included in Herrero et al. [7] and GLOBIOM	Countries or regions covered by case studies
Middle East and North Africa (MNA)	Middle East and North Africa (MENA)	*Algeria, Bahrain, Egypt, Iran, Iraq, Isreal, Jordan, Kuwait, Lebanon, Libya, Morocco, Oman, Qatar, Saudi Arabia, Syria, Tunisia, United Arab Emirates, Yemen*	
	Turkey	*Iran*	
Sub-Saharan Africa (SSA)	Congo Basin	*Cameroon, Central African Republic, Congo Republic, Democratic Republic of Congo, Equatorial Guinea, Gabon*	
	Eastern Africa	*Burundi, Ethiopia, Kenya, Rwanda, Tanzania, Uganda*	
	South Africa		
	Southern Africa (Rest of)	*Angola, Botswana, Comoros, Lesotho, Madagascar, Malawi, Mauritius, Mozambique, Namibia, Swaziland, Zambia, Zimbabwe*	
	West and Central Africa	*Benin, Burkina Faso, Cape Verde, Chad, Cote d'Ivoire, Djibouti, Eritrea, Gambia, Ghana, Guinea, Guinea Bissau, Liberia, Mali, Mauritania, Niger, Nigeria, Senegal, Sierra Leone, Somalia, Sudan, Togo*	
World (WORLD)			

CHAPTER 4

Bioethanol potential from miscanthus with low ILUC risk in the province of Lublin, Poland

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ABSTRACT

Increasing production of biofuels has led to concerns about indirect land use change (ILUC). So far, significant efforts have been made to assess potential ILUC effects. But limited attention has been paid to strategies for reducing the extent of ILUC and controlling the type of LUC. This case study assesses five key ILUC mitigation measures to quantify the low-ILUC-risk production potential of miscanthus-based bioethanol in Lublin province (Poland) in 2020. In 2020, a total area of 196 to 818 thousand hectare agricultural land could be made available for biomass production by realizing above-baseline yield developments (95-413 thousand ha), increased food chain efficiencies (9-30 thousand ha) and biofuel feedstock production on under-utilized lands (92-375 thousand ha). However, a maximum 203 to 269 thousand hectare is considered legally available (not protected) and biophysically suitable for miscanthus production. The resulting low-ILUC-risk bioethanol production potential ranges from 12 to 35 PJ per year. The potential from this region alone is higher than the national Polish target for 2nd generation bioethanol consumption of 9 PJ in 2020. Although the sustainable implementation potential may be lower, the province of Lublin could play a key role in achieving this target. This study shows that the mitigation or prevention of ILUC from bioenergy is only possible when an integrated perspective is adopted on the agricultural and bioenergy sectors. Governance and policies on planning and implementing ILUC mitigation are considered vital for realizing a significant bioenergy potential with low ILUC risk. One important aspect in this regard is monitoring the risk of ILUC and the implementation of ILUC mitigation measures. Key parameters for monitoring are land use, land cover and crop yields.

4.1 INTRODUCTION

From 2002 to 2012, the production of biofuels has expanded significantly in the EU [1]. This growth is largely policy driven, based on the idea that biofuels can play an important role in reducing GHG emissions and mitigating climate change [2]. However, in recent years, this assumption has been widely debated. One of the main topics of concern is land use change (LUC), and especially indirect lands use change (ILUC). Here, ILUC is defined as a change in land use that takes place if biofuel feedstock production displaces agricultural production of food, feed and fibres and this displacement results in 1) food, feed and fibres being produced elsewhere to continue to meet the demand, or 2) more land being taken into agricultural production because of increased food prices [3-5]. When ILUC entails the conversion of high carbon stock lands, e.g. forests or grasslands, this can lead to increased GHG emissions which reduces or even cancels out the GHG benefits of biofuels compared to fossil fuels [5]. Since the first publication on the negative effects of ILUC by Searchinger et al. [5], multiple studies have attempted to model and quantify the extent of (I)LUC and the level of related GHG emissions caused by biofuel production [6-10]. However, the modeling of LUC and (I)LUC-related GHG emissions is characterized by major limitations and challenges [11]. Results vary significantly between studies and outcomes are expected to remain uncertain [3,4,12]. Therefore, investigating how ILUC can be mitigated or prevented may be more important than assessing the scale of ILUC under current assumptions [3].

ILUC of biofuels can only be prevented when the direct LUC (DLUC) of the displaced activity is addressed as well. Therefore, it is necessary to take an integrated perspective on all land use, whether for food, feed, fibre and fuels. Previous research has identified the following key measures to reduce the extent of ILUC and control the type of land use change: above-baseline yield development, improved integration of food and biofuel chains, increased chain efficiencies, biofuel feedstock production on under-utilized lands and land zoning [3,13-15]. Very few studies, however, have investigated the potential of producing biofuels with low ILUC risk [14]. For the assessment of low-ILUC-risk biofuel potentials, regional analyses are of great importance because of several reasons. First, a regional analysis considers the specific characteristics of a region such as, for example, biophysical conditions, agricultural practices and the socio-economic context. Such factors are needed to define a feasible and suitable biofuel target for the region and develop appropriate policy strategies for realizing this target and mitigating ILUC. Second, a regional analysis is important to assess the availability and quality of data and to translate this into parameters for monitoring the implementation of ILUC mitigation measures and ILUC risks. Monitoring is required for correct certification of low-ILUC-risk biofuels.
The aim of this case study is 1) to assess how much additional biofuel can be produced

in 2020 by implementing ILUC mitigation measures (i.e. the low-ILUC-risk biofuel production potential), and 2) to identify parameters required for monitoring the risk of ILUC and the implementation of ILUC mitigation measures. The case study focuses on bioethanol production from miscanthus, in the Polish province of Lublin (Lubelskie voivodship). Lublin is located in the south-east of Poland. Diverse studies have shown that this province has a significant technical and economic potential for biomass production [16-20]. In addition, the development level of agricultural systems and the agricultural yields in Eastern Poland are lower compared to Western regions [21,22]. This suggests that agricultural productivity can improve significantly and thereby make land available for bioenergy feedstock production without ILUC. The choice to conduct the case study at province level is based on the good availability of data and regional differences in agricultural characteristics in Poland. Miscanthus is chosen because it has the potential to contribute to the development of the rural economy by the diversification of farms, which often enhances their economic resilience and profitability [23]. In addition, crop diversity helps to maintain or improve the agroecosystem [24].

4.2 METHODS AND MATERIALS

The case study presented here is based on a report by Gerssen-Gondelach et al. [25]. The general method to quantify ILUC mitigation measures was developed by Brinkman et al. [26]. This section describes the main aspects of the method and provides case specific details. For more details, the reader is referred to Gerssen-Gondelach et al. [25] and Brinkman et al. [26].

4.2.1 Assessment of low-ILUC-risk biofuel potential

The assessment of the low-ILUC-risk biofuel production potential is based on a combination of a top-down and bottom-up approach and distinguishes three main components, see Figure 4-1. Below, these components are shortly described.

Step 1: Top down assessment of agricultural production in the baseline and target scenario in 2020

From an economic model used to analyse ILUC factors (top-down approach), a biomass production baseline (without additional biofuels) and target (with a biofuel mandate) for the case study region in 2020 are established. The current study uses the outputs from the computable general equilibrium model MIRAGE-BioF (Modeling International Relationships in Applied General Equilibrium for Biofuel, hereafter referred to as MIRAGE) as generated for a study for DG Trade of the European Commission [9]. The baseline indicates the production of biomass for food, feed and fibre applications in the absence of

the biofuels mandate (i.e., assuming current biofuel production to remain approximately constant). The target refers to the total biomass production when a biofuels mandate is implemented; it includes food, feed and fibre demand as well as the extra feedstocks for biofuels needed to meet the biofuels mandate. The difference between the target and baseline is the extra agricultural production induced by the mandate (whether directly caused by increased demand for meeting the mandate or induced by increased crop prices). In MIRAGE, this amount is projected to cause LUC (both direct and indirect).

The MIRAGE study [9] includes two biofuel mandate scenarios which differ in their assumptions regarding future trade policy (business as usual or BAU vs. free trade). In the present study, the BAU scenario is applied, which means that all existing import tariffs on biofuels remain unchanged in 2020. The mandate includes first generation biofuels: biodiesel from oil palm, rapeseed, soybean and sunflower and bioethanol from maize, wheat, sugar cane and sugar beet [9].

This study considers both crop and cattle (beef and milk) production. The MIRAGE model outputs for crop production volumes in the baseline and target scenario are only available on the EU27 level. Therefore, the outputs are disaggregated to the case study region, based on the current share of crop production in Lublin compared to the EU27 [26]. The production of beef and milk in 2020 cannot be derived from the MIRAGE model. Therefore, the production in 2020 is estimated by assuming that the production trend will be in line with the recent trend in the European Union (1991-2012). It is assumed that the production is unaffected by a biofuel mandate and thus equal in the baseline and target scenario. In both the baseline and target scenario for 2020, the production volumes for crops and beef are projected to be lower than in 2010, see appendix A4.2. The production volume of milk is projected to remain constant compared to 2010.

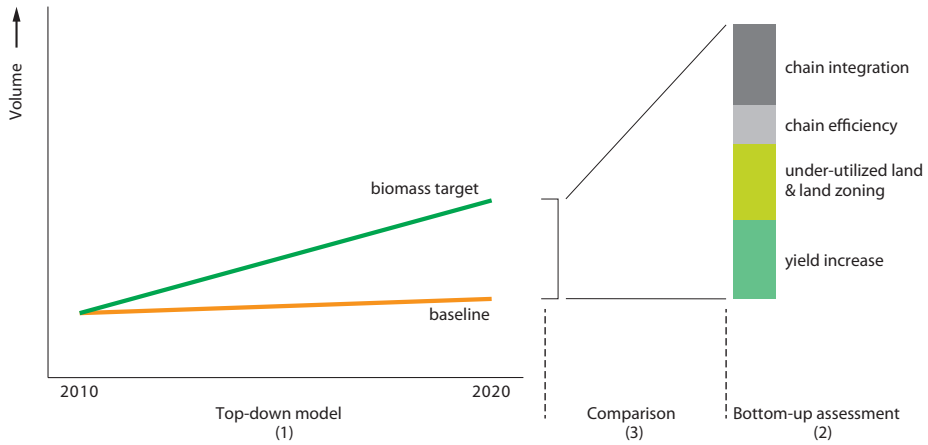

FIGURE 4-1 | General approach to analyze and quantify the biomass production potential with low ILUC risks [26]. The three steps (1-3) of the analysis are described in the main text.

Step 2: Bottom-up assessment of low-ILUC-risk miscanthus-based ethanol production potential

It was shown that in MIRAGE, no mandate for miscanthus-based bioethanol is included. Therefore, the production of miscanthus-based ethanol further increases the biomass production volume above the level of the target scenario. Only when the total biomass production potential with low ILUC risk is higher than the production induced by the target scenario, production of low-ILUC-risk miscanthus-based ethanol is possible. To determine the potential to produce ethanol from miscanthus with low ILUC risk, it is assessed how much agricultural land can be made available for miscanthus cultivation by implementing ILUC mitigation measures. First, a baseline yield scenario is defined to determine the initial total agricultural land area required in 2020 for the projected biomass production volume in the target scenario. Then, it is assessed to what extent the different ILUC mitigation measures can contribute to reducing this land requirement and making the surplus land (i.e. land no longer needed for the targeted biomass production) available for bioenergy (sections 4.2.2.1-4.2.2.5). To this aim, this study takes an integrated view on all land uses and looks for synergies between agriculture, forestry and bioenergy. The following ILUC mitigation measures are assessed (bottom-up approach): above-baseline yield development, improved chain integration, increased food chain efficiency, biofuel feedstock production on under-utilized lands and land zoning. The latter measure, land zoning, is distinct from the first four measures. It does not reduce land requirements for agricultural production, but establishes constraints on future production areas to avoid the conversion of protected and biophysically unsuitable areas to miscanthus cultivation.

Finally, the total surplus land resulting from integrating all ILUC mitigation measures and the potential bio-ethanol production on this land are calculated (section 4.2.2.6). This potential is the technical potential, which takes into account the demand for land for food and feed production, legal requirements regarding environmental conservation and minimal biophysical requirements for miscanthus cultivation. In the present chapter, this potential is called the low-ILUC-risk potential. Although the implementation of ILUC mitigation measures reduces the risk of ILUC, it not necessarily decreases the risk to zero. The ILUC risk will only be zero if it is guaranteed that biofuel feedstocks are only produced on land that is made available by one of the ILUC mitigation measures. This requires legislation and enforcement of regulations.

In the general approach, the baseline yield scenario for crops is derived from MIRAGE [26]. However, in this case study, the MIRAGE projections are not in line with recent yield trends in Lublin. As it is found that crop yields often follow a linear trend over time [27,28], the baseline scenario in this case study is based on linear extrapolation of historical yield trends (1999-2012) in Lublin. For cattle, the selected parameters for yield are the beef

and milk productivity (beef or milk production per animal per year) and the cattle density on meadows and pastures. The productivity and density values in 2020 are defined similar to crop yields. Currently, the total agricultural area needed for the production of the selected crops and for cattle in Lublin covers 1,224 thousand ha (87% of the utilized agricultural land area). Because of the increasing yields and the projected reduction in the total agricultural production volume in the target scenario compared to the level in 2010, the total land use reduces to 944 thousand ha in 2020 (see appendix A4.2).

Step 3: Comparison of low-ILUC-risk bioethanol potential to biofuel production target

The low-ILUC-risk biofuel production potential is compared to the biofuel production target. In the general method, this target is derived from MIRAGE's biofuel mandate scenario. As MIRAGE includes no target for miscanthus-based bioethanol, the production potential can only be compared to MIRAGE's production target for first generation bioethanol. The bioethanol production target from MIRAGE for the EU27 is disaggregated to the Polish national level based on the share of bioethanol production in Poland compared to the EU27 (Table 4-1). The target is not disaggregated to the level of Lublin province because no information is available about current biofuel production levels at the provincial level. In addition to the bioethanol production target from MIRAGE, the low-ILUC-risk biofuel potential is compared to the targets for biofuel consumption in 2020 as set in the Polish National Renewable Energy Action Plan (NREAP) for meeting the requirements of Directive 2009/28/EC (Table 4-1).

TABLE 4-1 | Current and targeted production of first and second generation bioethanol in Poland.

		Current production[a]	2020 projected baseline production without mandate[b]	2020 production target with mandate[b]	2020 consumption target Poland[c]
1st gen bioethanol					
	Million liter	177.5	174.9	567	760
	PJ	4.2	4.1	13.3	17.8
2nd gen bioethanol					
	Million liter	-	n/a	n/a	376
	PJ	-	n/a	n/a	8.8

[a] current: average 2009-2011 [29-31]; [b] derived from MIRAGE output for EU27; [c] targets as set in 2010 in the National Renewable Energy Action Plan [32]

4.2.2 Assessment of ILUC mitigation measures

The contribution of the ILUC mitigation measures to the miscanthus-based ethanol production potential is investigated for three scenarios; *low, medium* and *high*. Each scenario includes all ILUC mitigation measures and for each measure assumptions are

made about how this measure contributes to the generation of surplus land compared to the target scenario from step 2. For example, the scenarios assume more rapid developments in agricultural productivity and food chain efficiency compared to the baseline projections in the target scenario. The rates of development increase from the low to the high scenario to indicate the variability and uncertainty in the data and to test the effect on the low-ILUC-risk potential. Sections 4.2.2.1 to 4.2.2.5 explain the assumptions per measure for the different scenarios. Where the methods to assess the ILUC mitigation measures deviate from the general approach [26], this is also explained in these sections. In addition, section 4.2.2.6 describes how the total low-ILUC-risk biofuel potential in each scenario is calculated by integrating the results of all individual measures.

4.2.2.1 Above-baseline yield development

Increases in crop yield, beef and milk productivity and cattle density above the baseline projection result in a reduction of agricultural land required for crop and livestock production (assuming the production volume remains constant). On the resulting surplus land area, biomass can be produced with low ILUC risk (see Brinkman et al. [26] for a detailed description of the calculation). The baseline yield scenario was defined based on the finding that crop yields often follow a linear trend over longer terms [27,28]. Over shorter time periods, however, higher yield increases are possible, especially when the yield gap is still large [28]. The average yields and management levels in Lublin are lower compared to regions in Western Poland and to Germany (see Table 4-10 in appendix A4.1). Therefore, measures such as scaling up of farms, mechanization and improved use of chemicals, as already applied in these other regions, can enable higher annual yield growth rates compared to the baseline. Also, when crop yields increase and less land is needed for production, the use of lower quality land for production is likely to decrease which has a positive effect on the average yield levels. The crop yields in Western Poland and Germany give an indication of what yields can be attained in Lublin, but also maximum attainable crop yields based on e.g. climate and land suitability are taken into account. Similarly, for cattle production, cattle density and productivity levels in Poland and Germany [21,33] are assumed to be appropriate indicators of what improvements are attainable in Lublin. Data on historical and current crop-specific yields, cattle density and beef and milk productivity are collected or derived from the Central Statistical Office in Poland (CSO) [21] and the FAO [33]. The agro-ecological potential crop yield is derived from the Global Agro-Ecological Zones database [34]. As an example, Table 4-2 compares the current average wheat yield to the yield level in the baseline and the low, medium and high scenarios.

TABLE 4-2 | Current and projected average wheat yields and losses in transport and storage. Projections are given for the baseline and the low, medium and high scenarios.

	Average wheat yield Lublin ($t\ ha^{-1}\ yr^{-1}$)	Wheat losses (%)
Current[a]	3.7	5.0
Baseline scenario 2020	4.1	5.0
Low scenario 2020	4.5	3.8
Medium scenario 2020	5.7	2.5
High scenario 2020	7.5	0.8

[a] current wheat yield: average 2008-2012; current wheat losses: average 2008-2011

4.2.2.2 Improved chain integration

The production of second generation bioethanol generates various by-products such as lignin, proteins and carbon dioxide released during fermentation. These by-products can be used to produce a variety of value-added co-products [35]. Depending on the potential uses of these co-products and following the principles of consequential LCA [26,36-38], these co-products could be argued to reduce land demand and thereby help to mitigate ILUC. For example, when co-products can substitute livestock feed from crops, a certain amount of land can be freed-up from crop cultivation. In this case study, no co-products are included that have the potential to generate surplus land in 2020 (see the biofuel chain design in appendix A4.3). Therefore, this measure is not further investigated.

4.2.2.3 Increased food chain efficiency

This ILUC mitigation measure addresses the reduction of food losses in transport, storage, (un)loading, etc., such that a higher share of the produced goods reaches the consumer. Thereby, less land is needed to deliver the same amount of goods (see Brinkman et al. [26] for a detailed description of the calculation). In Poland, 27% of all food losses and food waste take place in the stages between farms and consumers [39]. These losses are equal to 15% of the total national food production; for comparison, the average percentage in the EU27 is 7% [39]. To analyze the land saving potential of food chain efficiency improvements, regional figures are not available. Therefore, national figures on food losses occurring during storage and transportation from FAO food balances are used [33]. These figures give estimated volumes of food losses for each separate agricultural product. In Table 4-2, the example of projected food losses is given for wheat in the baseline and the low, medium and high scenarios.

4.2.2.4 Biofuel feedstock production on under-utilized lands

Under-utilized land includes set-aside land, abandoned land, marginal lands or degraded land, which often has lower productivity than conventional agricultural land. The share of this land type that does not provide other services (e.g. agriculture, biodiversity, high carbon stocks or other ecosystem services) can be used for the production of biomass with low risk

of ILUC. The total area of under-utilized land in 2020 depends on the current area of under-utilized land and an increase in this area due to reduced agricultural land use from 2010 to 2020 as projected in the biofuel target scenario based on MIRAGE (section 4.2.1).

Regarding the amount of under-utilized land currently available in the case study area, the use of spatially explicit data about the location and extent of these types of land, its current uses and functions, and its suitability for the biofuel feedstock investigated in the case study is ideal. For Lublin province, however, spatially explicit data about the location of under-utilized lands is not available. Therefore, the current area of under-utilized agricultural land is estimated based on statistical data about set-aside, fallow and marginal land from the Central Statistical Office of Poland [21,40,41], Eurostat [42] and FAO [43]. In addition, it is estimated what part of the agricultural land not under agricultural activity can be considered as abandoned land potentially available for miscanthus. This is based on statistics and own estimates (for more details, see appendix A4.4). Additional abandoned land available in 2020 is estimated based on the area of agricultural land no longer required because of the projected reduction in agricultural production and increase in yields (section 4.2.1).
Often, the bioenergy crop yield on under-utilized lands is expected to be lower than average. However, not in all cases yields on under-utilized land are actually lower than on agricultural land as it depends on the soil and climate conditions. As the location and biophysical characteristics of under-utilized lands are unknown, the suitability and the attainable miscanthus yield on these lands cannot be assessed. Therefore, the impact of the yield level on the miscanthus production potential is assessed in a sensitivity analysis (see section 4.2.2.6).

4.2.2.5 Land zoning

While the previously described measures attempt to mitigate ILUC, land zoning aims at reducing the impacts of LUC, here especially the associated biodiversity losses and GHG emissions. This study includes land zoning in order to prevent the conversion of protected areas, including (primary and secondary) forest and high conservation value areas, for the production of biomass. In addition, in this case study, this measure also considers the land suitability for miscanthus production.

The land not excluded by land zoning for protection purposes is referred to as *legally available land. Suitable land* refers to land that is biophysically suitable for miscanthus production, considering minimal climate and soil requirements. The calculation of the legally available and suitable agricultural land area for miscanthus cultivation is based on the method applied by Pudelko et al. [17] to assess the technical potential of perennial energy crops in Poland. Spatial analyses for Lublin province are performed in the geographic

information system (GIS), using the following data sets: agricultural soil suitability [44], Corine Land Cover [45], digital elevation model, hydrogeological map [46], annual rainfall based on the Agroclimate Model of Poland [47] and protected areas [48,49].
First, to determine the legally available land, the following criterion is applied:

a. Soils located on protected areas of land are removed. Protected areas include all forests, national parks, landscape parks, nature reserves, strict protection areas, Natura 2000 sites, and their buffer zone.

Second, to assess what share of the total agricultural area is biophysically suitable for miscanthus cultivation, the following criteria are applied:

a. Miscanthus roots can extract water to a depth of approximately 2 meter [50]. Therefore, the ground water level is set at a depth up to 2 meter for all soils. The areas with a lower ground water table are excluded;
b. The minimal average annual precipitation is 550 mm yr^{-1} for all soils (see e.g. Kuś and Faber, 2009 in Sliz-Szkliniarz [51]). Areas where the precipitation did not exceed this minimum are removed;
c. Boggy and wet areas are excluded because the accessibility of machinery to waterlogged sites is limited and can cause soil damage. Also, the release of carbon dioxide due to land conversion will negatively affect the GHG emission balance of the biofuel;
d. Areas over 350 meter above sea level are excluded because production and transportation conditions are hampered in these regions.

Land not complying with these suitability criteria is only very marginally suitable for miscanthus production. On these lands, miscanthus yields would be significantly lower than on suitable lands (see appendix A4.3).
Finally, the criteria for legally available land and suitable land are combined, resulting in the total agricultural area legally available and suitable. Although considered suitable, the soil quality and degree of suitability of the areas included varies. Therefore, in the results, it is shown how land is distributed among suitability classes. This distribution is determined by using the Polish classification system that distinguishes twelve soil suitability classes or complexes [52]. Nine of these classes, apply to arable land and can be categorized into very good and good quality soils, lower quality soils and very weak soils. Three classes apply to grasslands (meadows and pastures) of various quality.
Pudelko et al. [17] excluded good and very good quality soils from their analysis, based on the guideline that bioenergy crops should not be cultivated on these lands. However, ILUC mitigation measures may free some areas that have good or very good quality soils while this would not result in displacement of crop production. Therefore, the present study includes all soil classes.

The land zoning criteria applied in this study do not include specific conditions on maximum carbon stocks to allow land use conversion. However, the analysis excludes all areas that are prohibited by the RED to be used for biomass production because of high carbon stocks (i.e. wetlands, forested areas, and peat land).

In the criteria described above, all protected areas currently under agricultural use (e.g. parts of the Natura 2000 network) are excluded from bioenergy production to ensure the conservation of biodiversity [53]. However, some of the protected areas may actually be designated as legally available for miscanthus cultivation because miscanthus can have a positive impact on the biodiversity of agricultural land. The biodiversity in miscanthus fields is found to be higher compared to annual crops [24,54]. This is potentially also true for grasslands, but the number of studies is limited yet and more research is needed [24,55]. In the medium and high scenarios it is assumed that a part of the suitable agricultural areas with high conservation value can be made legally available for miscanthus cultivation. Areas with high carbon stocks are excluded in all scenarios.

4.2.2.6 Integrated analysis of overall low-ILUC-risk biofuel potential

Table 4-3 provides a summary of the scenario assumptions per ILUC mitigation measure. Having evaluated the individual measures, the total potential biomass production without ILUC is analyzed. This is an integrated assessment that accounts for the interactions and feedback between different measures. An example of this is a reduction in food losses, that decreases the food production volume required for supplying the same amount of food, which influences the effect of above-baseline yield developments. The order in which the measures are considered in the integrated analysis influences the outcome of the assessment. In this study, the integration calculations are performed as following:

i. The agricultural land area required for food, feed, fuel and fibre production in 2020 as derived from the MIRAGE target scenario is taken as the initial land base.
ii. The measure *increased food chain efficiency* is implemented: the biomass production volume required after a reduction in food losses is calculated. The surplus area generated by this measure is calculated using the baseline yield development scenario (sections 4.2.2.1 and 4.2.2.3).
iii. The measure *above-baseline yield increases* is applied: based on the required food production as determined in step ii, the additional surplus area generated through above-baseline yield developments is calculated.
iv. The measure *use of under-utilized land* is taken into account: the area of under-utilized land is added to the total surplus land area from steps ii and iii.
v. The measure *land zoning* is implemented: the total surplus land area from steps ii to iv is compared to the total land area suitable and legally available for miscanthus production. In the case that the surplus land area is larger than the area suitable

TABLE 4-3 | Summary of scenario assumptions per ILUC mitigation measure sand per scenario

	Baseline	*LOW*	*MEDIUM*	*HIGH*
Above-baseline yield developments - crops[a]	Extrapolation of the historical linear yield trends in Lublin for the period 1999-2012 to 2020. The average annual yield increase is 1.8%, but varies between crops.	Annual yield increase of 2.3% for a period of ten years for all crops, based on the REFUEL projection for Central and Eastern European countries [56].	Crop specific yields are set to the current maximum yield level attained in Poland at the province level (average 2008-2012) [21]. The average annual yield increase is 3.2%, but varies between crops.	Crop specific yields are set to the current yield level attained in Germany (average 2008-2012) [33]. The average annual yield increase is 7.6%, but varies between crops.
Above-baseline yield developments - cattle	Extrapolation of the historical linear trends in cattle density and milk productivity in Lublin for the period 1999-2012 to 2020 [21]; In 2020, beef productivity attains current average productivity level of Poland in 2012 [40].	Beef and milk productivity are set equal to baseline scenario, cattle density is set equal to the current cattle density level in Germany (average 2008-2011) [33].	Both beef and milk productivity and cattle density are set equal to the current level attained in Germany (average 2008-2012 for beef and milk productivity; average 2008-2011 for cattle density) [33]	Both beef and milk productivity and cattle density are set equal to the current maximum level attained in Poland at the province level (average 2008-2012) [21].
Improved food chain efficiency[b]	Product specific food losses in 2020 are similar to average losses in Poland for the period 2008-2011 [33].	Product specific food losses reduce with 25%	Product specific food losses reduce with 50%	Product specific food losses reduce to the 15th percentile[c] of the loss percentages of all EU countries [33].
Biomass production on under-utilized land	n.a.	Low estimation of under-utilized land area, based on statistics.	Medium estimation of under-utilized land area, based on statistics.	High estimation of under-utilized land area, based on statistics.
Land zoning	n.a.	All protected areas are excluded.	Miscanthus cultivation is possible on a limited area of protected land where miscanthus cultivation could support improvements in biodiversity[d]	Miscant Miscanthus cultivation is possible on a limited area of protected land (larger than in the medium scenario) where miscanthus cultivation could support improvements in biodiversity[e]

[a] In the calculations it is assumed that yields will not decrease compared to the current level. If a yield in one of the scenarios is lower than the current yield, the current yield will be considered instead. If a yield in one of the scenarios is higher than the agro-ecological potential yield, this agro-ecological potential yield will be considered instead. The agro-ecological potential yield is derived from the Global Agro-Ecological Zones database [34]. The data reflect Polish average maximum attainable yields. Data was not found for all crops; in case no data was available, the maximum potential yield was not taken into account.
[b] It is assumed that losses do not increase. In case losses would increase in a certain scenario, the loss is set equal to the current level (average Poland 2008-2011), i.e. the reduction in losses is zero. In FAOSTAT, no food loss figures are given for rapeseed and beef. Therefore, estimations are made based on losses in Austria (rapeseed) and Hungary (beef); these are considered to be most comparable to levels in Poland.
[c] 15% of the EU countries attain this or a lower loss percentage, 85% of the EU countries attain a higher loss percentage.
[d] In the medium scenario, the following is assumed: of the area that is suitable for miscanthus production but not legally available according to the applied protection criteria, 50% will be made legally available because miscanthus cultivation has a positive impact on biodiversity. Protected areas that are not considered to be suitable are excluded for miscanthus production.
[e] In the high scenario, the following is assumed: of the area that is suitable for miscanthus production but not legally available according to the applied protection criteria, 100% will be made legally available because miscanthus cultivation has a positive impact on biodiversity. Protected areas that are not considered to be suitable are excluded for miscanthus production.

and legally available, the use of surplus land for biomass production is limited by land zoning restrictions. The total surplus land area resulting from applying all five measures is presented for a low, medium and high integrated scenario in which the low, medium and high scenarios of each measure are combined respectively. In addition, a distinction in the results is made between the surplus area of cropland and of meadow and pasture land.

vi. For each integrated scenario, the potential miscanthus and bio-ethanol production on the total surplus land area is calculated. These potentials depend on the miscanthus yield and the biofuel chain efficiency. Therefore, to assess the impact of the value chain design, the total chain productivity is defined for a medium scenario and two sensitivity scenarios (low and high), see Table 4-4.

TABLE 4-4 | Components and productivity of the value chain for miscanthus-based biofuel production (for detailed explanation see appendix A4.3).

Chain component	Assumptions	Parameter	Baseline	Sensitivity range (low-high)	References
Miscanthus cultivation and harvest	Spring yield, farming conditions are sub-optimal and plantations have not reached plateau yields yet	Yield (t dm/ha)	13	10-17	[57-60]
Storage	On-farm storage of bales in the open air covered with plastic sheeting or storage in a silo or under a bale tarp	Biomass loss (% dry matter)	3%	1-5% dry matter	[54,61,62]
Transport	Truck transport	Biomass loss (% dry matter)	0% [a]	-	-
Conversion	Biochemical conversion	Biomass-to-ethanol conversion efficiency (% HHV)	35%	35-40%[b]	[63-68]
Overall ethanol yield			**84 GJ/ha**	**64-129 GJ/ha**	Own calculation[c]

[a] Biomass losses during transport are assumed to be negligible; [b] The low conversion efficiency for the sensitivity analysis is equal to the baseline efficiency, because newly build plants already attain the baseline efficiency [63-68]; [c] Calculated from combining the miscanthus yield, storage and transportation losses and conversion efficiency.

4.3 RESULTS: ILUC MITIGATION POTENTIALS

4.3.1 Above-baseline yield development

Table 4-5 presents the land savings for the low, medium and high above-baseline yield developments in crop and cattle production compared to the target scenario. The saving potentials of crops are higher compared to cattle. This is in line with the fact that the cropland area is larger than the area of meadows and pastures (Table 4-9, appendix

A4.1). In all three scenarios, wheat yield improvements account for the largest area saved, followed by barley, triticale and rapeseed. For potatoes, sugar beets and apples, the yields in the low and medium above-baseline scenarios are actually lower compared to the baseline projection, because extrapolation of the recent yield trend results in a high yield increase. The additional area required for these crops compared to the target scenario is lower than the area saved by other crops, but reduces the total area saved. With regard to cattle, increasing the cattle density on meadows and pastures has a larger effect on the area saved than increasing the beef and milk productivity. The impacts of improvements in beef and milk productivity are comparable to each other.

TABLE 4-5 | Land saved by crops and cattle in the above-baseline yield scenarios and increased food chain efficiency scenarios.

Product	**Area saved (1000 ha)**					
	Above-baseline yield scenarios			Increased food chain efficiency scenarios		
	Low	Medium	High	Low	Medium	High
Wheat	25	80	132	3.7	7.4	12.5
Rapeseed	12	16	26	0.7	1.5	2.0
Potatoes, sugar beets and apples	-6	-13	9	0.6	1.2	1.9
Other crops	35	67	126	3.4	6.7	11.4
Cattle[a]	30	74	132	0.8	1.7	2.8
Total	**96**	**224**	**426**	**9.2**	**18.4**	**30.5**

[a] the land saved by cattle is meadow and pasture land

4.3.2 Increased food chain efficiency

The agricultural area saved in the increased food chain efficiency scenarios is presented in Table 4-5. The potentials are significantly lower compared to the potential from above-baseline yield development. Improved chain efficiencies of crops result in considerably higher land saving compared to cattle. Similar to the above-baseline yield improvement scenarios, wheat has the highest land saving potential, followed by oats and rapeseed.

4.3.3 Biofuel feedstock production on under-utilized lands

The total area of set-aside and fallow land is estimated to be 45-75 thousand hectare (Table 4-6, see appendix A4.4 for a more detailed explanation). In addition, the area of agricultural land that is held by owners who do not conduct agricultural activities and that could potentially be considered as abandoned land suitable for miscanthus production is estimated to be 5-20 thousand hectare (Appendix A4.4). Finally, according to the biofuel target scenario for 2020 based on MIRAGE and own estimates for cattle production, a total area of 280 thousand hectare will be abandoned compared to 2010 (section 4.2.1).

However, the projected rate of reduction in the agricultural land area is high compared to what is expected based on recent developments in Lublin and Poland. In addition, several factors could result in more land use than projected. Both issues are considered in more detail in the discussion. For this measure, the area of abandoned land in the low and medium scenario is estimated to be significantly smaller than 280 thousand hectare. In the high scenario, the total abandoned land area of 280 thousand hectare is included.
In statistics, only 202 ha of land was defined as degraded land [40,41]. Therefore, the share of marginal land in the total area of agricultural land is considered to be negligible. The resulting total estimated area of under-utilized in each scenario is presented in Table 4-6.

TABLE 4-6 | Estimated under-utilized land area available in 2020 in the low, medium and high scenarios.

	Area Lublin (1000 ha)			**References**
Scenario	LOW	MEDIUM	HIGH	
Set-aside and fallow land	45	60	75	[21,42,43]
Abandoned land not held for agricultural activity	5	15	20	[21] and own estimates
Abandoned land baseline scenario	42	98	280	MIRAGE projection [9] and own estimates
Of which:				
Cropland	34	80	229	
Grassland	8	18	51	
Marginal land	0	0	0	[40,41]
Total under-utilized land	**92**	**173**	**375**	

4.3.4 Land zoning

In section 4.2.2.5, criteria were given to assess both the legal availability and the biophysical suitability of agricultural land for miscanthus production. When only applying the protection criterion, the total agricultural area that is legally available is 1,267 thousand ha, see Table 4-7. When only considering the suitability criteria, the total agricultural area that is suitable for miscanthus production is 269 thousand ha. Other agricultural areas are considered very marginally suitable for miscanthus because of limited (soil) water availability, which is low in summer due to droughts [69]. Although miscanthus has a good water use efficiency compared to many other crops, it is found to be sensitive to water stress [70,71]. When combing the protection and suitability criteria, the total area suitable and legally available is 203 thousand hectare, which is equal to 12% of the total agricultural area. This value is used for the low scenario. Assuming that some protected areas could be made available for miscanthus production (as described in section 4.2.2.5), the suitable and legally available land area increases to 236 thousand ha in the medium scenario and 269 thousand ha in the high scenario.

TABLE 4-7 | Agricultural area legally available and suitable for miscanthus production.

Criteria applied	Resulting land area	Area (1000 ha)	% of total agricultural area	Area by soil quality (1000 ha) Arable land			Grassland
				Very good & good	Lower	Very weak	Total
None	Total agricultural land area[a]	1,745	100%	885	499	75	316
Protection	Total area legally available	1,267	73%	631	391	49	195
Suitability	Total area suitable	269	15%	55	134	0	80
Protection and suitability	Total area suitable and legally available for miscanthus	203	12%	40	114	0	50

[a] Equal to average of agricultural land area in 2010 and 2012

4.3.5 Integrated analysis

Figure 4-2 presents the combined potential surplus land area generated by the measures *increased food chain efficiency, above-baseline yield development* and *biofuel feedstock production on under-utilized lands* and compares this land area to the area suitable and legally available for miscanthus based on the measure *land zoning.*

The ILUC mitigation measures *above-baseline yield development* and *biofuel feedstock production on under-utilized lands* have the largest potential to make land available for biomass production. For the measure *biofuel feedstock production on under-utilized lands*, a large share of the potential is related to the projected reduction in demand for agricultural land in the biofuel target scenario of MIRAGE compared to the current situation. The largest share of the area saved is considered to be cropland. The suitability and legal availability criteria for agricultural land limit the use of the arable land area saved in all scenarios and of the grassland area saved in the medium and high scenario. The resulting ethanol production potential in each scenario is presented in Figure 4-3. This figure also presents the sensitivity of the biofuel potential to the miscanthus-ethanol chain productivity, as defined in section 4.2.2.6. The total bioethanol production potential ranges from 12.2 PJ per year in the case of a low ethanol yield in the low integrated scenario to 34.6 PJ per year in the case of a high ethanol yield in the high integrated scenario (i.e. 522-1,479 million liter per year). The 2nd generation bioethanol consumption target for Poland as set in the National Renewable Energy Action Plan is 8.8 PJ or 376 million liter in 2020. Thus, in all scenarios, the miscanthus-based ethanol production potential of only the province of Lublin is higher than this national target. In addition, in the National Renewable Energy Action Plan, the total target for the national consumption of all first and second generation biofuels in Poland is 60.5 PJ (2,582 million liter) in 2020 [32]. Thus, 20% to 57% of this target could be met by bioethanol production from miscanthus in the province of Lublin.

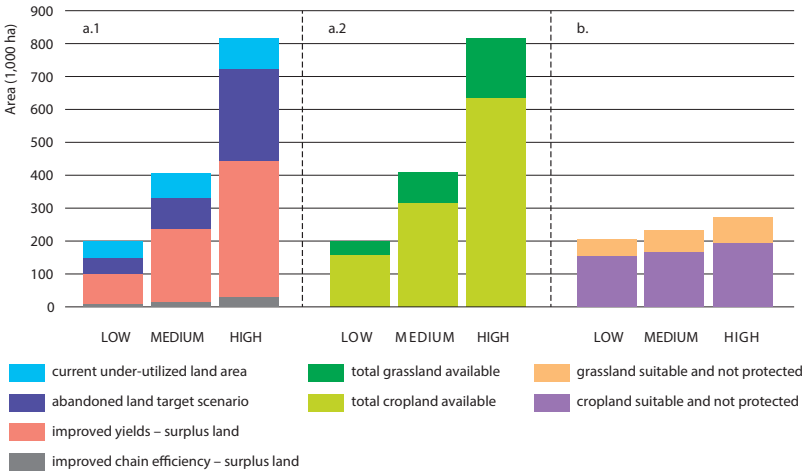

FIGURE 4-2 | Surplus area in the three integrated scenarios, breakdown by measures (a1) and agricultural cropland and grassland (a2); cropland and grassland area suitable for miscanthus cultivation and corrected for land zoning (b).

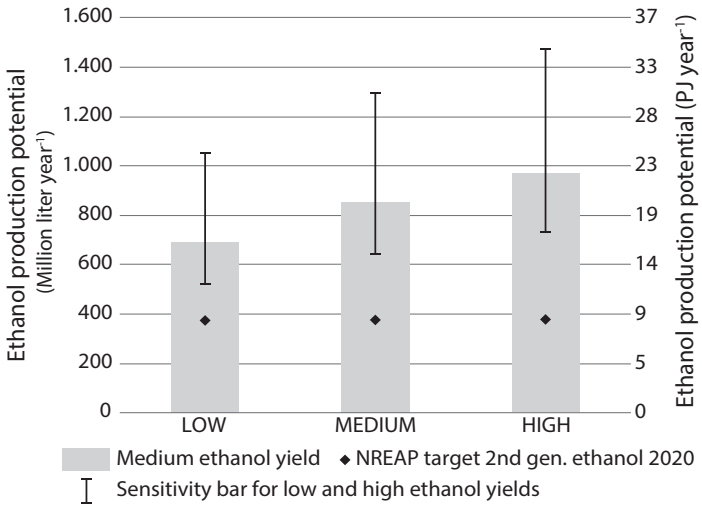

FIGURE 4-3 | Bioethanol production potentials in Lublin province in 2020 for medium ethanol yield and sensitivity bars for low and high ethanol yield, see table 3. The NREAP target is the Polish national consumption target for second generation bioethanol of 8.8 PJ in 2020, as stated in the Polish national renewable energy action plan.

4.3.6 Monitoring ILUC and ILUC mitigation measures

The analysis shows that technically it is possible to produce large additional amounts of biofuel in Lublin with a low risk of causing ILUC. For certification, it needs to be verified that biofuel feedstock is indeed produced with low ILUC risk, i.e. the risks of land conversion elsewhere or undesired land use change in the case study region as a result of biofuel feedstock production are within certain thresholds. Also, to control and manage the expansion of biofuel feedstock production, the implementation of the ILUC mitigation measures should be monitored.

For this case study, several parameters are identified that are important for monitoring ILUC risk (Table 4-8). First, the observation of land use (e.g. for agricultural or bioenergy production, or for forestry) and land use change over time is vital. For this, it is required to frequently compose land use and land cover maps. This can be done by using remote sensing (satellite monitoring), supplemented with field data for validation. Important is the detail of the land use and land cover maps. This means that maps should differentiate between, for example, forest and agricultural land, land under agricultural activity and abandoned or set-aside land, and between agricultural and bioenergy crops. In addition, appropriate spatial and temporal resolutions should be chosen. As farms in Lublin are often small, the spatial resolution should be high to enable the identification of differences in land use and land cover. It is especially important to observe areas that are excluded from bioenergy production through land zoning regulations. When land use change would occur in these areas or their buffer zone, this is a sign for potential ILUC risk. The observation of land use and land use change could be supported by monitoring land management, as this is a good indicator of how and for what purpose(s) land is used. Aspects of land management include, amongst others, the type (e.g. tillage or no tillage), intensity (e.g. full tillage or reduced tillage) and timing of management. However, collecting this type of data could be very time consuming. Second, changes in food and trade balances could be an indicator for increasing ILUC risk. For example, when agricultural production volumes increase at a higher rate than expected, the land area required for food, feed and bioenergy production is potentially larger than the area available without ILUC. In addition, changes in imports or exports of agricultural products might indicate growing production demand in the region or relocation of local production to other regions, which can both cause ILUC. Small changes compared to the projected production and trade volumes, however, should be considered to be within the uncertainty range of the projection. It should therefore be assessed what an appropriate threshold would be.

The key indicators to monitor the implementation of the ILUC mitigation measures are as follows. First, for above-baseline yield development, the most important parameter is the crop-specific annual yield. As yields fluctuate over time, it is recommended to monitor the

TABLE 4-8 | Parameters for monitoring ILUC risk and implementation of ILUC mitigation measures and data availability and quality for these parameters.

Monitoring	Parameter for monitoring	Availability of data	Quality of available data
ILUC risk	Land use and land cover	Good [45]	It needs to be assessed if the spatial and temporal resolution of the data is appropriate for monitoring
	Land management	Some aggregated data on provincial level, e.g. about amount of machinery and level of fertilizer use [21]	Potentially data for individual farmers[d]
	Food balance	Available at provincial and national level [21,33][a]	To be assessed
	Trade balance	Available at national level [33][b]	To be assessed
Above-baseline yield development	Annual yield	Good for most important products, statistics are updated annually [21]. For other products, data is lacking or only provided on aggregated level and/or for selected years [21,40]	Only provincial averages, no data for individual farmers[c].
	Farm size and management	Only aggregated data on provincial level [21]	Potentially data for individual farmers[d]
Increased food chain efficiency	Food losses in supply chain	Available per agricultural product, but only at national level and not specified per process step in the supply chain [33]	Poor, only estimates are provided and specification per process step is lacking
Land zoning and use of under-utilized land	Land use and land cover	Good for protected areas [48,49], no spatially explicit data available for under-utilized lands	It needs to be assessed if the temporal resolution of the data is appropriate for monitoring protected areas.

[a] On provincial level, data is only available for production volumes and for the most important agricultural products [21].
[b] Trade figures on province level were not found during this study. It is recommended to further investigate whether such data is already collected and available and how missing data can be collected in the future.
[c] A national agricultural census is taken every 6 to 8 years, but does not include yields [21].
[d] A national agricultural census is taken every 6 to 8 years and includes data about agricultural machinery [21]. Currently, the data is only provided at aggregated levels [21]. It should be further assessed if data for individual farmers can be made available and used for monitoring.

five-year moving average yield. In addition, the targeted yield should be defined as a range within which the average yield should be in a certain year. Statistics on provincial average crop yields are generally made available annually [21]. However, information about the performance of individual farmers is lacking. This information is useful to identify where efforts and investments for yield improvements are needed most. In addition, monitoring developments in farm size and management (e.g. the level and efficiency of fertilizers use)

is valuable to assess whether subsidies and other stimulating policies are effective, and whether advances in farm management are substantial enough to realize the expected or targeted yield improvements. Second, monitoring of food chain efficiency requires data on food losses in the whole supply chain, specified per agricultural product and per process in the chain. In this study, data availability and quality is poor (see Table 4-8 and section 4.2.2.3). More and better data need to be collected periodically to set targets for chain efficiency and monitor whether developments are in line with these targets. Third, land zoning and the use of under-utilized lands can be mainly monitored by periodically assessing land use and land cover (see also above). When remote sensing is used, the ability to differentiate between miscanthus and other crops is very important, because miscanthus may be cultivated in areas where the production of other crops is undesired or prohibited. As the information on the location and size of under-utilized lands is much more limited compared to protected areas, remote sensing and improved field data collection are important to set a baseline for monitoring this measure.

4.4 DISCUSSION AND CONCLUSIONS

4.4.1 Potential surplus land area

This case study assessed the production potential of miscanthus-based bioethanol with low ILUC risk in the Polish province of Lublin in 2020. Five measures have been analyzed that reduce the extent of ILUC and control the type of land use change. The total potential of these measures has been investigated for a low, medium and high scenario that refers to developments above the baseline projections. In 2020, a total area of 197 to 818 thousand hectare agricultural land could become available for biomass production. This is equal to 11% to 47% of the total agricultural area in Lublin. The largest potential to generate surplus land comes from *above-baseline yield developments* (95-413 thousand hectare). Increasing especially wheat yields adds significantly to the total potential of this measure. Also, the projected area of under-utilized land, 92 to 375 thousand hectare, is considerable. The large effect of these two ILUC mitigation measures illustrates the importance of improving land management. This finding is supported by assessments of land availability in Eastern Romania, Hungaria and North-East Kalimantan (Indonesia) [72] and in Brazil [73].

The potentials differ substantially between the scenarios and also the feasibility and likelihood of the scenarios vary significantly. For example, the yields applied in the high scenario are considered to be feasible based on existing farming practices in Germany. But it is questionable if the adoption of these practices can take place in the limited timeframe to 2020. Second, based on the disaggregation of results from the MIRAGE model, the production of crops in Lublin is projected to decline in both the baseline

and biofuel target scenario. This reduction strongly affects land use. The decline in crop production is primarily caused by a reduction in the cultivation of potatoes and cereals (except wheat and maize) as projected by MIRAGE for the EU27. These crops account for a significant part of the agricultural production in Lublin. Furthermore, according to MIRAGE, the production of especially oil crops (e.g. rapeseed, sunflower) and also other first generation bioenergy crops (e.g. wheat, maize, sugar beet) will increase. In the province of Lublin, however, the current production of oil crops and maize is very small. Therefore, the total decline in the production of potatoes and cereals (except wheat and maize) is larger than the total growth in the production of wheat, sugar beet and other crops (see appendix A4.2). But the resulting reduction in land use is not in line with recent developments in Lublin and Poland [21]. In addition, other competitive uses for released land, such as afforestation, exist. These are not taken into account in this analysis. It is recommended to further assess the potential pathways for crop production and land use and specifying under which conditions each scenario could be realized.

4.4.2 Legally available and suitable area

Although the surplus land area available in 2020 is potentially very large, a limited area of 203 to 269 thousand hectare (12-15% of the total agricultural area) is considered to be legally available and biophysically suitable for miscanthus production based on the criteria for protecting high conservation areas and minimum requirements for land suitability. As a result, in all scenarios, the amount of surplus land that could be used for miscanthus production is restricted. The limitation on land use is mainly caused by the suitability criteria and especially the sensitivity of miscanthus to water stress. However, this study only assessed the land suitability based on a few simple criteria like the minimum ground water level. It did not take into account other parameters such as soil characteristics [74] or the influence of the current vegetation and the conversion to miscanthus on the water balance and water availability. It is therefore recommended to further investigate how these factors affect the land suitability and the potential yield for miscanthus. The insights can be used to set a maximum area for growing miscanthus. In addition, lands that are only very marginally suitable for miscanthus may be suitable for other crops that have a higher tolerance to water stress, e.g. reed canary grass and switchgrass [70,71,75]. Agro-ecological zoning data [34] shows that in Poland, the soil suitability for reed canary grass, and to a lesser extent also switchgrass, is considerably higher than the suitability for miscanthus. Fischer et al. [18] found that a total of 61% of the agricultural land in Poland is moderately to very suitable for reed canary grass, miscanthus and/or switchgrass. Thus, selection of the most appropriate crop for each area could significantly increase the use of the surplus land area and raise the total biomass production potential.

4.4.3 Low-ILUC-risk bioethanol potential

Depending on the productivity of the bioethanol value chain, the low-ILUC-risk bioethanol production potential ranges from 12 to 35 PJ per year (522 to 1,479 million liter per year). For comparison, the national Polish target for 2nd generation bioethanol consumption is almost 9 PJ. This means that the province of Lublin could play a key role in achieving this target and help Poland even become an exporter of second generation bioethanol. This potential, however, is the technical potential that accounts only for key environmental aspects such as the protection of high conservation value areas. However, the (sustainable) implementation potential may be lower than the technical potential. The implementation potential is the fraction of the technical potential that can be produced at economically profitable levels and implemented within the considered timeframe, taking into account local constraints and policies [76]. For Lublin, several factors are identified that could significantly affect the implementation potential. First, the agricultural sector in Lublin is characterized by a large number of small farms and low average management levels compared to regions such as Western Poland and Germany (see Table 4-10 in appendix A4.1). To realize above-baseline yield increases, scaling up, modernization and intensification of agricultural production is needed. However, farmers have little capital to invest, and land prices are considered too low for selling or leasing land. Second, when the ILUC mitigation measures are implemented and land is made available for biomass production, several hurdles exist for farmers to start cultivating bioenergy crops. For example, in recent years, the production and trade of biomass for heat and electricity in Lublin province has been constrained by the lack of a stable market. Large amounts of biomass were imported from the Ukraine and, according to local experts, biomass prices offered to farmers in Lublin were too low [77-79]. With regard to miscanthus, a potential additional hurdle may be the high establishment costs compared to other energy crops [75]. The sustainable biofuel potential is the fraction of the technical potential that can be implemented while delivering positive environmental, social and economic impacts. To assess the sustainability of biofuels, sustainability criteria and indicators have been developed, see e.g. Cramer et al. [80], Franke et al. [81], McBride et al. [82] and Dale et al. [83]. It is unknown yet what will be the environmental and socio-economic impacts of implementing ILUC mitigation measures in Lublin. These aspects should be addressed in future research.

To maximize the implementation potential, governance and policies are considered vital. First, this could, for example, include financial support to farmers to facilitate improved production practices. Such support is already included in European and Polish agricultural and rural development policies [84,85], but should be increased to realize the full potential. Second, in the medium and high scenarios, it was assumed that miscanthus production in some protected areas with high conservation value may actually lead to improved

biodiversity. Therefore, land use policies should clearly define which areas are allowed to take into production for biomass. Third, it is recommended to further assess the potential barriers for implementing ILUC mitigation measures and producing bioenergy crops and biofuels at large scale. In addition, it should be investigated how these hurdles could be addressed. Fourth, monitoring ILUC risks and the implementation of ILUC mitigation measures is important. This case study identified several parameters that are useful for monitoring, e.g. land use, land cover and annual yields. However, the availability and quality of the data required for monitoring varies for the different parameters. Especially data about losses in the food supply chain and under-utilized lands should be improved. Finally, the assessment of the ILUC mitigation measures and the miscanthus-based bioethanol production potential with low ILUC risk in Lublin province in Poland shows that the mitigation or prevention of ILUC from bioenergy is only possible when the close link between the agricultural and bioenergy sectors is recognized. Therefore, an integrated perspective on these sectors in planning and implementing policies on ILUC prevention specifically (as well as on land use in general) is essential. Doing so would allow realizing a significant bioenergy potential with a low risk of causing ILUC while boosting the performance of the agricultural sectors as a whole.

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REFERENCES

[1] Observ'ER. Biofuels barometer. Observ'ER, Observatoire des énergies renouvelables: 2015.

[2] European Parliament and Council of the European Union. Directive 2003/30/EC of the European Parliament and of the Council of 8 May 2003 on the promotion of the use of biofuels or other renewable fuels for transport. Official Journal of the European Union L 123 2003;46:42-46.

[3] Wicke B, Verweij P, van Meijl H, et al. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 2012;3(1):87-100.

[4] Plevin RJ, O'Hare M, Jones AD, et al. Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. Environmental science & technology 2010;44(21):8015-8021.

[5] Searchinger T, Heimlich R, Houghton RA, et al. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. Science 2008;319(5867):1238-1240.

[6] Tyner WE, Taheripour F, Zhuang Q, et al. Land use changes and consequent CO2 emissions due to US corn ethanol production: a comprehensive analysis. Center for Global Trade Analysis, Purdue University: West Lafayette, IN, USA; 2010.

[7] Al-Riffai P, Dimaranan B and Laborde D. Global trade and environmental impact study of the EU biofuels mandate. International Food Policy Research Institute: Washington, DC, USA; 2010.

[8] Hertel TW, Golub AA, Jones AD, et al. Effects of US Maize ethanol on global land use and greenhouse gas emissions: Estimating market-mediated responses. Bioscience 2010;60(3):223-231.

[9] Laborde D. Assessing the land use change consequences of European biofuels policies. International Food Policy Research Institute: Washington, DC, USA; 2011. Available from: http://trade.ec.europa.eu/doclib/docs/2011/october/tradoc_148289.pdf

[10] EPA. Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis. Environmental Protection Agency: Washington, DC, USA; 2010. EPA-420-R-10-006.

[11] Warner E, Zhang Y, Inman D, et al. Challenges in the estimation of greenhouse gas emissions from biofuel-induced global land-use change. Biofuels, Bioproducts and Biorefining 2014;8(1):114-125.

[12] Plevin RJ, Beckman J, Golub AA, et al. Carbon Accounting and Economic Model Uncertainty of Emissions from Biofuels-Induced Land Use Change. Environmental science & technology 2015;49(5):2656-2664.

[13] Brinkman M, Wicke B, Gerssen-Gondelach S, et al. Methodology for assessing ILUC prevention. Copernicus Institute of Sustainable Development, Utrecht University: Utrecht, The Netherlands; 2014.

[14] van de Staaij J, Peters D, Dehue B, et al. Low Indirect Impact Biofuel (LIIB) Methodology - version Zero. 2012.

[15] Witcover J, Yeh S, Sperling D. Policy options to address global land use change from biofuels. Energy Policy 2013;56:63-74.

[16] de Wit M and Faaij A. European biomass resource potential and costs. Biomass and Bioenergy 2010;34(2):188-202.

[17] Pudełko R, Borzęcka-Walker M, Faber A, et al. The technical potential of perennial energy crops in Poland. Journal of Food, Agriculture & Environment 2012;10(2):781-784.

[18] Fischer G, Prieler S, van Velthuizen H, et al. Biofuel production potentials in Europe: Sustainable use of cultivated land and pastures. Part I: Land productivity potentials. Biomass and Bioenergy 2010;34(2):159-172.

[19] Faber A, Pudełko R, Borek R, et al. Economic potential of perennial energy crops in Poland. Journal of Food, Agriculture & Environment 2012;10(3&4):1178-1182.

[20] Szymańska D and Chodkowska-Miszczuk J. Endogenous resources utilization of rural areas in shaping sustainable development in Poland. Renewable and Sustainable Energy Reviews 2011;15(3):1497-1501.

[21] CSO. Local Data Bank [Internet: updated 2014, accessed 2013, 2014]. Available from: http://www.stat.gov.pl/bdlen/app/strona.html?p_name=indeks

[22] Eurostat. Eurostat - Data Explorer. Key farm variables: area, livestock (LSU), labour force and standard output (SO) by agricultural size of farm (UAA), legal status of holding and NUTS 2 regions [Internet: updated 2013, accessed 29 July 2014]. Available from: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ef_kvaareg&lang=en

[23] Agricultural Sustainability Institute U. What is Sustainable Agriculture? [Internet: accessed May 7 2015]. Available from: http://www.sarep.ucdavis.edu/about-sarep/def

[24] Dauber J, Jones MB, Stout JC. The impact of biomass crop cultivation on temperate biodiversity. GCB Bioenergy 2010;2(6):289-309.

[25] Gerssen-Gondelach S, Wicke B and Faaij A. ILUC prevention strategies for sustainable biofuels. Case study on the bioethanol production potential from miscanthus with low ILUC risk in the province of Lublin, Poland. Utrecht University: Utrecht, The Netherlands; 2014.

[26] Brinkman MLJ, Wicke B, Gerssen-Gondelach SJ, et al. Methodology for assessing and quantifying ILUC prevention options. Copernicus Institute of Sustainable Development, Utrecht University: Utrecht, The Netherlands; 2015.

[27] Ray DK, Ramankutty N, Mueller ND, et al. Recent patterns of crop yield growth and stagnation. Nat Commun 2012;3:1293.

[28] Gerssen-Gondelach S, Wicke B, Faaij A. Assessment of driving factors for yield and productivity developments in crop and cattle production as key to increasing sustainable biomass potentials. Food and Energy Security 2015;4(1):36-75.

[29] Observ'ER. Biofuels barometer. Systémes solaires le journal des énergies renouvelables 2011;204:68-93.

[30] Observ'ER. Biofuels barometer. Systémes solaires le journal des énergies renouvelables 2012;210:42-62.

[31] Observ'ER. Biofuels barometer. Systémes solaires le journal des énergies renouvelables 2013;216:48-63.

[32] Ministry of Economy. National Renewable Energy Action Plan. Ministry of Economy: Warsaw, Poland; 2010.

[33] FAO. FAOSTAT [Internet: updated 2014, accessed 2014]. Available from: http://faostat.fao.org

[34] FAO and IIASA. GAEZ Global Agri-Ecological Zones [Internet: updated 2014, accessed 19 February 2014]. Available from: http://gaez.fao.org/Main.html#

[35] Patton J. Value-added Coproducts from the Production of Cellulosic Ethanol. North Dakota State University: n.d. Available from: http://www.ag.ndsu.edu/CentralGrasslandsREC/biofuels-research-1/Cellulosic_Ethanol%20_Coproducts.pdf

[36] Finnveden G, Hauschild MZ, Ekvall T, et al. Recent developments in Life Cycle Assessment. Journal of environmental management 2009;91(1):1-21.

[37] Ekvall T and Weidema BP. System boundaries and input data in consequential life cycle inventory analysis. The International Journal of Life Cycle Assessment 2004;9(3):161-171.

[38] Reinhard J and Zah R. Consequential life cycle assessment of the environmental impacts of an increased rapemethylester (RME) production in Switzerland. Biomass and Bioenergy 2011;35(6):2361-2373.

[39] Rutten M, Nowicki P, Boogaardt M-, et al. Reducing food waste by households and in retail in the EU: A prioritisation using economic, land use and food security impacts. LEI Wageningen UR: The Hague; 2013. 2013-035.

[40] CSO. Statistical yearbook of agriculture 2013. Central Statistical Office (CSO), Agricultural department: Warsaw, Poland; 2014.

[41] CSO. Rural Areas in Poland - National Agricultural Census 2010. Central Statistical Office (CSO): Warsaw, Poland; 2013.

[42] Eurostat. Eurostat - Data Explorer. Fallow land and set-aside land: number of farms and areas by size of farm (UAA) and size of arable area [Internet: updated 2012, accessed 15 August 2014]. Available from: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ef_lu_ofsetasid&lang=en

[43] FAO. Chapter 1. Introduction. In: Chapter 1. Introduction. Fertilizer use by crop in Poland. Food and Agriculture Organisation of the United Nations (FAO): 2003, p. 1-14.

[44] IUNG. Agricultural Soil Suitability Map of Poland. Scale 1:100,000. 1974.

[45] Nunes de Lima MV. Image 2000 and CLC 2000 – Products and Methods. CORINE Land Cover updating for the year 2000. 2005.

[46] Institute of Geology. Hydrogeological Map of Poland. Scale 1:300,000. 1957.

[47] Górski T and Zaliwski A. Agroclimate model of Poland (in Polish). Pamiêtnik Puławski 2002;130:251-260.

[48] Ministry of the Environment. National System of Protected Areas (KSOCH) 2003. 2003.

[49] European Commission. Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Official Journal of the European Communities 1992;206:7-50.

[50] Caslin B, Finnan J, Easson L. 1 Crop production. In: Caslin B, Finnan J, Easson L, editors. Miscanthus best practice guidelines. Teagasc and Agri-Food and Bioscience Institute: Carlow and Hillsborough, Ireland; 2011, p. 5-26.

[51] Sliz-Szkliniarz B. 4 Methodological Approach for Bioenergy Potential Assessment. In: Sliz-Szkliniarz B, editor. Energy Planning in Selected European Regions - Methods for Evaluating the Potential of Renewable Energy Sources. Volume 2013, Karlsruher Institut für Technologie (KIT): 2013, p. 27-116.

[52] Terelak H and Witek T. Poland. In: Zinck JA, editor. Soil survey: perspectives and strategies for the 21st century. Land and Water Development Division, FAO: Rome; 1995, p. 100-103.

[53] European Parliament and Council of the European Union. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union 2009;140:16-62.

[54] Smeets EMW, Lewandowski IM, Faaij APC. The economical and environmental performance of miscanthus and switchgrass production and supply chains in a European setting. Renewable and Sustainable Energy Reviews 2009;13(6-7):1230-1245.

[55] Donnelly A, Styles D, Fitzgerald J, et al. A proposed framework for determining the environmental impact of replacing agricultural grassland with Miscanthus in Ireland. GCB Bioenergy 2011;3(3):247-263.

[56] de Wit M, Londo M, Faaij A. Productivity developments in European agriculture: Relations to and opportunities for biomass production. Renewable and Sustainable Energy Reviews 2011;15(5):2397-2412.

[57] Stampfl PF, Clifton-Brown JC, Jones MB. European-wide GIS-based modelling system for quantifying the feedstock from Miscanthus and the potential contribution to renewable energy targets. Global Change Biology 2007;13(11):2283-2295.

[58] Borkowska H and Molas R. Yield comparison of four lignocellulosic perennial energy crop species. Biomass and Bioenergy 2013;51:145-153.

[59] Matyka M and Kus J. Yielding and biometric characteristics of selected miscanthus genotypes. Problemy Inżynierii Rolniczej 2011;2:157-163.

[60] van Dam J, Faaij APC, Lewandowski I, et al. Biomass production potentials in Central and Eastern Europe under different scenarios. Biomass and Bioenergy 2007;31(6):345-366.

[61] Monti A, Fazio S, Venturi G. The discrepancy between plot and field yields: Harvest and storage losses of switchgrass. Biomass and Bioenergy 2009;33(5):841-847.

[62] Shinners KJ, Boettcher GC, Muck RE, et al. Harvest and Storage of Two Perennial Grasses as Biomass Feedstocks. Transactions of the ASABE 2010;53(2):359-370.

[63] Bansal A, Illukpitiya P, Singh SP, et al. Economic competitiveness of ethanol production from cellulosic feedstock in Tennessee. Renewable Energy 2013;59:53-57.

[64] Hamelinck CN and Faaij APC. Outlook for advanced biofuels. Energy Policy 2006;34(17):3268-3283.

[65] Tao L and Aden A. The economics of current and future biofuels. In Vitro Cellular & Developmental Biology - Plant 2009;45(3):199-217.

[66] Aden A, Ruth M, Ibsen K, et al. Lignocellulosic Biomass to Ethanol Process Design and Economics Utilizing Co-Current Dilute Acid Prehydrolysis and Enzymatic Hydrolysis for Corn Stover. National Renewable Energy Laboratory: Golden, Colorado, USA; 2002. Available from: http://www.nrel.gov/docs/fy02osti/32438.pdf

[67] POET-DSM Advanced Biofuels. First commercial-scale cellulosic ethanol plant in the U.S. opens for business - POET-DSM Advanced Biofuels [Internet: updated 2014, accessed 19 December 2014]. Available from: http://poetdsm.com/pr/first-commercial-scale-cellulosic-plant

[68] Abengoa Bioenergy. Abengoa celebrates grand opening of its first commercial-scale next generation biofuels plant [Internet: updated 2014, accessed 19 December 2014]. Available from: http://www.abengoabioenergy.com/web/en/prensa/noticias/historico/2014/abg_20141017.html

[69] Mioduszewski W. Small (natural) water retention in rural areas. Journal of Water and Land Development 2014;20(1):19-29.

[70] Lewandowski I, Clifton-Brown JC, Scurlock JMO, et al. Miscanthus: European experience with a novel energy crop. Biomass and Bioenergy 2000;19(4):209-227.

[71] Richter GM, Riche AB, Dailey AG, et al. Is UK biofuel supply from Miscanthus water-limited? Soil Use and Management 2008;24(3):235-245.

[72] Wicke B, Brinkman M, Gerssen-Gondelach S, et al. ILUC prevention strategies for sustainable biofuels: Synthesis report from the ILUC Prevention project. Copernicus Institute of Sustainable Development, Utrecht University: Utrecht, The Netherlands; 2015.

[73] Woods J, Lynd LR, Laser M, et al. Chapter 9. Land and Bioenergy. In: Souza GM, Victoria RL, Joly CA, Verdade LM, editors. Bioenergy & Sustainability: bridging the gaps. Scientific Committee on Problems of the Environment (SCOPE): Paris, France; 2015, p. 258-300.

[74] van der Hilst F, Dornburg V, Sanders JPM, et al. Potential, spatial distribution and economic performance of regional biomass chains: The North of the Netherlands as example. Agricultural Systems 2010;103(7):403-417.

[75] Lewandowski I, Scurlock JMO, Lindvall E, et al. The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. Biomass and Bioenergy 2003;25(4):335-361.

[76] Smeets EMW, Faaij APC, Lewandowski IM, et al. A bottom-up assessment and review of global bio-energy potentials to 2050. Progress in Energy and Combustion Science 2007;33(1):56-106.

[77] Gradziuk P [personal communication]. 2014. Polish Chamber of Biomass, Poland.

[78] Galczynska M [personal communication]. 2014. Foundation for Lubelskie Development, Poland.

[79] Faber A [personal communication]. 2014. Institute of Soil Science and Plant Cultivation (IUNG), Pulawy, Poland.

[80] Cramer J, Wissema E, de Bruijne M, et al. Testing framework for sustainable biomass, final report from the project group "Sustainable production of biomass". Project group Sustainable Production of Biomass: The Hague, The Netherlands; 2007.

[81] Franke B, Reinhardt G, Malavelle J, et al. Global Assessments and Guidelines for Sustainable Liquid Biofuel Production in Developing Countries. A GEF Targeted Research Project. Heidelberg/Paris/Utrecht/Darmstadt; 2012.

[82] McBride AC, Dale VH, Baskaran LM, et al. Indicators to support environmental sustainability of bioenergy systems. Ecological Indicators 2011;11(5):1277-1289.

[83] Dale VH, Efroymson RA, Kline KL, et al. Indicators for assessing socioeconomic sustainability of bioenergy systems: A short list of practical measures. Ecological Indicators 2013;26:87-102.

[84] Ministry of Agriculture and Rural Development. Program Rozwoju Obszarów Wiejskich 2014-2020 [Internet: accessed 26 September 2014]. Available from: https://www.minrol.gov.pl/Wsparcie-rolnictwa-i-rybolowstwa/PROW-2014-2020

[85] European Commission. Rural development 2014-2020 - Agriculture and rural development [Internet: updated 2014, accessed 29 September 2014]. Available from: http://ec.europa.eu/agriculture/rural-development-2014-2020/index_en.htm

[86] Zastawny J. Country Pasture/Forage Resouce Profiles. Poland. 3. Climate and Agro-ecological zones [Internet: updated 2000, accessed 5 September 2014]. Available from: http://www.fao.org/ag/AGP/AGPC/doc/counprof/poland.htm#3

[87] CSO. Agriculture in 2012. Central Statistical Office (CSO), Agricultural department: Warsaw, Poland; 2013.

[88] Statistical Office in Lublin. Agriculture and Forestry. In: Agriculture and Forestry. Statistical yearbook of Lubelskie Voivodship 2012. Statistical Office in Lublin: 2012.

[89] Statistical Office in Lublin. Chapter XV: Agriculture, Hunting and Forestry. In: Statistical yearbook of Lubelskie Voivodship 2013. Statistical Office in Lublin: 2013, p. 292-320.

[90] Eurostat. Number of agricultural holdings [Internet: updated 2014, accessed 23 October 2014]. Available from: http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&language=en&pcode=tag00001&plugin=1

[91] Searle SY and Malins CJ. Will energy crop yields meet expectations? Biomass and Bioenergy 2014;65:3-12.

[92] Allen D [personal communication]. 2014. Shell.

[93] Gerssen-Gondelach SJ, Saygin D, Wicke B, et al. Competing uses of biomass: Assessment and comparison of the performance of bio-based heat, power, fuels and materials. Renewable and Sustainable Energy Reviews 2014;40:964-998.

[94] Dale BE, Allen MS, Laser M, et al. Protein feeds coproduction in biomass conversion to fuels and chemicals. Biofuels, Bioproducts and Biorefining 2009;3(2):219-230.

[95] Bals BD, Dale BE, Balan V, et al. Recovery of Leaf Protein for Animal Feed and High-Value Uses. In: Recovery of Leaf Protein for Animal Feed and High-Value Uses. Biorefinery Co-Products. John Wiley & Sons, Ltd: 2012, p. 179-197.

[96] Norman AC and Murphy MR. Feed Value And In Situ Dry Matter Digestibility Of Miscanthus X Giganteus And Corn Stover [abstract]. 2005.

APPENDICES

A4.1 Characteristics of case study region

The province of Lublin is characterized by lowlands in the North and uplands in the South. The climate is continental and temperate and is characterized by warm summers and cold winters. The winters are colder and longer than in Western Polish regions, which have a more maritime climate [86]. Table 4-9 presents the importance of different land uses in the province. More than 70% of the total area is designated as agricultural land, but not all of this land is under agricultural activity. Within the province, there are large variations in the slopes of agricultural lands. Also, the land is characterized by a mosaic structure of diverse soil suitability classes [52]. Table 4-10 compares the characteristics of the agricultural sector in Lublin to Poland and Germany.

TABLE 4-9 | Land uses in Lublin province [40,41].

	2010 (1000 ha)		**2012 (1000 ha)**
Total area	**2415**	**Total agricultural area**	**1767**
Total agricultural area	1725	Area under agricultural use[a]	1407
Build-up and urbanized	65	Area in good condition	1377
Forests	561	-Sown	1027
Woody & bushy land	19	-Fallow	34
Ecological areas	4	-Permanent crops	76
Wasteland	22	--Of which orchards	73
Land under waters	17	-Kitchen gardens	9
Other	2	-Permanent meadows	210
		-Permanent pastures	22
Legally protected areas	570	-Other	30
Devastated and degraded land	3.1		

[a] Total agricultural land area excluding owners of agricultural land who do not conduct agricultural activities and owners of less than 1 ha of agricultural land who conduct agricultural activities on a small scale [87-89].

TABLE 4-10 | Characteristics of agriculture in Lublin and comparison to Poland and Germany.

	Lublin	Poland	Range Polish provinces	Germany
Farm size 2010 [22]				
Total number of farms	190,070	1,506,620		299,130
Less than 2 ha	21%	24%		5%
2 - 4.9 ha	35%	31%		4%
5 - 9.9 ha	27%	22%		16%
10 - 19.9 ha	13%	15%		21%
20 - 29.9 ha	3%	4%		10%
30 - 49.9 ha	1%	2%		15%
50 - 99.9 ha	1%	1%		17%
100 ha or over	0%	1%		11%
Average farm size (ha agricultural land per holding)	7.1	9.6	3.7-29.4	55.8
Technology status 2010 [21,33]				
No. of tractors per 100 ha[a]	12.5	9.9 (9.6)	3.5-19.2	(4.5)
No. of tractors per holding[a]	0.7	0.8 (0.6)	0.5-1.1	(2.1)
No. of other machinery per holding	2.3	2.3	1.1-3.6	n.a.
Total NPK fertilizer use (kg ha^{-1} yr^{-1})[b,c]	104	115 (134)	56-182	(150)
Nitrogen fertilizer use (kg ha^{-1} yr^{-1})[b,c]	57	66 (75)	30-100	(107)
Agricultural employment 2012 [21]				
Employment in agriculture (persons per 100 ha)[c]	21.7	15.6(19.3)	5.1-47.1	(3.7)
% of labor force employed in agricultural sector[c]	26.5	12.0(16.2)	2.6-26.5	(1.4)
Average yields 2008-2012 [21]				
Wheat (t ha^{-1} yr^{-1})	3.7	4.2	3.2-5.7	7.5
Sugarbeet (t ha^{-1} yr^{-1})	51.2	52.9	45.2-59.5	67.5

n.a. not available
[a] Figures between brackets are for 2007 and derived from FAOSTAT [33] and Eurostat [90]
[b] NPK: nitrogen (N), phosphorus (P), and potassium (K), total fertilizer use is total amount of N, P_2O_5 and K_2O
[c] Figures between brackets are derived from FAOSTAT [33]

A4.2 Projection agricultural production

Table 4-11 presents the projected production and land use of crops, beef and milk in Lublin in the baseline and target scenario for 2020, based on MIRAGE and the baseline yield projections. In the target scenario, the production of crops required for first generation biofuels under a biofuel mandate is included.

Currently, the total agricultural area needed for the production of the selected crops and for cattle covers 87% of the utilized agricultural land area (1,409 thousand ha) in Lublin province. When applying the baseline projections for crop yields, beef and milk productivity and cattle density, agricultural land use reduces by 290 thousand ha in the baseline scenario and 280 thousand ha in the target scenario. The largest share of this decrease can be attributed to cropland.

TABLE 4-11 | Current crop production in Lublin and projected production in 2020 in the MIRAGE-based baseline (without biofuel target) and biofuel target scenarios.

		Current[a]	**2020 baseline**[b]	**2020 target**[b]
		Production (kt/yr)	Production (kt/yr)	Production (kt/yr)
Crops	Sugar beets	1,601	1716	1991
	Wheat	1,052	1210	1203
	Potatoes	831	528	527
	Barley	445	283	282
	Mixed cereals	403	256	256
	Apples	365	408	407
	Triticale	328	209	208
	Rye	208	132	132
	Oats	185	118	117
	Maize	110	124	123
	Rape and turnip rape	94	128	146
	Total Crops	5,621	5,112	5,390
Livestock	Beef	55	46	46
	Milk (Mliter/yr)	779	780	780
		Total land use (kha)	Total land use (kha)	Total land use (kha)
Crops[c]		983	745	755
Livestock[d]		240	189	189
Total		1,224	934	943

[a] The current production and land area are defined as the average over 2008-2012, and are applied as the production and land area level in 2010.
[b] The production and land use for crops include the production of crops for biofuels. Because of the introduction of the biofuel mandate in the target scenario, the total crop production is larger in the target scenario compared to the baseline. For cattle, no additional production takes place in the target scenario compared to the baseline scenario.
[c] Land use for crops: area of cropland required for the projected production of crops.
[d] Land use for cattle: area of meadows and pastures for the projected production of beef and milk.

A4.3 Biofuel chain design and efficiencies

For this case study, the miscanthus variety *Miscanthus x giganteus* is considered. *M. x giganteus* grows well in a temperate climate zone, but is sensitive to extreme cold and hot temperatures [91]. *M. x giganteus* is especially sensitive to frost in the first year of establishment [75]. Cold winters and late frost in Lublin can result in high losses and a significant yield reduction. Yet, the main problem related to weather conditions in Lublin is late snowfall. The snow causes lodging, i.e. breaking or bending of the upper stem parts, which hinders the harvest and reduces harvest efficiency.

Miscanthus is either harvested in autumn or spring. In spring, dry biomass yields are about one third lower compared to autumn because of biomass decay, but the biomass quality is higher due to a reduced moisture and nutrient content [54,57]. Yields attain a plateau

from the third year after establishment and start to decline after a stand length of about 10 years [91]. In Poland, the maximum attainable plateau yield for miscanthus is about 30 t dm/ha in autumn and 20 t dm/ha in spring, see Table 4-12. This is attainable on very suitable lands and under optimal farming conditions (i.e. optimal farming practices, perfect weed control, no nutrient limitation, no pests or diseases, high quality rhizomes, no stand loss during first winter) [92]. On lands of lower suitability, miscanthus yields will be lower. Because of the time needed to establish plantations and optimize farming conditions, it is very likely that yields will be below the maximum yield in 2020. Therefore, here an average yield of 13 t dm/ha is assumed based on [57-60]. For the sensitivity analysis, a low and high yield of respectively 10 and 17 t dm/ha are applied.

TABLE 4-12 | Miscanthus yield potentials for various classes of land suitability.

Land suitability	Autumn yield range ($t\ dm\ ha^{-1}$)	Spring yield range ($t\ dm\ ha^{-1}$)	References
Very good	Up to 30	Up to 20	[57,58]
Good	15-21	10-14	[59,60]
Moderate	n/a	7-11	[60]
Very marginal	n/a	≤3	[60]

n/a, no data available

Miscanthus can be converted to biofuel via two main processes: biochemical or thermochemical conversion. A short review of recent literature and technological developments shows that the biochemical route is most advanced at this moment [67,68,93]. This process is therefore selected for this case study. In current commercial bio-ethanol plants, the production scale is 76-95 million liter per year [67,68].

Biochemical conversion of miscanthus (or other lignocellulosic feedstocks) to ethanol has several by-products. Lignin, a major component of the biomass in terms of weight, cannot be fermented. Currently, the main use is the combustion of lignin for the coproduction of steam and electricity. Proteins from herbaceous biomass sources can potentially be used as feed in livestock diets, as is already done with protein-rich product streams from corn and sugarcane ethanol plants [94]. The process for protein recovery, however, is not mature yet. Also, the extraction of proteins seems more promising for other herbaceous grasses than for miscanthus because proteins are translocated back from the leaves to the roots during winter and the feed quality of Miscanthus x giganteus is lower compared to other feed sources [94-96].

A4.4 Estimations of under-utilized land area

To estimate the current area of under-utilized land, the following data is used. First, Eurostat data shows that 2.4% of the total utilized agricultural area (UAA) in Poland was set-aside under an incentive scheme in 2007 [42]. Also, 1.8% of the UAA was classified as non-subsidized fallow land [42]. When it is assumed that the same percentages can be applied to the province of Lublin, 33 thousand ha of the utilized agricultural land in the case study area would be set-aside land under an incentive scheme, including 15 thousand hectare fallow land (Table 4-13). Another 26 thousand ha would be non-subsidized follow land. Then, the total area of fallow land is 40 thousand ha, which is higher compared to the 33 thousand ha designated as fallow land in 2012 [21]. Therefore, the area of fallow land not under an incentive scheme is estimated to be in the range of 15-25 thousand hectare. Second, figures from FAO and CSO suggest that the area of set-aside land in the province of Lublin declined after 2004 (since Poland entered the EU, see Table 4-13). Depending on how this trend developed after 2006, it is assumed that the total area of set-aside land in 2012 ranged between 30-50 thousand ha.

In addition to set-aside and fallow land, there may be a significant area of agricultural land that is currently not considered to be under agricultural activity but is potentially available for miscanthus production. Agricultural statistics for the case study region show that 360 thousand ha of agricultural land was not in agricultural use in 2012. Of this, 75 thousand ha was built up and under ponds and ditches [41]. The remaining 285 thousand ha at least includes land held by owners who do not conduct agricultural activities and owners who possess less than 1 ha of agricultural land and only conduct agricultural activities on a small scale [88,89]. A part of the land held by owners who do not conduct agricultural activities could potentially be considered as abandoned land suitable for miscanthus production. In 2007, there were about 10,000 holdings with non-agricultural activities [21]. The area of land they possessed is not found in statistical data. Assuming that the average agricultural area owned by these holdings is similar or smaller compared to the average farm size in Lublin province, about 70 thousand ha or less is possessed by these holdings. It is assumed that about 10 to 25% of this area, i.e. approximately 5 to 20 thousand ha, can be considered as abandoned land.

TABLE 4-13 | Estimations of area of land potentially available and suitable for miscanthus production that is currently designated as set-aside land or agricultural land not under agricultural activity.

	Area Poland (1000 ha)	% of UAA[a]	Area Lublin (1000 ha)	Reference
Set-aside land, estimate 1				
Set-aside under incentive scheme	367	2.4	33	Estimate based on Eurostat [74]
Of which fallow land	160	1.0	15	Idem
Fallow land not under incentive scheme	281	1.8	26	Idem
Estimation 1a: Set-aside under incentive scheme			30-35	
Estimation 1b: Fallow land not under incentive scheme			15-25	
Set-aside land, estimate 2				
Historical set-aside land area:				
1999-2001			86	[75]
2004			93	[10]
2005			61	[10]
2006			57	[10]
Estimation 2: set-aside area 2012			30-50	
Total estimated set-aside and fallow land area (based on estimates 1 and 2)			45-75	
Abandoned land				
Total agricultural area not in use 2012			360	[10]
Built up and under ponds and ditches			75	[10]
Estimated land area possessed by non-agricultural holdings			<70	Own estimate[b]
Total estimated abandoned land area			5-20	
Total under-utilized land				
Total estimated area			**50-95**	

[a] UAA, utilized agricultural area; [b] see main text in this appendix.

CHAPTER 5

GHG emissions and other environmental impacts of ILUC mitigation

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ABSTRACT

The implementation of measures to increase productivity and resource efficiency in food and bioenergy chains as well as to more sustainably manage land use can significantly increase the biofuel production potential while limiting the risk of causing ILUC. However, the application of these measures may influence the GHG balance and other environmental impacts of agricultural and biofuel production. This study applies a novel, integrated approach to assess the environmental impacts of agricultural and biofuel production for three ILUC mitigation scenarios, representing a low, medium and high miscanthus-based ethanol production potential, and for three agricultural intensification pathways in terms of sustainability in Lublin province in 2020. Generally, the ILUC mitigation scenarios attain lower net annual emissions compared to a baseline scenario that excludes ILUC mitigation and bioethanol production. However, the reduction potential significantly depends on the intensification pathway considered. For example, in the medium ILUC mitigation scenario, the net annual GHG emissions in the case study are 2.3 $MtCO_2$-eq yr^{-1} (1.8 tCO_2-eq ha^{-1} yr^{-1}) for conventional intensification and -0.8 $MtCO_2$-eq yr^{-1} (-0.6 tCO_2-eq ha^{-1} yr^{-1}) for sustainable intensification, compared to 3.0 $MtCO_2$-eq yr^{-1} (2.3 tCO_2-eq ha^{-1} yr^{-1}) in the baseline scenario. In addition, the intensification pathway is found to be more influential for the GHG balance than the ILUC mitigation scenario, indicating the importance of how agricultural intensification is implemented in practice. Furthermore, when the net emissions are allocated to bioenergy, the ILUC mitigation scenarios often abate GHG emissions compared to gasoline. But sustainable intensification is required to attain GHG abatement potentials of 90% or higher. A qualitative assessment of the impacts on biodiversity, water quantity and quality, soil quality and air quality also emphasizes the importance of sustainable intensification.

5.1 INTRO

Emissions from indirect land use change (ILUC) as a result of expanding biofuel production can reduce or even cancel out the GHG mitigation potential of biofuels [1]. Therefore, mitigation or prevention of ILUC and its impacts is essential for sustainable biofuel production. The extent of ILUC can significantly be reduced and the type of LUC be controlled when taking an integrated perspective on all land use, whether for food, feed, fibre and fuels [2,3]. Through the implementation of different ILUC mitigation measures, a large amount of additional biofuels can be produced with a low risk of causing ILUC [2,3]. However, besides the reduction of ILUC and associated emissions, the implementation of the ILUC mitigation strategies may also have other environmental impacts (e.g. on GHG emissions from agriculture or on biodiversity). These impacts are not yet well understood.

Agricultural intensification is suggested to play a key role in mitigating ILUC [3-7]. The reason is that agricultural intensification reduces the area of land required for food and feed production, which potentially results in surplus agricultural land that can be used for biomass production. The GHG and environmental impacts of agricultural intensification, however, will significantly depend on the intensification pathway. For example, Valin et al. [8] assess the effects of crop yield and livestock feed efficiency scenarios on the GHG balance of agricultural production in developing countries. They find that when above-baseline gains in crop yield are attained by intensive fertilizer application, the global GHG savings compared to the baseline are about 450 $MtCO_2$-eq yr^{-1} in 2050. In the case of sustainable intensification, i.e. through practices that improve crop yields without additional synthetic fertilizer, the emission savings are one third higher [8]. In addition, Valin et al. [8] find that, on a global level, improvements in livestock production have a larger effect on GHG mitigation than the intensification of crop production. Yet, they do not investigate the effect of energy crop production expansion on the total GHG balance of the agricultural and bioenergy sector [8]. This effect is examined by De Wit et al. [9], Melillo et al. [10] and van der Hilst et al. [11]. De Wit et al. [9] and Melillo et al. [10] assess the net GHG impacts of bioenergy expansion while mitigating ILUC through agricultural intensification on a European and global scale, respectively. But they include only one intensification pathway for crop production, and they exclude intensification in the livestock sector. Also, their GHG balances are not very detailed as these only account for nitrogen emissions, net soil organic carbon (SOC) fluxes and abated fossil emissions [9,10]. Van der Hilst et al. [11] perform a more detailed regional study of the net GHG balance in Ukraine. Valin et al. [8] show that such a regional study is important because the GHG impacts of agricultural intensification depend on region specific factors such as, for example, the degree of intensification possible based on the current yield gap. Van der Hilst et al. [11] include two intensification pathways for crops. The second, sustainable

intensification pathway includes a few GHG mitigation measures like reduced tillage, but both pathways assume balanced fertilization of crops. Therefore, the impacts of different agricultural intensification pathways, such as the change in GHG emissions due to different nutrient use efficiencies, are not properly evaluated. In other regional case studies that perform a detailed assessment of the GHG balance of bioenergy production, the impacts of agricultural intensification are not, or only partly, taken into account [12,13] or assumed to be negligible because intensification only takes place through improved management [14]. But in addition to GHG emissions, these case studies do assess other environmental impacts of bioenergy production [12-14]. Agricultural intensification can also have other environmental impacts that can be both positive and negative, and these impacts should be investigated as well. Therefore, the aim of this chapter is to assess for a specific region and biofuel case 1) the net GHG balance of agricultural and biofuel production under different ILUC mitigation scenarios, and 2) the influence of different intensification pathways on the GHG balance and other environmental impacts. This study is conducted for the case of miscanthus-based ethanol production in Lublin province, Poland.

5.2 METHODS

5.2.1 Inputs from case study ILUC mitigation

The present study builds on a previous study by Gerssen-Gondelach et al. [3] who assessed the low-ILUC-risk production potential of miscanthus-based ethanol in the province of Lublin in 2020. Gerssen-Gondelach et al. [3] analyzed five ILUC mitigation measures that reduce the extent of ILUC or control the type of land use change and calculated how these strategies could contribute to the availability of surplus agricultural land on which miscanthus can be produced without causing undesired LUC. As a reference, a baseline scenario was defined for agricultural land use in 2020. This baseline accounted for the agricultural production required to fulfill the projected demand for food, feed and fibres and the mandated production of first generation biofuels in the EU [15]. The baseline scenario also assumed that agricultural yields would develop in line with recent historical trends [3]. To assess the ILUC mitigation potential, three scenarios (low, medium and high) were defined in which four ILUC mitigation measures were applied to different degrees. These measures are: above-baseline yield development, improved food chain efficiency (i.e. reduced losses during storage and transportation), use of under-utilized land and land zoning. The fifth measure, increased integration of the food and biofuel chains, was not applicable to this case study. Table 5-1 presents the resulting land uses in the different scenarios for 2020 compared to 2010. The resulting miscanthus-based ethanol production potential in the three ILUC mitigation scenarios ranged from 16 to 23 PJ yr^{-1} for an average miscanthus yield of 13 t dm ha^{-1} yr^{-1} [3].

TABLE 5-1 | Current land use in agriculture (2010) and projected land use change in 2020 in the baseline and ILUC mitigation scenarios [3].

	(1000 ha)	2010	2020			
			Baseline[c]	ILUC mitigation		
				Low	Medium	High
Cropland	Remaining cropland	983	755-949	875	740	445
	Conversion to miscanthus	0	0	108	171	189
	Conversion to abandoned land	0	34-229	0	72	350
Grassland[a]	Remaining grassland	240	189-233	158	114	56
	Conversion to miscanthus	0	0	39	65	80
	Conversion to abandoned land	0	8-51	0	18	51
Under-utilized land	Remaining under-utilized	50-95	50-95	4	75	95
	Conversion to miscanthus	0	0	46	0	0
	TOTAL[b]	1,274-1,319	1,274-1,319	1,274	1,298	1,319

[a] Meadows and pastures; [b] The total agricultural area in Lublin was 1,745 thousand ha in 2010. The agricultural area included here accounts for the most important crops in terms of land use and for meadows and pastures; [c] Based on the projected agricultural production and yield developments in the baseline scenario, a large area of cropland and grassland (280 thousand ha) was calculated to be abandoned. However, this is a large reduction in a short time frame. Therefore, in the low and medium ILUC mitigation scenarios, the baseline reduction in agricultural land use is assumed to be lower.

5.2.2 GHG emissions

To assess the impact of ILUC mitigation on the total GHG balance of agricultural and bioenergy production, the annual emissions from these sectors are calculated for 2010 and for each scenario in 2020. This study does not perform a complete life cycle inventory, but includes the key GHG sources for which the emissions change due to the implementation of ILUC mitigation measures[1] and bioenergy production (Figure 5-1). The calculations of the GHG emissions are further explained in the following sub-sections.

As above-baseline yield developments (i.e. intensification) in crop and livestock production were found to play an important role in ILUC mitigation, it is also assessed how three different intensification pathways in term of sustainability influence the GHG balance of each ILUC mitigation scenario.

1 Changes in GHG emissions resulting from increased chain efficiencies in food storage and transport are only partly included. First, as a result of higher efficiencies, food production can be reduced to fulfill the same demand. The changes in GHG emissions due to reduced production are taken into account. Second, efficiency improvements also lead to reduced fuel consumption during storage and transport. However, this is not included because the food losses during storage and transport are generally low (2.5% or less of the total food weight [3,16]) and reductions in these losses will thus have a limited effect on fuel use. As the share of emissions from storage and transport in the total GHG balance of the food chain is considered to be small as well [17], the reductions in GHG emissions are expected to be negligible.

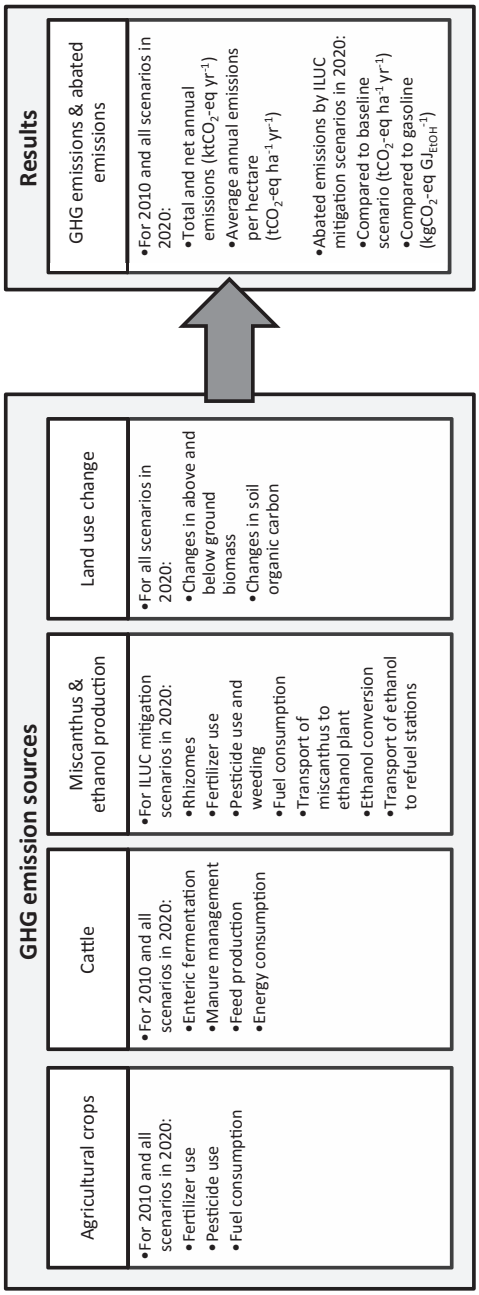

FIGURE 5-1 | Overview of GHG emission sources included in the GHG balances and results.

These pathways are defined as:

- Conventional intensification (CI): yield increases are attained by applying more fertilizers, pesticides, and mechanization. The nutrient, pesticide and energy use efficiency are not improved [8]. Also, conventional agricultural practices such as full tillage are applied.
- Intermediate sustainable intensification (II): yield increases are attained while the nutrient, pesticide and energy use efficiency are enhanced to some extent through improved agricultural practices like reduced tillage.
- Sustainable intensification (SI): yield increases are principally attained without increased input, but through adopting practices which optimize the resource use efficiency and strengthen the productive capacity of the soil [8,18,19]. These practices include good fertilizer management, integrated pest protection, reduced or no tillage and the prevention of monocultures. The aim of good fertilizer management is to optimize crop yields while minimizing nutrient losses, e.g. through the use of the right nutrient source and application of the fertilizer at the right rate, time and place [20]. This could also include the use of improved fertilizer types such as nitrification inhibitors and slow-release fertilizers [21,22]. Integrated pest protection or management integrates all available pest control strategies, including biological, physical and other non-chemical methods, with the aim to minimize the use of chemical pesticides [23].

Based on the total GHG balances, it is assessed whether the ILUC mitigation scenarios for 2020 abate emissions compared to 2010, the baseline scenario for 2020, and gasoline. Finally, a sensitivity analysis is performed to assess the influence of different assumptions on the results.

5.2.2.1 Agricultural crops

The projected growth in crop yields, as defined in Gerssen-Gondelach et al. [3], increases from the baseline scenario to the high ILUC mitigation scenario. The higher the yield growth, the more advances in the production system are required to attain the projected yields. For example, Figure 5-2 presents crop yields of maize, wheat and rapeseed in 2010 and all scenarios in 2020 and shows the related management level, based on agro-climatically attainable yields for different management systems as described in the Global Agro-Ecological Zones (GAEZ) methodology [24]. In 2010, the case study region is characterized by an intermediate management level. This means that the production is partly subsistence based and partly market oriented, is medium labor intensive, applies both manual labor, animal traction and some machinery and uses moderate levels of fertilizer and pesticides [24]. In the medium and/or high ILUC mitigation scenarios, an advanced management level is required which is characterized by commercial production, full mechanization, low labor intensity and optimal

use of fertilizers and pesticides [24]. Below, it is described how the levels of fertilizer, pesticide and fuel consumption are defined in each scenario and how the related GHG emissions are calculated. GHG emissions from seeds are excluded because the amount of seeds are assumed to remain unchanged, while yields increase in the different scenarios.

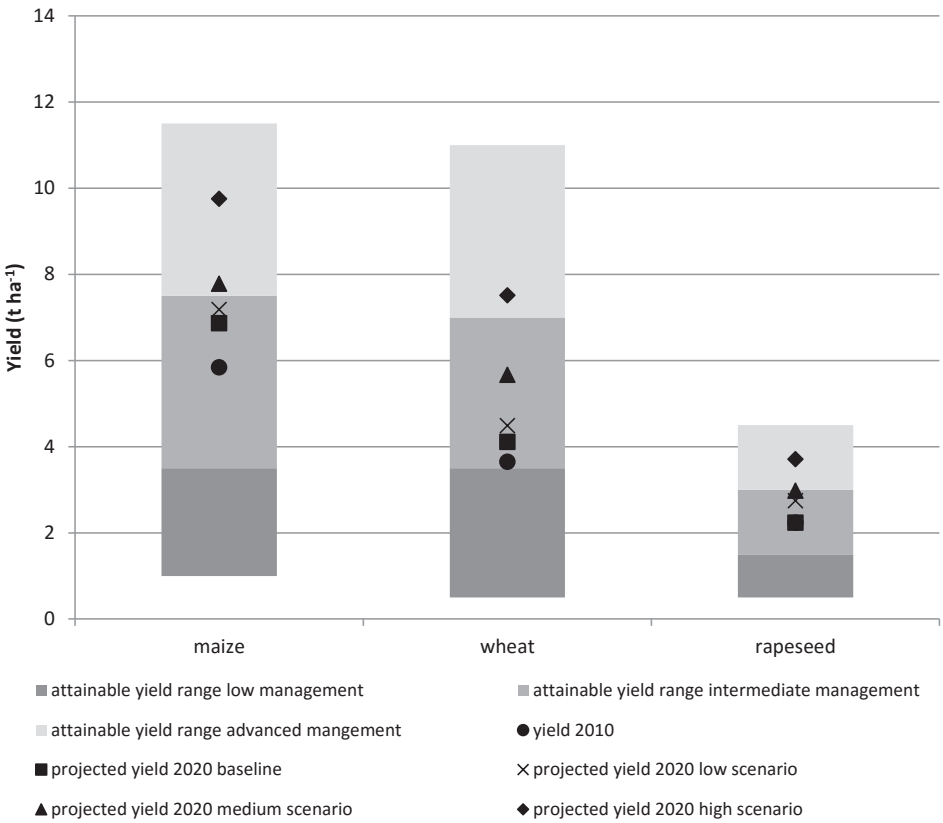

FIGURE 5-2 | Maize, wheat and rapeseed yield levels in 2010 and 2020 scenarios and attainable yield ranges for low, intermediate and advanced management levels, based on GAEZ data [24]. For rapeseed, the yields in 2010 and the baseline scenario for 2020 are equal (data points overlap).

Fertilizers

This study includes the fertilizer nutrients nitrogen (N), phosphate (P) and potash (K). The emissions related to fertilizer include emissions from fertilizer production, direct N_2O emissions from the soil and indirect N_2O emissions from volatization, leaching and runoff. For the ILUC mitigation scenarios, the emissions are calculated for the three intensification pathways. For these pathways, different assumptions are made with regard to nutrient use efficiency (NUE) and emission factors for N_2O emissions from N inputs, see below. The

measure of nutrient use efficiency applied in this study is the partial factor productivity (PFP), expressed in units of crop yield per unit of nutrient applied [25].

To determine the emissions related to fertilizer application, first the amount of fertilizer consumed is calculated. For three crops, i.e. wheat, maize and rapeseed, the fertilizer consumption level is based on the correlation between yield, fertilizer use (in kg ha^{-1}) and NUE, which is derived from historical data points for Lublin, Poland, Germany, the EU and Ukraine [26-30]. For example, Figure 5-3 presents historical nitrogen use levels in maize cultivation. From the data points, three isolines for constant nutrient use efficiency are derived. These three levels of NUE are designated as low, medium and high. Low NUE reflects sub-optimal management conditions and potential over-fertilization [25]. High NUE is attained through optimized management but there might also be a risk that the nutrient supply is limiting the crop yield [25].

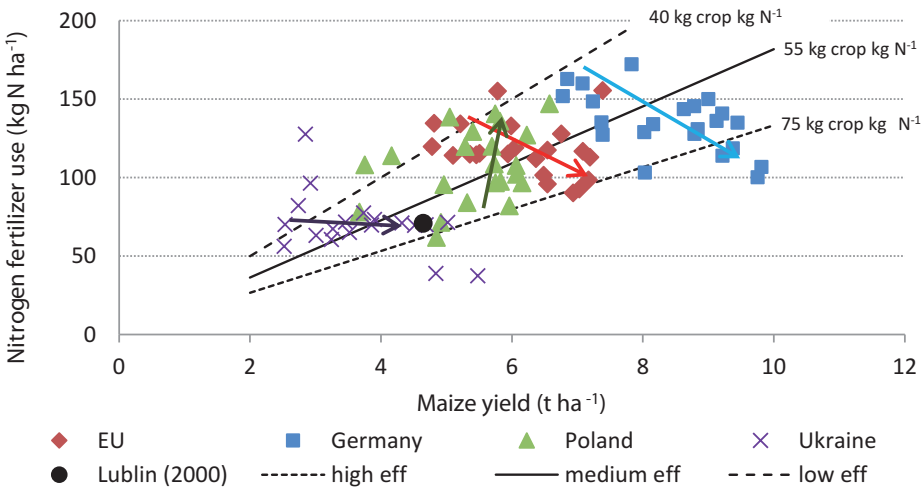

FIGURE 5-3 | Historical maize yields and fertilizer application levels in Lublin, Poland, Germany, EU and Ukraine (1990-2010). Isolines indicate the correlation between yield and fertilizer level for constant nutrient use efficiencies (low: 40 kg crop kg N^{-1}, medium: 55 kg crop kg N^{-1}, high: 75 kg crop kg N^{-1}). Arrows indicate the direction in yield development, fertilizer level and nutrient use efficiency over time in Poland, Germany, EU and Ukraine.

In the year 2000, the fertilizer levels applied and the crop yields attained in Lublin and Poland were lower compared to Germany and the EU, but the resulting NUE was higher (see data point for Lublin). However, in the following decade, Poland accessed the EU, fertilizer use per hectare increased and the NUE decreased, while the efficiencies in Germany and the EU improved over time (Figure 5-3). For Lublin, no data on fertilizer use by crop is available for years after 2000. Therefore, the NUE in Lublin in 2010 is based on

the trend in Poland and assumed to be low. Also, in the baseline scenario, this efficiency is assumed to remain low. In the ILUC mitigation scenarios, low NUE is assumed in the case of conventional intensification, medium NUE in the case of intermediate sustainable intensification and high NUE in the case of sustainable intensification.

Based on the calculated fertilizer application per hectare for wheat, maize and rapeseed and the total cultivation area of these crops in Lublin in 2010 [31], the total fertilizer consumption for these crops is calculated. This consumption is subtracted from the total fertilizer use in Lublin in 2010. The remaining fertilizer consumption is divided by the cultivation area of other agricultural crops, resulting in the average fertilizer use per hectare for these crops in 2010. Then, the fertilizer use for these other crops in the baseline scenario is based on the historical (2002-2014) average annual increase in fertilizer use per hectare in Lublin (in % per year relative to 2010). In the ILUC mitigation scenarios, the increase in fertilizer use for the other crops is assumed to develop in the same way as the increases for wheat, maize and rapeseed relative to the baseline scenario.

To calculate the emissions from fertilizer production, emission factors (gCO_2-eq $kg_{nutrient}^{-1}$) from the JEC E3-database for biofuel GHG calculations are applied as presented in the BioGrace GHG calculation tool [32], see Table 5-2. Direct and indirect N_2O emissions are calculated by applying the IPCC Tier 1 methodology [33] and assuming different emission factors for the three intensification pathways, based on the uncertainty range for default values (Table 5-2).

TABLE 5-2 | Key values for the calculation of GHG emissions from agricultural production.

Process	Aspect	Value	Unit	References
General	GWP CH_4	25	g CO_2-eq g CH_4^{-1}	
	GWP N_2O	298	g CO_2-eq g N_2O^{-1}	
Cultivation	Fertilizer production, N	5,881	g CO_2-eq kg N^{-1}	JEC E3-database [32]
	Fertilizer production, P	1,011	g CO_2-eq kg P^{-1}	idem
	Fertilizer production, K	576	g CO_2-eq kg K^{-1}	idem
	Direct N_2O emission factor[a]	0.015/0.01/0.006	kg N_2O-N kg $N_{applied}^{-1}$	IPCC [33]
	Volatization fraction[a]	0.15/0.1/0.06	kg NH_3-N+NO_x-N kg $N_{applied}^{-1}$	idem
	Volatization emission factor[a]	0.02/0.01/0.006	kg N_2O-N kg NH_3-N + NO_x-N^{-1}	idem
	Leaching and runoff fraction[a]	0.5/0.3/0.2	kg N kg $N_{applied}^{-1}$	idem
	Leaching and runoff emission factor[a]	0.015/0.0075/0.0050	kg N_2O-N kg $N_{leached}^{-1}$	idem
	Pesticide production	10,971	g CO_2-eq kg a.i. $^{-1}$	JEC E3-database [32]
	Diesel	38.3	MJ $liter^{-1}$ (HHV)	[34]
		87.6[b]	g CO_2-eq MJ_{diesel}^{-1}	JEC E3-database [32]

GWP, global warming potential; a.i., active ingredient.
[a] Values for respectively conventional/intermediate sustainable/sustainable intensification; [b] Emission factor for diesel includes indirect emissions.

Pesticides

Comparable to fertilizers, pesticides use per hectare and crop yields appear to be positively correlated [35]. However, for pesticides, not enough historical data is available to derive this correlation per crop as was done for fertilizers. Therefore, based on the limited data available, assumptions are made about low, average and high pesticide use efficiencies. This approach is applied to wheat, rapeseed, apples and the aggregated groups cereals (including oats, triticale, rye, barley, mixed cereals) and other crops (including potatoes, sugarbeet and maize). The pesticides levels in 2010 are based on most recent (2002-2008) data for Poland [36,37]. Because the management levels in Lublin are lower than the Polish average, it is assumed that the figures for Poland in the period 2002-2008 are appropriate for Lublin in 2010 and represent a medium pesticide use efficiency. For the emission factor of pesticides, see Table 5-2.

Machinery fuel use

Mechanization is an important driver for increasing yields and thus results in higher fuel use. Lorencowicz and Uziak [38] assessed the fuel consumption at family farms in Lublin and found that the fuel consumption is linearly correlated to the farm size (in hectare). They derived relationships for different groups of farms: farms with only one tractor, farms with more than one tractor, and all farms. In 2010, the average amount of tractors per farm in Lublin was less than 1 (Table 5-3). This is projected to remain true in the baseline scenario for 2020. For these cases, the formula derived for farms with only one tractor is applied to calculate the fuel use. In the ILUC mitigation scenarios, the average number of tractors per farm is projected to be 1 or higher than 1, but there may be a mix of farms with one or more tractors.

TABLE 5-3 | Data inputs on crop and cattle management levels in 2010 and 2020 scenarios.

		2010	2020 Baseline	ILUC mitigation		
				Low	Medium	High
Crops	Avg. # tractors per farm[a]	0.7	0.8	1.0	1.1	2.1
	Avg. farm size (ha)[a]	7	10	15	29	56
Cattle	Productivity (kg product $animal^{-1}$ yr^{-1})[a]					
	Dairy (milk)	4,600	5,165	5,165	7,070	6,490
	Beef (beef)	200	225	225	315	290
	Total cattle density (animals ha^{-1})[a,b]	1.5	1.5	1.8	1.8	4.0

[a] The values for 2010 are based on statistics for Lublin [3,39]. In each scenario for 2020, the values are related to the assumptions for yield improvements: for example, as the yield projections were based on yields attained in other Polish provinces and in Germany, the farm size is based on the average farm size in the same region [3]; [b] the cattle density is expressed as the number of animals per ha of meadows and pastures.

Therefore, for these scenarios the formula for all farms is applied and assumed to be true for intermediate sustainable intensification. For conventional and sustainable intensification, a higher and lowel fuel consumption level are estimated. The emission factor for diesel is presented in Table 5-2. The production of machinery is not included in the emission balance.

5.2.2.2 Cattle

The total emissions from dairy and beef production include emissions from enteric fermentation (CH_4), manure management (CH_4 and N_2O), feed production (N_2O and CO_2) and energy consumption (CO_2). For 2010 and each scenario, as well as for the three intensification pathways in each ILUC mitigation scenario, the emissions for each component (in kg CO_2-eq kg_{milk}^{-1} or kg CO_2-eq kg_{beef}^{-1}) are estimated based on a review of case studies by Gerssen-Gondelach et al. [40] and FAO data on average emissions from enteric fermentation and manure management in European countries and regions [41]. Then, the total annual emissions are calculated by multiplying the emissions per kg product by the total annual milk or beef production. Cattle production in 2010 was primarily characterized by a pasture-based production system. In this system, some of the feed comes from grazing on pastures and a larger part from fodder [31]. However, grasslands are often extensively managed and their productivity is lower compared to some Western European regions because of the lower annual precipitation [42,43]. Therefore, forage maize constitutes the largest part of the feed mix [43]. With increasing animal productivity and density in the scenarios for 2020 (see Table 5-3), the feed quality is initially improved by more intensive management of meadows and pastures, followed by including more feed crops and concentrates in the feed mix and reducing the time at pasture. Thus, the importance of grazing decreases, similar to dairy production in Germany [44]. The three intensification pathways in each ILUC mitigation scenario illustrate how GHG emissions are influenced by different levels of energy use efficiency and nutrient use efficiency in pasture management and feed production, and by different manure management practices. As meadows, pastures and forage maize were not included in the GHG balance of agricultural crops, no double counting occurs when including feed production in the GHG balance of cattle in 2010 and the baseline scenario. In the ILUC mitigation scenarios, the share of crops in the feed mix increases, but is assumed that the effect of potential double counting on the total GHG balance will be small because the share of emissions from feed production is limited to 5%-12% of the total emissions from cattle. In addition, the emissions from feed production also include manure applied to feed crops and pasture or directly deposited on pastures, which is not included in the GHG balance of agricultural crops.

5.2.2.3 Miscanthus and ethanol production

In the present study, a lifetime of 20 years for miscanthus is assumed. This lifetime includes one year of establishment followed by 19 years of spring harvesting [14]. The average yield is 13 t dm ha^{-1}, see Gerssen-Gondelach et al. [3] for a detailed review of miscanthus yields in Lublin and Poland.

Rhizomes

Rhizomes are planted in the year of establishment [12]. Regarding the emission factor for rhizomes, a wide range is found in the literature [12,14]. This range is used to define emissions factors for the three intensification pathways, see Table 5-4.

Fertilizer application

Miscanthus requires less fertilizer than annual crops. It is characterized by efficient nitrogen use and a large part of the nutrients is translocated to the roots during the senescence of the leaves during winter [45]. In the literature, there is no consensus on the response of miscanthus yield to fertilization rates [12,46]. Therefore, the amount of nutrient (N, P and K) removed with harvested biomass is used as a proxy for the required fertilizer application level [12,47]. Recommended application rates range between 3.0-4.9 kg N t dm^{-1}, 0.47-0.6 kg P t dm^{-1} and 6.5-7.0 kg K t dm^{-1} [12,47]. These ranges are used to define the application levels for the different intensification pathways. For nitrogen and potash, the resulting fertilizer consumption per hectare is in line with field studies in Lublin [48-51]. But for phosphate, the calculated application rate is significantly lower than reported in literature [49-51]. Therefore, the impact of the phosphate level is assessed in the sensitivity analysis. Miguez et al. [52] state that miscanthus cultivation may result in lower nitrate leaching compared to annual crops, but the fraction of N lost through leaching and runoff is uncertain [47]. Therefore, for each intensification pathway, the present study assumes lower emission factors compared to those applied for agricultural crops but still within the uncertainty range from the IPCC Tier 1 methodology [33], Table 5-4.

Fuel use

During miscanthus cultivation, machinery is used for soil preparation, planting of rhizomes, fertilizer application, weeding, harvesting (including baling) and removing the plantation at the end of the plantation's lifetime [12]. Additional fuel is consumed for storage of the bales, transport of bales to the bioethanol plant and transport of ethanol to fuel stations [12]. For all operations except final transport, the required machinery, work capacity (hours ha^{-1}) and fuel use are derived from Smeets et al. [12] and EcoInvent datasets [53], Table 5-4. For final ethanol transport to the fuel stations, three options are considered depending on the transportation distance. First, in case ethanol is used locally, transport can be performed by truck [60]. This option is included in the ILUC mitigation scenarios.

Second, in the case ethanol is exported, train transport or shipping can be chosen. The effects of these options are assessed in the sensitivity analysis. The estimates on rail and shipping distances are based on the assumption that ethanol is exported to Western European countries [60], Table 5-4.

Conversion

The conversion of miscanthus to ethanol requires chemicals and energy, but also co-generates electricity, see Table 5-4. To account for electricity co-generation in the GHG balance, system expansion is applied. Thus, the GHG balance includes the savings of not generating electricity in a conventional power plant. This credit is based on the Polish electricity mix, which heavily relies on coal-based power generation (Table 5-4).

5.2.2.4 Land use and land use change

Carbon stock changes take place due to land use and land use change (LULUC). In this case study, five types of land use change are considered as presented in Table 5-1. Carbon stock changes are calculated according to the IPCC Tier 1 approach [62,63] and the guidelines as published in the EU Commission Decision [64], equations 1-3.

$$\Delta CS = CS_A - CS_R \qquad \text{Equation 1}$$

$$CS_i = SOC + C_{BM} \qquad \text{Equation 2}$$

$$C_{BM} = \left(\frac{Y}{HI} - Y\right) \times CF_B \qquad \text{Equation 3}$$

Where: ΔCS = carbon stock change due to land use change, CS_i = carbon stock associated with land use i (t C ha^{-1}), A = actual land use, R = reference land use, SOC = soil organic carbon (t C ha^{-1}), C_{BM} = carbon stock in above and below ground biomass (t C ha^{-1}), Y = yield (t dm ha^{-1}), HI = harvest index, based on the harvestable yield compared to the total above and below ground biomass, CF_B = carbon fraction of dry matter in biomass (t C ha^{-1}).

Changes in the carbon stock of dead organic matter (DOM) are excluded as DOM stocks are considered zero for non-forest land [64]. For 2010 and the baseline scenario, region specific data on the SOC stocks of croplands and grasslands are used [65,66]. For other land uses, the SOC is a function of the standard SOC (SOC_{ref}) specified for the applicable climate and soil type multiplied by three factors related to the land use (F_{LU}), soil management (F_{MG}) and carbon inputs (F_I), Equation 4 and Table 5-5 [62,64].

$$SOC = SOC_{ref} \times F_{LU} \times F_{MG} \times F_I \qquad \text{Equation 4}$$

TABLE 5-4 | Key values for the calculation of GHG emissions from the miscanthus and bioethanol value chain.

Process	Aspect	Value	Unit	References
Cultivation	Miscanthus rhizomes[a]	280/200/110	kg CO_2-eq/ha^{-1}	[12,14]
	Fertilizer use, N[a]	5.0/4.0/3.0	kg N t dm^{-1}	idem
	Fertilizer use, P[a]	0.6/0.55/0.5	kg P t dm^{-1}	idem
	Fertilizer use, K[a]	7.0/6.5/6.0	kg K t dm^{-1}	idem
	Direct N_2O emission factor[a]	0.01/0.006/0.003	kg N_2O–N kg $N_{applied}^{-1}$	IPCC [33]
	Volatization fraction[a]	0.1/0.06/0.03	kg NH_3–N+NO_x–N kg $N_{applied}^{-1}$	idem
	Volatization emission factor[a]	0.01/0.006/0.001	kg N_2O–N kg NH_3–N + NO_x–N^{-1}	idem
	Leaching and runoff fraction[a]	0.3/0.2/0.1	kg N kg $N_{applied}^{-1}$	idem
	Leaching and runoff emission factor[a]	0.0075/0.005/0.0025	kg N_2O–N kg $N_{leached}^{-1}$	idem
	Diesel consumption, small tractor (60 kW) [a]	12/7.5/5	l h^{-1}	[12,53]
	Diesel consumption, medium tractor (75 kW) [a]	20/15/7.5	l h^{-1}	idem
	Diesel consumption, large tractor (100 kW) [a]	22/20/15	l h^{-1}	idem
Storage of bales	Biomass loss	3	% dm	[12,54,55]
	Fuel use	0.9	l t dm^{-1}	[12]
Truck transport	Truck transport, max load	27	tonne	idem
	Average distance	50	km^{-1}	Assumption
	Fuel empty	0.2	l km^{-1}	[12]
	Fuel full	0.4	l km^{-1}	idem
Conversion	Primary energy use	0.1	MJ_P MJ_{EtOH}^{-1}	[56]
		34	g CO_2-eq MJ_P^{-1}	idem
	Electricity co-generation	0.1	MJ_{Elec} MJ_{EtOH}^{-1}	[57-59]
Truck transport to fuel station	Maximum load[b]	25	tonne	[60]
	Diesel consumption	18.1	MJ km^{-1}	idem
	Average distance	100	km	idem
Train transport to fuel station	Maximum load[b]	1000	tonne	idem
	Electricity use	163	kWh km^{-1}	idem
	Average distance	800	km	idem
Ship transport to fuel station	Maximum load[b]	4000	tonne	idem
	Diesel consumption	647	MJ km^{-1}	idem
	Average distance	1100	km	idem
Energy	Electricity, EU mix (2009)	125.5	g CO_2-eq MJ_{Elec}^{-1}	idem
	Electricity, Polish mix (2009)	285.9	g CO_2-eq MJ_{Elec}^{-1}	idem
	Ethanol	23.4	MJ liter^{-1} (HHV)	[61]
		0.794	tonne m^{-3}	[32]

[a] Values for respectively conventional/intermediate sustainable/sustainable intensification; [b] the maximum load is restricted by mass, thus the maximum volume is not entirely utilized

With regard to the ILUC mitigation scenarios, it is assumed that the SOC of each land use can change due to altered management practices, depending on the intensification pathway. In the case of intermediate sustainable intensification, the SOC values remain equal to 2010 and the baseline. For conventional and sustainable intensification, the SOC values are assumed to respectively decrease or increase by 4%-5% based on IPCC [62,63] and annual mitigation potentials for these lands as estimated by Smith et al. [22]. In absolute terms, the carbon stock change is 0.3 t C ha^{-1} yr^{-1} for croplands and 0.4 t C ha^{-1} yr^{-1} for grasslands.

TABLE 5-5 | Default values for the calculation of carbon stock changes due to land use change.

Process	Aspect	Specification	Value	Unit	References
General	Climate	Cool temperate, moist			
	$SOC_{cropland}$	Average	75	t C ha^{-1}	[65]
	$SOC_{grassland}$	Average	88	t C ha^{-1}	[65]
	SOC_{ref}	Clay soils	95	t C ha^{-1}	[64]
Arable crops	C_{BM}		0	t C ha^{-1}	idem
Apples	F_{LU}	Perennial	1	Factor	[62,64]
	F_{MG}	Full tillage	1	Factor	idem
	F_I	High input without manure	1.11	Factor	idem
	$C_{BM,apples}$		43.2	t C ha^{-1}	[64]
Grassland	$C_{BM,grass}$	Grassland	6.8	t C ha^{-1}	idem
Abandoned land	F_{LU}	Set-aside	0.82	Factor	[62]
	F_{MG}	No tillage	1.15	Factor	idem
	F_I	Low input	0.92	Factor	idem
Miscanthus	F_{LU}	Grassland[a]	1	Factor	[64]
	F_{MG}	Improved[a]	1.14	Factor	idem
	F_I	Medium input[a]	1	Factor	idem
	HI		0.4	Factor	[14,67]
	CF_B		0.47	t C tdm^{-1}	[64]
Equilibrium time	D	Time needed to reach equilibrium soil C stock	20	yr	idem

SOC, soil organic carbon; C_{BM}, carbon stock in above and below ground biomass; $F_{LU,}$ land use factor; $F_{MG,}$ management factor; F_I, input factor; HI, harvest index; CF_B, carbon fraction of dry matter in biomass; D, equilibrium time. [a] Perennial herbaceous crops like miscanthus are not included as a separate land use in in the guidelines of the IPCC [62] and European Commission [64]. The factors of improved grassland and medium input are assumed to be most appropriate for miscanthus, see van der Hilst et al. [14]

5.2.2.5 Net GHG balances and GHG abatement levels

To attain the net annual emissions from agricultural and bioenergy production in Lublin in 2010 and all scenarios in 2020, the annual emissions from all GHG sources as discussed in sections 5.2.2.1 to 5.2.2.4 are added up. Based on these net GHG balances, it is assessed whether the ILUC mitigation scenarios abate emissions compared to 2010 and the baseline scenario for 2020. In addition, when the net emissions are fully allocated to bioenergy, it is assessed whether the production of bioethanol in the ILUC mitigation scenarios abates emissions compared to gasoline and whether the GHG reduction levels comply with EU

GHG savings requirements [68]. The emission factor for gasoline is 90 g CO_2-eq $MJ_{gasoline}^{-1}$ [69].

5.2.2.6 Sensitivity analysis

To assess how the results are influenced by different assumptions on e.g. the electricity mix and the ethanol chain efficiency, a sensitivity analysis is performed for the medium ILUC mitigation scenario with intermediate sustainable intensification. Table 5-6 gives an overview of all variables included and the assumptions made in the sensitivity analysis.

TABLE 5-6 | Assumptions for sensitivity analysis of the medium ILUC mitigation scenario (intermediate sustainable intensification pathway).

	Variable	Medium scenario: original assumptions	Sensitivity low	Sensitivity high
Miscanthus – ethanol chain	Phosphate level adapted to field studies in Lublin	7 kg P ha^{-1}	-	33 kg P ha^{-1}
	Conversion, electricity credit	Electricity mix Poland (285.9 g CO_2-eq MJ_{elec}^{-1})	Electricity mix EU27 (125.5 g CO_2-eq MJ_{elec}^{-1})	-
	Truck distance	100 km	50 km	150 km
	Transport mode	Truck	Train	Ship
	Chain efficiency (miscanthus and ethanol yield)	13 t dm ha^{-1}, 84.4 GJ_{EtOH} ha^{-1}	10 t dm ha^{-1}, 63.6 GJ_{EtOH} ha^{-1}	17 t dm ha^{-1}, 128.7 GJ_{EtOH} ha^{-1}
	Equilibrium time (LUC)	20 year	10 year	40 year

5.2.3 Assessment of other environmental impacts

To evaluate the environmental performance of ILUC mitigation and bioethanol production, this chapter applies the principle that the ILUC mitigation scenarios should not only save GHG emissions compared to fossil fuels, but should also have a positive or at least neutral impact on other parts of the environment. This means that the implementation of the ILUC mitigation measures and the cultivation and processing of miscanthus must not be at the expense of biodiversity, ground and surface water quantity and quality, soil quality and air quality [13,70]. This study qualitatively assesses the risk that this principle cannot be met. Based on literature, several indicators are selected to discuss and evaluate the potential changes in biodiversity, water quantity and quality, soil quality and air quality for the three intensification pathways in 2020. First, with regard to biodiversity, a distinction is made between areas of high nature value (HNV) and other areas. For HNV areas, this study evaluates the extent to which it is possible to continue meeting the habitat functions requirements of species living in these areas [14]. For other areas, it is assessed how their species abundance might be affected by changes in land use and land management

[13,14]. Second, with regard to water, the water availability is discussed based on the rates of precipitation and evapotranspiration [12,14]. The water quality is related to the risk of leaching of fertilizers and pesticides [14]. Third, the soil quality is considered by assessing the risk of reducing the productive capacity of the soil and the risk of erosion [14,70]. The productive capacity of the soil is defined by aspects such as the amount of soil organic matter (SOM), water holding capacity, nutrient retention and soil structure [14]. As soil organic carbon (SOC) is closely related to these aspects, SOC is used as a proxy indicator for the productive capacity of the soil [14]. Soil erosion includes wind and water erosion [12,14,70]. Salinization is excluded, as the risk of this type of erosion is considered negligible in Lublin [71]. Finally, indicators for air quality are emissions of non-GHG pollutants causing acidification (SO_2, NH_3 and NO_x) and emissions of fine particles (PM_{10}) [70]. Also, the risk of contamination of the air due to pesticides is discussed. The potential impact of each intensification pathway on these indicators is evaluated based on literature. Also, the expected impacts are qualified by using symbols, ranging from -- for a high risk of negative effects to ++ for no risk and high positive effects. It should be noted that this qualification is not always straightforward as it reflects the interpretation of the authors.

5.3 RESULTS

5.3.1 GHG emissions

The annual GHG balances for 2010 and 2020 are presented in Figure 5-4. In the baseline scenario for 2020, the net annual emissions are slightly lower compared to 2010. Emissions from machinery and cattle are reduced in the baseline scenario, but emissions from fertilizers increase. Carbon stock changes due to land use and land use change (LULUC) are negligible. In 2010 and the baseline scenario, the largest GHG emission sources are fertilizer use on cropland and enteric fermentation from cattle production. The ILUC mitigation scenarios generally reduce emissions compared to the baseline, see also Figure 5-5A. The only exception is the low ILUC mitigation scenario following the conventional intensification pathway. This can especially be explained by reductions in the SOC stocks of all land uses due to the application of conventional management practices. These land-use related soil carbon emissions are larger than the total carbon sequestration related to the conversion of agricultural lands to miscanthus, which results in positive net carbon emissions. In the medium and high ILUC mitigation scenarios following the conventional intensification pathway, the negative emissions related to LUC are higher and counteract the land-use related reduction in carbon stocks, resulting in negative net carbon emissions.

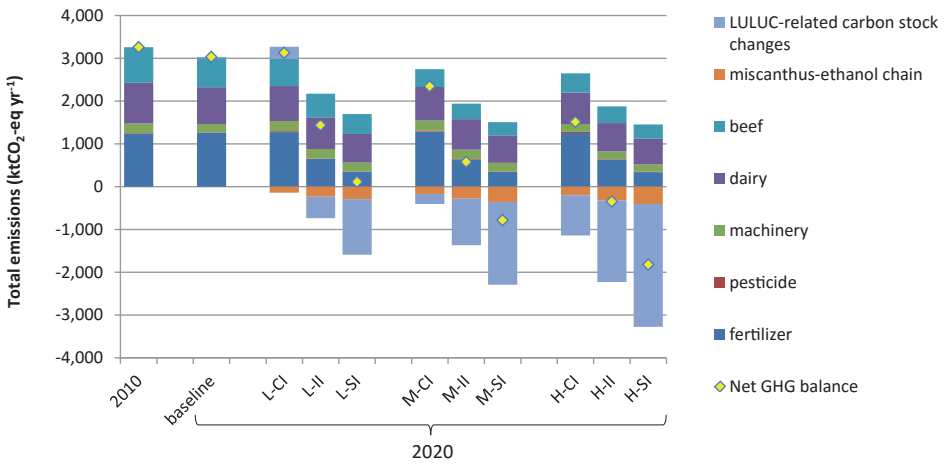

FIGURE 5-4 | Total and net annual GHG emissions for 2010 and the baseline and ILUC mitigation scenarios in 2020. Emissions from the miscanthus-ethanol value chain are based on truck transport of ethanol to local fuel stations. The equilibrium time for soil carbon stock changes is 20 years. ILUC prevention scenarios: L, low; M, medium; H, high. Intensification pathways: CI, conventional intensification; II, intermediate sustainable intensification; SI, sustainable intensification.

It is found that the differences between net annual GHG emissions and GHG abatement levels are often larger for intensification pathways in the same ILUC mitigation scenario than for ILUC mitigation scenarios following the same intensification pathway. The main reason for this is that the intensification pathways have considerable influence on the emissions related to fertilizer consumption and soil carbon stock changes in each land use type. First, the emissions from fertilizer consumption significantly depend on the nutrient use efficiency and emission factors applied. Therefore, the emission reduction attained through improved nutrient use efficiencies and emission factors in more a sustainable intensification pathway is considerably higher compared to the GHG mitigation attained through increased yields and a reduction in the cropland area in a higher ILUC mitigation scenario. Second, the difference in SOC stocks between the intensification pathways due to different management practices is generally larger than the difference in LUC-related carbon stock changes between the ILUC mitigation scenarios.

In the miscanthus-ethanol chain, especially fertilizer use but also storage and transport of bales contribute most to the GHG emission balance. However, due to the high emission factor for the Polish electricity mix, the electricity credit from ethanol production is very large, which results in negative emissions for the bioethanol chain. Also, land use change results in carbon sequestration in the ILUC mitigation scenarios. Considering an equilibrium time of 20 years, soil carbon sequestration due to conversion to miscanthus is found to be 1.5-1.8 t C $ha^{-1} yr^{-1}$ for cropland, 1.0-1.1 t C $ha^{-1} yr^{-1}$ for grassland and 1.2-1.4 t C $ha^{-1} yr^{-1}$ for abandoned land, and is highest for the sustainable intensification pathways.

These figures are in line with results from de Wit et al. [9] and Borzęcka-Walker [72]. The total carbon sequestration in soil and biomass ranges from about 22 to 45 t C ha^{-1}. Only for apple orchards, conversion to miscanthus causes a significant biomass loss, resulting in carbon emissions of 26 to 37 t C ha^{-1}. On croplands and grasslands that are abandoned and not used for miscanthus cultivation, the total carbon sequestration in biomass and soil is 6 to 9 t C ha^{-1} and -12 to -13 t C ha^{-1} respectively.

When considering the net GHG emissions from the ILUC mitigation scenarios per Gigajoule (GJ) of ethanol produced, it is found that in most cases miscanthus-based ethanol abates emissions compared to gasoline (Figure 5-5B). Exceptions are the low and medium ILUC mitigation scenarios following the conventional intensification pathway. For these cases, the net emissions are respectively 114% and 31% higher compared to gasoline. In the low scenario with intermediate sustainable intensification and the high scenario with conventional intensification, the GHG emission reduction compared to gasoline is 1% and 26% respectively, which is not in compliance with the EU GHG savings requirements of 35% reduction now and 60% reduction in 2018 compared to fossil fuels [68]. The five remaining cases comply with these goals, and four of these also fulfill the even higher reduction objective of 80-90% as suggested by Cramer et al. [73]. The latter four cases include all ILUC mitigation scenarios following the sustainable intensification pathway and the high scenario following the intermediate sustainable intensification pathway. The medium and high ILUC mitigation scenarios that attain a negative net GHG balance due to carbon sequestration in biomass and soil even realize an emission reduction of more than 100% compared to gasoline.

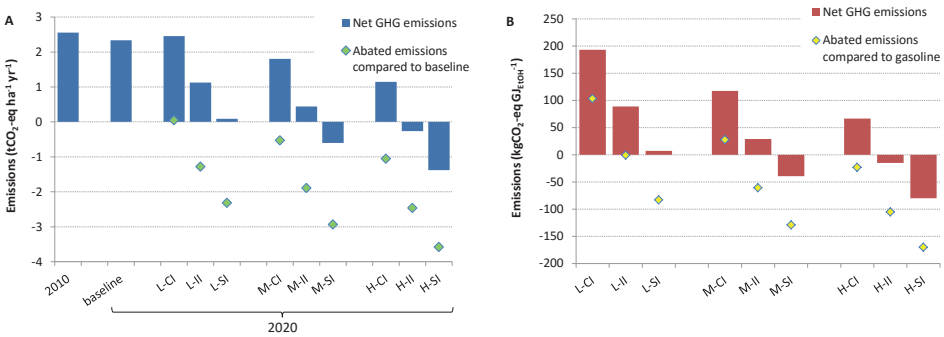

FIGURE 5-5 | Net GHG emissions and abated emissions compared to A) baseline scenario (tCO_2-eq ha^{-1} yr^{-1}) and to B) gasoline ($kgCO_2$-eq GJ_{EtOH}^{-1}). Positive abatement values mean higher emissions and no emission abatement compared to the reference, negative abatement values mean lower emissions and emission abatement compared to the reference. ILUC prevention scenarios: L, low; M, medium; H, high. Intensification pathways: CI, conventional intensification; II, intermediate sustainable intensification; SI, sustainable intensification.

5.3.2 Sensitivity analysis

Figure 5-6 presents the results of the sensitivity analysis. A significant effect is found for the electricity credit from ethanol production, the equilibrium time for carbon stock changes and the miscanthus-ethanol chain efficiency. For these three variables, the impact is highest for the emission abatement level compared to gasoline. The phosphate application rate for miscanthus production and the truck distance and transport mode to deliver bioethanol to fuel stations have negligible impact on the GHG emission abatement level compared to the baseline scenario and to gasoline (less than 1% change) and are therefore not included in Figure 5-6.

The equilibrium time is found to have the largest impacts on the results. Increasing the equilibrium time from 20 to 40 years has no influence on which cases reduce emissions compared to the baseline, but the difference between the medium ILUC mitigation scenario following the conventional pathway and the baseline scenario will become very small. Also, the difference in net annual GHG emissions between the three ILUC mitigation scenarios will become smaller. With regard to GHG emission reductions compared to gasoline, fewer cases will attain the GHG savings requirements of 60% or higher. Emission reductions of more than 100% will only be attained by the medium and high ILUC mitigation scenarios following the sustainable intensification pathway.

In this sensitivity analysis, the electricity credit from ethanol production was changed by applying the EU electricity mix instead of the Polish mix. Of course, the electricity credit also depends on the amount of electricity co-generated. For example, when electricity co-generation is halved, the total credit is also reduced to half of the initial credit. The effect on to the net GHG emissions and the emission abatement level compared to the baseline will be equal compared to halving the emission factor for the electricity mix. However, when the reduction in electricity co-generation is accompanied with a higher conversion efficiency from miscanthus to ethanol, the net GHG emissions can be allocated to a larger amount of ethanol, which decreases the reduction in the emission abatement level compared to gasoline.

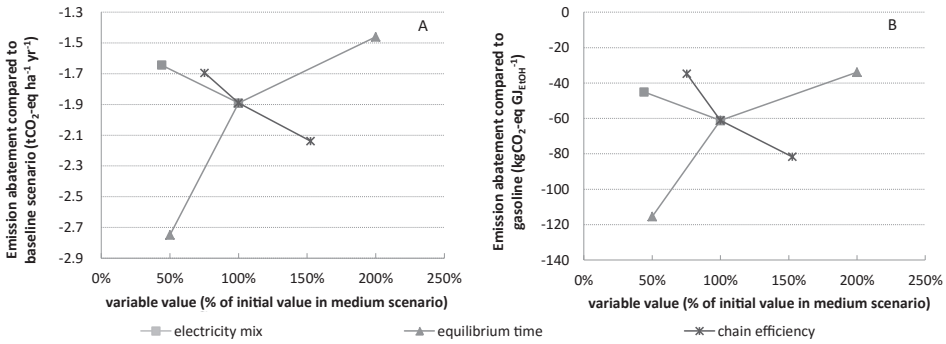

FIGURE 5-6. Sensitivity analysis of the abated emissions in the medium ILUC mitigation scenario following the intermediate sustainable intensifcation pathway: A) compared to the baseline scenario, and B) compared to gasoline.

5.3.3 Qualitative assessment of other environmental impacts

Key environmental impacts are discussed in the following sections, more information can be found in the Appendix.

5.3.3.1 Biodiversity

In unprotected agricultural areas, intensification is expected to lead to scaling up of farms and potentially also specialization. In the case of conventional intensification, this causes an increase in monocultures and loss, modification and fragmentation of habitats. This, in turn, leads to a decrease in species abundance. Also, inefficient fertilizer and pesticide usage results in increased leaching, soil degradation and water pollution, which are risk factors for species abundance [14].

As miscanthus is an extensively managed crop, the conversion of arable lands to miscanthus is often found to have a positive impact on species abundance [12,74]. The impacts of the conversion of grasslands to miscanthus are more uncertain and not yet well understood [74,75], but the risk of biodiversity loss is estimated to be higher. This is especially because many grasslands are semi-natural and extensively managed [42] and the conversion of extensive pastures has a higher risk of biodiversity loss than the conversion of intensively managed pastures [14].

As Poland supports a significant share of farmland bird populations in the EU [42], special attention should be paid to the conservation of these birds. Risk factors for the abundance and sometimes also the species richness of farmland birds are reductions in low-intensity farmland cover [42] and the conversion of areas with high bird densities to miscanthus cultivation [14].

5.3.3.2 Water

In Poland, annual precipitation is low, but monthly precipitation is highest during summer when the evapotranspiration is also peaking. However, the evapotranspiration exceeds the precipitation and water deficits occur during the summer [76]. In Lublin, the average annual precipitation ranges between approximately 500 and 600 mm depending on the location [77]. Figures from literature (Table 5-7) show that the water requirements of agricultural crops may exceed the annual precipitation, which results in a negative annual water balance and the occurrence of droughts. Conventional agricultural intensification increases the risk of groundwater deficiencies and droughts [76]. Risk factors include monocultures and irrigation [21]. Sustainable intensification practices that improve soil moisture, e.g. by increasing soil organic carbon through reduced tillage, may help crops to better withstand droughts [21,70].

Despite its high water use efficiency, the rate of evapotranspiration of miscanthus is found to be higher compared to traditional annual crops and pastures [12,14], see Table 5-7. Large scale cultivation of miscanthus, especially in the case of monocultures, will thus

contribute to the risk of groundwater depletion. In the assessment of the low-ILUC-risk miscanthus potential by Gerssen-Gondelach et al. [3], minimal water requirements (550 mm yr^{-1}) were already taken into account. Thus, areas with a high risk of depleting water bodies and competition with other water uses are excluded from miscanthus cultivation. However, more site-specific data and analysis is required to assess the impacts of miscanthus production on water availability in Lublin. This should take into account variables like the soil texture, rainfall pattern, wind speed, cropping pattern and the location-specific crop evapotranspiration factors for arable crops and miscanthus [78].

TABLE 5-7 | Crop evapotransiration coefficient Kc (dimensionless) and water use efficiency for miscanthus and agricultural crops.

	K_c growth period (low-high)	Duration growth period (days)	Water requirement growth period (mm)	WUE (g_{DM} kg_{water}^{-1})	References
Miscanthus (spring harvest)	0.85-1.2	365	n/a	n/a	van der Hilst et al. [14]
Miscanthus (autumn harvest)	0.3-1.6	215	900	4.2	Triana et al. [79]
Wheat	0.25-1.15	100-170	450-650	0.69-0.86	van der Hilst et al. [14], Dornburg et al. [80]
Maize	0.3-1.2	100-170	500-800	0.70-1.41	idem
Potato	0.5-1.15	100-175	500-700	1.08-1.89	idem
Sugarbeet	0.35-1.2	160-215	550-750	1.00-1.60	idem
Pasture	0.70-1.05	200	n/a	n/a	Szejba [81]
Meadows (extensive)	0.76-1.41	200	n/a	n/a	idem

K_c is the ratio between the actual non-water limited water demand and the reference evapotranspiration (ET_0)
n/a, not available

5.3.3.3 Soil

The GHG balances for the ILUC prevention scenarios showed that the chosen intensification pathway has a significant influence on the SOC balance and on the net annual emissions. This is confirmed by the literature. For example, Squire et al. [82] find that increased fertilizer and pesticide use in the UK resulted in reduced SOC and also in lower water holding capacity of the soil. Thus, sustainable intensification practices are important to prevent SOC losses [70]. In addition to reduced or no tillage, measures to increase SOC include the use of cover crops and replanting native vegetation on abandoned land [21-23].

Based on average SOC values used in the GHG emission calculations, the conversion of croplands and grasslands to miscanthus improves the soil carbon stocks. But the variation in SOC values for grasslands is large. Extensively managed grasslands have higher SOC stocks and the risk of carbon loss increases when converting these lands [14]. The risk of converting high carbon stock pastures increases as more pastures are converted in the

higher scenarios. Although grasslands with the highest risk of carbon losses (i.e. boggy and wet areas) were already excluded from miscanthus cultivation in the underlying study determining the surplus land area available for miscanthus [3], it needs to be carefully assessed which of the remaining grasslands can be converted to miscanthus while achieving SOC sequestration and which grasslands should be maintained to prevent SOC losses.

5.3.3.4 Air

Important sources of non-GHG pollutants causing acidification are fertilizer use (NH_3), manure management (NH_3) and tractor fuel combustion (NO_x) [83]. Also, pesticides have the potential to contaminate the air and pose a risk on the environment and human health [84]. Thus, sustainable intensification, including increased fertilizer, pesticide and fuel use efficiency and adoption of improved manure management technologies, is important to maintain or improve the air quality.

5.3.3.5 Synthesis

Based on the discussion in sections 5.3.3.1 to 5.3.3.4, Table 5-8 presents a qualification of the environmental impacts for to three intensification scenarios. It shows that conventional intensification could especially pose high risks on biodiversity, water quantity and quality and air quality. In the case of sustainable intensification, impacts are positive for almost all indicators.

5.4 DISCUSSION

In this chapter, we conducted a detailed region- and biofuel-specific assessment of the environmental impacts of agricultural and biofuel production for three ILUC mitigation scenarios and for three intensification pathways. By investigating the impacts of both agricultural intensification and biofuel production, we approach the subject of ILUC mitigation and biofuel production in a novel, integrated manner. While already a few previous studies have investigated the net GHG balance of biofuel and agricultural production, our approach is more detailed in terms of the GHG emission sources included, the impacts covered, the impacts of different intensification pathways and different ILUC mitigation (or biofuel potential) scenarios. This more detailed and integrated approach allows giving more comprehensive insights into the GHG and other environmental impacts of biofuel production.

Our study focused on the environmental impacts of ILUC mitigation. Other dimensions of sustainable bioenergy production, i.e. social and economic impacts are not included and need to be addressed in future research. This should also include an assessment of the trade-offs between environmental and socio-economic impacts, see e.g. Smeets and Faaij

TABLE 5-8 | Potential impacts on current levels of biodiversity, ground and surface water quantity and quality, soil quality and air quality in the case of conventional, intermediate sustainable and sustainable intensification toward 2020.

	Risk factor	Conventional intensification	Intermediate sustainable intensification	Sustainable intensification
Biodiversity				
Habitat functions in HNV areas	Conversion to miscanthus[a]	-	+/-	+
	Agricultural intensification	+/-	o	+
Species abundance in non-HNV areas	Agricultural intensification	--	+/-	+
	Cropland conversion to miscanthus	+/-	+	++
	Grassland conversion to miscanthus	--	-	+/-
Water				
Water quantity	Agricultural and miscanthus production	--	-	+/-
Water quality	Agricultural intensification	--	+/-	+
	Miscanthus cultivation	+/-	+	++
Soil				
SOC	Management and conversion of agricultural land	+/-	+	++
Soil erosion	Water erosion	-	+/-	+
	Wind erosion	+/-	+/-	+
Air				
Air quality	Airborne emissions of non-GHG pollutants causing acidification	--	-	+
	Emissions of PM_{10}	-	+/-	+
	Pesticides	--	-	+

[a] Only in selected areas where miscanthus cultivation is expected to have a positive impact on biodiversity
Symbols:
-- high risk of negative effects
- considerable risk of negative effects
0 low risk, no effects expected
+/- some risk, impacts may be either positive or negative
+ no risk, positive effects expected
++ no risk, high positive effects expected

[78]. The calculations of GHG emissions rely on a lot of uncertain data and assumptions. Key aspects are the projections of fertilizer and pesticide consumption levels, N_2O emission factors for the different intensification pathways and land use change. Below, these are discussed in more detail. Finally, the assessment of other environmental impacts only provides a first indication of these impacts and further quantification is needed. Because of high spatial heterogeneity in biophysical and climate conditions, this could, for example, be done by applying our approach to a spatially explicit analysis of environmental impacts. Van der Hilst et al. [14] provide an illustrative example of spatially explicit analysis, which could be further developed to integrate impacts of agricultural intensification.

5.4.1 Projections fertilizer and pesticide use

To project fertilizer consumption in 2020, correlations between fertilizer level and crop yield were derived from historical data. Yet, the number of data points found varies significantly between crops and fertilizer types. Therefore, the underpinning of assumptions about ranges in nutrient use efficiencies varies considerably. Also, the actual application levels will depend on local biophysical conditions. To determine fertilizer requirements more exactly, an environmental assessment model could be used that takes into account site-specific agro-ecological circumstances and crop specific nutrient demand (such as the MITERRA model used in de Wit et al. [9]). With regard to N_2O emission factors, the assumptions are based on IPCC values and might not fully correspond with the local situation. Nevertheless, the results clearly illustrate the considerable impact of different intensification pathways on the level of GHG emissions from fertilizer consumption and the importance of sustainable intensification to minimize these emissions.
Statistical data on pesticide use by crop is very limited and ranges significantly between countries [85-87]. The projections from the present chapter are within these wide ranges, but it is difficult to assess the suitability of the projected pesticide levels specifically for Lublin in 2020. Based on the ranges in pesticide use efficiency specified for different crops, the average annual increase in pesticide consumption is considered to be 0.06 kg a.i. ha^{-1} yr^{-1} in the case of sustainable intensification, 0.14 kg a.i. ha^{-1} yr^{-1} in the case of intermediate sustainable intensification and 0.40 kg a.i. ha^{-1} yr^{-1} in the case of conventional intensification. For comparison, the recent (2005-2012) national average growth trend in the total agricultural sector is 0.09 kg a.i. ha^{-1} yr^{-1}. Although the uncertainties about pesticide application levels are high, the impact on the total GHG balance will be limited as the share of emissions from pesticides is small. However, with regard to air quality and health, the impact of pesticides can be much more significant and sustainable intensification pathways are required to limit the use and negative effects of pesticides.

5.4.2 Land use change

In the present study, surplus agricultural land that is not converted to miscanthus cultivation is considered as abandoned land. In the case of cropland, this means that the resulting soil carbon sequestration is lower compared to the sequestration level in case of conversion to miscanthus. In the case of grasslands with high carbon stocks, the conversion to abandoned land may even result in soil carbon emissions. However, the land that is now assumed to be abandoned could also be converted to other land uses and this would impact the carbon stocks. For example, afforestation or other revegetation could, especially in the long run, significantly increase the carbon stocks in soil and biomass (see e.g. Schierhorn et al. [88]). In the past ten years, afforestation in Lublin has been limited to a few hundred hectares per year. But substantial improvements in agriculture, a reduction

in agricultural land use or financial support from the government can potentially increase the afforestation rate.

5.5 CONCLUSIONS

This study conducts a detailed analysis of the annual GHG emissions of agricultural and miscanthus-based ethanol production in Lublin province under three ILUC mitigation scenarios (low, medium and high) and three pathways for agricultural intensification in terms of sustainability. In addition, it qualitatively assesses other environmental impacts from agricultural and biofuel production for the three intensification pathways. The annual GHG balances for the ILUC mitigation scenarios in combination with different agricultural intensification pathways in 2020 are compared to the GHG balances of agricultural production in 2010 and 2020 without ILUC mitigation and miscanthus-based ethanol production (baseline). Generally, the ILUC mitigation scenarios attain lower net emissions compared to the baseline scenario. However, the reduction potential significantly depends on the intensification pathway considered due to differences in nutrient, pesticide and fuel use efficiencies, cattle production practices and agricultural management practices. For example, in the medium ILUC mitigation scenario, the GHG abatement potential ranges from 690 ktCO_2-eq yr^{-1} (0.5 tCO_2-eq ha^{-1} yr^{-1}) for conventional intensification to 3,815 ktCO_2-eq yr^{-1} (2.9 tCO_2-eq ha^{-1} yr^{-1}) for sustainable intensification. The intensification pathways have considerable influence on the emissions related to fertilizer consumption and soil carbon stock changes in each land use type. The deviation in net GHG emissions and GHG abatement levels is often larger for intensification pathways in the same ILUC mitigation scenario than for ILUC mitigation scenarios following same intensification pathway. For example, when considering the sustainable intensification pathway, the GHG abatement potential ranges from 2,920 ktCO_2-eq yr^{-1} (2.3 tCO_2-eq ha^{-1} yr^{-1}) for the low ILUC mitigation scenario to 4,850 ktCO_2-eq yr^{-1} (3.6 tCO_2-eq ha^{-1} yr^{-1}) for the high scenario. The variation in net annual emissions between different ILUC mitigation scenarios following the same intensification pathway can largely be explained by carbon sequestration due to land conversion.

In most cases, the ILUC mitigation scenarios also abate GHG emissions compared to gasoline. However, when following the conventional intensification pathway, no reduction is found (low and medium scenario) or the abatement potential is too low to comply with the EU GHG savings requirements of 35% or higher compared to fossil fuels (high scenario). In the case of the medium and high ILUC mitigation scenarios following the sustainable intensification pathway and the high ILUC mitigation scenario under intermediate sustainable intensification, the GHG reduction potential is higher than 90% compared to gasoline.

The importance of sustainable intensification is also recognized when considering other environmental impacts of ILUC mitigation and miscanthus cultivation. While conventional intensification could pose moderate to high risks on biodiversity, water quantity and quality, soil quality and air quality, sustainable intensification could mainly have positive impacts on each of these aspects. Based on the assessment of both the GHG balances and other environmental impacts, key measures for sustainable intensification and production in Lublin include optimizing the size of fields and the variation in vegetation types, applying good fertilizer management, integrated pest management, reduced or no tillage, replanting native vegetation or increasing afforestation on abandoned lands, and conserving grasslands with high SOC stock and/or high biodiversity value. When these measures are applied, a significant amount of bio-ethanol can be produced with low ILUC risk, high GHG abatement potentials and positive environmental impacts. Finally, the results from this study show that agricultural intensification has not only considerable influence on biomass potentials but also on the GHG and other environmental impacts of biomass production. Our novel and integrated approach allows assessing these impacts in a detailed, region-specific manner.

REFERENCES

[1] Searchinger T, Heimlich R, Houghton RA, et al. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. Science 2008;319(5867):1238-1240.

[2] Wicke B, Brinkman M, Gerssen-Gondelach S, et al. ILUC prevention strategies for sustainable biofuels: Synthesis report from the ILUC Prevention project. Copernicus Institute of Sustainable Development, Utrecht University: Utrecht, The Netherlands; 2015.

[3] Gerssen-Gondelach SJ, Wicke B, Borzęcka-Walker M, et al. Bioethanol potential from miscanthus with low ILUC risk in the province of Lublin, Poland. GCB Bioenergy 2015.

[4] Dornburg V, van Vuuren D, van de Ven G, et al. Bioenergy revisited: Key factors in global potentials of bioenergy. Energy & Environmental Science 2010;3(3):258-267.

[5] van Vuuren DP, van Vliet J, Stehfest E. Future bio-energy potential under various natural constraints. Energy Policy 2009;37(11):4220-4230.

[6] Erb K, Haberl H, Krausmann F, et al. Eating the Planet: Feeding and fuelling the world sustainably, fairly and humanely– a scoping study. Commissioned by Compassion in World Farming and Friends of the Earth UK. Institute of Social Ecology: Vienna, Austria; 2009. Social Ecology Working Paper No. 116.

[7] Slade R, Saunders R, Gross R, et al. Energy from biomass: the size of the global resource. Imperial College Centre for Energy Policy and Technology and UK Energy Research Centre: Londen, UK; 2011.

[8] Valin H, Havlik P, Mosnier A, et al. Agricultural productivity and greenhouse gas emissions: trade-offs or synergies between mitigation and food security? Environmental Research Letters 2013;8(3):9.

[9] de Wit MP, Lesschen JP, Londo MHM, et al. Greenhouse gas mitigation effects of integrating biomass production into European agriculture. Biofuels, Bioproducts and Biorefining 2014;8(3):374-390.

[10] Melillo JM, Reilly JM, Kicklighter DW, et al. Indirect Emissions from Biofuels: How Important? Science 2009;326(5958):1397-1399.

[11] van der Hilst F, Verstegen JA, Zheliezna T, et al. Integrated spatiotemporal modelling of bioenergy production potentials, agricultural land use, and related GHG balances; demonstrated for Ukraine. Biofuels, Bioproducts and Biorefining 2014;8(3):391-411.

[12] Smeets EMW, Lewandowski IM, Faaij APC. The economical and environmental performance of miscanthus and switchgrass production and supply chains in a European setting. Renewable and Sustainable Energy Reviews 2009;13(6-7):1230-1245.

[13] van Dam J, Faaij APC, Hilbert J, et al. Large-scale bioenergy production from soybeans and switchgrass in Argentina: Part B. Environmental and socio-economic impacts on a regional level. Renewable and Sustainable Energy Reviews 2009;13(8):1679-1709.

[14] van der Hilst F, Lesschen JP, van Dam JMC, et al. Spatial variation of environmental impacts of regional biomass chains. Renewable and Sustainable Energy Reviews 2012;16(4):2053-2069.

[15] Laborde D. Assessing the land use change consequences of European biofuels policies. International Food Policy Research Institute: Washington, DC, USA; 2011. Available from: http://trade.ec.europa.eu/doclib/docs/2011/october/tradoc_148289.pdf

[16] FAO. FAOSTAT [Internet: updated 2014, accessed 2014]. Available from: http://faostat.fao.org

[17] Moll S and Watson D. Environmental Pressures from European Consumption and Production. A study in integrated environmental and economic analysis. European Topic Centre on Sustainable Consumption and Production: Copenhagen, Denmark; 2009.

[18] Garnett T, Appleby MC, Balmford A, et al. Sustainable Intensification in Agriculture: Premises and Policies. Science 2013;341(6141):33-34.

[19] Hochman Z, Carberry PS, Robertson MJ, et al. Prospects for ecological intensification of Australian agriculture. European Journal of Agronomy 2013;44:109-123.

[20] Reetz HFJ, Heffer P, Bruulsema TW. Chapter 4. 4R nutrient stewardship: A global framework for sustainable fertilizer management. In: Drechsel P, Heffer P, Magen H, Mikkelsen R, Wichelns D, editors. Managing Water and Fertilizer for Sustainable Agricultural Intensification. 1st ed., International Fertilizer Industry Association (IFA), International Water Management Institute (IWMI), International Plant Nutrition Institute (IPNI), and International Potash Institute (IPI): Paris, France; 2015, p. 65-86.

[21] Smith P, Bustamante M, Ahammad H, et al. Agriculture, Forestry and Other Land Use (AFOLU). In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change . Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA.; 2014, p. 811-922.

[22] Smith P, Martino D, Cai Z, et al. Greenhouse gas mitigation in agriculture. Philosophical transactions of the Royal Society of London.Biological Sciences 2008;363(1492):789-813.

[23] Möckel S. 'Best available techniques' as a mandatory basic standard for more sustainable agricultural land use in Europe? Land Use Policy 2015;47:342-351.

[24] FAO and IIASA. GAEZ Global Agri-Ecological Zones [Internet: updated 2015, accessed 24 April 2015]. Available from: http://gaez.fao.org/Main.html#

[25] Fixen P, Brentrup F, Bruulsema TW, et al. Chapter 2. Nutrient/fertilizer use efficiency: Measurement, current situation and trends. In: Drechsel P, Heffer P, Magen H, Mikkelsen R, Wichelns D, editors. Managing Water and Fertilizer for Sustainable Agricultural Intensification. 1st ed., International Fertilizer Industry Association (IFA), International Water Management Institute (IWMI), International Plant Nutrition Institute (IPNI), and International Potash Institute (IPI): Paris, France; 2015, p. 8-38.

[26] FAO. Fertilizer use by crop in Poland. Food and Agriculture Organization of the United Nations (FAO): Rome, Italy; 2003.

[27] FAO. Fertilizer use by crop. Food and Agriculture Organization of the United Nations (FAO): Rome, Italy; 2006.

[28] FAO. FertiStat Fertilizer Use Statistics [Internet: updated 2007, accessed 21 April 2015]. Available from: http://www.fao.org/ag/agp/fertistat/index_en.htm

[29] Heffer P. Assessment of Fertilizer Use by Crop at the Global Level 2010-2010/11. International Fertilizer Industry Association (IFA): Paris, France; 2013.

[30] Rosas F. Fertilizer Use by Crop at the Country Level (1990–2010). Center for Agricultural and Rural Development, Iowa State University: Ames, Iowa; 2012.

[31] CSO. Local Data Bank [Internet: updated 2015, accessed 16 July 2015]. Available from: http://www.stat.gov.pl/bdlen/app/strona.html?p_name=indeks

[32] BioGrace Project. BioGrace excel tool. 2015.

[33] De Klein C, Novoa RSA, Ogle S, et al. Chapter 11. N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. The Intergovernmental Panel on Climate Change (IPCC): Japan; 2014, p. 11.1-11.54.

[34] Hydrogen Analysis Resource Center. Lower and Higher Heating Values of Fuels [Internet: updated 2015, accessed 23 June 2015]. Available from: http://hydrogen.pnl.gov/tools/lower-and-higher-heating-values-fuels

[35] Schreinemachers P and Tipraqsa P. Agricultural pesticides and land use intensification in high, middle and low income countries. Food Policy 2012;37(6):616-626.

[36] Surawska M and Kołodziejczyk R. The usage of plant protection products in Poland. Progress in Plant Protection 2006;46(1):470-483.

[37] Berent-Kowalska G and Stobiecki S. Pesticide sales and usage statistics in Poland. Poznan. 2009.

[38] Lorencowicz E and Uziak J. Fuel consumption in family farms. TEKA (Archives) of the Commission of Motorization and Power Industry in Agriculture - Polish Academy of Science Branch in Lublin 2009;9:164-171.

[39] CSO. Local Data Bank [Internet: updated 2014, accessed 2013, 2014]. Available from: http://www.stat.gov.pl/bdlen/app/strona.html?p_name=indeks

[40] Gerssen-Gondelach SJ, Lauwerijssen R, Havlik P, et al. Comparing intensification strategies for beef and dairy cattle production systems: impacts on GHG emissions, land occupation and land use change. Forthcoming.

[41] FAO. FAOSTAT [Internet: updated 2015, accessed 2015]. Available from: http://faostat.fao.org

[42] Sanderson FJ, Kucharz M, Jobda M, et al. Impacts of agricultural intensification and abandonment on farmland birds in Poland following EU accession. Agriculture, Ecosystems & Environment 2013;168:16-24.

[43] Scientific Committee on Animal Health and Animal Welfare. The welfare of cattle kept for beef production. European Commission, Health and Consumer Protection Directorate-General: 2001.

[44] Reijs JW, Daatselaar CHG, Helming JFM, et al. Grazing dairy cows in North-West Europe; Economic farm performance and future developments with emphasis on the Dutch situation. LEI Wageningen UR: The Hague, The Netherlands; 2013. LEI Report 2013-001.

[45] Lewandowski I, Clifton-Brown JC, Scurlock JMO, et al. Miscanthus: European experience with a novel energy crop. Biomass and Bioenergy 2000;19(4):209-227.

[46] Haines SA, Gehl RJ, Havlin JL, et al. Nitrogen and Phosphorus Fertilizer Effects on Establishment of Giant Miscanthus. BioEnergy Research 2015;8(1):17-27.

[47] Cadoux S, Riche AB, Yates NE, et al. Nutrient requirements of Miscanthus x giganteus: Conclusions from a review of published studies. Biomass and Bioenergy 2012;38:14-22.

[48] Borzęcka-Walker M, Faber A, Pudełko R, et al. Simulation of greenhouse gases from miscanthus cultivation in Poland using the DNDC model. Journal of Food, Agriculture & Environment 2012;10(2):1187-1190.

[49] Matyka M and Kus J. Yielding and biometric characteristics of selected miscanthus genotypes. Problemy Inżynierii Rolniczej 2011;2:157-163.

[50] Borkowska H and Molas R. Yield comparison of four lignocellulosic perennial energy crop species. Biomass and Bioenergy 2013;51:145-153.

[51] Borzęcka-Walker M. Nutrient content and uptake by *Miscanthus* plants. Electronic Journal of Polish Agricultural Universities 2010;13(3):.

[52] Miguez FE, Villamil MB, Long SP, et al. Meta-analysis of the effects of management factors on Miscanthus × giganteus growth and biomass production. Agricultural and Forest Meteorology 2008;148(8–9):1280-1292.

[53] Nemecek T and Kägi T. Life Cycle Inventories of Agricultural Production Systems. EcoInvent: Zurich and Dübendorf; 2007. 15.

[54] Monti A, Fazio S, Venturi G. The discrepancy between plot and field yields: Harvest and storage losses of switchgrass. Biomass and Bioenergy 2009;33(5):841-847.

[55] Shinners KJ, Boettcher GC, Muck RE, et al. Harvest and Storage of Two Perennial Grasses as Biomass Feedstocks. Transactions of the ASABE 2010;53(2):359-370.

[56] Hoefnagels R, Smeets E, Faaij A. Greenhouse gas footprints of different biofuel production systems. Renewable and Sustainable Energy Reviews 2010;14(7):1661-1694.

[57] Hamelinck CN and Faaij APC. Outlook for advanced biofuels. Energy Policy 2006;34(17):3268-3283.

[58] Aden A, Ruth M, Ibsen K, et al. Lignocellulosic Biomass to Ethanol Process Design and Economics Utilizing Co-Current Dilute Acid Prehydrolysis and Enzymatic Hydrolysis for Corn Stover . National Renewable Energy Laboratory: Golden, Colorado, USA; 2002. Available from: http://www.nrel.gov/docs/fy02osti/32438.pdf

[59] Tao L and Aden A. The economics of current and future biofuels. In Vitro Cellular & Developmental Biology - Plant 2009;45(3):199-217.

[60] Hamelinck CN, Suurs RAA, Faaij APC. International bioenergy transport costs and energy balance. Biomass and Bioenergy 2005;29(2):114-134.

[61] Oak Ridge National Laboratory. Conversion Factors for Bioenergy. North Carolina Cooperative Extension: 2008.

[62] Lasco RD, Ogle S, Raison J, et al. Chapter 5. Cropland. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. The Intergovernmental Panel on Climate Change (IPCC): Japan; 2007, p. 5.1-5.66.

[63] Verchot L, Krug T, Lasco RD, et al. Chapter 6. Grassland. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. The Intergovernmental Panel on Climate Change (IPCC): Japan; 2007, p. 6.1-6.49.

[64] European Commission. Commission Decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC. Official Journal of the European Union L 151 2010;53:19-41.

[65] Borzecka-Walker M [personal communication]. 2014. Institute of Soil Science and Plant Cultivation, Pulawy, Poland.

[66] Lugato E, Panagos P, Bampa F, et al. A new baseline of organic carbon stock in European agricultural soils using a modelling approach. Global Change Biology 2014;20(1):313-326.

[67] Himken M, Lammel J, Neukirchen D, et al. Cultivation of Miscanthus under West European conditions: Seasonal changes in dry matter production, nutrient uptake and remobilization. Plant and Soil 1997;189(1):117-126.

[68] European Commission. Sustainability criteria [Internet: updated 2015, accessed 24 July 2015]. Available from: http://ec.europa.eu/energy/node/73

[69] Fritsche UR, Rausch L and Schmidt K. Life Cycle Analysis of GHG and Air Pollutant Emissions from Renewable and Conventional Electricity, Heating, and Transport Fuel Options in the EU until 2030. Öko-Institut e.V.: Darmstadt, Germany; 2009.

[70] Franke B, Reinhardt G, Malavelle J, et al. Global Assessments and Guidelines for Sustainable Liquid Biofuel Production in Developing Countries. A GEF Targeted Research Project. Heidelberg/Paris/Utrecht/Darmstadt; 2012.

[71] Tóth G, Adhikari K, Várallyay G, et al. Updated map of salt affected soils in the European Union. In: Tóth G, Montanarella L, Rusco E, editors. Threats to Soil Quality in Europe. European Commission, Joint Research Centre, Institute for Environment and Sustainability: Ispra (VA), Italy; 2008, p. 65-78.

[72] Borzęcka-Walker M, Faber A, Mizak K, et al. Soil Carbon Sequestration Under Bioenergy Crops in Poland. In: Özkaraova Güngör EB, editor. Principles, Application and Assessment in Soil Science. InTech: 2011, p. 151-166.

[73] Cramer J, Wissema E, de Bruijne M, et al. Testing framework for sustainable biomass, final report from the project group "Sustainable production of biomass". Project group Sustainable Production of Biomass: The Hague, The Netherlands; 2007.

[74] Dauber J, Jones MB, Stout JC. The impact of biomass crop cultivation on temperate biodiversity. GCB Bioenergy 2010;2(6):289-309.

[75] Donnelly A, Styles D, Fitzgerald J, et al. A proposed framework for determining the environmental impact of replacing agricultural grassland with Miscanthus in Ireland. GCB Bioenergy 2011;3(3):247-263.

[76] Mioduszewski W. Small (natural) water retention in rural areas. Journal of Water and Land Development 2014;20(1):19-29.

[77] Statistical Office in Lublin. Statistical yearbook of the Lubelskie voivodship 2014. Statistical Office in Lublin: Lublin, Poland; 2014.

[78] Smeets EMW and Faaij APC. The impact of sustainability criteria on the costs and potentials of bioenergy production – Applied for case studies in Brazil and Ukraine. Biomass and Bioenergy 2010;34(3):319-333.

[79] Triana F, Nassi o Di Nasso N, Ragaglini G, et al. Evapotranspiration, crop coefficient and water use efficiency of giant reed (Arundo donax L.) and miscanthus (Miscanthus ɔgiganteus Greef et Deu.) in a Mediterranean environment. GCB Bioenergy 2015;7(4):811-819.

[80] Dornburg V, Faaij A, Verweij P, et al. Biomass Assessment. Assessment of global biomass potentials and their links to food, water, biodiversity, energy demand and economy: Inventory and analysis of existing studies. Supporting document. Netherlands Environmental Assessment Agency: Bilthoven, The Netherlands; 2008.

[81] Szejba D. Evapotranspiration of Grasslands and Pastures in North-Eastern Part of Poland. In: Irmak A, editor. Evapotranspiration - Remote Sensing and Modeling. InTech: Rijeka, Croatia; 2012, p. 179-196.

[82] Squire GR, Hawes C, Valentine TA, et al. Degradation rate of soil function varies with trajectory of agricultural intensification. Agriculture, Ecosystems & Environment 2015;202:160-167.

[83] Galbraith D, Smith P, Mortimer N, et al. Review of Greenhouse Gas Life Cycle Emissions, Air Pollution Impacts and Economics of Biomass Production and Consumption in Scotland. Scottish Executive, Environment and Rural Affairs Department: Edinburgh, UK; 2006.

[84] Sattler C, Kächele H, Verch G. Assessing the intensity of pesticide use in agriculture. Agriculture, Ecosystems & Environment 2007;119(3–4):299-304.

[85] CBS. Gebruik gewasbeschermingsmiddelen in de landbouw (Use of crop protection agents in agriculture, in Dutch) [Internet: updated 2015, accessed 16 July 2015]. Available from: http://statline.cbs.nl/

[86] Fera. Pesticide usage tables [Internet: updated 2015, accessed 16 July 2015]. Available from: https://secure.fera.defra.gov.uk/pusstats/myindex.cfm

[87] Fernandez-Cornejo J, Nehring R, Osteen C, et al. Pesticide Use in U.S. Agriculture, 21 Selected Crops, 1960-2008. U.S. Department of Agriculture, Economic Research Service: 2014. EIB-124.

[88] Schierhorn F, Müller D, Beringer T, et al. Post-Soviet cropland abandonment and carbon sequestration in European Russia, Ukraine, and Belarus. Global Biogeochemical Cycles 2013;27(4):1175-1185.

[89] European Commission. Management of Natura2000 sites: Best practices [Internet: updated 2015, accessed 10 July 2015]. Available from: http://ec.europa.eu/environment/nature/natura2000/management/best_practice_en.htm

[90] Panagos P, Ballabio C, Borrelli P, et al. Rainfall erosivity in Europe. Science of The Total Environment 2015;511:801-814.

[91] Borrelli P, Panagos P, Ballabio C, et al. Towards a pan-European assessment of land susceptibility to wind erosion. Land Degradation & Development 2014.

APPENDIX

Qualitative assessment of other environmental impacts

Biodiversity

In Gerssen-Gondelach et al. [3] it was assumed that biomass production can take place in a small part of HNV areas, where the conversion to miscanthus cultivation improves biodiversity. However, to realize this, sustainable production practices are important. In some HNV areas, such as Natura2000 sites, agricultural production takes place. In Natura2000 areas, management practices are promoted that are economically sustainable and conserve the present species and habitats [89]. When strict protection criteria are lacking, regional developments towards conventional agricultural intensification might pose some risks for these areas. Alternatively, the promotion of sustainable intensification practices in the region could also give an impulse for further improvements in HNV areas. In other agricultural areas, intensification is expected to lead to scaling up of farms and potentially also specialization. In the case of conventional intensification, this causes an increase in monocultures and loss, modification and fragmentation of habitats. This, in turn, leads to a decrease in species abundance. Also, inefficient fertilizer and pesticide usage results in increased leaching, soil degradation and water pollution, which are risk factors for species abundance [14]. In the case of sustainable intensification, the optimization of the size of fields and the variation in vegetation types, and the application of good fertilizer management and integrated pest protection reduce negative impacts on biodiversity [12,20,23].

One specific aspect that should be considered when investigating the most suitable practices for sustainable intensification in Lublin, is the conservation of farmland birds. Poland supports a significant share of farmland bird populations in the EU and should therefore play an important role in the conservation of these birds [42]. However, Sanderson et al. [42] find that reductions in low-intensity farmland cover result in decreased abundance of farmland birds and sometimes also in their species richness. The effects are more negative when low-intensity farmland is lost because of intensification than because of abandonment [42].

The conversion of arable lands to miscanthus is often found to have a positive impact on species abundance as miscanthus is an extensively managed crop [12,74]. The effect is expected to be largest when sustainable production practices are applied. The impacts of the conversion of grasslands to miscanthus are more uncertain and not yet well understood [74,75], but the risk of biodiversity loss is estimated to be higher. In Poland, many grasslands are semi-natural and extensively managed [42] and the conversion of extensive pastures has a higher risk of biodiversity loss than the conversion of intensively

managed pastures [14]. In addition, Van der Hilst et al. [14] find that biodiversity in the North of the Netherlands is at risk when converting areas with high densities of important bird species. When these considerations are taken into account in sustainable production pathways, the location of miscanthus cultivation should be chosen with the aim to avoid areas of significant biodiversity value [70].

Water

In Poland, annual precipitation is low, but monthly precipitation is highest during summer when the evapotranspiration is also peaking. However, the evapotranspiration exceeds the precipitation and water deficits occur during the summer [76]. In Lublin, the average annual precipitation ranges between approximately 500 and 600 mm depending on the location [77]. Figures from literature (Table 5-7) show that the water requirements of agricultural crops may exceed the annual precipitation, which results in a negative annual water balance and the occurrence of droughts. Conventional agricultural intensification increases the risk of groundwater deficiencies and droughts [76]. Risk factors include monocultures and irrigation [21]. Sustainable intensification practices that improve soil moisture, e.g. by increasing soil organic carbon through reduced tillage, may help crops to better withstand droughts [21,70]. Despite its high water use efficiency, the rate of evapotranspiration of miscanthus is found to be higher compared to traditional annual crops and pastures [12,14], see Table 5-7. Large scale cultivation of miscanthus, especially in the case of monocultures, will thus contribute to the risk of water depletion. In the assessment of the low-ILUC-risk miscanthus potential by Gerssen-Gondelach et al. [3], minimal water requirements (550 mm yr^{-1}) were already taken into account. Thus, areas with a high risk of depleting water bodies (groundwater reservoirs, streams and lakes) and competition with other water uses are excluded. However, more site-specific data and analysis is required to assess the impacts of miscanthus production on water availability in Lublin. This should take into account variables like the soil texture, rainfall pattern, wind speed, cropping pattern and the location-specific crop evapotranspiration factors for arable crops and miscanthus [78].

Excessive use of fertilizers and pesticides causes contamination of ground and surface water and increases the risk of euthrophication. In sustainable intensification, good fertilizer management and integrated pest protection reduce leaching and improve the water quality [20,23]. As miscanthus is managed more extensively than annual crops, the conversion of arable lands to miscanthus has a positive impact on water quality [14]. In the case of conversion of semi-natural, low-intensity grasslands, the fertilizer use may actually increase and sustainable fertilizer practices are required to avoid negative effects.

Soil

The GHG balances for the ILUC prevention scenarios showed that the chosen intensification pathway has a significant influence on the SOC balance and on the net emissions. This is confirmed by the literature. For example, Squire et al. [82] find that increased fertilizer and pesticide use in the UK resulted in reduced SOC and also in lower water holding capacity of the soil. Thus, sustainable intensification practices are important to prevent SOC losses [70]. In addition to reduced or no tillage, measures to increase SOC include the use of cover crops and replanting native vegetation on abandoned land [21-23].
Based on average SOC values used in the GHG emission calculations, the conversion of croplands and grasslands to miscanthus improves the soil carbon stocks. But the variation in SOC values for grasslands is large. Extensively managed grasslands have higher SOC stocks and the risk of carbon loss increases when converting these lands [14]. The risk of converting high carbon stock pastures increases as more pastures are converted in the higher scenarios. Although grasslands with the highest risk of carbon losses (i.e. boggy and wet areas) were already excluded from miscanthus cultivation in the underlying study determining the surplus land area available for miscanthus [3], it needs to be carefully assessed which of the remaining grasslands can be converted to miscanthus while achieving SOC sequestration and which grasslands should be maintained to prevent SOC losses.

With regard to soil erosion, the risk of water erosion is higher than wind erosion. The average annual precipitation in Lublin is low, but rainfall has a relatively high erosive effect compared to regions in Western Poland [90]. This is related to the occurrence of droughts, as rainfall on dry soils causes great damage. As a result, the risk of soil erosion due to rainfall is medium to high and is highest in the central and south-western part of Lublin [90]. Also, conventional agricultural intensification and unification of land cover decrease the water retention capacity of the soil and increase the risk of soil erosion due to precipitation [76]. In conversion, practices like reduced tillage that improve soil organic carbon may reduce water erosion [21]. Theoretically, the soil in Lublin is also susceptible to wind erosion. Major risk factors are the low precipitation levels and the large area of arable land, which land type is considered to be most susceptible to wind erosion [91]. However, in practice, the risk of wind erosion is low because of a limited occurrence of high wind speeds [91].

When converting agricultural lands to miscanthus, the risk of soil erosion is expected to decrease in the case of cropland conversion [12,14]. However, in the case of grassland conversion, the risk of soil erosion increases [12,14]. This is especially true in the first year of miscanthus establishment and for extensive pastures [14].

Air

Important sources of non-GHG pollutants causing acidification are fertilizer use (NH_3), manure management (NH_3) and tractor fuel combustion (NO_x) [83]. In addition, fuel combustion causes emissions of fine particles. Thus, sustainable intensification, including increased fertilizer, pesticide and fuel use efficiency and adoption of improved manure management technologies, is important to maintain or improve the air quality.

Another risk factor for air quality is pesticide usage as pesticides have the potential to contaminate the air and pose a risk on the environment and human health [84]. The risks depend on the type of pesticide (ingredients) applied. Thus, although the contribution of pesticides to the total GHG balance is small, sustainable intensification and integrated pest protection are important to minimize other environmental impacts. Integrated pest protection includes, for example, the use of natural predators and parasitoids instead of pesticides [21,23]. When pesticides are used, these should be selected carefully and applied at the right rate, time and place.

CHAPTER 6

Competing uses of biomass: assessment and comparison of the performance of bio-based heat, power, fuels and materials

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ABSTRACT

The increasing production of modern bioenergy carriers and biomaterials intensifies the competition for different applications of biomass. To be able to optimize and develop biomass utilization in a sustainable way, this chapter first reviews the status and prospects of biomass value chains for heat, power, fuels and materials, next assesses their current and long-term levelized production costs and avoided emissions, and then compares their greenhouse gas abatement costs. At present, the economically and environmentally preferred options are wood chip and pellet combustion in district heating systems and large-scale cofiring power plants (75-81 $US\$_{2005}/tCO_2\text{-}eq_{avoided}$), and large-scale fermentation of low cost Brazilian sugarcane to ethanol (-65 to -53 $\$/tCO_2\text{-}eq_{avoided}$) or biomaterials (-60 to -50 $\$/tCO_2\text{-}eq_{avoided}$ for ethylene and -320 to -228 $\$/tCO_2\text{-}eq_{avoided}$ for PLA; negative costs represent cost effective options). In the longer term, the cultivation and use of lignocellulosic energy crops can play an important role in reducing the costs and improving the emission balance of biomass value chains. Key conversion technologies for lignocellulosic biomass are large-scale gasification (bioenergy and biomaterials) and fermentation (biofuels and biomaterials). However, both routes require improvement of their technological and economic performance. Further improvements can be attained by biorefineries that integrate different conversion technologies to maximize the use of all biomass components.

6.1 INTRODUCTION

In the last decade, biomass use for the production of modern bioenergy and biomaterials grew significantly in order to oppose the depletion of fossil resources (and associated increasing energy prices) and to reduce greenhouse gas (GHG) emissions [1]. For both energy and material application of biomass, it is expected that this growth will continue or even accelerate. For example, the Intergovernmental Panel on Climate Change (IPCC) reviewed recent literature and scenarios on long-term biomass deployment potentials and biomass demand for bioenergy [2,3]. In 2008, global bioenergy use accounted for a primary biomass supply of 50 exajoule (EJ_p) per year. By 2050, the global biomass demand for bioenergy is projected to reach about 77 EJ_p/year in the absence of climate policies (median case of baseline scenarios) and about 155 EJ_p/year under the most stringent GHG mitigation scenarios [3]. In addition, Saygin et al. [4] estimate an economic potential of biomass use of almost 20 EJ_p/year for substitution of synthetic organic material in the chemical industry in 2050. Hence, a total biomass supply of 100-175 EJ/year would be required to meet projected demand for both bioenergy and biomaterials in 2050. By the same year, the technical biomass deployment potential is estimated to be in the range of 100-300 EJ_p/year [2].

The increasing demand for biomass will intensify the competition between biomass feedstocks as well as their applications; not only between food and non-food uses, but also between non-food applications for energy and materials. Thus, to ensure sustainable expansion of biomass use, we need insight in which routes (biomass value chains) are most promising for producing heat, power, fuels and materials in terms of their technological, economic and environmental performance. This requires: i) a clear view on the status and prospects of potential value chains; and ii) assessment and comparison of their economic and environmental performance on the short and longer term. Assessment of the performance over time is important, because biomass value chains are in different stages of development and have different potentials for improvement. For example, on the short term, new technologies may be more expensive than established technologies. But, as capacity deployment increases, with technological learning, they could become cheaper in the longer term. Key indicators for the economic and environmental performance of biomass value chains are levelized production costs, avoided greenhouse gas emissions and GHG abatement costs.

Although these aspects have been assessed widely in literature, earlier (review) work mainly considers bioenergy, and especially biofuels [5-8]. This literature generally considers either environmental or economic aspects [5,7-10]. In addition, most studies that consider biomaterials focus on environmental impacts (see, e.g., [11-13]), while the number of economic assessments is limited [14,15]. Comparative work between bioenergy and biomaterials only includes environmental aspects [16] or biomass use in

the manufacturing sector [17]. However, as energy and material applications in different sectors are competing for biomass feedstocks, only an assessment that includes both their economic and environmental impact can generate better insight in the overall performance of the various biomass value chains. Finally, for various bioenergy systems and their components, literature has analyzed the role of technological learning in historical cost developments [18-20]. For a good understanding of economic improvement potentials over time, and of potential speeds of technological development and deployment, these insights need to be included in a comparative assessment.

In order to address some of the shortcomings identified in the existing literature, the aim of this chapter is to evaluate existing and potential biomass value chains for heat, power, fuels and materials. This includes a review of the current status and prospects of these biomass value chains, an assessment of current and projected levelized production costs and GHG emission reductions, and a comparison of the GHG abatement costs of these chains for different time frames.

The remainder of this chapter is organized as follows. In section 2, we explain the methodology of our review and assessment. We discuss the status and prospects of existing and novel biomass value chains in section 3. Next, we present collected cost and GHG emission data in section 4, and select a number of biomass value chains for further comparison. In section 5, we present the results and perform a sensitivity analysis. We discuss important assumptions and limitations in our work in section 6, and draw conclusions in section 7.

6.2 METHODOLOGY

Our work examines four components of the biomass value chains: biomass production, feedstock pretreatment, transportation and conversion. First, we describe the options for each component (e.g. different conversion technologies) and the status and prospects of these options based on a literature review. Secondly, we collect cost and GHG emission data for all components of the biomass value chain as well as for the fossil reference chains, all based on literature review. The review includes data for both the present level of technology and projections for 2030. Thirdly, we define biomass value chains by selecting an option for each of its components. By using the cost and emission data from the preceding review, we calculate and compare the levelized production costs, avoided emissions and GHG abatement costs of these value chains and their fossil reference.

Regarding feedstock production, we also consider yields in various geographical regions. In addition, we assess the potential effect of technological learning on future cost developments. Direct comparison of the levelized costs and GHG emissions requires a uniform functional unit, e.g. the unit of input biomass or the unit of output. Because we consider diverse products with different functions and from different feedstock types (including agricultural and forestry residues), we cannot select one common functional unit

[21,22]. Therefore, we first evaluate the levelized production costs and GHG emissions (per gigajoule (GJ) or tonne product) in relation to the levelized costs and GHG emissions of the fossil equivalent products, respectively. Thereafter, we compare the different value chains by calculating their avoided GHG emissions per hectare (tonnes of carbon dioxide equivalent (tCO_2-$eq_{avoided}$) per hectare (ha)) and GHG abatement costs (in real 2005 US Dollars ($US\$_{2005}$) per tCO_2-$eq_{avoided}$). We make all calculations on the basis of higher heating values (HHV).
The explicit inclusion of co- and/or by-products is an important methodological aspect in calculating levelized costs and GHG emissions of e.g. combined heat and power production (CHP) or biodiesel production with glycerin as a by-product. In scientific literature, system expansion is a commonly used and generally preferred allocation method [23,24]. To calculate levelized costs, we apply system expansion by taking into account a revenue for by-products (for CHP, we consider heat as a by-product). We only collect GHG emission data from life cycle inventory studies that apply system expansion.

6.2.1 Cost data standardization

In order to make a fair comparison, we standardize the collected cost data and calculate levelized production costs. We perform the following procedure:

i. Indexation. We convert cost data to $US\$_{2005}$ using gross domestic product (GDP) inflators [25] and annual currency exchange rates [26].
ii. Feedstock cost. We choose an average cost to deliver the biomass feedstock to a conversion plant in the selected region, based on our review work. For the fossil reference chains, we use present and projected fossil resource price data as approximation of the delivery costs.
iii. By-product revenue. We choose a fixed by-product revenue for each by-product type, based on values found in literature.
iv. Calculation of levelized cost. We apply the methodology as described and used in Bruckner at al. [27]:

$$LCOP = \frac{\alpha \cdot I + OM_{fixed} + OM_{\text{var, non feed}} + F - R}{P} \qquad (1)$$

$$\alpha = \frac{r}{1-(1+r)^{-L}} \qquad (2)$$

where *LCOP* = levelized cost of product [$/GJ product or $/t product]; α = capital recovery factor [%/yr]; *I* = investment cost [$]; OM_{fixed} = fixed annual operation cost [$/yr]; $OM_{var,non\ feed}$ = non-feed variable operation costs [$/yr]; *F* = feedstock cost [$/yr]; *R* = By-product revenue [$/yr]; *P* = annual production [GJ product or tonne product per year]; *r* = annual discount rate (7%); *L* = economic lifetime [yr].

The fixed annual operation costs (OM_{fixed}) consist of labor, maintenance, plant overhead and insurance. Variable operating costs are composed of feedstock costs (F), by-product revenues (R), and non-feed variable operating costs ($OM_{var,non\ feed}$), which include utilities,

auxiliaries and catalysts. In order to reflect the varying lifetimes and capacity factors in actual conversion facilities, we do not standardize these parameters.
We calculate levelized costs for all biomass value chains, and for heat, power and material production from oil, natural gas and coal. For fossil gasoline and diesel, we use the wholesale gasoline and diesel fuel prices as the surrogate for the gasoline and diesel fuel production costs. We adopt the methodology from the US Environmental Protection Agency [28] to estimate how the crude oil price affects the wholesale gasoline and diesel prices:

$$\mathrm{COG} = (P_{oil} \cdot 2.65) + 27 \qquad (3)$$

$$\mathrm{COD} = (P_{oil} \cdot 3.38) - 11.7 \qquad (4)$$

Where *COG* and *COD* are the cost of gasoline and diesel [US¢/gallon] respectively, and P_{oil} is the crude oil price [$/bbl].

6.2.2 Avoided GHG emissions and GHG abatement costs

To calculate avoided GHG emissions, we collect GHG emission data from life cycle inventories which include all activities from resource extraction (cradle) to final end use (grave)[1], and cover both direct and indirect emissions (i.e. emissions that are a result of the consumption of purchased materials and energy, and of upstream activities such as the production and transport of purchased materials and energy). We use the data without any harmonization in background assumptions. Effects of direct and indirect land use change (DLUC and ILUC) are space and time dependent and case specific [29,30]. As we want to compare the biomass value chains on a more general level, we exclude LUC induced GHG emissions from our analysis, but we consider this topic in our discussions. To calculate GHG abatement costs, we apply the methodology as described and used in Damen et al. [31]:

$$GHG\text{abatement cost} = \frac{LCOP_{bio} - LCOP_{ref}}{m_{GHG,\,ref} - m_{GHG,\,bio}} \qquad (5)$$

Where *LCOP* is the levelized cost ($/GJ or $/tonne) and *m* the GHG emission factor (tCO_2-eq/GJ or tCO_2-eq/tonne) of the biomass (*bio*) and reference (*ref*) chain.

The calculations of levelized costs, avoided emissions, and GHG abatement costs can be found in the supplementary material.

6.2.3 Sensitivity analysis

We apply a sensitivity analysis to evaluate how GHG abatement costs are affected by feedstock costs, fossil energy prices and annual discount rates.

1 For bio-materials, it is assumed that these are transported to waste-to-energy-facilities after their end of life and that they are incinerated without energy recovery.

6.3 CHAIN DESCRIPTION

Many possible compositions exist for biomass value chains (Figure 6-1). In this section we discuss the status and prospects of the different options for feedstock production, logistics, pre-treatment and conversion.

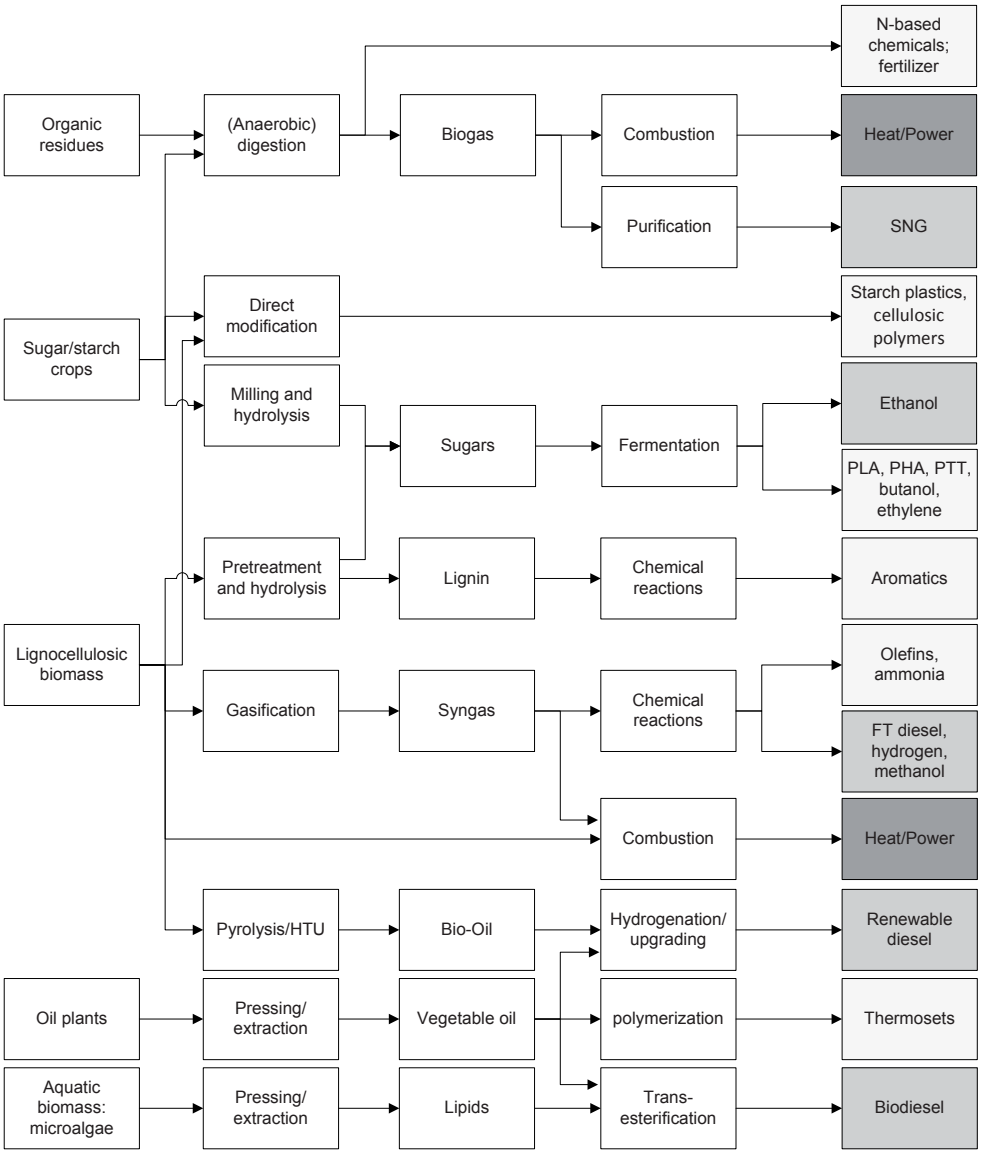

FIGURE 6-1 | Main routes from biomass to bioenergy and biomaterials discussed in this chapter.

6.3.1 Feedstocks

A wide range of feedstocks is used for the production of bioenergy and biomaterials. Heat and power systems rely mostly on wood (e.g. logs, sawdust), and residues and wastes from agriculture, food processing and municipal solid waste (MSW) [6]. Sugar, starch and oil crops are currently the main feedstocks for (liquid) biofuels and for biomaterials [6,32]. Other feedstocks that are expected to become important for the production of biofuels and biomaterials are dedicated energy crops, including short rotation crops (SRC, e.g. willow, poplar, eucalypt) and perennials (switchgrass, miscanthus), and lignocellulosic wastes and agricultural and forestry residues [33,34]. In addition, organic residues like manure are applied for the production of gaseous biofuel. Aquatic biomass is potentially a highly productive source of oil (microalgae) or polysaccharides (macroalgae). Yet, the technology and process chain need substantial improvement before commercial production will be economically feasible [35].

The technical supply potential of biomass resources depends on numerous factors. Agricultural and forestry residues and organic wastes ultimately depend on the demand for conventional agricultural and forestry products and on ecological considerations. Based on literature review, Chum et al. [2] estimate that these resources have prospects to provide between 25 and 280 EJ/yr by 2050. For the same year, the technical potential for conventional agricultural crops and dedicated energy crops ranges from 0-810 EJ/yr [2]. The potential is mainly determined by land availability (and suitability) and biomass yields (for current and projected yields, see section 6.4.1). While conventional crops are especially suitable for cultivation on agricultural land, dedicated crops are also considered to be suitable for cultivation on marginal and degraded lands [34]. Microalgae are cultivated in open ponds or closed photobioreactors (PBRs), and thus do not compete for arable land [35].

6.3.2 Biomass logistics

When feedstock production and conversion are located in the same country, the raw feedstock is directly transported to the conversion plant or to an intermediate storage facility, from where further distribution takes place [28]. For corn in the USA, for example, the feedstock transport distance from the farm to the ethanol plant is below 80 km [19]. In Indonesia, oil palm mills are located close to the cultivation areas, and crude palm oil is transported to biodiesel plants that are more distant [36]. In contrast, wood pellets for power and heat production are traded over long distances. These pellets are mainly transported from Canada, the USA and Europe to the USA and Europe. The main transport modes are ocean freight, short sea shipping and truck transport [37].
National and international trade of biomass feedstocks, especially of pellets, has expanded

in recent years [38]. As a result of the introduction of lignocellulosic biomass for biofuels and biomaterials, Latin America and Sub-Saharan Africa are considered to be potential large net exporters of biomass and international trade is expected to play a pivotal role in further development of the biomass sector [39,40]. Optimization of international freight costs and of storage and handling at seaports are key prerequisites to achieve low cost feedstock supply [38].

6.3.3 Biomass pretreatment technologies

As handling and (long distance) transport of raw biomass is inefficient and economically unattractive, pretreatment or densification is applied. At biofuel plants, pretreatment basically refers to mechanical treatment (e.g. milling) of the feedstock to clean and size the biomass and to destroy the cell structure. Below, we discuss pretreatment technologies that aim at efficient and cheap transport of feedstocks. Due to growing international trade, the importance of these technologies will increase in the future [40].

Chipping, briquetting and pelletization

Chipping, briquetting and pelletization are wide-spread methods to pretreat wood resources. When chipping, the feedstock is reduced in size. During briquetting or pelletization, biomass is compressed and extruded in screw or piston presses [2]. Compared to chips, briquettes or pellets have a uniform size, and moisture and heat content, which enhances handling [41]. Pellets have a higher heating value (HHV) of about 20.5 GJ/t_{dm}, compared to 19.5 GJ/t_{dm} for chips [42]. For pelletization, Uslu et al. [43] report a net energetic efficiency of 84-88% (LHV).

Most pellet plants are located in Europe, Canada and the USA. These plants purchase their feedstock from nearby or adjacent sawmills, or from the logging industry [44,45]. Briquetting plants can usually be found in India and Thailand, and use a range of by-products from the food and forest processing industry (secondary residues). Chips are generally produced from wood waste, as a byproduct of conventional forestry [2].

Torrefaction

Torrefaction is the anaerobic heating of biomass at temperatures between 200 and 300°C. During the process, the feedstock loses water but maintains 90% of its original energy content [46]. Torrefied biomass is a solid uniform product with a HHV of 20-24 GJ/t_{dm} [43]. But, it is also a porous product with a low volumetric density, and further densification is desirable. Torrefaction combined with pelletisation (TOP) is considered to be an attractive option. Torrefaction technology is in the demonstration phase [43,47]. Plants that have been built at commercial scale are not operating at full capacity yet [48]. The net efficiency (LHV) is 92% for torrefaction and 90-95% for TOP [43].

Pyrolysis & hydrothermal upgrading

Under anaerobic conditions and at a high temperature (500 ^{o}C), biomass decomposes into liquid bio-oil, charcoal (biochar) and a mixture of gas (syngas). A distinction can be made between slow and fast pyrolysis, depending on the residence time in the reactor. Biochar production is maximized by slow pyrolysis, bio-oil production by fast pyrolysis [6]. Because bio-oil has a higher energy density compared to pellets and torrefied biomass, fast pyrolysis is considered to be an attractive pretreatment technology [6]. The net efficiency (LHV) of pyrolysis is 64% [43].

Bio-oil can also be produced by hydrothermal upgrading (HTU, also hydrothermal liquefaction). At high pressure (120-200 atm), a temperature of 300-400 ^{o}C and in the presence of water, biomass is liquefied [6]. While slow pyrolysis is commercially applied throughout the world, fast pyrolysis is an early commercial technology [49,50]. HTU technology is in the demonstration phase. Commercial application of pyrolysis and HTU bio-oil is closest to being realized in heat and power generation. In addition, research activities consider different routes for bio-oil upgrading to biofuels such as diesel, gasoline and kerosene [6,43,49]. Biomass feedstock for pyrolysis and HTU will mainly be lignocellulosic material, but also algal lipids could be converted to bio-oil by these technol

6.3.4 Biomass conversion technologies

Figure 6-2 shows the development phases of biomass conversion technologies for heat, power, fuels and materials. This overview is not meant to be exhaustive, but gives an overview of possible technologies that are regularly considered in literature. Especially with regard to biomaterials, the options are numerous. Below, we discuss the current status and prospects of the most important technologies.

6.3.4.1 Heat and power

A wide variety of heat and power generation routes are available, which can be classified into dedicated combustion (biomass feedstock only), cofiring, gasification and anaerobic digestion. These routes are applied for heat or power only, as well as for CHP (also called co-generation). Cogeneration is mainly applied to increase the overall efficiency of power production. CHP plants play a significant role in the production of heat and power, and it is expected that they will maintain this position in the future [2,6].

Dedicated combustion

The leading technologies in dedicated biomass combustion are pellet boilers or chip burners. These firing systems are commercially applied to produce hot water or steam. Main applications are domestic and district heating, and power and CHP generation

	R&D	Demonstration	Early commercial	Commercial
Heat			Gasification	Combustion
			Small scale gasification	Domestic/district/industrial
Power/CHP		Combustion		
		Stirling engine	ORC	Steam engine Steam cycle
	Gasification			Anaerobic digestion
	Gas turbine IGFC	Gas engine IGCC	Steam cycle	Landfill gas Manure digesters
		Cofire		
		Indirect	Parallel	Direct
Fuels		Fermentation		
		Lignocellulosic to ethanol		Sugar & starch to ethanol
	Esterification			
	Biodiesel from microalgae			Biodiesel
	Hydrogenation			
	Renewable diesel from microalgae		Renewable diesel	
		Anaerobic digestion		
		Biogas upgrading (methane) Biogas reforming (hydrogen)		Biogas
	Gasification			
		FT (diesel)		
		Catalytic synthesis:		
	SNG, hydrogen		Methanol	
Materials /chemicals	Fermentation			
	PA6 & PA66		PTT, PHA	PLA Ethylene (Butanol[a])
		PP, PVC	Ethylene derivatives:	PE, PET
	Lignin processing		Esterification	Polyol polymerization
	Aromatics		ECH	Lubricants, polymers, surfactants
	Gasification			Direct modification of natural polymers
	FT (olefins)		MTO (olefins)	Cellulosic polymers Starch plastics
Combined / Biorefinery products	Biorefinery – Integrated technologies			

FIGURE 6-2 | Development phase of biomass conversion technologies for energy and materials. adapted from Bauen et al. [6] and Chum et al. [2], with information from [14,32,41,51-53].

[a] fermentative production of butanol has been commercially applied in the past, but has ceased.
R&D, research and development; ORC, organic rankine cycle; AD, anaerobic digestion; IGFC, integrated gasification fuel cell; IGCC, integrated gasification combined cycle; FT, Fischer-Tropsch; SNG, substitute natural gas; PTT, polytrimethylene terephtalate; PHA, polyhydroxyalkanoates; PLA, polylactide; PP, polypropylene; PVC, polyvinylchloride; PE, polyethylene; PET, polyethylene terephtalate; ECH, epichlorohydrin; MTO, methanol-to-olefins

[41,54]. In the case of power or CHP production, biomass combustion is combined with a steam cycle. The produced steam is pressurized and expanded in a steam turbine. The steam turbine converts the thermal energy into rotary motion. At present, typical capacities are 5-100 kW_{th} for domestic heating, 0.5-5 MW_{th} for district heating, and 2.5-100 MW_e for steam cycle technologies [[8,54];[55] in:[27]]. Domestic and district heating (including both heat and small-scale CHP production) is mainly employed in Scandinavia and Austria. Steam cycles are widely applied in stand-alone power plants and in the pulp and paper industry, which derives process heat from waste incineration [56]. At the mentioned capacities, investment costs are 300-1,200 $/$kW_{th}$ for domestic heating, and 500-800 $/$kW_{th}$ for district heating. For power and CHP systems, the investment costs are 1,850-6,200 $/$kW_e$ (all for wood chips and pellets) [[54,57];[8,55] in:[27]]. The thermal efficiency for domestic and district heating is 79-88% [54,57], the electric efficiency for

power and CHP is 18-28% [[8];[55] in:[27]]. Combustion of MSW instead of wood pellets results in higher investment costs and a lower efficiency [54]. In addition, two types of direct combustion boiler systems can be distinguished on the basis of how the feedstock is fed into the boiler: fixed-bed (stoker) and fluidized-bed [41]. However, investment cost data for power generation does not significantly differ between these two systems [[58];[55] in:[27]].

Other power or CHP technologies based on dedicated biomass combustion are the Stirling engine and Organic Rankine Cycle (ORC). The ORC works analogous to the steam cycle, but uses an organic fluid instead of water. This fluid has a lower boiling point and allows for low temperature heat conversion to mechanical power [6]. The Stirling engine uses combustion heat to directly heat a gaseous working fluid in the engine. Both technologies are in the demonstration phase [6]. Because of the relatively small capacity of 50 kW_e-1.6 MW_e, these technologies show good potential for domestic or distributed cogeneration [6]. The investment costs, however, are currently very high at 5,800-9,800 $/$kW_e$ [[54];[8] in:[27]]. Also, current electric efficiencies are 9-16%, and improvement is needed [6,8,54].

Cofiring

Cofiring, or co-combustion, mostly involves combustion of biomass and pulverized coal [6]. Currently, direct cofiring is successfully applied in existing coal furnaces for power or CHP production. The scales of these plants range from 5-100 MW_e. The electrical efficiencies are relatively high, 26% for CHP and 36-41% for power, and investment costs are low at about 200-500 $/$kW_e$ [[54,56];[55] in:[27]]. The major bottleneck is the biomass cofiring ratio, which is limited to 5-10%. An important reason is that biomass ashes differ from coal ashes. Deposition of biomass ashes on surfaces in the boiler and in catalysts affects the efficiency of the plant [6]. Technologies that avoid this issue are parallel and indirect cofiring. Parallel cofiring involves biomass combustion in a separate boiler, while the produced steam is inserted into the main steam circuit of the coal plant. This technology is in the stage of early commercialization [6]. Through indirect cofiring, gasified biomass is cleaned-up to so-called syngas which is combusted with (pulverized or gasified) coal or natural gas [6,59], see the discussion about gasification below. Indirect cofiring systems are in the demonstration phase [6,41]. Parallel cofiring requires 2-4 times higher investment costs compared to direct cofiring, for indirect cofiring this factor is even higher [[7,8,54];[55] in:[27,58]].

Gasification

Gasification occurs when biomass is heated with a sub-stoichiometric amount of oxygen, resulting in partial oxidation of the biomass [2,6]. If the gasification is performed at 900-1000 ^{o}C a mixture of hydrogen (H_2), carbon monoxide (CO), carbon dioxide (CO_2), water vapor (H_2O), as well as methane and tars is obtained (product gas). If air is used as oxygen

source, large amounts of nitrogen (N_2) are also present. If gasification is performed at >1500 ^oC a gas mixture of mainly H_2, CO, and also CO_2 and H_2O is obtained (syngas). This is always performed with enriched or (almost) pure oxygen. Syngas can also be obtained by cleaning the product gas. Although both product gas and syngas can be used for heat and power production, product gas is preferred because of the higher energy efficiency of low temperature gasification compared to syngas production. In general, the preferred technology for power production is atmospheric circulating fluidized bed (CFB) gasification [59].

Most gasifiers use wood and other lignocellulosic biomass, but can also be designed to convert other biomass materials and municipal waste [59]. Commercial technologies burn the product gas directly in a boiler to (co-)produce steam. These systems can be applied for, for example, district heating [41]. Another option is to combust the product gas in an internal combustion engine (ICE), which is either a gas turbine or gas engine, to produce power or CHP. These plants are in the R&D and demonstration phase [6,41]. A technology based on this concept is the Integrated Gasification Combined Cycle (IGCC), in which a steam boiler and steam turbine use exhaust heat from the gas turbine to generate additional electricity [58,59]. The main advantage of power and CHP production through gasification, compared to dedicated biomass combustion, is the higher electric efficiency of 28-34% for gas engines and 40-45% for an IGCC [8,54,59]. However, the presence of contaminants in the product gas (e.g. tars, nitrogen, chlorine compounds) requires a certain degree of gas cleaning (depending on the type of feedstock and gasifier), which proves to be a technological challenge [59,60]. Cost estimations (e.g. in [8,54,61,62]) suggest that gasification can attain lower investment costs compared to dedicated biomass combustion. Yet, initial investments are high and make the commercialization of demonstration plants difficult [56,59]. Also, upscaling of low temperature CFB gasification has only limited effect on the investment costs (the amount of circulating bed material has to be increased). Because of these technological and economic hurdles, Kirkels and Verbong [59] expect that commercialization of the IGCC will be very difficult. Indirect cofiring is considered to be a more feasible option, because of lower requirements for gas cleaning and lower costs [59].

Anaerobic digestion

During anaerobic digestion, organic matter undergoes biological degradation in absence of oxygen or air. Biogas, the main product of anaerobic digestion, is a gas mixture of methane (CH_4) and carbon dioxide (CO_2) [6]. Biogas combustion technologies for power and CHP production are similar to syngas combustion [2,6]. Commercial biogas production takes place in biodigestors or through landfill gas recovery. Anaerobic digestion in biodigestor units is mainly related to agricultural activities and is employed on large scale in China and Germany. Recovery of landfill gas is successfully applied in mainly the UK,

Spain and Italy [6]. Investment costs for CHP production based on anaerobic digestion in biodigestors range from 2,500 $/kW$_e$ at 3.4 MW$_e$ to 5,100 $/kW$_e$ at 0.3 MW$_e$, which is lower than gasification at similar scales. But, electric efficiencies are lower as well (13-19%) [57]. A large variety of biomass materials, both wet and dry, can be applied to anaerobic digestion. Lignocellulosic biomass, however, is not suitable for anaerobic digestion because the component lignin is not biodegradable [2,6].

6.3.4.2 Fuels

Current commercially applied biofuel production plants are based on so-called first-generation technologies and biomass feedstocks. These include ethanol from the fermentation of sugar and starch crops, biodiesel from transesterification of vegetable oils, and biogas production through anaerobic digestion. Also the production of renewable diesel by hydrogenation of oils and fats is considered to be a first-generation technology. In many projections, including the WEO scenarios [63], these first-generation feedstock and conversion technologies will mainly represent the biofuel production in the next decade. In 2030, however, second-generation feedstocks and technologies are projected to achieve a share of 60% in the total biofuel production [52,63]. Second-generation technologies use lignocellulosic feedstocks, applying hydrolysis and fermentation to ethanol, gasification and processing to various biofuels or pyrolysis to pyrolysis oil and other fuels.

Fermentation (+hydrolysis)

The leading technologies in biochemical fuel production are the fermentation of monosaccharides (C6 sugars) and of the polysaccharide starch. These technologies are commercially applied, mainly to produce corn-based ethanol in the USA and sugarcane ethanol in Brazil. Ethanol is used as a gasoline substitute, and minor shares of ethanol can be blended with gasoline without the need for modifications to the vehicle [6]. At present, fermentation units have capacities up to 550 MW$_{feed}$ for corn and up to 1000 MW$_{feed}$ for sugarcane. Ethanol production from sugarcane, however, is often combined with sugar production and only about 50% of the feedstock is converted to ethanol [18,64]. During the fermentation process, microorganisms like yeast and bacteria metabolize sugars obtained from the feedstock [2]. C6 sugars can easily be extracted from sugar crops. Starch, however, first needs to be depolymerized through hydrolysis, an enzymatic conversion process [2,65]. For corn ethanol production, a distinction can be made between dry and wet milling processes. Most corn ethanol plants apply dry milling, in which the corn grain is milled mechanically, and the complete milling product is fed into the hydrolysis and fermentation process. In wet milling, the corn is soaked in a mixture of water and SO_2, which allows for separation of the kernel components. Hydrolysis and fermentation are then only applied to the starch stream [64]. Currently, the investment

costs for large-scale conversion are about 160 $/kW$_{feed}$ for corn (dry milling) and 100 $/kW$_{feed}$ for sugarcane. Costs and efficiencies can be improved through the co-production of electricity from feedstock processing residues.

An alternative option to biochemically produce ethanol is the fermentation of lignocellulosic feedstock. This process is more complex compared to conversion of sugar and starch feedstocks. Currently, the production of lignocellulosic ethanol is explored and demonstrated at pilot, demonstration and commercial scales [66]. First, a pretreatment process (e.g. hydrolysis or steam explosion) separates the biomass into cellulose, hemicellulose and lignin. This is followed by acid or enzymatic hydrolysis of the cellulose, which consists of long chains of C6 sugars. Hemicellulose contains a mix of C5 and C6 sugars and is easier to breakdown than cellulose. Finally, the C5 and C6 sugars are fermented. Lignin is not a carbohydrate and cannot be hydrolysed and fermented. Yet, it can be combusted for power co-generation [56,65]. The major bottleneck is the pre-treatment of the feedstock, which is relatively expensive and inefficient. To make the production process more efficient and cost-effective, existing processes are improved and novel processes are developed [6,56,66,67]. Also, options for process integration are investigated and developed [6,66,67]. The most mature configuration, which is now in the demonstration phase, is called Separate Hydrolysis and Fermentation (SHF). In SHF, all hydrolysis and fermentation steps take place in separate reactors. Simultaneous Saccharification and Fermentation (SSF) combines hydrolysis with the fermentation of C6 sugars. In Simultaneous Saccharification and Co-Fermentation (SSCF), hydrolysis and co-fermentation of C5 and C6 sugars take place simultaneously in one reactor. Consolidated Bioprocessing (CBP) combines all processes, including enzyme production, in a single vessel. It is considered to be the endpoint in the evolution of biomass conversion technology. Due to the level of development, SHF, SSF and SSCF may commercialized in the short (<5 yr) or medium term (5-15 yr). CBP will only be attainable in the longer term [6,67]. Investment costs are estimated to be 500-700 $/kW$_{feed}$ for near-term commercial plants (SHF, SSF and SSCF) [7,28,60] and 240-510 $/kW$_{feed}$ in 2030 [54,60]. Much effort is also made in the development of biorefineries, which co-produce other energy and material products next to ethanol (see section 3.4.4 for a discussion on biorefineries) [66].

Transesterification

Transesterifcation is the major commercial chemical process to produce biofuels from vegetable oils and animal fats. The vegetable oils contain triglycerides. During transesterification, the triglycerides react with alcohols (often methanol) to form an alkyl ester of fatty acids. These esters are referred to as biodiesel. Glycerine is formed as byproduct [2,6]. Present biodiesel plants have maximum production capacities of more than 400 MW$_{feed}$. At these capacities, investments costs of 160 $/kW$_{feed}$ can be attained

[7,27]. Biodiesel can be blended with fossil-based diesel at a blending rate of at least 20% biodiesel without modifications to the vehicle's engine or fuel system [6]. As an alternative to vegetable oils, algal oil is considered to be a potential (third-generation) feedstock in the future [28].

Hydrogenation

An alternative to transesterification, hydrogenation (or hydrotreating) is a chemical process that produces renewable diesel. The process involves the reaction of vegetable oils and animal fats with hydrogen [6]. An alternative feedstock is bio-oil derived from pyrolysis or hydrothermal liquefaction, which enables the application of second- or third-generation biomass feedstocks as well. Hydrogenation is applied at commercial scale, but development and deployment are at an earlier stage than transesterification [68]. The production costs are potentially lower compared to transesterification. Bain [7] reports investment costs of less than 70 $\$/kW_{feed}$ at a production capacity of about 120 MW_{feed}. Renewable diesel is chemically similar to fossil-based diesel, making blending possible in any proportion [6].

Anaerobic digestion

Biogas from anaerobic digestion can be upgraded to methane or reformed to hydrogen. Biomethane (or substitute natural gas, SNG) complies with natural gas standards and can be injected into the natural gas grid and/or used as transport fuel [6]. Several technologies for biogas upgrading are commercially available (e.g. pressure swing absorption and water scrubbing), while other technologies like cryogenic upgrading are at earlier stages of development [2,6,69]. Most upgrading plants are located in Europe, where capacities mainly range between 100 and 1000 Nm^3 biogas/h (0.6-5.6 MW_{biogas}). A number of large-scale upgrading plants with capacities up to 13,000 Nm^3 biogas/h (72 MW_{biogas}) are located in the USA [69]. Persson [70] reports investment costs of about 600 $\$/kW_{biogas}$ at a capacity of 200 Nm^3 biogas/h and of approximately 420 $\$/kW_{biogas}$ at 600 Nm^3 biogas/h.

Gasification

After high temperature gasification of (lignocellulosic) biomass or bio-oil, a synthesis process can be applied to convert H_2 and CO from the syngas into biofuels. Firstly, a widely considered technology to produce fuels like diesel, gasoline and kerosene (jet fuel) is Fischer-Tropsch (FT) synthesis [71]. The production of FT diesel, ethanol and also other fuels is demonstrated at pilot to commercial scales [72]. Secondly, a range of fuels can be produced by catalysed synthesis, including methane (synthetic natural gas: bio-SNG), methanol, dimethyl ether (DME), ethanol and butanol. A third route is water gas shift and separation to hydrogen [2].

Compared to product gas combustion for heat and power, effective gas cleaning is even more important for the synthesis of biofuels. The issue that current gas cleaning technology is insufficient hinders the commercialization of these conversion processes [59,60,72]. Kirkels and Verbong [59] believe that once clean syngas is available, existing technologies (for fuel production after coal gasification) can be applied to produce biofuels. The investments costs of high temperature gasification are higher compared to atmospheric CFB gasification. Studies by EPA [28], Bain [7] and Hamelinck et al. [60] find short-term investment costs from 500 $\$/kW_{feed}$ for ethanol to almost 700 $\$/kW_{feed}$ for FT diesel, at production scales of about 400 MW_{feed}. To make gasification economically feasible, Kirkels and Verbong [59] state that production scales of more than 1000 MW_{syngas} are required.

6.3.4.3 Materials

Biomass serves as raw material for a large variety of non-energy products [73]. We distinguish between two product groups. The first covers traditional wood, paper and textile products (e.g. cotton). This also includes the use of wood as substitute for steel and concrete in construction, which is considered to be an important option to reduce the use of fossil energy and non-biomass materials [74,75]. The second category covers synthetic chemicals produced from biomass feedstock. Some materials have been produced from biomass for decades, e.g. alkyd resins from vegetable oils [76]. Bio-based materials that may replace current petrochemical products or that allow for new applications and markets can be considered as a group of novel biomaterials. In this study, we focus on these new biomaterials. We discuss the production routes for these novel biomaterials in this section.

Today, synthetic organic and inorganic chemicals (e.g. plastics, fibres and nitrogen fertilizers) are produced from a limited number of platforms chemicals (mainly olefins and aromatics, but also ammonia, methanol, carbon black, oxygen and chlorine), see Table 6-1 [4,77,78]. The largest share of these platform chemicals is produced from hydrocarbon feedstocks (primarily crude oil and natural gas) [79]: in the organic chemical industry this share is 90%, the remaining 10% is derived from vegetable oils and biomass [4]. Olefins and aromatics are the main resource for plastics and fibers. In 2007, plastics and fibers represented about 75% and 13%, respectively, of the product mix in the organic chemical industry [4]. Petrochemical ammonia is the key source for the production of synthetic nitrogen fertilizers [78]. In 2009, about 33% of methanol was converted into formaldehyde, which is used for resins but also for other products. Other main uses of methanol are the production of gasoline additives like MTBE (methyl tertiary butyl ether) or as fuel [80]. In many cases, the fossil-based materials can be substituted by a chemically identical bio-based material (Table 6-1). In addition, it is possible to convert biomass into

chemicals and materials that have unique structures and properties [77]. Depending on their material properties, these can be considered as a (partial) substitute for current fossil-based chemicals or they can allow for new applications and new markets [32,73]. In literature, dozens of materials that can be produced from biomass are investigated [15,32,73,81], see Figure 6-3 for some key production routes and examples of products (partly already produced from biomass, partly in development). Many materials, however, are high value added, but low volume materials (e.g. certain engineering plastics). In this section, we will focus on high volume materials like bulk plastics.

TABLE 6-1 | Main platform chemicals derived from fossil resources, and their derivatives [32,78,79,82-84].

Platform chemical from fossil feedstock	Global production 2009 (Mt/yr)	Important derivatives [a]	Global production 2009 (Mt/yr)	Chemically identical bio-based counterpart?
Olefins				
Ethylene	112.6	PE	60.0	Yes
		PVC	37.5	Yes[d]
		Ethanol	2.6 (1998)[b]	Yes
Propylene	53.0	PP	27.6	Yes
		Epoxy resins	1.2 (2007)[c]	Yes[d]
		Butanol	2.8 (2004)	Yes
Butylene	20.3 (2004)	MTBE	30.2	Yes[d]
Aromatics				
Benzene	49.2	PS	18.0	No
		PA6/PA66	6.6 (2007)[c]	Yes
		phenol	8.3	
Toluene	19.6	Fuel additives	n/a	Yes
		PUR	12.3 (2007)[c]	No[d]
Xylenes	35.6 (2004)	PET/PBT	12.4	Yes[d]
Other				
Ammonia	152.4 (Mt NH_3)	Urea	147.5	
		Nitric acid	52.3	Yes
Methanol	40.6	Formaldehyde	28.7	Yes
		MTBE	30.2	Yes[d]
		Acetic acid	5.2	Yes
Chlorine	61.2	Epoxy resins	1.2 (2007)[c]	Yes[d]
		PVC	37.5	Yes[d]

PE: polyethylene; PVC: polyvinylchloride; PUR: polyurethanes; PP: polypropylene; PS: polystyrene; PA: polyamide; PET: polyethylene terephthalate; PBT: polybutylene terephthalate; n/a: not available

[a] Derivatives can be mentioned more than once, because the production of these chemicals often involves the use of different platform chemicals; e.g. PVC (ethylene and chorine), epoxy resins (chlorine and propylene), and MTBE (methanol and butane)

[b] Installed capacity in 1998/1999 [79], fossil synthetic ethanol only; production of bioethanol production for fuel use not included

[c] Global consumption in 2007 [32]

[d] The bio-based counterparts are partially bio-based chemicals

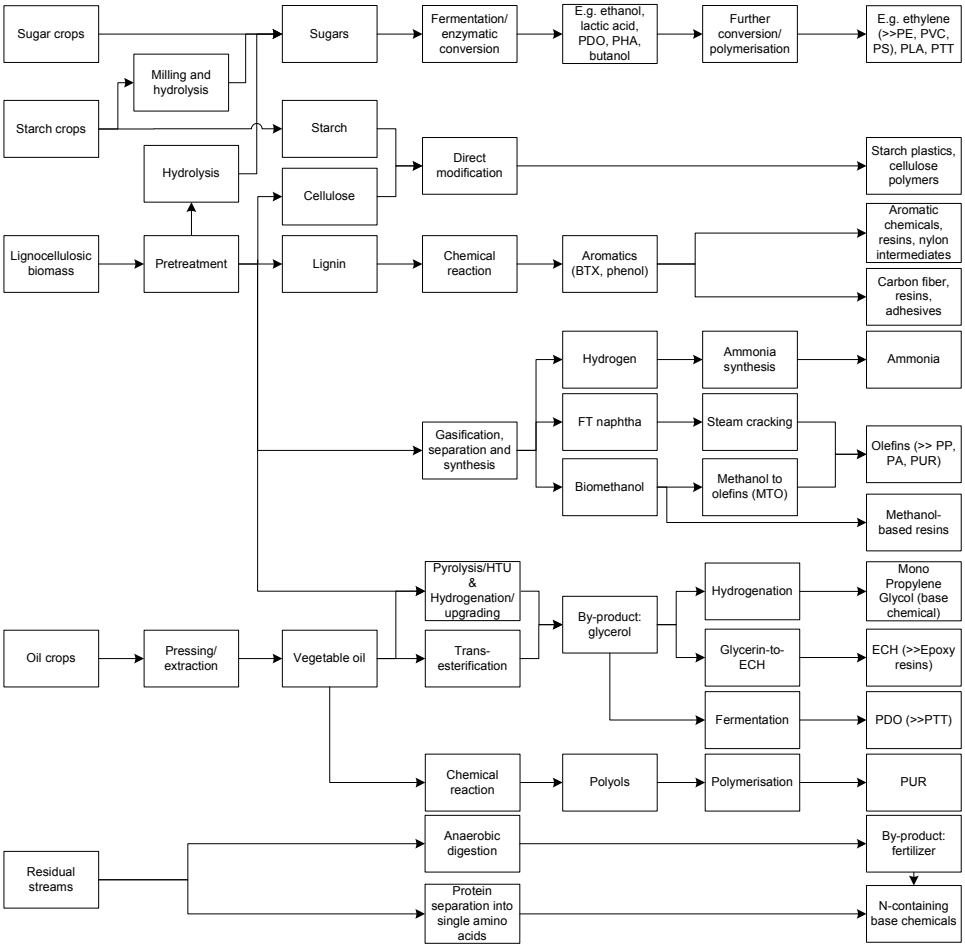

FIGURE 6-3 | Possible routes to use different biomass feedstock types to produce materials and chemicals that can substitute current fossil-based chemicals [73,85-88].

PE, polyethylene; PUR, polyurethanes; PP, polypropylene; PS, polystyrene; PA, polyamide; PDO, 1,3-propanediol; BTX, benzene, toluene, and xylenes; PHA, polyhydroxyalkanoates; PTT, polytrimethylene terephtalate; PLA, polylactide; PVC, polyvinylchloride; ECH, epichlorohydrin; MTO, methanol-to-olefins

These will have most impact on biomass demand, and the largest potential to reduce fossil fuel use and GHG emissions. In addition, many biomaterials and their production processes are too innovative to properly discuss their status and prospects, and to give economic data. Therefore, we limit our discussion to the following conversion technologies: direct modification of natural polymers, fermentation, transesterification, polymerization of natural oils, gasification, pyrolysis, and catalytic conversion.

Direct modification of natural polymers

Two important natural polymers present in biomass are starch and cellulose. Natural cellulose is extracted or chemically modified to produce cellulosic polymers [89]. Cellulosic fibres like viscose, modal and tencel are widely produced. In 2008, the global production amounted to 3.0 Mt [90]. Cellulose films like cellophane dominated the packaging field in the past, but lost their market dominance in the 1950s [89]. The production capacity was 36 kt in 2010, and is not expected to increase by 2015 [91,92].

Native starch has some disadvantages like brittleness and poor thermal processability [89]. However, these can be (partially) overcome by thermal, chemical or mechanical processing of the starch. A variety of products can be derived from these processes which are referred to as starch plastics [89]. The production capacity of starch plastics was 25 kt in 2003 and grew to 155 kt in 2007. As a result, starch plastics are one of the most important bio-based plastics in the present market [32,91]. The projected growth in global production capacity from 2010 to 2015, however, is limited to 6% [91,92]. Starch plastics can partially replace fossil-based plastics like poly-ethylene (PE). Yet, based on their technical properties, the substitution potential of starch plastics is estimated to be limited to 5% of the global plastic consumption in 2007 [32].

Fermentation

Through fermentation of biomass, not only ethanol but also other chemicals can be produced[2]. Some are suitable for direct end use application, others serve as building blocks for materials [73,89,93]. Prominent examples that can play an important role in replacing the main fossil-based chemicals identified in Table 6-1 are:

> *Ethanol*: In addition to the use of bio-based ethanol as fuel, this chemical can also be used for the production of ethylene and other platform chemicals [94].
>
> *Ethylene*: Ethylene is a building block for a variety of bulk materials (Table 6-1) [32]. It is produced by catalytic dehydration of ethanol (called ethanol-to-ethylene or ETE) [87]. The production of bio-based ethylene has been commercial in

2 The application of fermentation and enzymatic processes is also called white or industrial biotechnology (in Europe and USA resp.) [15]

the past, ceased in the early 1990s, but is receiving renewed attention. As the conversion can be carried out via an established technology, bio-ethylene production is already economically competitive in Brazil, where sugarcane prices are low and experience in ethanol production is extensive [95]. Also, biobased PE is already produced on a commercial scale (200 kt in 2010), and applied for packaging [76,91]. The production of other derivatives like PVC is in the stage of demonstration and early commercialization.

Propylene: Propylene is an important platform chemical for materials like polypropylene (PP). Compared to bio-based ethylene, the production of bio-based propylene is technologically more challenging [96]. One production route is metathesis, which is a reaction of ethylene with 2-butylene to form propylene [96]. This is a commercial technology for fossil-based propylene [97]. The Brazilian company Braskem expects to bring an ethylene-based bio-polypropylene plant online in 2013 [98]. The plant will have a production capacity of at least 30 kt/yr [98]. An alternative route may be the production of 2-propanol via fermentation, followed by dehydration (comparable to ETE) [96]. This route is neither very advanced, nor widely explored [77].

Butanol: Butanol is mainly used as solvent and thinner or in plasticizers [73,93]. The fermentative production of butanol was an established process in the first half of the 20th century, but was abandoned because the production from fossil resources was cheaper [93]. In 2007, Haveren et al. [77] expected that the production could become economically viable in 5-10 years.

Polylactide (PLA): PLA is formed through polymerization of lactic acid, a direct fermentation product. It can partially substitute fossil-based plastics like PE and PET. PLA is commercially available and is considered to be an important plastic in the future biobased plastic sector [32,92]. European Bioplastics [91,92] projects the production capacity to grow from 113 kt in 2010 to 216 kt in 2015.

Polyhydroxyalkanoates (PHA): PHA is a direct product of fermentation, and is a potential substitute for various fossil-based plastics [32]. With an installed capacity of 70-88 kt/yr (2010) the production is still in early commercial stages. Until recently, PHA was expected to be one of the main bioplastics in the next decade [32,92]. This has, however, become uncertain since the joint venture Telles LLC was ended in January 2012 [99,100].

Polytrimethylene terephtalate (PTT): PTT is produced by polycondensation of 1,3-propanediol (PDO – a product of biomass fermentation) and purified terephthalic acid (a petrochemical product). The production process of PTT is commercial. The material properties of PTT are similar to PET but have some advantages for certain applications. Currently, PTT is mainly applied in fibres for carpets and textiles [76].

Next to these products, fermentation yields by-products that can be of value for other purposes. Crop residues and by-products from crop processing (e.g. dried distillers grain soluble (DDGS) from corn ethanol production) are rich of proteins. The amino acids that form these proteins are only found in biological sources, and are a potential source for nitrogen containing chemicals like polyamides and urea, which is used for fertilizers [85,101,102]. Besides, the biological wastes could be utilized in anaerobic digestion; the digestate can be applied as fertilizer. Finally, lignocellulosic biomass contains lignin. Lignin cannot be fermented and is primarily used for power generation. However, as the building block molecules of lignin are of aromatic nature, lignin is considered as source for high volume production of bio-based aromatics and aromatic-based chemicals [73,77].

Transesterification

The transesterification of (vegetable) oils to biodiesel produces an important by-product: glycerol. Glycerol is considered to be a base chemical for other chemicals and materials. First, epoxy resins (thermosets) can be derived from epichlorohydrin (ECH), which is made via a glycerin-to-ECH process [32]. Besides, ethylene glycol and propylene glycol can be derived from glycerol and converted to alkyd resins or the olefins ethylene and propylene [77]. Another route is the fermentative process to PDO, which can be used as building block for polymers like PTT [93]. Different companies have announced to start up ECH plants in 2012 with a combined capacity of 300 kt refined glycerin [103].

Polymerization of natural oils

Natural oils that contain two or more hydroxyl (-OH) groups are called natural oil polyols (NOPs) [76]. Castor oil is one of the few vegetable oils that contain these hydroxyl groups by nature. Other oils, like soy, rapeseed and sunflower oil, can be converted to polyols by a chemical reaction [76]. Currently, the production and application of NOPs take place on a commercial scale [76]. Also, the production of alkyd resins from polyols is an established process. A more recent application of polyols is the production of partially bio-based polyurethanes (PUR) [76,104].

Gasification

In the petrochemical industry, CO and H_2 from syngas have a high economic significance as synthetic component for the manufacture of important intermediates like ammonia, methanol and acetic acid [79]. The interest in gasification for biochemicals focuses on the production of olefins and methanol [105,106]. We distinguish between two routes. The first option is FT synthesis, which not only produces liquid fuels but also FT naphtha (up to 30%). FT naphtha can be converted to olefins by steam cracking, which is applied at large scale in the petrochemical sector [107]. The second route is catalysed synthesis of the syngas to produce methanol. The methanol can be converted to olefins [107]. Currently,

thermochemical production of bio-based methanol from glycerin is commercially applied [106]. Recently, the first industrial methanol-to-olefins (MTO) production units (where methanol is produced from natural gas or coal) were put into operation. Activities aiming at commercialization of thermochemical biomass conversion are not linked yet to the production of olefins [95].

Pyrolysis

As we discussed in section 6.3.3, pyrolysis not only produces bio-oil but also biochar. The biochar contains a large amount of nutrients and is considered to be an interesting resource for fertilizer production [50].

Catalytic conversion

Many chemical reactions are catalytic conversions, i.e. a catalyst is used –but not consumed– to augment the reaction. Examples of catalytic conversions mentioned earlier in this section are the conversion of ethanol to ethylene, depolymerization of lignin (in R&D), and methanol production from syngas. The number of catalytic reactions to produce chemicals from biomass and bio-based components is large, see for example [94,108]. One interesting option is the production of furans from cellulose and hemicellulose fractions of biomass. Furans (e.g. furfural) are considered as aromatic building blocks for polymers and as precursors for hydrocarbon fuels like diesel and jet fuel [76,108].

6.3.4.4 Biorefinery

Biorefinery is defined as "the sustainable processing of biomass into a spectrum of marketable products (food, feed, materials, chemicals) and energy (fuels, power, heat)" [109]. The biorefining facility integrates different conversion technologies to maximize both the use of all biomass components and the displacement of fossil resources. Also, the combined production of high volume, low value energy products and lower volume, high value chemicals improves the cost effectiveness of biomass processing [73,110,111]. Elements of the biorefining concept are already used in many of today's biomass converting industries. The co-production of ethanol and animal feed from sugar and starch crops, or the co-production of biodiesel, glycerol and animal feed from oil crops are well-established [109]. IEA Bioenergy task 42 on biorefineries [109] expects that a variety of biorefineries will be introduced in the short term by valorizing side products in existing biomass conversion plants. New, advanced biorefinery concepts that are now in the R&D, pilot or small-scale demonstration phase, are expected to be commercialized in the medium term. Wijffels and Barbosa [35] expect that commercial production of algal biofuels and protein-based co-products (chemicals, food, feed ingredients) will be economically feasible in 10-15 years. Given the variety of conversion technologies, and the dependence on feedstock and location (determining the circumstances including

prices), process configurations of biorefineries are expected to be less uniform compared to petroleum refineries.

Recently, several studies have been performed to assess the economic and environmental performance of different biorefinery concepts. The BIOREF-INTEG project [112] compares the calculated cost of the main product of 8 reference cases in six biomass processing sectors (e.g. food, feed, biofuels) to 12 related biorefinery configurations. They find that for 9 out of 12 biorefinery concepts, the main product cost declines compared to the reference cost level. For example, a high cost reduction of 41% is found for bioethanol production when combined with the production of lactic acid. In general, biorefinery projects that have best prospects to improve the economics of the reference cases are found to be projects that 1) need no or little changes to the reference process, 2) aim at a better valorization of co-products (e.g. ECH production from glycerol), and/or 3) apply a fermentation process [112]. Laser et al. [113] compare the efficiency and environmental and economic performance of fourteen mature biorefinery configurations that include biochemical and thermochemical processes to produce fuels, power and/or animal feed protein from lignocellulosic biomass. They find that scenarios which integrate bio- and thermochemical processing (i.e. the feedstock is first biochemically converted to ethanol, and the lignin-rich residue is converted thermochemically) can achieve overall process efficiencies of 70-80%, compared to 61-73% for biochemical biofuel production combined with power and/or protein production and 55-64% for thermochemical biofuel production combined with power generation [113]. With regard to process economics, configurations that involve biochemical production of ethanol are found to be most profitable (i.e. lowest levelized production costs, highest internal rate of return, and lowest minimum selling price compared to other configurations). While protein coproduction configurations especially have a good economic potential at crude oil prices below 50 $/bbl, biorefineries that integrate bio- and thermochemical processing become most profitable at higher oil prices [113]. Also, when a large share of the grid power is generated from coal, configurations that include power co-generation avoid most GHG emissions. When a future electricity system becomes less dependent on fossil fuels, however, the amount of avoided GHG emissions is largest for biorefineries that integrate bio- and thermochemical processing [113].

6.4 COST AND EMISSION DATA

In this section, we present the cost and GHG emission data gathered for biomass production, feedstock pretreatment, transportation and conversion, based on review of state-of-the-art-literature. For the purpose of completeness, we complement the cost data of conversion technologies for bio-based heat, power and fuel as compiled by the IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation [27] with cost and GHG emission data from other sources for bioenergy conversion technologies and fossil reference technologies, for bio-based materials and their fossil reference products, and for feedstock supply (production, transport and pretreatment).

6.4.1 Feedstock yields

Typical yields and projected yield improvements are summarized in Table 6-2 (sugar, starch and oil crops) and Table 6-3 (lignocellulosic biomass and MSW). Large differences in yields exist between crop types. For example, sugarcane production attains significantly higher yields compared to other conventional crops [114]. Among oil crops, oil palm attains the highest yields. With regard to algal oil, the potential yield is reported to be 40-50 thousand liter/(ha.yr), resulting in a neutral lipid yield that is several times higher than the palm oil yield [2,115]. For each crop type, yields vary widely according to local climatic conditions, land suitability, and management levels (e.g. irrigation, fertilizer and pesticide use) [116]. Conventional cropping systems in Western Europe and other developed countries are already highly-input intensive and have attained considerable yield growth in the past 50 years [117]. In these regions, future productivity increases will be limited. Through the adaptation of advanced management practices, considerable gains in productivity could also be attained in other regions, particularly Sub-Saharan Africa, Latin America, Eastern Europe and Central Asia. Under the condition that current yield trends are maintained, Jaggard et al. [118] expect that global yields of agricultural crops will increase by a factor 0.9-2.4 between 2007 and 2050. This is equal to yield improvement rates of -0.2-2.1 %/yr depending on the crop type and cultivation region. Average yield growth rates for crops considered in Table 6-2 range between 0.5 %/yr for sugarcane and 1.3 %/yr for sugar beet. The FAO expects an average agricultural production growth of 1.5 %/yr for the next three decades [119]. Kindred et al. [120] project that yield growth rates can be increased to about 2%/yr under the most favorable conditions (e.g. high crop prices and high investments in agricultural research). Current yields of short rotation crops and perennial grasses are often similar or higher compared to yields of conventional crops, especially of starch and oil crops. In addition, yield projections show that lignocellulosic crops have potential for major yield improvements, both in relative and absolute terms [39,116]. In Europe, Ericsson et al. [121] expect improvement rates from 1.5%/yr for poplar and eucalyptus, up to 3.2%/yr for miscanthus between 2005 and 2020.

6.4.2 Feedstock production costs

The production or roadside costs (Table 6-2 and Table 6-3) refer to total costs to make the feedstock available at the roadside for transportation and storage. Roadside costs depend on yields, the costs of land and labour, prices of inputs, and on the management system (e.g. use of machinery) [121]. For oil crops that have a relatively low yield, i.e. rapeseed, sunflower and soy, the roadside costs are 5.3-16 $/GJ$_{oil}$. This is rather high compared to 3-6 $/GJ$_{oil}$ for oil palm and 1.6-4.4 $/GJ$_{oil}$ for Jatropha, which can attain higher yields [2,122,123]. Low roadside costs are also found for high yield sugarcane (2.5-7.5 $/GJ) and lignocellulosic crops (0.7-8.7 $/GJ) [18,121-127]. In the roadside costs of agricultural and forestry residues, only collection and field transport are included. There are no costs related to production, because these residues are considered a residual stream from crop or wood production. As a result, roadside costs in Europe can be as low as 1 $/GJ [123,128].

Future roadside cost projections depend on assumed improvements in the production system (e.g. improved fertilization) and expected market developments (e.g. growing demand for biomass resources may result in higher land prices). Cost projections by Ericsson et al. [121] suggest that the potential for cost reductions in Europe is smaller for starch and sugar crops than for SRC and perennials. Also, cost-supply curves by De Wit and Faaij [123] project that the major share of the European production of oil, starch and sugar crops in 2030 can be accomplished at a cost of 5-12 $/GJ. Cost projections for SRC and perennials in the same year and region range between 2 and 7 $/GJ [123]. With regard to potential future algae lipid production, medium-term costs are estimated to be 30-80 $/GJ for open pond reactors and 50-140 $/GJ for PBRs [28].

TABLE 6-2 | Sugar, starch and oil crops: feedstock yields, projected yield improvement, current and projected roadside costs.

	Feedstock	Region	Present average yields (GJ/(ha.yr))[a]	Projected yield improvement (%/yr)	Roadside costs ($/GJ feed)[b]		References
					Current	2030	
Starch	Mixed	Europe			6.6-11.3	6-10	[123]
	Wheat	Europe	31-160	1.3			[114,118,121]
		USA	41-47	1.0			[114,118]
		Canada	36-45	1.0			[114,118]
		Argentina	27-44	1.1			[114,118]
	Corn	Europe	35-147	1.1			[114,118]
		USA	140-156	1.0	6.7		[19,114,118]
		Canada	126-136	0.8			[114,118]
		Argentina	84-115	1.4			[114,118]
		China	78-84	1.4			[114,118]
	Triticale (whole crop)	Europe	144-216	0.3			[121]
	Cassava	China	97-103				[114]
		Africa	10-74				[114]
		Africa, semi-arid	14-56		10 - >50		[125]
		Africa, arid	1-28		15 - >50		[125]
Sugar crops	Sugarcane	Brazil	355-415	1.0	2.5		[18,114,118,129]
		USA	375-425	0.1			[114,118]
		Mexico	365-405				[114]
		C. America	390-410				[114]
		Colombia	490-530				[114]
		Argentina	440-445	0.8			[114,118]
		India	340-360	0.9			[114,118]
		Australia	420-465	0.4			[114,118]
		Mozambique	350-522		7.5		[122]
	Sugar beet	Europe	82-232	1.0	5.3-10	5.3-9.3	[118,123,130]
	Sorghum (whole crop)	Europe	23-180	0.3-1.1			[114,118,121]

TABLE 6-2 | Continued

	Feedstock	Region	Present average yields (GJ/(ha.yr))[a]	Projected yield improvement (%/yr)	Roadside costs ($/GJ feed)[b]		References
					Current	2030	
Oil crops	Rape seed	Europe	30-70	1.0	7.2-16		[2,114,118]
	Sunflower	Europe	20-44	0.6			[114,118]
	Rape/sunflower	Europe			5.3-13.3	8-12	[123]
	Soy	Brazil	18-21	1.2			[2,118]
		Argentina	19-30	1.2	22.7		[118,122]
		North America	16-19	0.6-0.9	11.7		[2,118]
	Palm	Brazil	170				[2]
		Colombia	165				[122]
		Asia (Indonesia/Malaysia)	110-180		3-6		[39,122,131]
		C./W. Africa	27				[39]
	Jatropha	World	17-88		3.2		[2]
		Tanzania	17-46		1.6-4.4		[122]
		Africa, semi-arid	36-48		26- >40		[125]
		Africa, arid	3-37		29- >50		[125]

Roadside costs: total costs to make the feedstock available at the roadside for transportation and storage.
Conversion factors used to convert yield data to GJ/(ha.yr) can be found in the supplementary material.
[a] For oil crops: GJ oil/(ha.yr)
[b] for oil crops: $/GJ oil

TABLE 6-3 | Lignocellulosic biomass and MSW: feedstock yields, projected yield improvement, current and projected roadside costs.

	Feedstock	Region	Average yields (GJ/(ha.yr))		Projected yield improvement (%/yr)	Roadside costs ($/GJ feed)		References
			Present	Projected		Current	Projected	
SRC	Mixed	World					<2.3-4.5 (2050)	[132]
		Europe		355-710 (2050)		2.7-6.2	2-5.3 (2030)	[116,121,123]
		USA					<2.5-3.1 (2030)	
		Africa, semi-arid	110-250	315-475 (2050)[a]		1.9 – 4.2		[116,125]
		Africa, arid	14-175			2.2 – 23.8		[125]
		Northern America		375-535 (2050)				[116]
		Latin America		315-455 (2050)				[116]
	Willow	World	190	290 (2030)				[39]
		USA	210-230			2.5		[127]
		Europe	90-250		2.3	1.8-4.7		[121,124,130,133]
	Poplar	World	180	265 (2030)				[39]
		Europe	180-375		1.5	1.8-4.7		[121,124,133]
		USA	110	385-545 (2050)				[116,134]
	Eucalyptus	USA	245					[116,134]
		Latin America	340-435			1.3		[43,124,133]
		Europe	230		1.5			[121]
		Mozambique	136-485			0.7-1.4		[126]
Perennial grasses	Mixed	Europe				4-8.7	3.3-8 (2030)	[121,123]
		USA					<2.1-3.3 (2030)	[135]
	Miscanthus	World	190	380 (2030)				[39]
		Europe	17-340		3.2			[121,130]
	Switchgrass	World	215	290 (2030)				[39]
		S. Europe	235		2.3			[121]
		USA	235					[116,134]
		Argentina	90-180					[116,136]
	Reed Canary Grass	Europe	140		2.3			[121]

TABLE 6-3 | Continued

	Feedstock	Region	Average yields (GJ/(ha.yr))		Projected yield improvement (%/yr)	Roadside costs ($/GJ feed)		References
			Present	Projected		Current	Projected	
Fuel wood	Forest wood	World	6					[116]
	Forest wood+res.	World	12-69		0.8-1.3			[137]
				14-86 (2030)				
				16-106 (2050)				
	Forest residues	Europe	2-22			0.7-5.3	1-4 (2030)	[2,123,124,128]
		USA				<2-3		[135]
Residues	Agric. residues	Europe	5-125			1-5.3		[123,128]
		USA				<2.2-2.8		[135]
		World	7-31	9-35 (2030)				[137]
				11-38 (2050)				
	Straw	USA	65			<2.2-3.3		[134,135]
	Wheat straw	Europe	60					[2]
		USA	7-75			<2.2-3.3		[2,135]
	Sugarcane straw	Brazil	90-126					[2]
	Corn stover	N. America	15-155			<2.2-3.3		[2,135]
		India	22-30					[2]
	Sorghum stover	World	85					[2]

Roadside costs: total costs to make the feedstock available at the roadside for transportation and storage.
Conversion factors used to convert yield data to GJ/(ha.yr) can be found in the supplementary material.
[a] projection for Sub-Saharan Africa.

6.4.3 Pre-treatment and transportation costs

The costs of delivering biomass to the conversion plant are the total costs of feedstock production, pretreatment and transportation. The delivering costs are thus determined by spatial distribution of biomass resources, transport distance, mode of transport and the type of biomass pretreatment. Below, we discuss the costs of various pretreatment and transportation options.

6.4.3.1 Costs of pretreatment

For pelletization of wood residues like logs and sawdust, Uslu et al. [43] and Sikkema et al. [44] give costs between 1.7 and 5.4 $/GJ (excluding feedstock costs). Uslu et al. [43] find a pelletization cost of 1.3 $/GJ for energy crops. The pellet production costs are mainly defined by feedstock drying, pressing and cooling [44]. For torrefaction of wood chips, Uslu et al. [43] estimate the costs to be 3.9 $/GJ, and for the TOP process 3.0 $/GJ. Batidzirai et al. [48] find TOP costs of 4.1 $/GJ for small-scale, and 2.8 $/GJ for large-scale production. For pyrolysis, Uslu et al. estimate the costs to be 7.3-14.6 $/GJ for wood chips in Europe and 1.8-4.5 $/GJ for energy crops in Latin America [43].

6.4.3.2 Costs of logistics

Regional or national truck transport of raw biomass costs 0.3-1.6 $/GJ for sawdust and shavings [44], and 2.4 $/GJ for corn stover in the USA [28]. In Europe, national transportation costs for pellets range from 0.3 $/GJ delivered for inland river shipping to 1.2 $/GJ for small volume truck transport. Train and large volume truck transport costs about 0.8 $/GJ [37,44]. Costs for international transport of pellets range from 1.1 $/GJ for truck transport to 1.7 $/GJ for short sea shipping. Intercontinental ocean shipping of pellets from Canada to Western Europe (16,500 km) costs 1.8-4.8 $/GJ [37,44], and from Latin America to Western Europe (11,500 km) about 1.0 $/GJ [43]. For torrefied pellets and pyrolysis oil, Uslu et al. [43] estimate the costs of ocean shipping to be 1.6 $/GJ and 1.8 $/GJ respectively. They also show that ocean shipping is more expensive for torrefied biomass (55 $/tonne dry) compared to torrefied pellets (37 $/tonne dry) [43].

6.4.4 Biomass conversion costs

Below, we discuss the cost data found for biomass conversion to bioenergy and biomaterials. Important factors that determine the biomass conversion costs are: scale of conversion, load factors, and the production volume and value of co-products.

6.4.4.1 Heat

We give an overview of present and projected cost data for domestic and district heat production from biomass and fossil resources in Table 6-4. Comparison of bioenergy data

TABLE 6-4 | Present and projected cost data for heat production from biomass and fossil resources[a].

		Typical size of device (MW_{th})			Investment cost ($/ kW_{th})			O&M fixed annual ($/($kW_{th}$.yr))			Feedstock conversion efficiency heat (%)			Capacity factor (%)	Economic design lifetime (years)			References
Biomass present																		
Domestic	boiler	0.005	-	0.1	310	-	1200	13	-	43	80	-	88	13-29	10	-	20	[57]
		0.01	-	0.05	564	-	951	19	-	29	79	-	82	18	15			[54]
District	boiler	0.5	-	5	532	-	783	24	-	147	79	-	83	46	15	-	20	[54]
	gasifier	0.15			874			82			83			37	15			[54]
Biomass projection 2030																		
Domestic	boiler	0.01	-	0.05	523	-	899	19	-	29	83	-	85	18	15			[54]
		0.005	-	0.1	310	-	1200	13	-	43	88	-	95	13-29	12	-	24	[57]
District	boiler	0.5	-	5	476	-	740	21	-	145	83	-	86	46	15	-	20	[54]
	gasifier	0.15			821			80			87			37	15			[54]
Fossil reference present[b]																		
Domestic	NG boiler	0.008	-	0.04	41	-	675	2	-	17	88			29	10			[139-141]
District	Oil boiler	50			132			9[c]			90			68	30			[139]

NG: natural gas

[a] If a range in size of the system is given, the higher cost data and the lower conversion efficiency, capacity factor and lifetime refer to small-scale production and vice versa.

[b] No cost projections for 2030 were found in literature. We assume investment costs decline with 5% from current to 2030, based on rate found for biomass systems.

[c] O&M fixed annual 2 $/($kW_{th}$.yr) + O&M non feed variable 1 $/$MWh_{th}$.

from the International Energy Agency (IEA) [57] and the European Environment Agency (EEA) [54] to the fossil reference shows that the investment and operation & maintenance (O&M) costs are significantly higher for all biomass systems. This is also true for 2030, as the projected reductions in investment and O&M costs are limited for both domestic and district heating. For domestic chip and pellet systems the reduction in investment costs is 0-7% [54,57]. The reductions for district heat systems are projected to be about 10% for a 5 MW boiler, and 5-6% for a 0.5 MW boiler and the 150 kW gasifier option [54]. Yet, the data from the IEA and EEA show that economies of scale have a significant effect on both the investment and O&M costs. Although this is especially true for domestic heating, the possibilities for upscaling of domestic heating systems are limited. A study by Weiss et al. [138], however, suggests that cost reductions can be attained in the production of the boiler systems. They find that, amongst others, economies of scale in both boiler assembly and component manufacturing played an important role in cost reductions of fossil-based condensing gas boilers. Unlike the cost components, the feedstock conversion efficiencies of biomass heating systems are comparable to the fossil reference systems. By 2030, technological improvements in biomass conversion result in efficiency increases of 2-4%.

6.4.4.2 Power

Cost data for power generation from biomass and fossil fuels are provided in Table 6-5 (power generation in non-CHP plants) and Table 6-6 (power production through CHP). The typical capacities in biomass systems are generally lower compared to fossil reference systems. As a result, mainly because of economies of scale, the specific investment costs ($/kW$_e$) are higher. In both tables, the costs for cofiring only cover retrofit costs to make an existing coal power plant suitable for biomass cofiring. As a result, cofiring options have substantially lower investment costs than dedicated biomass systems. Direct cofiring is a commercial technology, and has lower investment costs compared to parallel and indirect cofiring. Projections, however, do not show cost reductions for direct cofiring [[55] in:[27];[54]]. The projected reduction in the investment costs of indirect cofiring are 13% [54].

For both dedicated biomass combustion and gasification technologies (both power and CHP production), we find large divergence in investment costs between literature sources. Regarding combustion, this may be explained by a difference in costs between established and advanced technologies. In the case of gasification, the variance can partially be explained by differences between specific gasification technologies [7,8,142], but a comparison between Obernberger et al. [8] and EEA [54] shows that estimates of further investments do also vary. Considering the lower investment levels for gasification, we find that a present 50 MW$_e$ IGCC may have lower specific investment costs than a 100 MW$_e$ combustion-based steam cycle [[55] in:[27];[54]]. Yet, at the higher level the

TABLE 6-5 | Present and projected cost data for power generation from biomass and fossil fuels (power only, CHP not included)[a].

	Typical size of device (MW_e)	Investment cost ($2005/$kW_e$)	O&M fixed annual ($2005/($kW_e$·yr))	Non-feed variable operating cost ($c/$kWh_e$)	Feedstock conversion efficiency electricity (%)	Capacity factor (%)	Econ. life-time (yrs)	References
Biomass present								
Dedicated combustion/Gasification								
SC (fixed/ fluidized bed)	25 - 100	2,600 - 4,100	84 - 87	0.34 - 0.4	27 - 28	70 - 80	20	[[55] in:[27]]
SC	25 - 100	1,850 - 2,800						[6,144][e]
IGCC[d]	50	1,900 - 3,500	88 - 99		45	68 - 80	20	[54,62][b,e]
	250	1,200 - 2,200	55 - 61		45	68 - 80	20	
Cofire								
Direct	20 - 100	430 - 500	12	0.18	36	70 - 80	20	[[55] in:[27]]
	35 - 70	186	118		41	68	30	[54]
Parallel	20 - 100	760 - 900	18		36	70 - 80	20	[[58] in:[27]]
Indirect (CC)	450	901 - 1037	46 - 51		44	68	20	[54][b]
Biomass projection 2030								
Gasification								
IGCC	50	1,600 - 3,000	94 - 104		45	68	20	[54,62][b,d]
	250	1000 - 1,900	58 - 64		45	68	20	
Cofire								
Direct	35 - 70	186	118		47	68	30	[54]
Indirect (CC)	450	798 919	40 45		47	68	20	[54][b]

TABLE 6-5 | Continued

	Typical size of device (MW_e)	Investment cost ($2005/$kW_e$)	O&M fixed annual ($2005/($kW_e$·yr))	Non-feed variable operating cost ($c/$kWh_e$)	Feedstock conversion efficiency electricity (%)	Capacity factor (%)	Econ. life-time (yrs)	References
Fossil reference								
Present								
NGCC	150 - 870	472 - 1133	24 - 45[c]	0 - 0.02	38 - 60	85	30	[143,145]
coal	300 - 1300	732 - 2505	26 - 60[c]		39 - 46	85	40	[145]
PC	n.a	2060	75	0.36	46	85	30	[143]
2030								
NGCC	n.a	784	21	0.02	58	89	30	[143]
PC	n.a	1867	64	0.33	46	89	30	[143]

SC: steam cycle; IGCC: integrated gasification combined cycle; CC: combined cycle; NGCC: natural gas combined cycle; PC: pulverized coal; n.a. not available

[a] If a range in size of the system is given, the higher cost data and the lower conversion efficiency, capacity factor and lifetime refer to small-scale production and vice versa.

[b] For gasification routes from EEA [54], only costs of electricity production were given (taking gasified biomass as feedstock). For comparison reasons, gasification costs were added based on Bain [7] and Obernberger et al. [8] (see supplementary material). For 300 MW output (clean product gas) the current investment cost is estimated to be 80-110 M$. In the longer term, the investment costs are assumed to reduce with 10% to 72-99 M$. Current gasifier efficiency is estimated to be 80%. The projected gasifier efficiency is 85%. The scaling factor is 0.7 and O&M costs are 4% of investment costs.

[c] Fossil references from IEA [145]: O&M costs include both variable and fixed costs.

[d] based on the cost projection of EEA [54], a cost reduction of about 14% was applied to current investment cost levels

[e] investment costs given for another size were scaled using a scaling factor n=0.7

TABLE 6-6 | Present and projected cost data for power generation by CHP[a].

	Typical size of device (MW_e)	investment cost ($/$kW_e$)	O&M fixed annual ($/($kW_e$·yr))	O&M non-feed var. ($c/$kWh_e$)	Feedstock conversion efficiency electricity (%)	Power-heat ratio	Capacity factor (%)	Econ. life-time (yrs)	References
Biomass present									
Combustion									
SC (fixed bed)	25 - 100	2,800 - 4,200	86	0.35	24	0.67	70 - 80	20	[[55] in:[27]]
SC	2.5 - 10	4,100 - 6,200	54	3.5	18	0.28	55 - 68	20	[8,27]
SC	2.5 - 10	2,500 - 4,000							[146]
Direct cofiring (SC)	5 - 100	184 - 249	113 - 153		26	0.48	68	25	[54][h]
Steam engine	0.8	4,830 - 5,587	563 - 586		12 - 13	0.19 -0.21	68	20	[54]
ORC	0.65 - 1.6	6,500 - 9,800	59 - 80	5.1	14	0.19	55 - 68	20	[8,27]
	0.8	5,806 - 6,710	513 - 542		9 - 12	0.14 -0.19	68	20	[54]
Stirling engine	0.05	6,086	205		16	0.27	68	10	[54]
MSW (SC)	10	11,125	671		14	0.4	80	15	[54]
AD - gas engine	0.3 - 3.36	2,487 - 5,098	554 - 702		13 - 19	0.67	68 - 91	15-25	derived from IEA [57]
Gasification									
Gas engine	0.1	9,000 - 15,000	494 582		30	0.71	68	20	[8,54,61][b,f]
	1	4,500 - 7,500	245 - 291		28 - 34	0.82	68	20	
	10	2,400 3,700	65	1.1-1.9	30	1.52	68	20	
IGCC	50	2,000 - 3,500	96 - 107		40	0.88	68 - 80	20	[54,61][b,f]
	250	1,200 - 2,300	59 - 66		40	0.88	68 - 80	20	
Micro gas turbine	0.1	6,676 - 8,835	460 - 574		31	0.65	68	15[c]	[54][b]
Biomass projection 2030									
Combustion									
Steam engine	0.8	3,941 - 4,728	535 - 561		15 - 16	0.24 -0.26	68	20	[54]
ORC	0.8	4,889 - 5,660	486 - 510		14	0.22	68	20	
Stirling engine	0.05	4,541	159		20	0.34	68	10	[54]
Direct cofiring (SC)	5 - 100	184 - 249	113 153		28 - 30	0.56	68	25	[54][h]
MSW (SC)	10	11,125	671		17	0.40	80	15	[54]

TABLE 6-6 | Continued

	Typical size of device (MW_e)	investment cost ($/$kW_e$)	O&M fixed annual ($/($kW_e$.yr))	O&M non-feed var. ($c/$kWh_e$)	Feedstock conversion efficiency electricity (%)	Power-heat ratio	Capacity factor (%)	Econ. life-time (yrs)	References
Gasification									
Gas engine	0.1	8,000 - 13,000	456 530		35 -	0.88	68	20	[8,54,61][b,e]
	1	3,900 - 6,600	211 - 246		40	1	68	20	
	10	2,000 - 3,200	65	1.1-1.9	40	1.52	68	20	
IGCC	50	1,700 - 3,000	96 106		42	1	68 - 80	20	[54,61][b,g]
	250	1,000 - 2.000	59 - 66		42	1	68 - 80	20	
Micro gas turbine	0.1	5,434 - 7,193	410 481		38	0.87	68	15[c]	[54][b]
Fossil reference									
Present									
NGCC	<250	930	48		32		66	30	[143]
gas turbine	25	1262	48		32		64	30	[143]
gas engine	<2	709	38		41		46	30	[143]
NGGT	60 - 405	715 - 1683	30[d] - 104		38		85	30	[145]
2030									
NGCC	<250	1341	59		43		67	30	[143]
gas turbine	8 - 45	1252 - 1895	45 - 93		25 - 28		75	30	[143]
gas engine	<2	745	35		41		46	30	[143]

SC: steam cycle; CC: combined cycle, ORC: Organic Rankine Cycle; IGCC: integrated gasification combined cycle; NGCC: natural gas combined cycle; NGGT, natural gas gas turbine.
[a] If a range in size of the system is given, the higher cost data and the lower conversion efficiency, capacity factor and lifetime refer to small-scale production and vice versa.
[b] For gasification routes from EEA [54], only costs of electricity production were given (taking gasified biomass as feedstock). For comparison reasons, gasification costs were added based on Bain [7] and Obernberger et al. [8] (see supplementary material). For 300 MW output (clean product gas) the current investment cost is estimated to be 80-110 M$. In the longer term, the investment costs are assumed to reduce with 10% to 72-99 M$. Current gasifier efficiency is estimated to be 80%. The projected gasifier efficiency is 85%. The scaling factor is 0.7 and O&M costs are 4% of investment costs. Note that this approach leads to high investment and fixed O&M costs for small-scale gasification routes (0.1-1 MW_e), in EEA the 'feedstock' costs were not significantly higher compared to larger-scale units.
[c] in EEA [54] an economic lifetime of 10 years was found, in this work 15 years is used to account for the gasifier lifetime
[d] Fossil references from IEA [145]: O&M costs include both variable and fixed costs.
[e] based on the cost projection of EEA [54], a cost reduction of about 12% was applied to current investment cost levels
[f] investment costs given for other sizes were scaled using a scaling factor n=0.7
[g] based on the cost projection of EEA [54], a cost reduction of about 15% was applied to current investment cost levels
[h] investment costs given for another size were scaled using a scaling factor n=0.9

(scaled) investment costs are lower for dedicated biomass combustion. In addition, gasification combined with a gas engine attains lower investment costs compared to an IGCC at similar scales [54].

Of small-scale CHP technologies, Stirling engines have lowest investment costs [54]. For commercial CHP production through MSW combustion, investment costs are very high compared to all other biomass technologies. Also, projections for 2030 do not show any cost reductions [54]. Projected reductions in investment costs of other power and CHP technologies range from 11% for indirect cofiring, and 12-19% for gasification-based technologies to 24% for a Stirling engine [54]. For larger-scale dedicated biomass combustion technologies (> $2MW_e$), no projections were found. The investment costs of natural gas and coal-based power plants are projected to decrease by about 10% until 2030 [143]. In contrast, the investment costs of natural gas-based CHP technologies are projected to increase by 5-6%. Yet, while the electric efficiency of NGCC-based CHP production is projected to increase by more than 30%, the feedstock conversion efficiencies of other fossil reference technologies are not projected to improve [143]. The efficiencies of bioenergy technologies are expected to increase by 1-7% for large-scale gasification (IGCC and indirect cofire) and 17-25% for small-scale CHP [[55] in:[27];[8,27,54,57]].

6.4.4.3 Fuels

In Table 6-7, we give an overview of cost data for the production of first and second-generation biofuels. As no appropriate data on investment and O&M costs of an oil refinery for gasoline and diesel production is available, we cannot compare cost data for biofuels and fossil fuels. Therefore, only final production cost data will be compared in section 6.6.1.1.

Currently, the highest production capacity is attained for sugarcane ethanol. Scaling suggests that the investment costs for corn and wheat ethanol production at this capacity level (1000 MW feed) would be just above the costs for sugarcane (about 120-135 $/$kW_{feed}$). This is true for data from Bain [7] for wheat ethanol production in Canada, Argentina and the USA. For Europe, much higher investment costs are found by the EEA [54]. For combined ethanol and sugar production, which is often the case in Brazil, only approximately half of the feedstock is used for ethanol production. As a result, the feedstock conversion efficiency is only 17%. When only taking into account the feedstock share for ethanol, the efficiency is 38-42% [64,144]. Biodiesel production from soy and palm oil attains a very high feedstock conversion efficiency of 103%, because methanol is incorporated into the product. The same is true for renewable diesel production (hydrogen added), which attains a conversion efficiency of 95% [7]. In addition, at relatively low capacity, the hydrogenation route results in the lowest investment costs of

all biofuels (hydrogen production excluded; hydrogen input is included in variable O&M costs). If investments costs for methanol or hydrogen production respectively would be included, investments would be considerably higher. At capacities between 300 and 700 MW_{feed}, the production of second-generation fuels requires very high investment costs [7,28,60]. The investments are projected to decline, partly as a result of upscaling, but remain higher than first-generation investment costs [7,28,60]. This is especially the case for FT diesel. During the production of second-generation fuels, however, often more power is co-generated than needed for the process [7,28,54,60]. By-product revenues for exported electricity will have a positive effect on the production costs. It is expected that improved technology for sugarcane ethanol production will also result in increasing sale of electricity [147].

6.4.4.4 Materials

Table 6-8 shows cost data for current olefins production from Ren et al. [14]. In their study, Ren et al. [14] also made cost projections for these production routes. However, the study does not provide sufficient data to reproduce future levelized cost data. Cost figures for biotechnological conversion routes from sugarcane and corn to ethylene and other white biotechnology chemicals are derived from Hermann and Patel [15][3]. In Hermann and Patel, the costs are based on the conversion of fermentable sugar to materials (i.e. the conversion costs do not depend on the type of feedstock and the required process steps, but on the price of fermentable sugar). We have recently updated these calculations[4],[5] (see Saygin et al. [4] for more details) and will present the results in section 5.2. We did not find production cost data of other materials (amongst which are starch plastics, cellulose polymers, ECH and epoxy resin). The figures from Ren et al. [14] show that the cost figures vary with the specific process design. When electricity is co-generated, the investment and O&M costs are generally higher, and feedstock conversion efficiencies are lower. The investment costs are lowest for biomass gasification combined with the methanol-to-olefins process. Gasification combined with the FT naphtha process results

3 Published as production costs plus profit (PCPP)

4 The work of Hermann and Patel was conducted within the BREW project (2003-2006), for which confidential data was made available. The confidential data were not used in our update. Wherever the original calculations involved the use of confidential data, extrapolation and triangulation was applied using publicly data for various crude oil prices published in Patel et al. [93], Hermann and Patel [15] and Hermann et al. [150].

5 To estimate the production costs, we assumed that the production costs are 5-25% below the product value (PCPP). This was based on investment and production cost data for bio-ethylene published by IEA-ETSAP [95]: the capital costs of bio-ethylene plants are stated to be 278 M$ at 200 kt/yr and 400 M$ at 350 kt/yr. At a capital charge of 11% (CRF, depreciation only), this means that the investment costs are 125-155 $/tonne. The report gives production costs of 970-1,630 $/tonne (Brazilian sugarcane) and 1,700-2,730 $/tonne (US corn). At a capital charge of 30% (depreciation plus profits), the investment costs would be 340-420 $/t and the production costs would increase to 1,185-1,895 $/tonne (sugarcane) and 1,915-1,995 $/tonne (corn). So, to come back to the production costs we would need to reduce the product value by 7-21%. This is consistent with the sensitivity analysis for the capital charge in Hermann and Patel [15].

TABLE 6-7 | Present and projected cost data for first and second-generation biofuel production[a].

	Typical size of device (MW_{feed})	Investment cost ($2005/$kW_{feed}$)	O&M fixed annual ($2005/($kW_{feed}$.yr))	O&M non feed variable ($/$GJ_{feed}$)	Feedstock conversion efficiency fuel (%)	Capacity factor (%)	Economic lifetime (years)	References
Current								
Ethanol – 1st generation								
Sugarcane[b]	172	259 - 358	25 - 35	0.87	17	50	20	[[7,27];[64,144,148] in:[2]]
	1,024	83 - 115	16 - 22	0.87	17	50	20	
Corn[h]	138	241 - 310	17 - 27	1.98	54	95	20	[[7,27];[64] in:[2]]
	554	158 - 203	9 - 13	1.98	62	95	20	
Wheat	152	220 - 282	16 - 25	1.41	49	95	20	[[7,27];[6,149] in:[2]]
	166 - 498	419 - 552	22 - 28		54	86	15	[54][c]
	607	144 - 185	8 - 12	1.41	53	95	20	[[7,27];[6,149] in:[2]]
Diesel - 1st generation[d]								
Biodiesel soy oil	44	301 - 323	27 - 46	2.58	103	95	20	[7,27]
	440	159 - 171	9 - 13	2.58	103	95	20	
Biodiesel palm oil	44	299 - 339	34 - 46	2.58	103	95	20	[7,27]
	440	158 - 180	10 - 13	2.58	103	95	20	
Biodiesel rape-seed	102 - 401	142 - 167	30 - 35		49 - 62	91	20	[54][e]
Renewable diesel soy oil	117 - 235	49 - 67	2 - 3	2.29	92	95	20	[7]
BC fuels - 2nd generation								
Ethanol SHF	440	497	19	3.09	37[g]	96	20	[28]
Ethanol SSF	400 - 2000	528 - 683	34 - 44		35	91	20	[60][f]
Ethanol SSCF	334	562	37	1.11	44	95	20	[7]
	667	470	27	2.11	44	95	20	
TC fuels – 2nd generation								
Ethanol	440	550	32	0.69	32[g]	96	20	[28]
	374 - 747	420 - 510	25 - 33	0.17 - 0.18	40	95	20	[7]
Methanol	400 - 2000	394 - 552	16 - 22	0.47	59	91	20	[60][f]
Hydrogen	400 - 2000	427 - 580	17 - 23		35	91	20	[60][f]
FT diesel	400 - 2000	539 - 686	24 - 30		42	91	20	[60][f]

TABLE 6-7 | Continued

	Typical size of device (MW_{feed})	Investment cost ($2005/$kW_{feed}$)	O&M fixed annual ($2005/($kW_{feed}$.yr))	O&M non feed variable ($/$GJ_{feed}$)	Feedstock conversion efficiency fuel (%)	Capacity factor (%)	Economic lifetime (years)	References
Projection 2030								
1st generation								
Biodiesel rape-seed	80 - 401	167 - 181	35 - 38		62	91	20	[54] [e]
Ethanol wheat	179 - 490	362 - 436	19 - 23		54 - 59	86	15	[54] [c]
BC fuels - 2nd generation								
Ethanol	179	243	101		56	86	15	[54]
Ethanol CBP	400 - 2000	383 - 512	14 - 18		47	91	20	[60] [f]
TC fuels – 2nd generation								
Methanol	400 - 2000	341 - 442	14 - 18		57	91	20	[60] [f]
Hydrogen	400 - 2000	388 - 486	16 - 19	0.01	41	91	20	[60] [f]
FT diesel	400 - 2000	434 - 552	19 - 24		42	91	20	[60] [f]
	1075	925	56		47	86	20	[54]

BC, biochemical; TC, thermochemical

[a] If a range in size of the system is given, the higher cost data and the lower conversion efficiency, capacity factor and lifetime refer to small-scale production and vice versa.

[b] integrated sugar mill: 50% ethanol & 50% sugar production (sugar is considered to be a by-product)

[c] Cost data from EEA [54] for large-scale (289 MW ethanol) derived using scaling factor of 0.75 [60].

[d] feedstock conversion efficiency is given for vegetable oil to diesel, except for rapeseed biodiesel where the efficiency is given for rapeseed to biodiesel; methanol or hydrogen inputs are included in O&M costs.

[e] Original capacity 12.5 MW biodiesel, scaled to 50 and 250 MW biodiesel; cost data derived using a scaling factor of 0.95 [60].

[f] Scaling was used to derive cost data for larger scale (present cost data) or smaller scale (projected cost data). For scaling factors, see Hamelinck and Faaij [60].

[g] derived from production capacity and feedstock use, assuming HHV ethanol is 23.4 MJ/liter and HHV lignocellulosic biomass is 18 GJ/t dm

[h] dry milling

in the highest investment costs. Regarding the production routes of ethylene through biomass fermentation, the investment and fixed O&M costs are highest for the conversion of lignocellulosic biomass. This is because more complex technology is required for hydrolysis and fermentation. When production scales would be increased, fermentation-based processes become modular beyond a certain size, while cost reductions may still be attained for gasification processes.

TABLE 6-8 | Present cost data for olefin production [107].

		Investment cost ($/t product)	O&M ($/t product)	Non-feed variable operating cost ($/t product)	Power co-generation (GJ/t product)	Feedstock conversion efficiency olefin (wt%)	Capacity factor (%)	Economic lifetime
Fermentation-ethylene ETE	Sugarcane	275	38.8	0	16	11.6	91	15
	Corn	275	38.8	816	0	28.1	91	15
	Ligno	251	83.4	176	4	18.9	91	15
		353	213.6	0	22	15.3	91	15
		353	213.6	88	5	16.5	91	15
Gasification-Olefins[a]	FT naphtha	395	122.5	0	16	9.0	91	15
		462	140.6	0	50	10.5	91	15
	MTO	221	84.8	309	0	21.1	91	15

[a] mainly ethylene and propylene

TABLE 6-9 | Current and projected conversion efficiencies for fermentative biochemicals and -materials, derived from [11].

		Conversion efficiency (t product/t dm feed)[a]		
		Sugarcane	Corn	Corn stover
Ethanol	Current	0.16	0.44	
	Future	0.17	0.45	0.25
Butanol	Current	0.12	0.32	
(ABE[b])	Future	0.14	0.38	0.21
Ethylene	Current	0.10	0.26	
	Future	0.10	0.27	0.16
PLA	Current	0.17	0.68	
	Future	0.26	0.68	0.40
PHA	Current	0.11	0.31	
	Future	0.14	0.40	0.21
PTT	Current	0.39	1.02[c]	
	Future	0.51	1.36[c]	0.70

[a] Values based on sugar contents: sugarcane 0.42 t sucrose/t dm [147,151]; corn 0.94 t sucrose/t dm; corn stover 0.56 t sucrose/t dm [11]; future values represent the technical potential after 20-30 years of R&D [11]
[b] Butanol is produced through ABE fermentation, by which also acetone and ethanol are produced.
[c] The biomass feedstock conversion efficiencies for corn-based PTT are higher than 1 because the feedstock for PTT is only partially bio-based.

Current and projected conversion efficiencies for various biochemicals are compared in Table 6-9. The efficiencies are highest for starch feedstock (corn) and lowest for sugar feedstock (sugarcane). This is consistent with Ren et al. [14]. The efficiencies for PTT are higher than for other products, because the feedstock for PTT only partially consists of biomass.

6.4.4.5 Technological learning

Typically, the increasing diffusion of a technology into the market results in a decrease of the unit cost of the technology [20]. This can be explained by mechanisms like learning-by-doing, technological innovation, and economies of scale. The combined effect of these mechanisms is referred to as technological learning [138]. Empirical observations show that costs tend to decline at an almost fixed rate with each doubling of the cumulative production [20]. This rate is called the learning rate, and the relationship between the unit cost and cumulative production can be described by the so-called experience (or learning) curve.

Historic learning in biobased systems

In recent literature, a number of analyses has been performed to quantify learning in bioenergy systems, see Table 6-10. Attempts to do the same for biomaterials have not been successful. The major reasons are the lack of time series for prices due to the early stage of development and the non-existence of a bulk market price (due to the small production quantities).

The learning system of bioenergy systems can be split into three separate parts, i.e. the feedstock supply system, O&M of the conversion plant, and investment in the plant. Cost reductions in feedstock production are assessed for various crop types [18,19,133,152]. In all cases, the development of improved plant varieties (increased yields) was found to be an important driver for historic cost reductions. In addition, sugarcane production costs were significantly reduced by increasing the length of the ratoon system, and improving the management system (increased harvesting efficiency) [18]. In corn production, costs were highly influenced by the increased size of farms (economies of scale) [19]. The production costs of dedicated energy crops decreased significantly through mechanization in harvesting [133]. With regard to biomass logistics, Van den Wall Bake et al. [18] show that transportation costs for sugarcane mainly declined because of upscaling truck loads, automation of logistic systems and improved infrastructure. De Wit et al. [133] find similar mechanisms for eucalyptus in Brazil. Junginger et al. [153] find that transport cost for forest wood chips in Sweden have remained stable. However, technical improvement of the chippers has significantly reduced the supply costs of wood chips. Investment and O&M costs of the conversion process decline due to various learning mechanisms. In

many cases, however, increasing scales of the conversion plants are found to be a key driver for cost reductions [18-20]. Findings of Junginger et al. [20] suggest that this is especially true for technologies which are developed on a global scale. For technologies developed on a local scale, learning-by-using and learning-by-interacting are indicated to play a major role in the reduction of costs.

Projections on learning induced cost reductions

Extrapolation of experience curves provides the opportunity to investigate potential future cost developments. Under a continued growth scenario for Brazilian sugarcane ethanol, Van den Wall Bake et al. [18] project that sugarcane production costs can decrease by 35-45% in 2020 compared to 2000-2004. Ethanol production costs are projected to decrease by 17-48% in the same time period. Hettinga et al. [19] estimate that corn production costs will decrease by approximately 30% until 2020, and ethanol production costs will decrease by about 46%, both compared to the 2000-2004 level. These projections, however, are only based on further technological progress and do not take into account minimum attainable cost levels [19]. Therefore, studies by, for example, de Wit et al. [133] and IEA [52] do combine the experience curve approach with bottom-up analyses. In the combined approach, bottom-up analysis provides insight in the improvement potential of every supply chain component and derives minimum cost levels. The learning rate and the cumulative production, which is related to market volume, determine the speed at which costs decline and when minimum costs are achieved. Based on this approach, de Wit et al. [133] project that when the demand for SRC increases to an upper limit of 38 EJ in 2030, minimum production costs could be achieved in 2021-2025. At a demand of 20 EJ in 2030, the minimum cost level is projected to be reached between 2024-2030 [133]. In the Biofuels Roadmap, the IEA [52] assesses future production costs for different biofuels. Under favorable conditions (low impact of rising oil prices, rapid increase in market volume, high learning rate), advanced biofuels such as lignocellulosic ethanol and FT diesel can become cost competitive to petroleum gasoline and diesel by about 2030. In a more pessimistic scenario, costs decline more slowly and will permanently remain higher than in the optimistic scenario. By 2050, only first-generation ethanol and second-generation bio-SNG will be cost competitive to their fossil equivalents [52].

Finally, in the longer term, it can be expected that the increasing implementation of biorefinery concepts will influence the effect of technological learning. On the one hand, the variety in process configurations of biorefineries may slow down the speed at which the cumulative production doubles and costs decline. On the other hand, biorefineries may be split into different components with their own investment and O&M costs, and associated progress ratios. These progress ratios may be derived from analogue technologies that are operated at larger scale.

TABLE 6-10 | Learning rates for major components of bioenergy systems and final energy carriers.

	LR (%)	Time frame	Region	N	R^2	References
Feedstock production						
Sugarcane (tonnes sugarcane)	32±1	1975–2005	Brazil	2.9	0.81	[18]
Corn (tonnes corn)	45±1.5	1975–2005	USA	1.6	0.87	[19]
Rapeseed (tonnes rapeseed)	20±1	1971-2006	Germany		0.97	[152]
Eucalyptus (EJ eucalyptus)	37	1955-2010	Brazil			[133]
	27		World			
Poplar (EJ poplar)	22-29		Italy			[133]
Logistic chains						
Forest wood chips (Sweden)	12–15	1975–2003	Sweden/ Finland	9	0.87–0.93	[153]
Investment and O&M costs						
CHP plants	19–25	1983–2002	Sweden	2.3	0.17–0.18	[20]
Biogas plants	12	1984–1998		6	0.69	[20]
Ethanol production from sugarcane	19±0.5	1975–2003	Brazil	4.6	0.8	[18]
Ethanol production from corn (only O&M costs)	13±0.15	1983–2005	USA	6.4	0.88	[19]
Final energy carriers						
Ethanol from sugarcane	7	1970–1985	Brazil			[154]
	29	1985–2002		~6.1	n.a.	
Ethanol from sugarcane	20±0.5	1975–2003	Brazil	4.6	0.84	[18]
Ethanol from corn	18±0.2	1983–2005	USA	7.2	0.96	[19]
Electricity from biomass CHP	8–9	1990–2002	Sweden	~9	0.85–0.88	[20]
Electricity from biomass	15	Unknown	OECD	n.a.	n.a.	[155]
Biogas	0–15	1984–2001	Denmark	~10	0.97	[20]

LR, Learning rate; N, number of doublings of cumulative production; R^2, coefficient of determination; n.a. not available

6.4.5 GHG emissions

In this section we present GHG emission ranges from life cycle inventories reported in literature, Table 6-11 to Table 6-14. These include both current data and projections for 2030. The total life cycle emissions include the emissions from feedstock production and supply, and, wherever relevant, from energy inputs in the conversion process.

A literature review by NREL [156] shows that the GHG emissions from power generation do not differ significantly between combustion and gasification. In the case of CHP production, however, the power-heat ratio is smaller for biomass combustion than for gasification. As a result, combustion receives more credits and attains lower net emissions compared to gasification [54,157]. In addition, the applied allocation method causes the net emissions for current power generation in CHP plants to be smaller than zero (except for CHP from wheat- and corn-based biogas). This allocation method thus suggests that CHP, and more specifically combustion-based CHP, is the preferred option. However, this is not always the case. CHP is only attractive when a demand for heat exists

in the immediate surroundings. Gasification has a higher electric efficiency compared to combustion, and may be preferred because electricity has a higher energy quality than heat. Also, through effective gas cleaning, impurities can be removed from the product gas. This results in reduced corrosion problems, cleaner combustion and lower non-GHG emissions of for example sulphur and chlorine compounds [158].

The cultivation of wheat, corn, soybean and rapeseed is associated with high GHG emissions, and affects the life cycle emissions of biogas CHP, biofuels and biomaterials. The major reason are N_2O emissions, which result from the use of fertilizers and agrochemicals [11,14,16,23,157,159]. However, there are large ranges for cultivation-induced emissions because of the sensitivity to the specific cultivation area and management system [23]. In contrast to cultivation, by-product credits from milling rapeseed and soybean can result in negative emissions for the processing and logistics step. This is to also true for jatropha. For palm oil, the credits are too small to attain negative emissions [23]. For sugarcane, perennials and short rotation crops, the emissions related to cultivation are relatively small compared to the crops discussed above. In the data gathered by Hoefnagels et al. [23], it is assumed that inland transport distances are higher for perennials and SRC (260-500 km) compared to corn, wheat and sugarcane (50 km). This results in up to 5 times higher emissions related to logistics. Still, the total upstream emissions for perennials and SRC are smaller than for corn and wheat, and smaller or comparable to the upstream emissions of sugarcane.

A few biofuel chains produce surplus power during the conversion process and can attain negative life cycle emissions. These are short-term ethanol from eucalyptus and FT diesel from miscanthus and eucalyptus, and future sugarcane ethanol [23,157]. With regard to biomaterials, the production of surplus power can result in negative life cycle emissions for ethylene, future butanol, and for olefins production via FT synthesis. This is however only true for sugarcane and lignocellulosic feedstocks. Environmental assessments of PHA find a relatively wide range in life cycle emissions. This reflects the variety of options for the production process. Although negative emissions are found in some cases, a review by Chen and Patel [96] indicates that high processing requirements for PHA production (downstream processing and wastewater treatment) involve substantial direct and indirect energy use, which results in an unattractive GHG balance.

TABLE 6-11 | Cradle-to-grave GHG emissions of heat and power generation.

			Timeframe	Upstream (gCO_2-eq/ MJ feed)		Life cycle (gCO_2-eq/MJ product)		Reference
				min	max	min	max	
Heat	Domestic	Wood pellets	EU present	1.4	4.7	4.3	11.6	[157]
			EU	8.8	10.3	10.4	12.1	[44]
			EU 2030	2.3	4.0	6.1	9.1	[157]
		Wood chips	EU present	2.0	2.0	6.0	9.4	[157]
			EU 2030	1.9	2.0	6.3	7.6	[157]
		SRC	EU present	5.9	7.0	11.0	14.5	[157]
			EU 2030	3.9	4.3	8.9	10.3	[157]
	District	Wood pellets	EU	2.5	2.6	2.7	2.8	[44]
		Lignocellulosic	EU present	4.8	7.0	8.5	13.7	[157]
		(SRC & perennials)	EU 2030	2.8	9.4	6.3	14.8	[157]
Power[c]	Cofire[b]	All feedstocks - power	1980-2010			-1	47	[156]
		Wood pellets (Canada to NL) - power	present	8.6	16	21.1	39.1	[44]
		Wood chips - power	EU present	2.0	2.0	11.8		[157]
		Wood chips - CHP	EU present	2.0	2.0	-113.7		[157]
	Combustion	All feedstocks - power	1980-2010			1	42	[156]
		Wood pellets Stirling - CHP	EU present	1.4	4.7	-274.3		[157]
		Wood chips ORC - CHP	EU present	2.0	2.0	-440.7		[157]
		Wood chips steam engine - CHP	EU present	2.0	2.0	-360.3		[157]
		Biogas - CHP (manure, landfill)	EU present	6.0	13.3	-67.6	-3.3	[157]
		Biogas - CHP (corn, wheat, sorghum)	EU present	15.1	36.5	-6.1	52.8	[157]
			EU 2030	11.5	23.4	-14.2	14.4	[157]
	Gasification[a]	All feedstocks - power	1980-2010			1	36[d]	[156]
		Wood chips IGCC - power	EU present	5.4	10.6	58.9		[157]
		Wood chips IGCC - CHP	EU present	5.4	10.6	-25.6		[157]
		Wood chips indirect cofiring - power	EU present	5.4	10.6	60.1		[157]
		Wood chips gasification ICE/GT - CHP	EU present	5.4	10.6	-74.3	-52.4	[157]
		SRC gasification ICE - CHP	EU present	10.7	16.0	-68.3	-59.4	[157]
			EU 2030	9.2	11.2	-68.8	-63.9	[157]

SC, steam cycle; ORC, Organic Rankine Cycle; ICE, internal combustion engine; GT, gas turbine; CC, combined cycle.
[a] For power generation through biomass gasification, the emissions from gasification are included in the upstream life cycles [157]
[b] Cofiring: only emissions for cofired biomass are given (average emissions for total generation would also include the part from coal).
[c] In the case of CHP, the net GHG emissions for electricity are derived by subtracting the emissions of a replaced natural gas-fired heating system from the total emissions of the cogeneration system.
[d] one estimate reaching 83 gCO_2-eq/MJ_e

TABLE 6-12 | Cradle-to-grave GHG emissions of biofuels.

(gCO_2-eq/MJ fuel)		Timeframe	Cultivation		Processing & logistics[a]		Conversion		Life cycle[d] (WTW)		Reference
			min	max	min	max	min	max	min	max	
Ethanol	Corn	Europe - present	34.1	74.9	0.5		-32.4	51.9	3.8	100.6	[23]
			54.5	*61.4*					55.9	61.4	[157]
		Europe - 2030 (whole crop)	*19.1*	*23.0*					19.6	23.6	[157]
		US - present	57.1		0.5		23.7		82.8		[23]
									56.4		[159]
	Wheat	Europe - present	39.0	67.5	0.6		-22.0	38.5	20.6	86.1	[23]
			45.9	*61.0*					45.9	61.1	[157]
		Europe - 2030 (whole crop)	*16.7*	*18.0*					17.3	18.6	[157]
	Sugarcane	Brazil - present	6.9	15.1	0.5	1.4	-5.8	2.2	10.8[c]	25.1[c]	[23]
									29.7[c]		[157]
		Brazil - future	6.2	6.9	0.5		-52.1	-9.7	-40.4[c]	2.6[c]	[23]
	Switchgrass	Europe - present	17.0		3.3		-6.7	-3.3	15.0	18.4	[23]
		Europe - 2030	12.3	12.4	3.3		-6.7		10.3	10.5	[23]
	Miscanthus	Europe - present	10.8	11.0	3.3		-3.3		11.8	12.0	[23]
		Europe - 2030	7.8	8.0	3.3		-3.3		8.8	9.0	[23]
	Eucalyptus	Brazil - present	0.9		1.5		-5.7		-1.0	-1.0	[23]
Biodiesel (FAME)	Oil palm	South East Asia – present	12.4	16.7	27.4[b]	36.2[b]	3.8	7.5	49.2	56.8	[23]
	Soy	US – present[b]	107.1		-10.4[b]		3.8		101.8		[23]
		Brazil – present	58.8	112.9	-10.4[b]	14.3[b]	3.8	9.6	81.6	107.6	[23]
	Rapeseed	Europe – present	33.7	161.4	-35.9	-27.4	1.4	3.9	10.5	130.6	[23]
			8.9	*35.0*					9.5	35.5	[23]
		Canada – present	101.4		-35.9		10.1		76.8		[23]
	Jatropha	Tanzania – present	33.9		-11.8[b]		7.1	10.0	30.5	33.4	[23]
FT diesel	Switchgrass	Europe - present	17.3		2.3		-8.8		11.3		[23]
		Europe - 2030	12.1		2.3		-8.8		6.7		[23]
	Miscanthus	Europe - present	10.7		2.3		-8.8		5.3	5.4	[23]
		Europe - 2030	7.7		2.3		-8.8		2.3	2.5	[23]
			-10.5						-9.9		[157]
	Eucalyptus	Brazil - present	2.0		3.5		-10.3		-0.5		[23]
	SRC	Europe - 2030	*-38.7*	*-9.7*					-35.6	-9.1	[157]

Italic: emission range given under cultivation is total of emissions for cultivation, processing and logistics.
FAME, fatty acid methyl ester; WTW, well-to-wheel
[a] Unless stated otherwise, logistics consists of inland transport of feedstock.
[b] Logistics includes transport of crude oil from the region of cultivation to the EU, where the conversion process takes place.
[c] Life cycle emissions include international ship transport of ethanol from Brazil to EU.
[d] Fuel combustion in a reference Otto or diesel engine

TABLE 6-13 | Cradle-to-grave GHG emissions of biomaterials production.

			Feedstock emissions (tCO_2-eq/t feed)	**Life cycle (Cradle-to-grave emissions w/o energy recovery, tCO_2-eq/t product)**				**Life cycle (With energy recovery, tCO_2-eq/t product)**		
	Process	Feedstock		Present		Projected: 2030 or beyond		Present	Projected	References
				min	max	min	max			
Ethylene	Fermentation - ETE	Sugarcane	-0.54	-0.9	-0.1	-1.4	-0.6	-1.3	-1.8	[4,11,13,14]
		Corn	0.4	2.1	3.1	2				[11,13,14]
	Hydrolysis + fermentation - ETE	Ligno	0.16	-1.4	0.5	0				[11,13,14]
Olefins	Gasification, FT, steam cracking	Ligno		-3.8	-0.1					[13,14]
	gasification, catalysed synthesis, MTO	Ligno		1						[13,14]
Butanol		Sugarcane	-0.54	0.2	1.3	-2	-0.6	0.4	-2.2	[4,11]
		Corn	0.4	3	3.5	0.4	1.8			[11]
		Ligno	0.16			-1	0.3			[11]
PHA	Fermentation	Sugarcane	-0.54	-1.1	3.4	-0.5	0.2	-1.3	-0.7	[4,11]
		Corn	0.4	1.9	6.4	1.8				[11]
		Ligno				0.5				[11]
PLA	Fermentation	Sugarcane	-0.54	1.7	2.8	0.9	1.5	1.9	1.0	[4,11]
		Corn	0.4	3	4.2	2.2	2.5			[11,16]
		Ligno	0.16			1.4	1.7			[11]
PTT	Fermentation	Sugarcane	-0.54	3.7	3.9	3.4	4	3.3	3.0	[4,11]
		Corn	0.4	4.5	4.8	4.1	4.7			[11]
		Ligno	0.16			3.7	4.3			[11]

6.4.6 Synthesis: selection of value chains

Considering the cost and emission data collected, we find that there are incongruences in the availability of data. The number of bioenergy technologies for which cost data is available is larger than the number for which GHG emission data is found. Also, we collected more current cost and emission data than projected data for both bioenergy and biomaterials. As a result, we can calculate levelized production costs for all technologies considered in section 6.4.4. But, the number of current and future biomass value chains for which we can calculate GHG abatement costs is limited by the availability of GHG emission data. Below, we define the biomass value chains (type of feedstock, pretreatment and logistics for each conversion technology) and select the input data needed to calculate the levelized costs.

TABLE 6-14 | Cradle-to-grave GHG emissions of fossil reference products.

Biomass product	Fossil reference product	Today 2010	Projection 2030	References
Energy		(gCO_2-eq/MJ product)		
Heat	Heat (hot water)	95.3[a] (EU-25)	91.5[a]	[21,27,157]
Power	Power	134.4 (EU-27 mix)	113.0	[21,157,160,161]
Ethanol	Gasoline	90	73.6[b]	[21,157,162-165]
Diesel (bio-, renewable, and FT diesel)	Diesel	86	70.3[b]	[21,157,162-165]
Materials		(tCO_2-eq/t product)[c]		
Ethylene/olefins	Ethylene/olefins	4.4	4.2	[11]
Butanol	Butanol	4.3	4.1	[11]
PHA	PE	4.7	4.5	[11]
PLA	PET	5.5	5.3	[11]
PTT	PTT	5.2	5.0	[11]

[a] Based on natural gas combustion
[b] For gasoline and diesel, we assume that the WTW emissions reduce with 1% per year [164,165]. If marginal fossil fuels (tar sands) would be used in the future, this would negatively affect the GHG emission balance of fossil reference products. We do not take this into account here.
[c] System boundary: cradle to grave (without use phase), incineration without energy recovery at end of lifetime [11]; for the projections, an emission reduction rate similar to heat is assumed.

6.4.6.1 Selection input cost and emission data

For each conversion technology, we compose a biomass value chain by defining the type of feedstock, pretreatment and logistics (Table 6-15). Based on the cost figures found for feedstock production, pretreatment and transportation (sections 6.4.2 and 6.4.3), Table 6-15 shows which values for the feedstock delivery costs are chosen to be used in the calculation of levelized production costs and in the sensitivity analysis. For most feedstocks, we use a cost level that is representative for a region where the feedstock is typically produced and converted. For wood pellets we assume ocean freight from Canada to Europe. As it is expected that the importance of pretreatment and international trade of lignocellulosic energy crops will increase in the future, it is preferred to take related biomass value chains into account. Due to a lack of emission data, however, our selection is limited to non-pretreated woody energy crops which are cultivated and converted in the same region. The production costs of biomaterials are based on the price of fermentable sugar. To give a representation of the production costs based on Brazilian sugarcane, we use a price of 141 $/tonne fermentable sugar (3.4 $/GJ sugarcane). For the USA and EU, we use a fermentable sugar price of 250 $/tonne sugar (13.3 $/GJ corn). Higher sugar price levels are represented by the world raw sugar price (contract 11): the 10 year average sugar price is 276 $/tonne, the 5 year average price is 347 $/tonne [166]. For the fossil feedstocks, we use present and projected fossil resource price data from the IEA World

TABLE 6-15 | Feedstock types and costs used for levelized cost calculations.

Product	Feedstock	Region feedstock cultivation	Transport mode	Region final conversion	Feedstock delivery cost ($/GJ)[a]				Feedstock yield (GJ/(ha.yr))	
Energy					present	sensitivity	2030	sensitivity	present	2030
Heat, combustion	Pellets (sawdust, shavings)	Canada	Ocean vessel	EU	9	5-12	4.5	3-7	20	25
Power/CHP, combustion	Pellets (sawdust, shavings)	Canada	Ocean vessel	EU	9	5 -12	4.5	3-7	20	25
	MSW	n/a	Local truck	EU	1.5	0.5-3	1.5	0.5-3	n/a	n/a
Power, AD	Manure, biowastes	n/a	Local truck	EU	1.5	0.5-3	1.5	0.5-3	n/a	n/a
Power/CHP, gasification	Pellets (sawdust, shavings)	Canada	Ocean vessel	EU	9	5 -12			20	25
	Woody energy crops	EU/LA	National truck	EU/LA			3.5	2-5		280
Ethanol (1st gen.)	Sugarcane	Brazil	National truck	Brazil	3.4	2-6.5			380	460
	Corn	USA	National truck	USA	9.0	4-10			145	175
	Wheat	EU	National truck	EU	17	15-20	15.5	13-18	130	235
Biodiesel	Soy oil	USA/Brazil	National truck	USA/Brazil	15	10-25			20	25
	Palm oil	Asia	National truck	Asia	23	13-27			160	195
	Rapeseed oil	EU	National truck	EU	26	15-32	13	11-15	50	60
Renewable diesel	Soy oil	USA	National truck	USA	15	10-25			20	25
2nd gen. fuels	Woody energy crops	EU/LA	National truck	EU/LA	4.5	2.5-6.5	3.5	2-5	190	280
Materials					**Feedstock costs ($/tonne)**					
Fermentation	Fermentable sugar	Brazil	National truck	Brazil	141	73-250	141	73-250	9.2 (t sugar/ha)[b]	11.1
		USA/EU	National truck	USA/EU	250	141-418	209	141-418	7.7 (t sugar/ha)[c]	9.3
	Woody energy crops	EU/LA	National truck	EU/LA	86	44-113			190	
Gasification	Woody energy crops	EU/LA	National truck	EU/LA	86	44-113			190	

n/a not applicable
[a] Feedstock delivery costs: total costs to deliver feedstock to the conversion plant (roadside costs + pretreatment + transportation)
[b] 0.42 tonne fermentable sugar/t dm sugarcane [147,151], current sugarcane yield 380 GJ/(ha.yr), projected 460 GJ/(ha.yr)
[c] 0.94 tonne fermentable sugar/t dm corn [11], current corn yield 145 GJ/(ha.yr), projected 175 GJ/(ha.yr)

Energy Outlook 2010 [63], Table 6-16. Finally, we select a fixed by-product revenue for each by-product type, based on economic values found in literature (Table 6-17).

The biomass value chains composed for biofuels and biomaterials correspond to the value chains defined in literature to calculate the life cycle emissions. For heat and power production, however, emissions are often given for other feedstock types like wood chips. As we consider wood pellets shipped from Canada to Europe, we recalculate the total life cycle emissions by assuming that only the upstream emissions change (Table 6-18).

TABLE 6-16 | Current and projected fossil resource prices [63].

Fossil resource	Applications	Price current		Price projection 2030[a]		
		Average	Sensitivity	Average	Sensitivity	
Oil	Heat, power, fuels, materials	10.4	8.3-12.5	15.3	12.5-18.1	$/GJ
		(75)	(60-90)	(110)	(90-130)	($_{2009}$/bbl)
Natural gas	Heat, power	6.4	3.5-8.1	11.1	8.1-13.7	$/GJ
		(7.4)	(4.1-9.4)	(12.9)	(9.4-15.9)	($_{2009}$/MMBTU)
Coal	Power	3.3	2.8-4.0	3.8	2.4-4.1	$/GJ
		(92)	(77-110)	(105.6)	(66.3-112.5)	($_{2009}$/tonne)

[a] Based on WEO 2010 scenarios: average for New Policies scenario, lower sensitivity value for 450 pp scenario, higher sensitivity value for Current Policies scenario [63].

TABLE 6-17 | By-product revenues used for levelized cost calculations.

Product		Byproduct(s)	By-product revenue	References
Heat		-	-	
Power		-	-	
CHP		Steam (large-scale CHP >25MW$_e$)	4.85 $/GJ steam[a]	[27]
		Hot water (small-scale CHP <10MW$_e$)	12.51 $/GJ hot water[b]	[27]
Ethanol	Sugarcane	Sugar[c]	4.26 $/GJ feed	[7,27]
	Corn	DDGS	1.56 $/GJ feed	[7]
	Wheat	DDGS	1.74 $/GJ feed	[7]
Biodiesel		Glycerol	0.58 $/GJ feed	[7]
2nd generation fuels		Electricity	0.054 $/kWh$_e$	[28]
Ethylene/olefins		Electricity	0.054 $/kWh$_e$	[28]

[a] 75% of heat output is sold
[b] 33% of heat output is sold
[c] In Brazil, sugarcane ethanol is often produced in an integrated sugar-ethanol plant; we treat sugar as byproduct, and assume that 50% of the sucrose is used for ethanol production and 50% for sugar production.

TABLE 6-18 | Total life cycle emissions for heat and power production based on combustion or gasification of wood pellets shipped from Canada to Europe[a].

				life cycle (gCO_2-eq/MJ product)	
				min	max
Heat	Domestic		present	13.4	17.6
			2030	12.2	15.5
	District		present	12.0	16.0
			2030	9.6	16.8
Power[b]	Cofire[c]	Power	present	25.3	21.4
		CHP	present	-102.3	-85.8
	Combustion	Stirling - CHP	present	-249.5	-228.5
		ORC - CHP	present	-406.5	-356.8
		Steam engine - CHP	present	-368.1	-308.9
	Gasification	IGCC - power	present	73.8	
		IGCC - CHP	present	-8.9	
		Indirect cofiring - power	present	75.4	
		Micro-gas turbine - CHP	present	-30.6	
		ICE - CHP	present	-52.1	-50.0

ORC, Organic Rankine Cycle; ICE, internal combustion engine; IGCC, integrated gasification combined cycle.
[a] Upstream emissions for wood pellets (pelletization of sawdust and shavings in Canada and pellet ocean shipping to Europe): 8.6 gCO_2-eq/MJ pellets [44] , no projected reduction in emissions [157]
[b] In the case of CHP, the net GHG emissions for electricity are derived by subtracting the emissions of a replaced natural gas-fired heating system from the total emissions of the cogeneration system.
[c] Cofiring: only emissions for cofired biomass are given (average emissions for total generation would also include the part from coal).

6.5 RESULTS

6.5.1 Levelized costs

6.5.1.1 Bioenergy

In Figure 6-4, the levelized costs of biomass-based heat, power and fuel production are compared to the price ranges of their fossil equivalent products. At a fossil oil price of 75 $\$_{2009}/bbl_{oil}$ and a natural gas price of 7.4 $\$_{2009}/MMBTU_{NG}$, large capacity wood pellet-fueled domestic and district heating (15 $/GJ for 100 kW domestic, and 18 $/GJ for 5 MW district heat) can only be cost competitive to capital intensive fossil heating systems.

With regard to power generation (power only and CHP), a significant variety in levelized costs is found. None of the technologies are cost competitive with fossil-based electricity (which costs up to 20 $\$/GJ_e$). Considering large-scale systems, we found that direct and parallel cofiring attain lower investments costs than the IGCC. At a wood pellet price of 9 $/GJ, the costs for cofiring (28-34 $\$/GJ_e$ at 5-100 MW_e) are comparable to a 250 MW IGCC (25-33 $\$/GJ_e$). Power production in a 50 MW_e IGCC costs 29-40 $\$/GJ_e$. Biomass combustion steam cycles can potentially also attain lower investment costs compared to an IGCC.

However, lower efficiencies for biomass combustion steam cycles affect the levelized costs significantly. At a feedstock price of 9 $/GJ, the costs for biomass combustion steam cycles are 45-47 $/GJ$_e$ at 100 MW$_e$ and about 50-56 $/GJ$_e$ at 20 MW$_e$. At a low feedstock cost level of 3 $/GJ (wood logs, residues) the cost for combustion would be 22-33 $/GJ$_e$. Similar cost levels are found by Bauen et al. [6]. Steam cycle technologies based on MSW combustion have both high investment costs and low efficiencies, resulting in levelized costs of 75 $/GJ$_e$. Small-scale CHP plants (gas engines and turbines, steam engine, ORC and stirling engines, <1 MW$_e$) are very costly at 78-122 $/GJ$_e$. Although projected cost reductions to 48-87 $/GJ$_e$ in 2030 are substantial (feestock cost 3.5 $/GJ for woody energy crops), and fossil energy prices are expected to increase, these technologies do not become cost competitive in the longer term. For a woody energy crop-fueled IGCC (feedstock 3.5 $/GJ) we find production costs of 11-26 $/GJ in 2030. In literature, power generation costs for an IGCC are projected to be 8-19 $/GJ [[144];[6] in:[2]]. Unfortunately, no projected levelized costs could be calculated for combustion steam cycles. A review study by the National Research Council finds no cost reductions between 2010 and 2020 [167].

Of the various first-generation biofuels, only large-scale ethanol production from low-cost Brazilian sugarcane (9-12 $/GJ$_{EtOH}$ at 3.4 $/GJ$_{sugarcane}$) attains lower production costs than petrochemical gasoline (16 $/GJ$_{gasoline}$). The costs of US corn ethanol (17-21 $/GJ$_{EtOH}$ at 9.0 $/GJ$_{corn}$), soy-based biodiesel and renewable diesel (17-19 $/GJ$_{diesel}$ at 15 $/GJ$_{soy\ oil}$) are comparable to the costs of fossil gasoline and diesel (16 $/GJ$_{diesel}$). Wheat ethanol (33-38 $/GJ$_{EtOH}$ at 17 $/GJ$_{wheat}$) and rapeseed biodiesel (39-49 $/GJ$_{deisel}$ at 26 $/GJ$_{rapeseed\ oil}$) have the highest levelized cost levels. In literature, the costs of commercial biofuels are largely comparable to our results (see table 2.7 in Chum et al. [2]). Disparities in production costs can mainly be explained by differences in feedstock costs used for the calculations.

The results for second-generation fuels show that methanol and hydrogen attain the lowest production costs, which are competitive to fossil diesel and gasoline. We find methanol production costs of about 12 $/GJ$_{MeOH}$ (4.5 $/GJ for woody energy crops). For comparison, a review by IEA-ETSAP [168] shows that cost estimates in literature mainly range between 9-36 $/GJ$_{MeOH}$ (no feedstock costs given), depending on plant setup, production capacity and local conditions. The productions costs of natural gas-based methanol are about 5-13 $/GJ$_{MeOH}$ [168]. By 2030 the costs of lignocellulosic-based fuels are projected to decline with rates up to 50% to a level comparable with present large-scale sugarcane ethanol production.

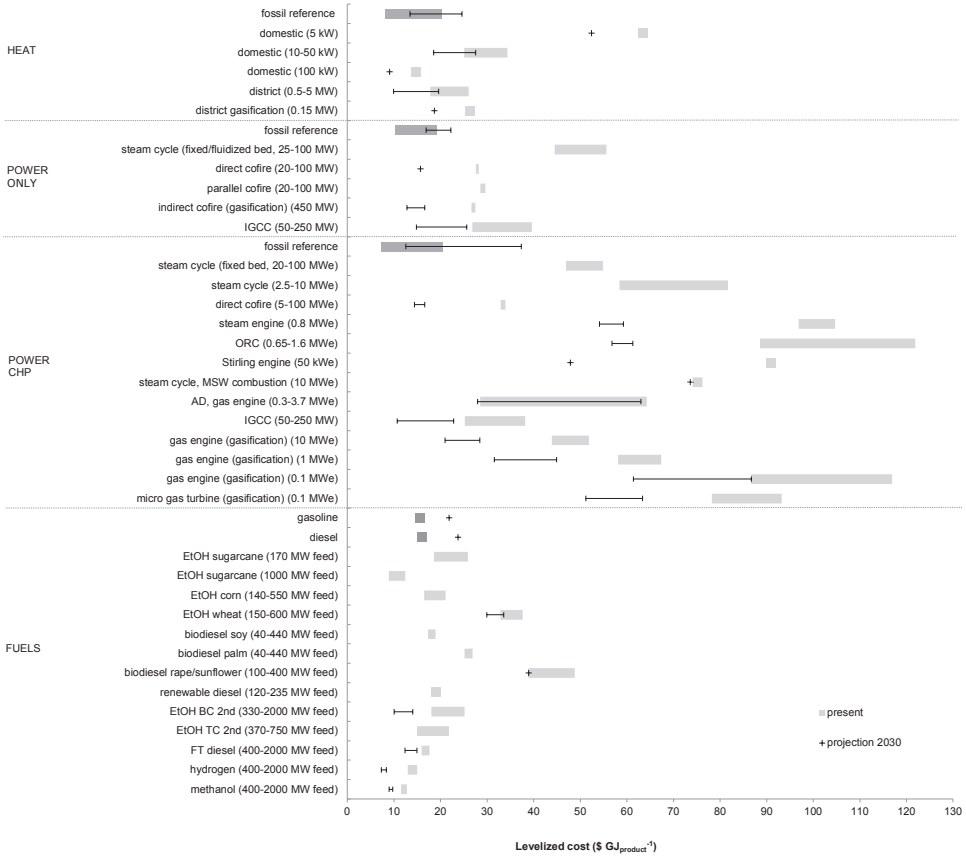

FIGURE 6-4 | Levelized costs of heat, power and fuels. Light grey bars: current costs of bio-energy; dark grey bars: current costs of fossil-based energy; black line: range of cost projections for 2030 based on multiple data sets; black plus sign: cost projection for 2030 based on one data set. CHP: levelized costs for power include a revenue for heat (Table 6-17). Fossil reference current: 75 $\$_{2009}/bbl_{oil}$; 2030: 110 $\$_{2009}/bbl_{oil}$ (Table 6-16).

6.5.1.2 Biomaterials

Figure 6-5 shows the (levelized) production costs of biomaterials. The figure indicates that only butanol and corn-based ethylene are not cost competitive to their fossil equivalent. Comparison with the cost estimates of Hermann and Patel [15] (based on fossil oil price of 25 $\$_{2009}/bbl_{oil}$) shows that the economic viability of biomaterials is substantially higher at a fossil oil price of 75 $\$_{2009}/bbl_{oil}$ because the oil price largely affects the costs of materials from fossil feedstocks. As the costs of fossil-based materials increase at a higher rate with the oil price than the costs of biomaterials, the production of butanol is projected to become economically feasible in the longer term, and all biomaterials are projected to improve their economic competitiveness.

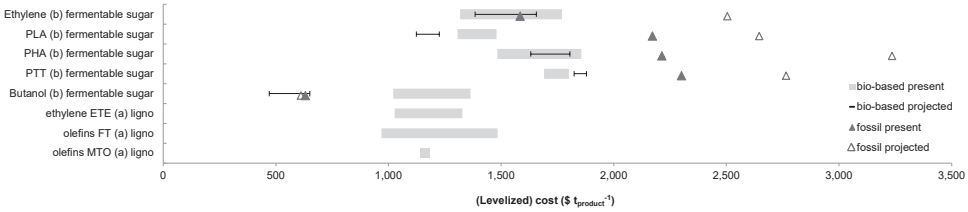

FIGURE 6-5 | Levelized cost for biomaterials and biochemicals. (a) levelized costs for technologies considered by ren et al. [14]; (b) estimated cost range for materials based on the work of Saygin et al. [4] using a fermentable sugar price range of 141 $/t$_{fermentable\ sugar}$ (Brazilian sugarcane, 3.4 $/GJ) to 250 $/t$_{fermentable\ sugar}$ (USA/EU, 13.3 $/GJ$_{corn}$). Light grey bars: current costs of bio-materials; black line: cost projections biomaterials for 2030; dark grey filled triangles: current cost of fossil equivalents; unfilled triangle: cost projection fossil equivalents for 2030. Fossil reference current: 75 $_{2009}$/bbl$_{oil}$; 2030: 110 $_{2009}$/bbl$_{oil}$ (Table 6-16).

For fermentation-based ethylene, our calculations based on data published in [15,93,150] result in higher production costs (1,320 $/tonne for 3.4 $/GJ$_{sugarcane}$ to 1,770 $/tonne for 13.3 $/GJ$_{corn}$) compared to the calculation based on data from Ren et al. [14] (1,030-1,330 $/tonne for 4.5 $/GJ$_{woody\ energy\ crops}$). IEA-ETSAP reports the production costs of sugarcane-based ethylene in Brazil and India to be around 1,100 $/tonne [95]. In other regions, the production costs are higher. For US corn-based ethylene the costs are just below 1,900 $/tonne [95]. The report also states that current costs for petrochemical ethylene 1,000-1,200 $/tonne in most regions (75 $_{2009}$/bbl oil) [95]. This is lower than the 1,600 $/tonne found in our analysis at the same oil price. Although the cost figures differ between sources, they show that ethylene production from biomass is only economically feasible at low feedstock prices and high oil prices.

For biochemical butanol production, Tao and Aden [64] estimate the future production costs to be just below 1,000 $/tonne. At 470-650 $/tonne, our long-term projection is significantly lower. For other materials, only selling prices are mentioned in literature. For PLA, we find production costs of 1,300 $/tonne (Brazilian sugar) to 1,480 $/tonne (US/EU sugar), while selling prices are about 1,800-3,500 $/tonne [76,104]. PHA prices are 3,500-4,700 $/tonne for bulk applications [76,104]. Our calculations result in costs of 1,480-1,860 $/tonne. The considerable difference between production costs found in our analysis and selling prices for PLA and PHA can be explained by risks of market introduction of new materials and the low volumes in which these materials are sold, resulting in high costs for selling efforts like marketing. Another reason could be the fermentable sugar prices used in our calculations, which are lower than the current world raw sugar price.

6.5.2 Avoided emissions

6.5.2.1 Bioenergy

Figure 6-6 shows that most bioenergy products do avoid emissions. Because system expansion is applied, the net emissions of CHP power are negative. As a result, we find

that CHP systems for power production avoid significantly more emissions compared to other bioenergy systems. Small-scale biomass combustion-based CHP technologies, which have low power-heat ratios, avoid most emissions. Biogas CHP based on manure and biowastes performs comparable to gasification CHP.

In section 4.5 life cycle emissions as low as 3.8 gCO_2-eq/MJ_{EtOH} were found for corn ethanol, resulting in a emission reduction of 86 gCO_2-eq/MJ. However, the figures also showed that a maximum reduction of 43 gCO_2-eq/MJ is more likely. Larger reductions were only found in Europe for high by-product credits for the conversion process. We do not include the high reduction level in Figure 6-6, and find that sugarcane ethanol (66-81 gCO_2-$eq_{avoided}$/MJ_{EtOH}) performs better compared to corn ethanol. Ethanol production from wheat performs comparable or slightly better than production from corn. The performance of biodiesel depends greatly on the feedstock. The use of soy (Brazil and US) does not result in emission reductions, or only a very small reduction in Brazil (4.4 gCO_2-$eq_{avoided}$/MJ_{diesel}). For rapeseed, a very large range is found (-45 to 77 gCO_2-$eq_{avoided}$/MJ_{diesel}) which is mainly caused by a high emission range found for feedstock cultivation (Table 6-12).

The highest emission reductions are achieved by lignocellulosic ethanol and FT diesel (72-91 gCO_2-$eq_{avoided}$/MJ_{fuel}). According to the projections for 2030 the emissions related to these fuels will decrease even further. However, as the WTW emissions of fossil-based fuels are projected to decrease as well due to improved engine efficiencies (tar sands are not taken into account), the reduction of emissions can become smaller (50-65 gCO_2-$eq_{avoided}$/MJ_{EtOH} and 63-106 gCO_2-$eq_{avoided}$/MJ_{diesel}). In the case of sugarcane ethanol, the increasing amount of electricity sold to the grid will reduce the emissions and the amount of emissions avoided will increase. At the same time, however, the implementation of renewable energy technologies increases and causes a reduction in the credit for exported electricity.

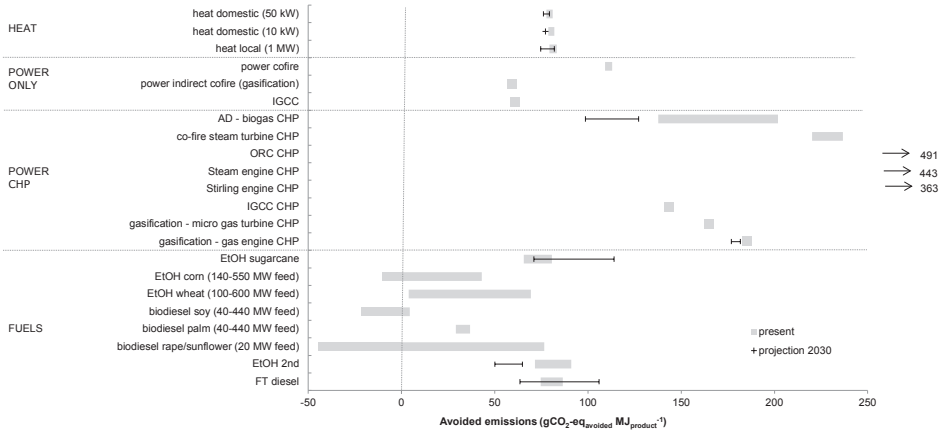

FIGURE 6-6 | Avoided emissions of bioenergy. Light grey bars: current avoided emissions; black line: range of avoided emissions for 2030 based on multiple data sets; black plus sign: emission reduction for 2030 based on one data set.

6.5.2.2 Biomaterials

In section 6.4.5, it was shown that, comparable to biofuels, the life cycle emissions of materials produced from sugarcane are lower than the emissions of corn-based materials. In Figure 6-7, this results in a higher emission reduction for materials from sugarcane. In the projections of Hermann et al. [11], reductions for lignocellulosic biomass are in between sugarcane and corn. Emission reductions are highest for ethylene from sugarcane (4.5-5.3 $tCO_2\text{-}eq_{avoided}/t_{ethylene}$) and lignocellulosic biomass (3.9-5.8 $tCO_2\text{-}eq_{avoided}/t_{ethylene}$), and for olefins production via FT synthesis (4.5-8.2 $tCO_2\text{-}eq_{avoided}/t_{olefins}$). We find a wide range in avoided emissions in the life cycle of PHA. The discussion in section 6.4.5 made clear that a low emission reduction level (or no emission reduction in the case of corn) is more likely than a high reduction.

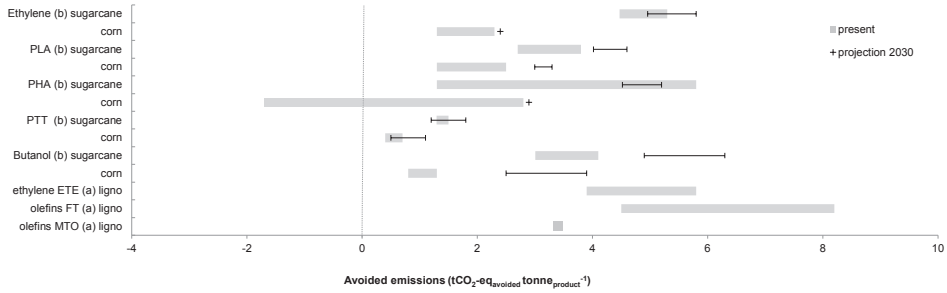

FIGURE 6-7 | Avoided emissions of biomaterials and biochemicals. (a) emission reduction for technologies considered by Ren et al. [14], (b) emission reduction for materials considered by Saygin et al. [4]: projected avoided emission levels for lignocellulosic energy crops lie between projected emission reduction levels of corn and sugarcane. Light grey bars: current avoided emissions; black line: range of avoided emissions for 2030 based on multiple data sets; black plus sign: avoided emissions for 2030 based on one data set.

6.5.2.3 Avoided emissions per hectare

Figure 6-8 presents the emission abatement for energy and materials on a hectare basis. As present heat and power production is based on combustion of wood residues, which have a low yield per hectare, the GHG emission abatement level per hectare is low (0.5-1.7 tCO_2-$eq_{avoided}$/ha). The projection for a gas engine CHP, however, shows that the abatement can increase significantly when lignocellulosic energy crops will be applied (17-20 tCO_2-$eq_{avoided}$/ha). Biofuels and biomaterials avoid most GHG emission per hectare when based on sugarcane (10-12 tCO_2-$eq_{avoided}$/ha for ethanol and 10-14 tCO_2-$eq_{avoided}$/ha for PLA). These are followed by fuels and materials from lignocellulosic energy crops (5-8 tCO_2-$eq_{avoided}$/ha for ethanol and 7-9 tCO_2-$eq_{avoided}$/ha for ethylene).

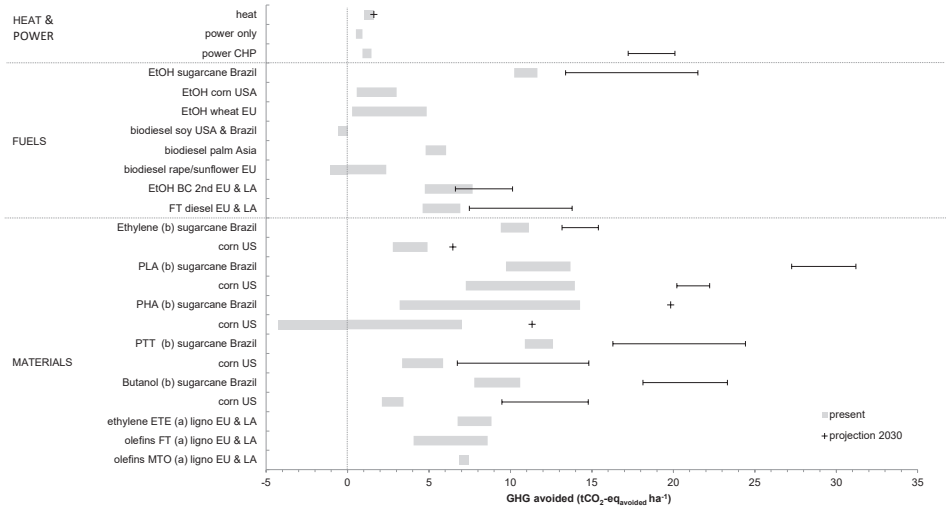

FIGURE 6-8 | Avoided emission per hectare. Materials: (a) emission reduction for technologies considered by Ren et al. [14], (b) emission reduction for materials considered by Saygin et al. [4]. Light grey bars: current avoided emissions; black line: range of avoided emissions for 2030 based on multiple data sets; black plus sign: avoided emissions for 2030 based on one data set.

6.5.3 GHG abatement costs

At the average production costs of fossil-based products (Figure 6-4), the majority of bioenergy products are not economically competitive compared to their fossil reference. On the other hand, Figure 6-6 shows that most value chains avoid emissions compared to their fossil equivalent. This results in positive abatement costs (Figure 6-9). Negative abatement costs are found for low cost Brazilian sugarcane used in large-scale production of ethanol (-65 to -53 \$/tCO_2-$eq_{avoided}$) and most biomaterials (e.g. -60 to -50 \$/tCO_2-$eq_{avoided}$ for ethylene). In the case of corn (USA) and wheat (EU) ethanol, soy and rapeseed

biodiesel (US/Brazil and EU respectively), and corn butanol, PHA and PTT[6], the emission reduction can be very small or bio-based processes can even cause larger emissions compared to the fossil reference. In case of very small emission reductions, the GHG abatement costs become very large (either positive or negative) and are therefore not meaningful in comparison with other biomass value chains. We also find high abatement costs for small-scale gasification technologies (383-548 $/tCO_2$-eq$_{avoided}$). This is a result of the high production costs for power generation compared to the fossil reference.

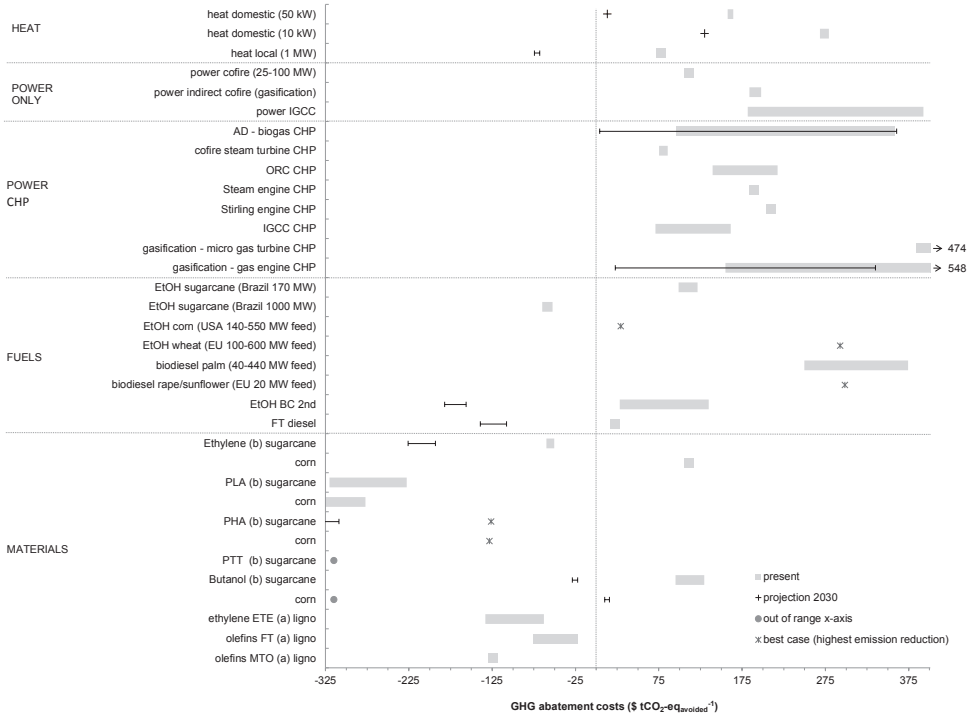

FIGURE 6-9 | GHG abatement costs. Materials: (a) abatement costs for materials considered by Ren et al. [14]; (b) abatement costs for materials considered by Saygin et al. [4]. L*ight grey bars*: current abatement costs; black line: range of abatement costs for 2030 based on multiple data sets; *black plus sign*: abatement cost for 2030 based on one data set; *asterisks*: best case, abatement cost at highest emission reduction: lower or no reduction found as well; *dot*: GHG abatement costs fall out of range x-axis (PTT sugarcane: lower than -325 $/tco_2$-eq$_{avoided}$; butanol corn: higher than 400 $/tco_2$-eq$_{avoided}$). Soy biodiesel and corn-based PTT attain very low and/or negative emissions and were left out of the figure. Fossil reference cost levels applied for bioenergy: heat 12.4 $/GJ, power 15.8 $/GJ, power (CHP) 15.1 $/GJ, gasoline 15.5 $/GJ, diesel 16 $/GJ.

6 For PTT, the relatively small share of bio-based compounds in the polymer limits the emission reduction potential.

As the prices of fossil products are projected to increase in the future, while production costs of bioenergy and biomaterials are likely to decline, more conversion routes can become economically feasible and attain negative abatement costs. The large negative abatement costs of future lignocellulosic biofuels (-182 to -108 $/tCO_2$-eq$_{avoided}$), for example, are not caused by small emission reductions, but by a high cost difference compared to the fossil reference.

6.5.4 Sensitivity analysis

In Figure 6-10 to Figure 6-12, we analyze the sensitivity of the GHG abatement costs for changes in biomass feedstock costs, in fossil resource prices and in the discount rate. We perform the analysis for a selection of biomass value chains that attained negative or relatively low positive abatement costs.

Figure 6-10 shows that the abatement costs of current heat and power production remain positive when the feedstock costs decrease by 44% from 9 to 5 $/GJ. Considering the most cost competitive technologies, the abatement costs of large-scale district heating decline from 79 to 15 $/tCO_2$-eq$_{avoided}$. The abatement costs of cofiring-based power generation drop from 113 to 24 $/tCO_2$-eq$_{avoided}$. Only in the case of future domestic heating, the feedstock costs determine whether the abatement costs become negative or positive. The system attains negative abatement costs (-10 $/tCO_2$-eq$_{avoided}$) when the costs of wood pellets decline from 4.5 to 3 $/GJ.

For all current biofuel and biomaterial value chains considered in Figure 6-10, the point where GHG abatement costs turn from negative to positive is within the feedstock cost range considered. The results suggest that the sensitivity to the feedstock costs is larger for the GHG abatement costs of first-generation biofuels than of second-generation fuels. The abatement costs of sugarcane ethylene are less sensitive to feedstock costs than the abatement costs of sugarcane ethanol. This can be explained by the longer chain of capital costs (more processing steps), due to which the importance of the feedstock delivery costs becomes smaller.

A change in the fossil resource prices has most impact on the production costs of fossil diesel and ethylene. The effect is lowest for power generation from coal (no CHP). However, only when the initial abatement cost level was found to be close to zero, the fossil resource price level determines whether the biomass value chain is cost competiveness to the fossil reference or not (Figure 6-11). This applies to current FT diesel and sugarcane ethylene production, and to future large-scale domestic heating and gas engine CHP.

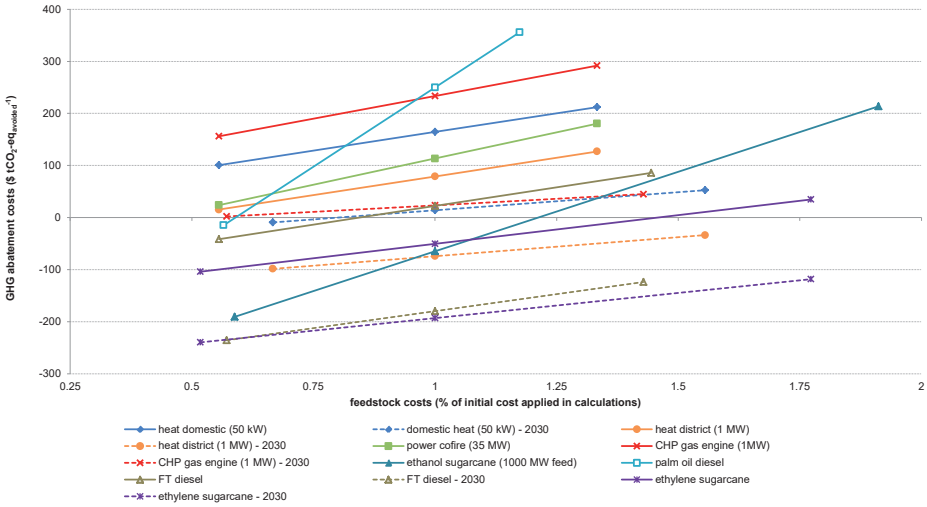

FIGURE 6-10 | GHG abatement costs as a function of the biomass feedstock delivery costs.

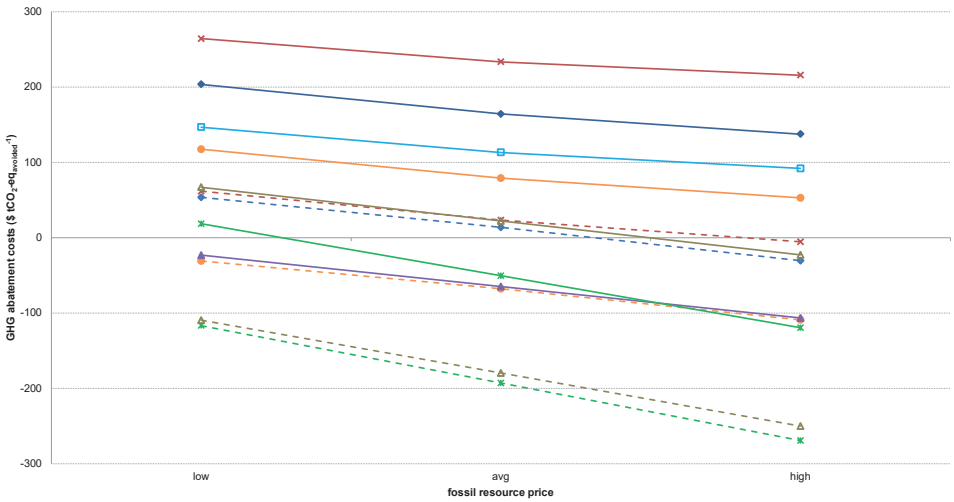

FIGURE 6-11 | GHG abatements costs as a function of the fossil resource prices (see Table 6-16 the applied fossil resource price ranges).

Within the range of 3-10% for the discount rate, only the longer-term abatement costs of domestic heating and gas engine CHP change from negative to positive (Figure 6-12). At a low discount rate of 3%, the abatement costs of current FT diesel production attain 1.6 $\$/tCO_2\text{-}eq_{avoided}$.

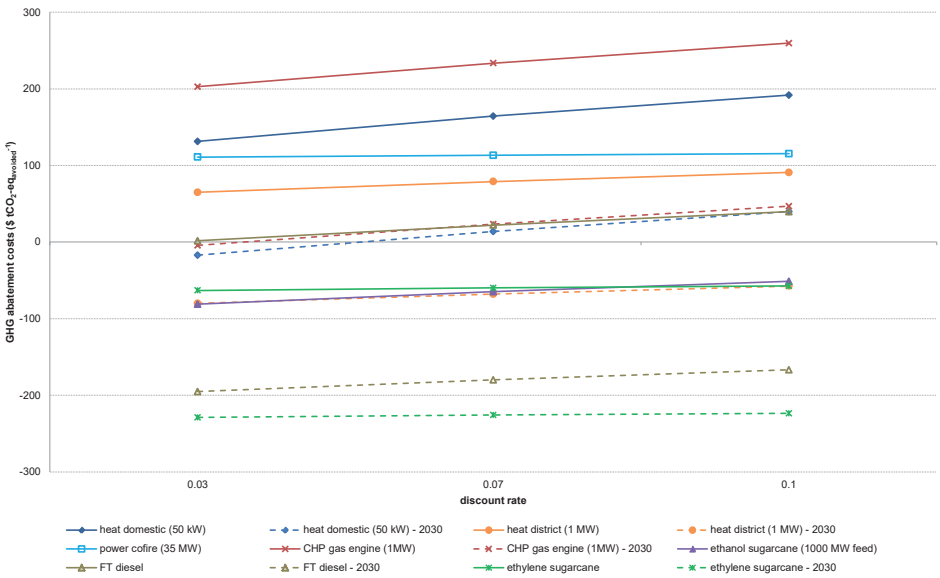

FIGURE 6-12 | GHG abatement costs as a function of the discount rate.

Figure 6-13a-d compares the impact of the different parameters on the GHG abatement costs of four biomass value chains. For heat and power production in the longer term, fossil fuel prices influence the abatement cost most (difference of 28-45 $/tCO_2$-eq$_{avoided}$). The impacts of the discount rate and feedstock costs are a bit smaller (difference of 21-40 $/tCO_2$-eq$_{avoided}$). For ethylene, the range of the fermentable sugar price is large, and has significant impact on the abatement costs (change up to 85 $/tCO_2$-eq$_{avoided}$). The feedstock costs have also the largest influence on the abatement costs of FT diesel (difference of 64 $/tCO_2$-eq$_{avoided}$). The discount rate has a relatively small effect on the abatement costs of FT diesel (change of 17-21 $/tCO_2$-eq$_{avoided}$) and ethylene (change of 3 $/tCO_2$-eq$_{avoided}$).

6.6 DISCUSSION

This study is the first extensive review on various uses of biomass and addresses the status and prospects as well as the economic and environmental performance of biomass value chains. Therefore, this study fills an important gap in current literature. Still, our results are subject to uncertainties due to for example the quality of available data and to the assumptions made. We discuss the most important topics below.

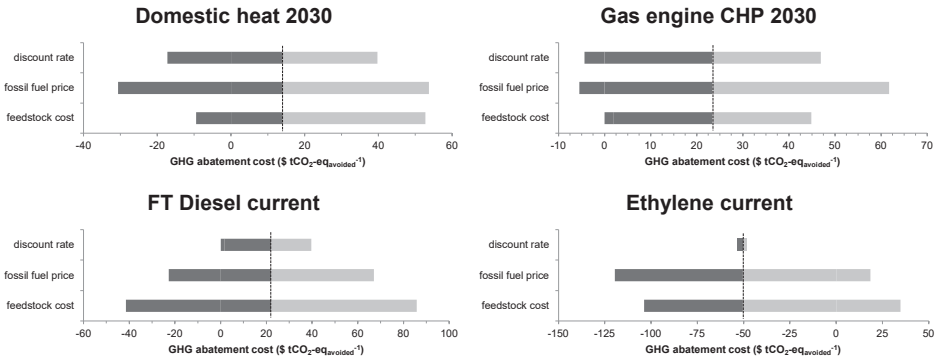

FIGURE 6-13 | Comparison of impact of feedstock cost, fossil fuel price and discount rate on the GHG abatement costs of (a) domestic heating 2030, (b) power generation by gas engine chp 2030, (c) FT diesel current, (d) sugarcane ethylene current. Dotted line indicates abatement level at initial levels of discount rate, fossil fuel price and feedstock cost; lower discount rate, higher fossil fuel price and lower feedstock cost decrease the abatement costs (dark grey) and vice versa (light grey).

6.6.1 Data availability and quality

The levelized production costs and GHG abatement costs were calculated based on cost and emission data from literature. The availability and quality of this data, however, varies. With regard to data availability for bioenergy, the availability of GHG emission data was the factor that limited the amount of value chains for which GHG abatements costs could be calculated. In contrast, for biomaterials, the amount of GHG emission data was larger than the amount of cost data. The lack of cost data for biomaterials limited the selection of value chains to fermentation- and gasification-based routes. In addition, less experience and empirical data is available for second-generation feedstocks than for established feedstock production and supply systems. Still, both costs and emissions of cultivation and logistics have been assessed in literature. New pretreatment technologies like torrefaction, however, are not yet considered in LCA studies [23,157,169].

Concerning the data quality, the level of GHG emissions for agricultural crops, especially for corn and rapeseed, is very case-specific [23]. For conversion to second-generation biofuels, we found some variation in current (and to a lesser extent longer-term) cost estimates between various sources, which cannot be explained by scaling. Finally, the production costs of bio-based and fossil materials calculated in this work varied between sources and from cost estimates in literature. This can mainly be explained by uncertainties in the underlying data. First, the economic assessment in the BREW project [15,93,150] was based on industry data, which are not publically available. To update this assessment, we applied triangulation or extrapolation based on publicly data provided for various crude oil prices. As the capital costs are not publically available, we needed to estimate

the difference between the production costs and the product values. This estimation was based on publicly available data from other sources, which may be inconsistent with the BREW methodology. Also, it was not possible to adapt the capital costs for the long-term cost projections. Secondly, Ren et al. [14] state that the investment costs for the production of biomaterials are rather uncertain. However, they consider the effect of these uncertainties on the total production costs to be small in comparison with the effect of fluctuating feedstock and utility costs. Taking these considerations into account, we consider the data for biomaterials to be less robust than cost data for bioenergy. The production costs and GHG abatement costs found for biomaterials should be considered as an indication of their cost effectiveness.

6.6.2 Technological learning

The data used to calculate longer-term levelized production costs include assumptions on technological learning [54,142]. These assumptions are, however, not always explicitly defined in literature. In section 4.4.5, we discussed that the effects of technological learning are pathway dependent, and depend on both the specific system and on market volume development. For several established biomass value chains, experience curves have been derived in literature. For novel feedstocks and conversion systems, which are still in early phases of production, it is not possible to empirically determine progress ratios. However, the progress ratio is often a sensitive parameter for the determination of future cost levels [170]. Therefore, alternative approaches need to be used to assess cost developments of new systems and technologies.

In addition, in section 4.4.5, we discussed that costs may decrease due to different learning mechanisms, triggered by factors such as R&D expenditures and technical improvements [3,171]. These different factors should be taken into account to make accurate estimates on learning rates and to assess future cost trends. But, the relationship between such factors and cost reductions is complex, and a better understanding of the reasons behind cost declines is still needed [[18,172,173],[174] in:[175]]. Finally, other factors than technological learning may influence costs as well. For example, competition for land and/or biomass feedstocks can cause biomass feedstock prices to go up. Such mechanisms are not included in this study. The sensitivity analysis in section 6.6.4 shows that higher feedstock prices may especially affect the levelized costs and GHG abatement costs of biofuels.

6.6.3 (I)LUC

Land use is an important factor in the production of biomass for energy and materials. Land use and changes in land use as a result of this production, however, can lead to

unintended environmental, ecological and social impacts. One of the main issues discussed recently is the increase of GHG emissions induced by both direct and indirect land use change (DLUC and ILUC) [30]. We did not account for LUC related GHG emissions in our study. DLUC is location specific and depends on previous land use. Its impact can be negative (conversion of peat land forest) or positive (conversion of degraded land). ILUC-induced GHG emissions are market-mediated effects of bioenergy that cause LUC outside the feedstock production area. Various modeling exercises indicate that ILUC can have significant impacts on the total GHG emission balance [30]. Still, there are a lot of uncertainties and shortcomings in these models and there is a debate on how policy could deal with ILUC. Options include using feedstocks with low ILUC risks, such as agricultural and forestry residues and lignocellulosic energy crops (preferably produced on degraded and marginal lands), increasing efficiencies in agricultural crop and livestock production, and applying sustainable land use planning [30].

6.6.4 Allocation of costs and emissions

In our study, we applied system expansion to allocate costs and GHG emissions among the primary product and its by-products. The selected allocation method can, however, have significant impact on the results. Hoefnagels et al. [23] investigated the impact of the allocation method on the GHG balance of biofuels. They show that the allocation method can influence the performance of a production chain. When allocated by mass, for example, the advantage is higher for systems that produce solid co-products than for systems that co-produce heat or electricity, which cannot be allocated on a mass basis. Also, system expansion benefits systems that produce a large amount of co-products, resulting in more by-product credits (see section 6.4.5).

6.6.5 Fossil energy price in bioenergy costs

In the cost calculations for bioenergy, we assumed that all cost parameters are independent of the fossil energy price. However, this is a simplification as fossil energy consumption takes place in every biomass value chain (e.g. during transportation). As the fossil energy prices are hidden in, for example, transportation and O&M costs, it is beyond the scope of our work to take into account how these prices effect our calculations.

6.7 CONCLUSIONS

In this study, we reviewed the status and performance of biomass value chains for heat, power, fuels and materials. Using normalized and harmonized cost and GHG emission data, we calculated and compared current and projected (2030) levelized costs, avoided emissions and GHG abatement costs of different biomass value chains.

Levelized costs

Large-scale biomass-to-*heat* value chains can become cost competitive with fossil reference systems. At a wood pellet price of 9 $\$_{2009}$/GJ, the levelized costs of heat are 15 $\$/GJ_{th}$ for a 100 kW domestic heating system and 18 $\$/GJ_{th}$ for 5 MW district heating (8-20 $\$/GJ_{th}$ for fossil reference systems at 75 $\$_{2009}/bbl_{oil}$). In 2030, these large scale systems also become competitive with cheaper fossil heating systems. At a pellet price of 4.5 $/GJ, the levelized costs are 9 $\$/GJ_{th}$ for the 100 kW domestic heating system and 10 $\$/GJ_{th}$ for 5 MW district heating (13-25 $\$/GJ_{th}$ for fossil reference systems at 110 $\$_{2009}/bbl_{oil}$).
For biobased *power and CHP*, cofiring of wood chips and pellets is currently the most economically feasible option. The costs of direct cofiring of pellets (5-100 MW) are about 28-34 $\$/GJ_e$ (fossil reference 7-20 $\$/GJ_e$). Considering novel technologies, gasification-based biomass value chains, which apply large-scale IGCC and indirect cofiring technologies, have levelized costs of 25-40 $\$/GJ_e$ for near-term commercial production and 11-26 $\$/GJ_e$ in 2030 (fossil reference 13-37 $\$/GJ_e$ in 2030).
With regard to first-generation *biofuels*, large-scale (1000 MW_{feed}) production of Brazilian sugarcane ethanol (in an integrated sugar mill) attains the lowest costs (9-12 $\$/GJ_{etOH}$ compared to 16 $\$/GJ_{gasoline}$ (75 $/bbl)). Biodiesel production costs in the range of 18-49 $\$/GJ_{biodiesel}$ and is nearly competitive in the cases of soy-based biodiesel and renewable diesel. Considering second-generation biofuels from lignocellulosic energy crops, short-term commercial production of thermochemical hydrogen (13-15 $\$/GJ_{hydrogen}$) and methanol (about 12 $\$/GJ_{methanol}$) may be cost competitive to gasoline and diesel. In the longer term, all value chains of second-generation biofuels are projected to become cost competitive and attain levelized production costs from 7-8 $\$/GJ_{hydrogen}$ for hydrogen to 12-15 $\$/GJ_{diesel}$ for FT diesel (22-24 $\$/GJ_{fossil\ fuel}$ for fossil fuels at 110 $/bbl).

The production of *biomaterials* from low cost Brazilian sugarcane (except butanol) is economically feasible. Currently, costs of ethylene, PLA, PHA and PTT are 270-865 $/t lower compared to their fossil reference products (sugar 141 $\$/t_{ferm.sugar}$, oil 75 $\$_{2009}/bbl_{oil}$). In the longer term, this difference increases to about 940-1600 $/t (sugar 141 $\$/t_{ferm.sugar}$, oil 110 $\$_{2009}/bbl_{oil}$). PLA has the best economic potential both today and in the longer term.

Avoided GHG emissions

The best option for domestic and district heating are wood chip and pellet boilers; both today (78-83 $gCO_2\text{-}eq_{avoided}/MJ_{th}$) and in 2030 (75-82 $gCO_2\text{-}eq_{avoided}/MJ_{th}$). Small-scale CHP technologies with low power-heat ratios receive a significant credit for heat and attain the highest GHG abatement levels for power generation. Gasification-based routes, which have higher power-heat ratios, attain abatement levels of 59-186 $gCO_2\text{-}eq_{avoided}/MJ_e$.
Sugarcane ethanol production is the best option amongst first-generation biofuels to abate GHG emissions (66-81 $gCO_2\text{-}eq_{avoided}/MJ_{etOH}$). Higher abatement levels can be

attained by biomass value chains of second-generation bio-ethanol and FT diesel (72-91 gCO_2-$eq_{avoided}$/MJ_{fuel}), specifically from short rotation crops. Sugarcane ethanol and FT diesel are also preferred in the longer term because of increasing emission abatement (71-114 gCO_2-$eq_{avoided}$/MJ_{fuel}). For current biomaterials, sugarcane ethylene avoids most emissions (4.5-5.3 tCO_2-$eq_{avoided}$/$t_{ethylene}$). In the longer term, this is butanol from sugarcane (4.9-6.3 tCO_2-$eq_{avoided}$/$t_{butanol}$). Also FT olefins can attain high abatement levels (4.5-8.2 tCO_2-$eq_{avoided}$/$t_{olefins}$).

GHG abatement costs

Currently, the following biomass value chains have negative abatement costs (i.e. are cost effective and avoid GHG emissions): ethanol (sugarcane, large scale production), ethylene (sugarcane and lignocellulosic feedstock), PLA (sugarcane and corn), and olefins (gasification of lignocellulosic feedstock). Heat and power production have positive abatement costs. For heat, district heating is currently the best option (75-79 \$/tCO_2-$eq_{avoided}$ at a fossil reference cost of 12.4 \$/GJ_{th}). For power generation, this is large-scale cofiring CHP (75-81 \$/tCO_2-$eq_{avoided}$ at 100 MW_e and a reference cost of 15 \$/GJ_e) and near-term commercial power production in a 250 MW IGCC-based CHP plant (71-115 \$/tCO_2-$eq_{avoided}$). Based on limited longer-term projections, the biomass value chains of at least the following applications are expected to achieve negative abatement costs: district heat, (indirect) cofiring, IGCC, lignocellulosic biochemical ethanol and thermochemical FT diesel, and sugarcane butanol.

Overall

At present, the combined economic and environmental performance is best for the production of sugarcane-based ethanol, ethylene and PLA, followed by wood chip combustion in district heating systems and in cofiring power plants. In the longer term, fermentation of low cost sugarcane remains attractive for the production of cost competitive ethanol and materials with high GHG abatement potentials. In addition, advanced conversion technologies can play a key role in producing bio-based products that have both a good economic and environmental performance. Biomass gasification can process various feedstocks and produce a wide range of products including heat, power, fuels and materials. Biochemical conversion of lignocellulosic biomass is promising for the production of both biofuels and biomaterials. Lignocellulosic energy crops can play a significant role in decreasing both the final production costs and life cycle emissions of bioenergy (especially biofuels) and biomaterials. As the resource availability varies across regions, the importance of international trade and of (novel) pretreatment technologies will increase. Finally, the integration of different conversion technologies in biorefineries can maximize the use of all biomass components and improve the economic and environmental performance of the combined value chains.

Although the results show good perspectives for second-generation feedstocks and technologies and for biomaterials, there is still uncertainty about their actual performance. Projections are based on many assumptions, including technological progress and cost developments in biomass feedstock production, supply, and conversion. Technological and economical improvements are both preconditions for large scale commercial implementation of biomass gasification and biochemical conversion of lignocellulosic biomass. The available data for production routes of biomaterials via fermentation and gasification give a good outlook for for example PLA and ethylene, especially in the longer term. Future research should pay extra attention to the economic performance of these and other biomaterials and on how and at what speed cost reductions can be achieved because data availability on this aspect is limited. As technological learning plays an important role in the reduction of costs, this should be an important topic in future research.

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Supplementary material

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References

[1] Harvey M and Pilgrim S. The new competition for land: Food, energy, and climate change. Food Policy 2011;36:S40-S51.

[2] Chum H, Faaij A, Moreira J, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 209-332.

[3] Fischedick M, Schaeffer R, Adedoyin A, et al. Mitigation potential and costs. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 791-864.

[4] Saygın D, Gielen DJ, Draeck M, et al. Assessment of the technical and economic potentials of biomass use for the production of steam, chemicals and polymers. Renewable and Sustainable Energy Reviews 2014;40:1153-1167.

[5] Gnansounou E, Dauriat A, Villegas J, et al. Life cycle assessment of biofuels: Energy and greenhouse gas balances. Bioresource technology 2009;100(21):4919-4930.

[6] Bauen A, Berndes G, Junginger M, et al. Bioenergy - A Sustainable and Reliable Energy Source: A Review of Status and Prospects. IEA Bioenergy: 2009. IEA Bioenergy ExCo:2009:06.

[7] Bain RL. World Biofuels Assessment, Worldwide Biomass Potential: Technology Characterizations. National Renewable Energy Laboratory: Golden, CO, USA; 2007. NREL/MP-510-42467.

[8] Obernberger I, Thek G and Reiter D. Economic Evaluation of Decentralised CHP Applications Based on Biomass Combustion and Biomass Gasification. BIOS Bioenergiesysteme GmbH: Graz, Austria; 2008.

[9] Larson ED. A review of life-cycle analysis studies on liquid biofuel systems for the transport sector. Energy for Sustainable Development 2006;10(2):109-126.

[10] von Blottnitz H and Curran MA. A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. Journal of Cleaner Production 2007;15(7):607-619.

[11] Hermann BG, Blok K, Patel MK. Producing Bio-Based Bulk Chemicals Using Industrial Biotechnology Saves Energy and Combats Climate Change. Environmental science & technology 2007;41(22):7915-7921.

[12] Weiss M, Haufe J, Carus M, et al. A Review of the Environmental Impacts of Biobased Materials. Journal of Industrial Ecology 2012;16:S169-S181.

[13] Ren T and Patel MK. Basic petrochemicals from natural gas, coal and biomass: Energy use and CO2 emissions. Resources, Conservation and Recycling 2009;53(9):513-528.

[14] Ren T, Daniëls B, Patel MK, et al. Petrochemicals from oil, natural gas, coal and biomass: Production costs in 2030–2050. Resources, Conservation and Recycling 2009;53(12):653-663.

[15] Hermann B and Patel M. Today's and tomorrow's bio-based bulk chemicals from white biotechnology. Applied Biochemistry and Biotechnology 2007;136(3):361-388.

[16] Dornburg V, Lewandowski I, Patel M. Comparing the land requirements, energy savings, and greenhouse gas emissions reduction of biobased polymers and bioenergy: An analysis and system extension of life-cycle assessment studies. Journal of Industrial Ecology 2004;7(3-4):93-116.

[17] Saygın D and Patel MK. Renewables for industry: An overview of the opportunities for biomass use. Utrecht University, Group of Science, Technology and Society / Copernicus Institute: Utrecht, Netherlands; 2010.

[18] van den Wall Bake JD, Junginger M, Faaij A, et al. Explaining the experience curve: Cost reductions of Brazilian ethanol from sugarcane. Biomass and Bioenergy 2009;33(4):644-658.

[19] Hettinga WG, Junginger HM, Dekker SC, et al. Understanding the reductions in US corn ethanol production costs: An experience curve approach. Energy Policy 2009;37(1):190-203.

[20] Junginger M, de Visser E, Hjort-Gregersen K, et al. Technological learning in bioenergy systems. Energy Policy 2006;34(18):4024-4041.

[21] Cherubini F, Bird ND, Cowie A, et al. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. Resources, Conservation and Recycling 2009;53(8):434-447.

[22] Cherubini F and Strømman AH. Life cycle assessment of bioenergy systems: State of the art and future challenges. Bioresource technology 2011;102(2):437-451.

[23] Hoefnagels R, Smeets E, Faaij A. Greenhouse gas footprints of different biofuel production systems. Renewable and Sustainable Energy Reviews 2010;14(7):1661-1694.

[24] Cherubini F, Strømman AH, Ulgiati S. Influence of allocation methods on the environmental performance of biorefinery products—A case study. Resources, Conservation and Recycling 2011;55(11):1070-1077.

[25] U.S. Bureau of Labor Statistics. Cunsumer Price Index [Internet: accessed 2011]. Available from: http://www.bls.gov/cpi/#data

[26] OANDA. Historical exchange rates [Internet: accessed 2011]. Available from: http://www.oanda.com/currency/historical-rates/

[27] Bruckner T, Chum H, Jäger-Waldau A, et al. Annex III: Recent Renewable Energy Cost and Performance Parameters. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC special report on renewable energy sources and climate change mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 1001-1022.

[28] EPA. Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis. Environmental Protection Agency: Washington, DC, USA; 2010. EPA-420-R-10-006.

[29] Fritsche UR, Hennenberg K and Huenecke K. The "iLUC Factor" as a Means to Hedge Risks of GHG Emissions from Indirect Land Use Change. Working Paper of the project "Bio-global: Sustainability Standards for internationally traded Biomass". Öko-Institut: Darmstadt, Germany; 2010.

[30] Wicke B, Verweij P, van Meijl H, et al. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 2012;3(1):87-100.

[31] Damen K, van Troost M, Faaij A, et al. A comparison of electricity and hydrogen production systems with CO2 capture and storage—Part B: Chain analysis of promising CCS options. Progress in Energy and Combustion Science 2007;33(6):580-609.

[32] Shen L, Worrell E, Patel M. Present and future development in plastics from biomass. Biofuels, Bioproducts and Biorefining 2010;4(1):25-40.

[33] Bacovsky D. How close are second-generation biofuels? Biofuels, Bioproducts and Biorefining 2010;4(3):249-252.

[34] Sims REH, Mabee W, Saddler JN, et al. An overview of second generation biofuel technologies. Bioresource technology 2010;101(6):1570-1580.

[35] Wijffels RH and Barbosa MJ. An Outlook on Microalgal Biofuels. Science 2010;329(5993):796-799.

[36] Wright A. Global-Bio-Pact Case Study. Socio-Economic Impacts of the Palm oil chain in Indonesia. Greenlight Biofuels Indonesia: 2011.

[37] Sikkema R, Steiner M, Junginger M, et al. The European wood pellet markets: current status and prospects for 2020. Biofuels, Bioproducts and Biorefining 2011;5(3):250-278.

[38] Junginger M, Bolkesjø T, Bradley D, et al. Developments in international bioenergy trade. Biomass and Bioenergy 2008;32(8):717-729.

[39] Fischer G, Hizsnyik E, Prieler S, et al. Biofuels and Food Security. OPEC Fund for International Development (OFID) & International Institute for Applied Systems Analysis (IIASA): Austria; 2009.

[40] Heinimö J and Junginger M. Production and trading of biomass for energy – An overview of the global status. Biomass and Bioenergy 2009;33(9):1310-1320.

[41] Peterson D and Haase S. Market Assessment of Biomass Gasification and Combustion Technology for Small- and Medium-Scale Applications. National Renewable Energy Laboratory: Golden, Colorado, U.S.; 2009.

[42] Institute of Chemical Engineering, TU Wien. BIOBIB - A database for biofuels [Internet: accessed 2011]. Available from: http://www.vt.tuwien.ac.at/biobib/EN/

[43] Uslu A, Faaij APC, Bergman PCA. Pre-treatment technologies, and their effect on international bioenergy supply chain logistics. Techno-economic evaluation of torrefaction, fast pyrolysis and pelletisation. Energy 2008;33(8):1206-1223.

[44] Sikkema R, Junginger M, Pichler W, et al. The international logistics of wood pellets for heating and power production in Europe: Costs, energy-input and greenhouse gas balances of pellet consumption in Italy, Sweden and the Netherlands. Biofuels, Bioproducts and Biorefining 2010;4(2):132-153.

[45] van Sleen P, Vis M, Abban-Mensah I, et al. Global-Bio-Pact Case Study: Socio-economic impacts of second generation conversion technologies in Canada. BTG and Proforest: 2011.

[46] Bergman PCA, Boersma AR, Zwart RWR, et al. Torrefaction for biomass co-firing in existing coal fired power plants: "Biocoal". ECN: Petten, The Netherlands; 2005.

[47] Kiel J. Torrefaction for upgrading biomass into commodity fuel: European developments. European Pellet Conference; March 2; Wels, Austria. The Netherlands: ECN; 2011.

[48] Batidzirai B, Mignot APR, Schakel WB, et al. Biomass torrefaction technology: Techno-economic status and future prospects. Energy 2013;62:196-214.

[49] Bridgwater AV. Review of fast pyrolysis of biomass and product upgrading. Biomass and Bioenergy 2012;38:68-94.

[50] Laird DA, Brown RC, Amonette JE, et al. Review of the pyrolysis platform for coproducing bio-oil and biochar. Biofuels, Bioproducts and Biorefining 2009;3(5):547-562.

[51] Obernberger I and Thek G. Combustion and gasification of solid biomass for heat and power production in Europe - State-of-the-art and relevant future developments. Proc. of the 8th European Conference on Industrial furnaces and boilers; April; Vilamoura, Portugal. 2008.

[52] IEA. Technology Roadmap: Biofuels for transport. International Energy Agency: 2011.

[53] European Bioplastics and Institute for Bioplastics and Biocomposites. Biopolymers production capacity 2011 (by type) [artwork]. 2012. Available from: http://en.european-bioplastics.org/market/market-development/;

[54] EEA. Maximising the environmental benefits of Europe's bioenergy potential. European Environment Agency (EEA): 2008.

[55] McGowin C. Renewable Energy Technical Assessment Guide - TAG-RE: 2008. Electric Power Research Institute (EPRI): Palo Alto, CA, USA; 2008.

[56] Faaij APC. Bio-energy in Europe: changing technology choices. Energy Policy 2006;34(3):322-342.

[57] IEA. Renewables for Heating and Cooling – Untapped Potential. International Energy Agency: Paris, France; 2007.

[58] Bain RL, Denholm P, Heath G, et al. Biopower Technologies. In: Hand MM, Baldwin S, DeMeo E, et al, editors. Renewable Electricity Futures Study. Volume 2, National Renewable Energy Laboratory: Golden, CO, USA; 2012, p. 6.1-6.58.

[59] Kirkels AF and Verbong GPJ. Biomass gasification: Still promising? A 30-year global overview. Renewable and Sustainable Energy Reviews 2011;15(1):471-481.

[60] Hamelinck CN and Faaij APC. Outlook for advanced biofuels. Energy Policy 2006;34(17):3268-3283.

[61] Kurkela E and Kurkela M. Advanced Biomass Gasification for High-Efficiency Power: Publishable Final Activity Report of BiGPower Project. VTT TIEDOTTEITA: Finland; 2009. Available from: http://www.vtt.fi/inf/pdf/tiedotteet/2009/T2511.pdf

[62] Klimantos P, Koukouzas N, Katsiadakis A, et al. Air-blown biomass gasification combined cycles (BGCC): System analysis and economic assessment. Energy 2009;34(5):708-714.

[63] IEA. World Energy Outlook 2010. International Energy Agency: Paris, France; 2010.

[64] Tao L and Aden A. The economics of current and future biofuels. In Vitro Cellular & Developmental Biology - Plant 2009;45(3):199-217.

[65] Cherubini F. The biorefinery concept: Using biomass instead of oil for producing energy and chemicals. Energy Conversion and Management 2010;51(7):1412-1421.

[66] EERE information center. Using Fermentation and Catalysis to Make Fuels and Products: Biochemical Conversion (fact sheet). US Department of Energy, Office of Energy Efficiency & Renewable Energy: 2010.

[67] Hamelinck CN, Hooijdonk Gv, Faaij APC. Ethanol from lignocellulosic biomass: techno-economic performance in short-, middle- and long-term. Biomass and Bioenergy 2005;28(4):384-410.

[68] Biofuels Digest. Advanced Biofuels & Chemicals Project [Database]. Biofuels Digest: 2011. Available from: http://www.biofuelsdigest.com/bdigest/2011/11/16/advanced-biofuels-chemicals-capacity-to-reach-5-11b-gallons-by-2015-207-projects-new-database/

[69] Petersson A and Wellinger A. Biogas upgrading technologies – developments and innovations. IEA Bioenergy: 2009.

[70] Persson M. Evaluation of upgrading techniques for biogas. Lund University: Lund, Sweden; 2003.

[71] Tijmensen MJA, Faaij APC, Hamelinck CN, et al. Exploration of the possibilities for production of Fischer Tropsch liquids and power via biomass gasification. Biomass and Bioenergy 2002;23(2):129-152.

[72] EERE information center. Using Heat and Chemistry to Make Fuels and Power: Thermochemical Conversion (fact sheet). US Department of Energy, Office of Energy Efficiency & Renewable Energy: 2010.

[73] Werpy T and Petersen G. Top Value Added Chemicals from Biomass: Volume I—Results of Screening for Potential Candidates from Sugars and Synthesis Gas. U.S. Department of Energy, Energy Efficiency and Renewable Energy (EERE): 2004.

[74] Eriksson L, Gustavsson L, Hänninen R, et al. Climate change mitigation through increased wood use in the European construction sector—towards an integrated modelling framework. European Journal of Forest Research 2012;131(1):131-144.

[75] Gustavsson L, Pingoud K, Sathre R. Carbon Dioxide Balance of Wood Substitution: Comparing Concrete- and Wood-Framed Buildings. Mitigation and Adaptation Strategies for Global Change 2006;11(3):667-691.

[76] Bolck C, Ravenstijn J, Molenveld K, et al. Biobased Plastics 2012. Wageningen UR Food & Biobased Research: Wageningen; 2011.

[77] Haveren Jv, Scott EL, Sanders J. Bulk chemicals from biomass. Biofuels, Bioproducts and Biorefining 2008;2(1):41-57.

[78] IEA. Tracking Industrial Energy Efficiency and CO2 Emissions. International Energy Agency (IEA): 2007.

[79] Weissermel K and Arpe H-. Industrial Organic Chemistry. 4th ed., Wiley-VCH: Weinheim, Germany; 2003.

[80] MMSA. MMSA Global Methanol Supply and Demand Balance, 2005-2010E [Internet: updated 2010, accessed 23 August 2012]. Available from: http://www.methanol.org/Methanol-Basics/Resources/MMSA-Global-Methanol-Supply-and-Demand.aspx

[81] Holladay JE, Bozell JJ, White JF, et al. Top Value Added Chemicals from Biomass: Volume II—Results of Screening for Potential Candidates from Biorefinery Lignin. U.S. Department of Energy, Energy Efficiency and Renewable Energy (EERE): 2007.

[82] International Fertilizer Industry Association. Production and trade statistics. Urea [Internet: updated 2012, accessed 23 August 2012]. Available from: http://www.fertilizer.org/ifa/HomePage/STATISTICS/Production-and-trade

[83] International Fertilizer Industry Association. Production and trade statistics. Ammonia [Internet: updated 2012, accessed 23 August 2012]. Available from: http://www.fertilizer.org/ifa/HomePage/STATISTICS/Production-and-trade

[84] Bhown AS and Freeman BC. Analysis and Status of Post-Combustion Carbon Dioxide Capture Technologies. Environmental science & technology 2011;45(20):8624-8632.

[85] Sanders JPM, Clark JH, Harmsen J, et al. Process intensification in the future production of base chemicals from biomass. Chemical Engineering and Processing: Process Intensification 2012;51:117-136.

[86] Hermann B. Opportunities for Biomaterials: Economic, environmental and policy aspects along their life cycle [Doctoral dissertation]. Utrecht University: The Netherlands; 2010.

[87] Shen L. Bio-based and Recycled Polymers for Cleaner Production. An assessment of plastics and fibres [Doctoral dissertation]. Utrecht University: The Netherlands; 2011.

[88] SynGest. BioAmmonia™ from Biomass: 100% Organic Nitrogen Fertilizer and Renewable Fuel. Our technology [Internet: updated 2009-2010, accessed 31 August 2011]. Available from: http://www.syngest.com/technology.html

[89] Shen L, Haufe J and Patel MK. Product overview and market projection of emerging bio-based plastics. PRO-BIP 2009. Utrecht University: The Netherlands; 2009.

[90] Oerlikon. The Fiber Year 2008/09. A World Survey on Textile and Nonwovens Industry. OC Oerlikon Corporation AG: 2009. Issue 9.

[91] European Bioplastics and University of Applied Sciences and Arts Hanover. Biopolymers production capacity 2010 by type [artwork]. 2011. Available from: http://en.european-bioplastics.org/market/market-development/

[92] European Bioplastics and University of Applied Sciences and Arts Hanover. Biopolymers production capacity 2015 [artwork]. 2011. Available from: http://www.tricorbraun.com/News/SustainabilityTimesNewsletter/BioplasticsRefillableContainersareTopStories.aspx;

[93] Patel M, Crank M, Dornburg V, et al. Medium and Long-term Opportunities and Risks of the Biotechnological Production of Bulk Chemicals from Renewable Sources - The potential of White Biotechnology: The BREW Project. Utrecht University: The Netherlands; 2006.

[94] Posada JA, Patel AD, Roes L, et al. Potential of bioethanol as a chemical building block for biorefineries: Preliminary sustainability assessment of 12 bioethanol-based products. Bioresour Technol 2013;135:490-499.

[95] IEA-ETSAP and IRENA. Production of bio-ethylene: technology brief. IEA-ETSAP and IRENA: 2013. IEA-ETSAP and IRENA Technology Policy Brief I13.

[96] Chen G and Patel MK. Plastics Derived from Biological Sources: Present and Future: A Technical and Environmental Review. Chemical reviews 2012;112(4):2082-2099.

[97] Tallman MJ and Eng CN. Catalytic routes to olefins. AIChE Spring National Meeting; 2008 April 7-10; New Orleans, Louisiana, USA.

[98] Braskem. Green Products - Green Polypropylene (Green PP) [Internet: accessed December 11 2012]. Available from: http://www.braskem.com.br/site.aspx/green-products-USA

[99] Plasteurope.com. METABOLIX: Telles bioplastics jv with Archer Daniels Midland to end / Metabolix seeks new partners [Internet: updated 16 January 2012, accessed 2 August 2012]. Available from: http://www.plasteurope.com/news/detail.asp?id=221297

[100] ICIS Green Chemicals. Metabolix, ADM cut bioplastic ties [Internet: updated 13 January 2012, accessed 2 August 2012]. Available from: http://www.icis.com/blogs/green-chemicals/2012/01/metabolix-adm-cut-bioplastic-t.html

[101] Lammens TM, Potting J, Sanders JP, et al. Environmental comparison of bio-based chemicals from glutamic acid with their petrochemical equivalents. Environmental science & technology 2011;45(19):8521-8528.

[102] Brehmer B, Bals B, Sanders J, et al. Improving the corn-ethanol industry: Studying protein separation techniques to obtain higher value-added product options for distillers grains. Biotechnology and bioengineering 2008;101(1):49-61.

[103] ICIS. Uncertain future for biodiesel-based glycerin. Regional issues concerning biodiesel complicate the global market for glycerin [Internet: updated 2010, accessed June 28 2012]. Available from: http://www.icis.com/Articles/2010/10/04/9397834/Uncertain-future-for-biodiesel-based-glycerin.html

[104] Endres H- and Siebert-Raths A. Engineering Biopolymers. Markets, Manufacturing, Properties and Applications. Hanser Publications: Munich, Germany; 2011.

[105] Torres Galvis HM, Bitter JH, Khare CB, et al. Supported Iron Nanoparticles as Catalysts for Sustainable Production of Lower Olefins. Science 2012;335(6070):835-838.

[106] BioMCN. Process [Internet: accessed 2 August 2012]. Available from: http://www.biomcn.eu/our-product/process.html

[107] Ren T. Petrochemicals from Oil, Natural Gas, Coal and Biomass: Energy use, Economics and Innovation [Doctoral dissertation]. Utrecht University: The Netherlands; 2009.

[108] Alonso DM, Bond JQ, Dumesic JA. Catalytic conversion of biomass to biofuels. Green Chemistry 2010;12(9):1493-1513.

[109] de Jong E, Langeveld H and van Ree R. IEA Bioenergy Task 42 Biorefinery [Internet: accessed 30 August 2011]. Available from: http://www.iea-bioenergy.task42-biorefineries.com/publications/brochures/

[110] de Jong E, Higson A, Walsh P, et al. Bio-based chemicals. Value added products from biorefineries. IEA Bioenergy task 42 Biorefinery: 2012.

[111] Bozell JJ and Petersen GR. Technology development for the production of biobased products from biorefinery carbohydrates-the US Department of Energy's "Top 10" revisited. Green Chemistry 2010;12(4):539-554.

[112] BIOREF-INTEG. Development of advanced biorefinery schemes to be integrated into existing industrial fuel producing complexes. ECN: The Netherlands; 2010.

[113] Laser M, Larson E, Dale B, et al. Comparative analysis of efficiency, environmental impact, and process economics for mature biomass refining scenarios. Biofuels, Bioproducts and Biorefining 2009;3(2):247-270.

[114] FAOSTAT. Production, Crops, Yield [Internet: updated 2011, accessed 17 August 2011]. Available from: http://faostat.fao.org/site/567/default.aspx#ancor

[115] Weyer K, Bush D, Darzins A, et al. Theoretical Maximum Algal Oil Production. BioEnergy Research 2010;3(2):204-213.

[116] Smeets EMW, Faaij APC, Lewandowski IM, et al. A bottom-up assessment and review of global bio-energy potentials to 2050. Progress in Energy and Combustion Science 2007;33(1):56-106.

[117] de Wit M, Londo M, Faaij A. Productivity developments in European agriculture: Relations to and opportunities for biomass production. Renewable and Sustainable Energy Reviews 2011;15(5):2397-2412.

[118] Jaggard KW, Qi A, Ober ES. Possible changes to arable crop yields by 2050. Philosophical Transactions of the Royal Society B: Biological Sciences 2010;365(1554):2835-2851.

[119] FAO, IFAD, IMF, et al. Price Volatility in Food and Agricultural Markets: Policy Responses. 2011.

[120] Kindred D, Sylvester-Bradley R, Garstang J, et al. Anticipated and potential improvements inland productivity and increased agricultural inputs with intensification. ADAS UK Ltd: UK; 2008. Version 3.4.

[121] Ericsson K, Rosenqvist H, Nilsson LJ. Energy crop production costs in the EU. Biomass and Bioenergy 2009;33(11):1577-1586.

[122] Franke B, Reinhardt G, Malavelle J, et al. Global Assessments and Guidelines for Sustainable Liquid Biofuel Production in Developing Countries. A GEF Targeted Research Project. Heidelberg/Paris/Utrecht/Darmstadt; 2012.

[123] de Wit M and Faaij A. European biomass resource potential and costs. Biomass and Bioenergy 2010;34(2):188-202.

[124] Hamelinck CN, Suurs RAA, Faaij APC. International bioenergy transport costs and energy balance. Biomass and Bioenergy 2005;29(2):114-134.

[125] Wicke B, Smeets E, Watson H, et al. The current bioenergy production potential of semi-arid and arid regions in sub-Saharan Africa. Biomass and Bioenergy 2011;35(7):2773-2786.

[126] Batidzirai B, Faaij APC, Smeets E. Biomass and bioenergy supply from Mozambique. Energy for Sustainable Development 2006;10(1):54-81.

[127] Tharakan PJ, Volk TA, Lindsey CA, et al. Evaluating the impact of three incentive programs on the economics of cofiring willow biomass with coal in New York State. Energy Policy 2005;33(3):337-347.

[128] Esteban LS and Carrasco JE. Biomass resources and costs: Assessment in different EU countries. Biomass and Bioenergy 2011;35(Supplement 1):S21-S30.

[129] Ipeadata. Production historical series: Área colhida & Produção de cana-de-açúcar - cana-de-açúcar [Internet: updated 2011, accessed 16 August 2011]. Available from: http://www.ipeadata.gov.br/Default.aspx

[130] van Dam J, Faaij APC, Lewandowski I, et al. Biomass production potentials in Central and Eastern Europe under different scenarios. Biomass and Bioenergy 2007;31(6):345-366.

[131] Harsono SS, Prochnow A, Grundmann P, et al. Energy balances and greenhouse gas emissions of palm oil biodiesel in Indonesia. GCB Bioenergy 2011;4(2):213-228.

[132] Hoogwijk M, Faaij A, de Vries B, et al. Exploration of regional and global cost–supply curves of biomass energy from short-rotation crops at abandoned cropland and rest land under four IPCC SRES land-use scenarios. Biomass and Bioenergy 2009;33(1):26-43.

[133] de Wit M, Junginger M, Faaij A. Learning in dedicated wood production systems: Past trends, future outlook and implications for bioenergy. Renewable and Sustainable Energy Reviews 2013;19:417-432.

[134] Gnansounou E and Dauriat A. Techno-economic analysis of lignocellulosic ethanol: A review. Bioresource technology 2010;101(13):4980-4991.

[135] U.S. Department of Energy. U.S. Billion-ton update: Biomass Supply for a Bioenergy and Bioproducts Industry. Oak Ridge National Laboratory: Oak Ridge, TN, USA; 2011. ORNL/TM-2011/224.

[136] van Dam J, Faaij APC, Hilbert J, et al. Large-scale bioenergy production from soybeans and switchgrass in Argentina: Part A: Potential and economic feasibility for national and international markets. Renewable and Sustainable Energy Reviews 2009;13(8):1710-1733.

[137] Fischer G and Schrattenholzer L. Global bioenergy potentials through 2050. Biomass and Bioenergy 2001;20(3):151-159.

[138] Weiss M, Dittmar L, Junginger M, et al. Market diffusion, technological learning, and cost-benefit dynamics of condensing gas boilers in the Netherlands. Energy Policy 2009;37(8):2962-2976.

[139] Karlsson Å and Gustavsson L. External costs and taxes in heat supply systems. Energy Policy 2003;31(14):1541-1560.

[140] Milieu Centraal. Nieuwe cv of combiketel kopen. Prijsvergelijk hr-ketel [Internet: updated 2011, accessed 2012]. Available from: http://www.milieucentraal.nl/themas/energie-besparen/verwarmen/centrale-verwarming/nieuwe-cv-ketel

[141] Nederlandse Installatie Maatschappij. CV prijslijst [Internet: updated 2012, accessed 2012]. Available from: http://www.nederlandseinstallatiemaatschappij.nl/prijslijst.html

[142] Hamelinck CN, Faaij APC, den Uil H, et al. Production of FT transportation fuels from biomass; technical options, process analysis and optimisation, and development potential. Energy 2004;29(11):1743-1771.

[143] van den Broek M, Veenendaal P, Koutstaal P, et al. Impact of international climate policies on CO2 capture and storage deployment: Illustrated in the Dutch energy system. Energy Policy 2011;39(4):2000-2019.

[144] IEA Bioenergy. Potential Contribution of Bioenergy to the World's Future Energy Demand. IEA Bioenergy: 2007. ExCo 2007:02.

[145] IEA and NEA. Projected costs of generating electricity. OECD, IEA and NEA: France; 2010.

[146] Obernberger I and Thek G. Techno-Economic evaluation of selected decentralised CHP plants based on biomass combustion in IEA partner countries. BIOS BIOENERGYSYSTEME GmbH: Graz, Austria; 2004.

[147] Macedo IC, Seabra JEA, Silva JEAR. Green house gases emissions in the production and use of ethanol from sugarcane in Brazil: The 2005/2006 averages and a prediction for 2020. Biomass and Bioenergy 2008;32(7):582-595.

[148] Beer T and Grant T. Life-cycle analysis of emissions from fuel ethanol and blends in Australian heavy and light vehicles. Journal of Cleaner Production 2007;15(8-9):833-837.

[149] Fulton L, Howes T and Hardy J. Biofuels for Transport - An International Perspective. Organization for Economic Cooperation and Development and International Energy Agency: Paris, France; 2004.

[150] Hermann BG, Dornburg V, Patel MK. 13 Environmental and Economic Aspects of Industrial Biotechnology. In: Soetaert W, Vandamme EJ, editors. Industrial Biotechnology: Sustainable Growth and Economic Success. Wiley-VCH: 2010, p. 433-455.

[151] Seabra JEA, Macedo IC, Chum HL, et al. Life cycle assessment of Brazilian sugarcane products: GHG emissions and energy use. Biofuels, Bioproducts and Biorefining 2011;5(5):519-532.

[152] Berghout NA. Technological Learning in the German Biodiesel Industry: An Experience Curve Approach to Quantify Reductions in Production Costs, Energy Use and Greenhouse Gas Emissions [Master thesis]. Utrecht University: The Netherlands; 2008.

[153] Junginger M, Faaij A, Björheden R, et al. Technological learning and cost reductions in wood fuel supply chains in Sweden. Biomass and Bioenergy 2005;29(6):399-418.

[154] Goldemberg J and Johansson TB. World Energy Assessment Overview: 2004 Update. United Nations Development Programme (UNEP), Bureau for Development Policy: New York, USA; 2004.

[155] IEA. Experience Curves for Energy Technology Policy. International Energy Agency: Paris, France; 2000.

[156] NREL. Biopower Results – Life Cycle Assessment Review [Internet: updated 2012, accessed 24 August 2012]. Available from: http://www.nrel.gov/analysis/sustain_lca_bio.html

[157] Fritsche UR, Rausch L and Schmidt K. Life Cycle Analysis of GHG and Air Pollutant Emissions from Renewable and Conventional Electricity, Heating, and Transport Fuel Options in the EU until 2030. Öko-Institut e.V.: Darmstadt, Germany; 2009.

[158] Knoef HAM. Chapter 1. Introduction. In: Knoef HAM, editor. Handbook Biomass Gasification. 1st ed., BTG Biomass Technology Group: The Netherlands; 2005.

[159] Kaliyan N, Morey RV, Tiffany DG. Reducing life cycle greenhouse gas emissions of corn ethanol by integrating biomass to produce heat and power at ethanol plants. Biomass and Bioenergy 2011;35(3):1103-1113.

[160] EIA. International energy statistics: Electricity prices [Internet: updated 2010, accessed July, 27 2011]. Available from: http://www.eia.gov/emeu/international/electricityprice.html

[161] EIA. International energy statistics: Electricity consumption [Internet: accessed July, 27 2011]. Available from: http://www.eia.gov/cfapps/ipdbproject/IEDIndex3.cfm?tid=2&pid=2&aid=2

[162] BP. BP Statistical Review of World Energy June 2011. BP: London, UK; 2011. Available from: http://www.bp.com/assets/bp_internet/globalbp/globalbp_uk_english/reports_and_publications/statistical_energy_review_2011/STAGING/local_assets/pdf/statistical_review_of_world_energy_full_report_2011.pdf

[163] JEC - Joint Research Centre-EUCAR-CONCAWE collaboration. Well-to-wheels Analysis of Future Automotive Fuels and Powertrains in the European Context. Version 3c. WTW APPENDIX 1: Summary of WTW Energy and GHG balances. European Commission, Joint research Centre, Institute for Energy and Transport: 2011. EUR 24952 EN - 2011.

[164] International Energy Agency (IEA). Transport. In: IEA, editor. Energy Technology Perspectives. International Energy Agency: Paris, France; 2010.

[165] Pickrell D. Fuel options for reducing greenhouse gas emissions from motor vehicles. U.S. Department of Transportation, John A. Volpe National Transportation Systems Center: Cambridge, MA, USA; 2003. DOT-VNTSC-RSPA-03-03.

[166] Index mundi. Sugar monthly price. January 2002-December 2011 [Internet: updated 2012, accessed 2012]. Available from: http://www.indexmundi.com/commodities/?commodity=sugar&months=120

[167] National Research Council. 4 Economics of Renewable Electricity. In: America's Energy Future Panel on Electricity from Renewable Resources, National Research Council, editor. Electricity from Renewable Resources: Satus, prospects, and impediants. The National Academies Press: Washington, DC, USA; 2010.

[168] IEA-ETSAP and IRENA. Production of bio-methanol: technology brief. IEA-ETSAP and IRENA: 2013. IEA-ETSAP and IRENA Technology Policy Brief I08.

[169] Ciolkosz D and Wallace R. A review of torrefaction for bioenergy feedstock production. Biofuels, Bioproducts and Biorefining 2011;5(3):317-329.

[170] de Wit M, Junginger M, Lensink S, et al. Competition between biofuels: Modeling technological learning and cost reductions over time. Biomass and Bioenergy 2010;34(2):203-217.

[171] IEA. Energy technology perspectives 2010. Scenarios and Strategies to 2050. International Energy Agency: Paris, France; 2010.

[172] Sagar AD and van der Zwaan B. Technological innovation in the energy sector: R&D, deployment, and learning-by-doing. Energy Policy 2006;34(17):2601-2608.

[173] Junginger M, Suurs R, Verbong G, et al. Putting experience curves in context; links to and between technology development, market diffusion, learning mechanisms and systems innovation theory. In: Junginger M, van Sark W, Faaij A, editors. Technological Learning in the Energy Sector: Lessons for Policy, Industry and Science. Edward Elgar Publishing Limited: Cheltenham, UK; Northampton, MA, USA; 2010, p. 36-47.

[174] Mukora A, Winskel M, Jeffrey HF, et al. Learning curves for emerging energy technologies. Proceedings of the Institution of Civil Engineers - Energy 2009;162(4):151-159.

[175] Wiser R, Yang Z, Hand M, et al. Wind energy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011.

CHAPTER 7

Summary and Conclusions

7.1 RESEARCH CONTEXT

Currently, fossil fuels (oil, coal and gas) account for a large share of the current global primary energy supply. However, this large scale fossil fuel consumption cannot be sustained in the long run. This is because of the finiteness of fossil resources and the high contribution of fossil fuels to anthropogenic GHG emissions [1]. Therefore, alternative resources for energy and fossil fuel-derived materials need to be found. Biomass is considered an important alternative as its applications are versatile, including the production of heat, power, liquid biofuels, biochemicals and biomaterials. In 2010, modern uses of biomass for heat, electricity and biofuels amounted to approximately 22 EJ_p [2], while biomass demand for novel synthetic organic materials is low (0.6 EJ) [3]. To attain a low carbon supply of energy and materials, the required contribution of biomass to the total primary energy supply is estimated to lay between 100-200 EJ_p yr^{-1} in 2050 [3-9], increasing to possibly 300 EJ towards 2100 [10]. At the same time, the IPCC estimates the global technical biomass supply potential to range from 100 to 300 EJ_p in 2050 [11,12]. Thus, to mitigate climate change and to avoid competition between biomass feedstocks for food, feed, energy and materials, large amounts of biomass need to be mobilized. In addition, it is important to deploy biomass value chains that are most promising for producing heat, power, fuels and materials in terms of their technological, economic and environmental performance. However, the recent increase in global biomass production for modern bioenergy and biomaterial purposes has also raised concerns about the role biomass can actually play in mitigating climate change. One of the main topics of concern is unwanted land use change (LUC), and especially indirect lands use change (ILUC). Therefore, to mobilize large amounts of biomass while reducing GHG emissions and mitigating climate change, it is important to investigate how ILUC can be mitigated or prevented [13].

Agricultural intensification is considered an important measure for making future surplus agricultural land available for bioenergy crop production, mitigating ILUC and thereby improving the GHG balances of biomass value chains. In addition, intensification in crop and livestock production may also contribute to improving the environmental performance of the agricultural sector itself. Many studies have investigated the effects of agricultural intensification on the availability of surplus agricultural land and biomass potentials. Various studies have also analyzed the GHG balance and other environmental impacts of agricultural intensification and biomass production, and/or the environmental and economic performance of biomass value chains. However, more research is needed on the regional possibilities for agricultural intensification, the impacts of different pathways for agricultural intensification, and the comparative performance over time of biomass value chains for energy and materials. Regional possibilities for agricultural

intensification include the rates at which intensification may take place and the influence of technological, economic and institutional factors on these rates. Because livestock is a major factor in agricultural land use and relatively poorly studied, it is especially needed to investigate the possibilities for livestock intensification and their impact on the regional availability of surplus agricultural land and biomass potentials. In addition, it needs to be understood better how different agricultural intensification pathways effect the GHG and environmental performance of both the agricultural and biomass sector. Finally, the comparison of biomass value chains for energy and materials should include both the economic and environmental performance of these chains.

7.1.1 Aim and research questions

This thesis examined the regional potentials, the environmental impacts and economic performance of different pathways for agricultural intensification, biomass production and biomass use. To this end, the following research questions were formulated:

1. What are regional possibilities for agricultural intensification and what are their impacts on the demand for agricultural land and biomass potentials?
2. What is the effect of different agricultural intensification pathways and biomass production on GHG emissions and the environment?
3. Which biomass value chains are preferred today as well as in the future with regard to their economic performance and GHG balance?

The research questions are addressed in Chapters 2 to 6 as shown in Table 7-1. Chapter 2 analyzed the pace and direction of historical yield developments (1961-2010) for various crops and livestock products in seven countries in different world regions, and examined the technological, economic and institutional driving factors behind these developments. In addition, this chapter discussed how yield projections are defined in models that assess biomass potentials and impacts of biomass production, and how these projections can be improved based on the findings of the historical analysis. Chapter 3 investigated two pathways for intensification of cattle production systems and compares the impacts of these two pathways on farm gate GHG emissions, land occupation and LUC-related emissions in nine world regions. Chapter 4 examined the potentials for agricultural intensification and four other measures to mitigate ILUC in the province of Lublin, Poland. In addition, it quantified how much miscanthus-based ethanol can be produced in this region with a low risk of causing ILUC. Chapter 5 assessed how the implementation of agricultural intensification and other ILUC mitigation measures influences the GHG balance and other environmental impacts of agricultural and biofuel production in Lublin province. It specifically investigated the effects of three pathways for agricultural intensification in terms of sustainability (e.g. with regard to nutrient use efficiency and tillage practices).

Chapter 6 reviewed the status and prospects of biomass value chains for heat, power, fuels and materials, assessed their current and long-term levelized production costs and avoided emissions, and compared their greenhouse gas abatement costs. The results of each chapter are summarized in the next section.

TABLE 7-1 | Overview of thesis chapters and the research question(s) addressed in them.

Chapter	Region(s)	Research question		
		1	2	3
Chapter 2: Assessment of driving factors for yield and productivity developments in crop and cattle production as key to increasing sustainable biomass potentials	Australia, Brazil, China, India, USA, Zambia, Zimbabwe	x		
Chapter 3: Intensification pathways for beef and dairy cattle production systems: Impacts on GHG emissions, land occupation and land use change	Europe, Brazil, North America (detailed bottom-up data) 9 world regions (disaggregated global data and model results)	x	x	
Chapter 4: Bioethanol potential from miscanthus with low ILUC risk in the province of Lublin, Poland.	Lublin (Poland)	x		
Chapter 5: GHG emissions and other environmental impacts of ILUC mitigation	Lublin (Poland)		x	x
Chapter 6: Competing uses of biomass: Assessment and comparison of the performance of bio-based heat, power, fuels and materials	Global		x	x

7.2 SUMMARY OF THE RESULTS

Chapter 2 addresses research question 1 by assessing the pace and direction of historical yield developments between 1961 and 2010 as well as their driving factors for five major crops, beef and cow milk in Australia, Brazil, China, India, USA, Zambia and Zimbabwe. In addition, it explores how yield projections are defined in models that assess biomass potentials and impacts, and how these projections can be improved based on the findings of the historical analysis. The analysis of historical yield trends and yield growth rates was based on statistical data. Temporal shifts were identified for each product on country level and differences between products within a country were described. Explanations were sought by comparing the observed changes with technological/management, economic, and institutional developments in the country. The study shows that historical yield growth (especially of crops) has often followed a linear trend. Average yield growth rates over the investigated period ranged in most cases between 0.7-1.6% yr^{-1} for crops, 1.0-1.5% yr^{-1} for milk and 0.4-0.8% yr^{-1} for beef (all relative to 2010). In all cases, yields and yield growth rates have fluctuated in varying degrees. Large fluctuations were found for crops especially when driving factors changed strongly (e.g. agricultural and trade

policies in Zambia). Also, in each case, the analysis revealed periods during which yields improved at a higher or lower rate compared to the long-term average. Periods of high yield growth, e.g. 8.5% yr^{-1} for soybean in Zimbabwe in the 1970s, show that relatively fast improvements can be attained in cases where the yield gap is large. Such significant improvements can be realized especially under favorable conditions with regard to well-functioning markets and governance that stimulate improvements in agricultural technology and management. The role of different drivers in yield developments is found to be region specific. Yet, supporting agricultural policies have played an important role in increasing yields in all countries, especially for agricultural crops. In cattle production, a key factor was the growing importance of national and export markets for commercial beef and milk production. Current models that assess biomass potentials and impacts only take into account one or a limited number of endogenous factors which influence yields. This chapter shows that, in order to improve the models and thereby our understanding of potential future pathways for agricultural yield developments and for sustainable biomass production, models should take into account more, often regionally specific drivers, yield gaps and (potential) policy pathways.

Chapter 3 addresses research questions 1 and 2 by assessing two intensification pathways in beef and dairy cattle production in nine world regions, and comparing their impact on GHG emissions, land occupation and LUC-related emissions. The pathways include i) incremental intensification within the existing production system (where it is distinguished between pasture-based, mixed and industrial production systems), and ii) transformational intensification by moving from one system to another. First, a review was conducted of bottom-up studies on farm gate emissions from dairy production in Europe and beef production in North America and Brazil. Then, a global data set on GHG emissions from cattle production was used to extrapolate the findings from this review to other world regions. Finally, the Global Biosphere Management Model (GLOBIOM) model was applied to perform a global assessment of land occupation and LUC-related emissions in all nine world regions. For dairy in Europe, farm gate emission reductions of 1%-14% are found for intensification within one system and 2%-26% for system transitions. Moving from mixed to industrial production either decreases emissions by 8%-14% or increases the GHG balance depending on, for example, the reduction in emissions due to improved feed quality or the increase in emissions related to manure management. Whether the influence on the overall GHG balance is larger for intensification within one system or for system transitions depends on the specific design of the initial and final production systems. This also applies to other developed regions. In developing countries, the differences within and between pasture-based and mixed systems are more significant and there is a large potential to reduce emissions by intensification within the pasture-based system. The additional GHG reduction potential of a transition from pasture-based

to mixed and industrial production is limited. Also, emission reductions of intensification within the mixed system are smaller compared to the pasture-based system. For beef production in Brazil, the emission reduction potential for intensification within the pasture-based system could exceed 50% of the original emissions when changing from extensive, natural grass-based pastures to improved pastures. Intensification of mixed systems in Canada reduces emissions by 6%-7%. Moving from pasture-based to mixed production in the USA decreases emissions by 16% and moving from mixed to industrial production results in an emission reduction of 9%. In every region, the difference in emissions within and between production systems is larger compared to dairy production. This implies that intensification within pasture-based beef production systems (through adoption of best practises available) cannot only attain considerable GHG reductions in developing regions but also in some developed regions. Also, the additional GHG reduction potentials of transitions from pasture-based to mixed systems compared to intensification within pasture-based systems, as well as the GHG mitigation potentials of intensification within mixed systems are considered larger for beef than for dairy. Although both the dairy and beef sector can often attain significant farm gate emission reductions through intensification within pasture-based systems, the transition to mixed systems is important to reduce land occupation and LUC-related emissions. LUC mitigation is considered to be the most important GHG mitigation strategy for cattle production in Sub-Saharan Africa and Latin America, where LUC-related emissions account for 20% to more than 50% of the total GHG balance. Regarding dairy, moving from pasture-based to mixed production could reduce the LUC-related emissions by more than 50% in both regions. For beef, the reduction could be 20% in Latin America and 50% in Sub-Saharan Africa. Important strategies to reduce both farm gate and LUC-related emissions include increasing the productivity of grassland and cropland and increasing the animal productivity through improved feed quality.

Chapter 4 addresses research question 1 by assessing the production potential of miscanthus-based bioethanol with a low risk of causing ILUC by implementing agricultural intensification and four other ILUC mitigation measures in the Polish province of Lublin in 2020. It combined a top-down approach to define a reference scenario for agricultural production and land use in Lublin in 2020 and a bottom-up approach to calculate how the ILUC mitigation strategies can contribute to the availability of surplus agricultural land on which biomass can be produced without causing undesired LUC. The total potential of the ILUC mitigation measures was investigated for a low, medium and high scenario. In 2020, a total area of 196 to 818 thousand hectare agricultural land, equal to 11% to 47% of the total agricultural area in Lublin, could be made available for biomass production by realizing above-baseline yield developments or agricultural intensification (95-413 thousand ha), increased food chain efficiencies (9-30 thousand ha) and biofuel feedstock

production on under-utilized lands (92-375 thousand ha). Although the surplus land area available in 2020 is potentially very large, a limited area of 203 to 269 thousand hectare (12-15% of the total agricultural area) is considered to be legally available and biophysically suitable for miscanthus production based on the criteria for protecting high conservation areas and minimum requirements for land suitability. As a result, in all scenarios, the amount of surplus land that could be used for miscanthus production is restricted. The limitation on land use is mainly caused by the suitability criteria and especially the sensitivity of miscanthus to water stress. However, the land suitability was assessed based on a limited number of criteria like the minimum ground water level, and should be analyzed in more detail. Depending on the productivity of the bioethanol value chain, the low-ILUC-risk bioethanol production potential ranges from 12 to 35 PJ yr^{-1} (522 to 1,479 million liter yr^{-1}). For comparison, the national Polish target for 2nd generation bioethanol consumption in 2020 is almost 9 PJ, and the total biofuel target in 2020 is 60 PJ. The bioethanol production potential, however, is the technical potential that accounts only for key environmental aspects such as the protection of high conservation value areas. Although the sustainable implementation potential may be lower, the province of Lublin could play a key role in achieving this target and help Poland even become an exporter of second generation bioethanol. Governance and policies on planning and implementing ILUC mitigation are considered vital for realizing a significant bioenergy potential with low ILUC risk. One important aspect in this regard is monitoring the risk of ILUC and the implementation of ILUC mitigation measures. Key parameters for monitoring are changes in land use, land cover and crop yields.

Chapter 5 addresses research questions 2 and 3 by assessing how the implementation of agricultural intensification and other ILUC mitigation measures, combined with the production of biofuel influences the net GHG balance and other environmental impacts of agricultural and biofuel production in the province of Lublin in 2020. It conducted a detailed analysis of the impacts for three ILUC mitigation scenarios, representing a low, medium and high miscanthus-based ethanol production potential, and for three agricultural intensification pathways in terms of sustainability. Generally, the ILUC mitigation scenarios attain lower net annual emissions compared to the reference scenario that excludes ILUC mitigation and bioethanol production. However, the reduction potential significantly depends on the intensification pathway considered. For example, in the moderate ILUC mitigation scenario, the net annual GHG emissions in the case study are 2.3 $MtCO_2$-eq yr^{-1} (1.8 tCO_2-eq ha^{-1} yr^{-1}) for conventional intensification and -0.8 $MtCO_2$-eq yr^{-1} (-0.6 tCO_2-eq ha^{-1} yr^{-1}) for sustainable intensification, compared to 3.0 $MtCO_2$-eq yr^{-1} (2.3 tCO_2-eq ha^{-1} yr^{-1}) in the reference scenario. The variation in net annual emissions between the intensification pathways is related to differences in nutrient, pesticide and fuel use efficiencies, cattle production practices and other agricultural management practices,

indicating the importance of how agricultural intensification is implemented in practice. The intensification pathway is found to be more influential for the GHG balance than the ILUC mitigation scenario. For example, when considering the sustainable intensification pathway, the net annual GHG emissions are 0.1 $MtCO_2$-eq yr^{-1} (0.1 tCO_2-eq ha^{-1} yr^{-1}) for the low ILUC mitigation scenario and -1.8 $MtCO_2$-eq yr^{-1} (-1.4 tCO_2-eq ha^{-1} yr^{-1}) for the high scenario. The variation in net annual emissions between different ILUC mitigation scenarios following the same intensification pathway can largely be explained by carbon sequestration due to land conversion. When the net emissions are allocated to bioenergy, the ILUC mitigation scenarios often abate GHG emissions compared to gasoline. But sustainable intensification is required to attain GHG abatement potentials of 90% or higher. A qualitative assessment of other environmental impacts shows that conventional intensification could pose moderate to high risks on biodiversity, water quantity and quality, soil quality and air quality. Sustainable intensification could mainly have positive impacts on each of these aspects.

Chapter 6 addresses research questions 2 and 3 by reviewing the status and prospects of biomass value chains for heat, power, fuels and materials, assessing their current and projected (2030) levelized production costs and avoided emissions and comparing their GHG abatement costs. The cost and GHG emission data for all components of the biomass value chains as well as for the fossil reference chains were derived from literature. At present, the economically and environmentally preferred options are wood chip and pellet combustion in district heating systems and large-scale cofiring power plants (75 to 81 $US\$_{2005}$ $tCO_2\text{-}eq_{avoided}^{-1}$), and large-scale fermentation of low cost Brazilian sugarcane to ethanol (-65 to -53 \$ $tCO_2\text{-}eq_{avoided}^{-1}$) or biomaterials (-60 to -50 \$ $tCO_2\text{-}eq_{avoided}^{-1}$ for ethylene and -320 to -228 \$ $tCO_2\text{-}eq_{avoided}^{-1}$ for PLA; negative costs mean that the biomass value chain is cost competitive compared to its fossil reference at an oil price of 75 $\$_{2009}$ bbl^{-1}). In the longer term, the cultivation and use of lignocellulosic energy crops can play an important role in reducing the costs and improving the emission balance of biomass value chains. Key conversion technologies for lignocellulosic biomass are large-scale gasification (bioenergy and biomaterials) and fermentation (biofuels and biomaterials). However, both routes require improvement and scaling up of the technology to achieve higher efficiencies and lower costs over time. Further improvements can be attained by biorefineries that integrate different conversion technologies to maximize the use of all biomass components.

7.3 MAIN FINDINGS AND CONCLUSIONS

Based on the main findings of chapters 2 to 6, the following answers to the three research questions of this PhD thesis can be given.

RESEARCH QUESTION 1:

What are regional possibilities for agricultural intensification and what are their impacts on the demand for agricultural land and biomass potentials?

This thesis assessed three key aspects of regional possibilities for agricultural intensification: the first is the extent to which crop and livestock yields (i.e. production per hectare) can be increased, for example, by adopting best practices for agricultural management. The second aspect considered is the rate at which agricultural intensification can take place. The third aspect includes the driving factors and barriers for agricultural intensification and modernization. The findings of this PhD thesis regarding these three aspects and their impacts are explained in detail below.

Yield improvement potentials

With regard to the possibilities of increasing yields, it is important to distinguish between crops and livestock due to their different characteristics with regard to management practices and land use. Focusing first on crops, the possible extent of intensification is primarily defined by the local agro-ecological situation, including factors such as the climate and soil conditions. In the absence of other reduction factors such as pests and diseases, the agricultural technologies and management practices applied define what yields can actually be realized and how large the yield gap compared to the agro-ecological attainable yield is. To assess the possibilities for yield improvements, it is useful to look at surrounding or comparable regions that have comparable agro-ecological yield potentials, but have already realized higher yields. This provides insights in what yields are attainable at a certain level of technology and agricultural practices. This approach was applied to the case study on ILUC mitigation potentials in Lublin province, Poland (Chapter 4). In this case study, three scenarios were defined for different levels of intensification, based on current crop yields in Western Poland and Germany. These scenarios showed that crop yields could be improved by about 15% up to 100% when adopting agricultural practices from these example regions. Based on the characteristics of agriculture in the Lublin, Western Poland and Germany, improvements in agricultural production in Lublin could include mechanization, scaling up of farms and increasing the use of fertilizers.

With regard to livestock, a maximum attainable yield is not commonly defined for local conditions and more difficult to establish. While crop yields are directly related to land

use and can therefore be used as an indicator for intensification potentials, livestock productivity is generally expressed as the amount of product per animal (e.g. kg_{milk} cow^{-1} yr^{-1}) or per unit feed intake (i.e. the feed conversion efficiency in terms of e.g. kg_{milk} kg_{feed}^{-1}), which is not directly related to land use. To assess regional intensification potentials, it is thus necessary to also take into account the feed consumption and the land use related to feed production. This land use includes both grassland and cropland. In Chapter 3, this was done for cattle production for both diary and beef by reviewing bottom-up data for Europe and the USA, and disaggregated global data for nine world regions. The bottom-up data indicated that intensification from a pasture-based to a mixed beef production system has the potential to increase yields by 0.02 t_{beef} ha^{-1} or around 125% in the USA and by 0.18 t_{beef} ha^{-1} or around 170% in the UK. For the same intensification strategy for milk production in Austria, an improvement potential of about 1.7 t_{milk} ha^{-1} or 125%-140% was found. The disaggregated data for the nine world regions studied suggested that the potentials to increase yields in North America and Europe through transformational intensification are even larger. For beef, the average improvement potential is around 0.3 t_{beef} ha^{-1} or 780% in Northern America and around 0.2 t_{beef} ha^{-1} (270%) in Europe. For milk in Europe the average improvement potential is about 2.0 t_{milk} ha^{-1} or 550%. In other regions the relative differences in yields between pasture-based and mixed systems are generally even larger. In addition, when considering either pasture-based or mixed systems, there are large differences in yields between regions. Although practices from one region cannot be adopted one-to-one in other regions, this suggests that yields can be improved in many regions by intensification within the current production system. Thus, similar to crops, information on land use and animal production practices in well-performing regions can provide insights in the intensification potentials for livestock in other regions. In the case study on ILUC prevention in Lublin (Chapter 4), intensification potentials for beef and milk cattle production were therefore also based on cattle production in Western Poland and Germany. Based on these regions, beef and milk production per animal could be increased by about 40%-55%, and the total cattle density on pastures and meadows could be improved by about 15%-160%. Strategies to realize these improvements include, for example, improving pasture productivity, increasing the amount of crops and concentrates in the feed ration and genetic selection of cattle. In Chapter 3 and 4, data limitations on intensification of cattle production from bottom-up studies were observed. Therefore, this bottom-up data was complemented with more aggregated data on country level or higher. Thus, to improve the assessments of regional intensification potentials, more bottom-up studies and monitoring are needed.

Yield growth rates

Depending on the extent of the intensification potential and, for example, the time required to adopt or develop improved technologies, part of the regional intensification potential can be realized in short term, while other parts are only realizable in the long term. To assess the rate at which intensification can take place, it is useful to look at historical developments. This was done in Chapter 2. The analysis in this chapter showed that the rate at which yields have increased over the past five decades (1961-2010), varied significantly between regions. Over these five decades, yield growth was generally linear. However, the chapter revealed shorter periods during which yields improved at a higher rate compared to the long-term average due to, for example, increased mechanization and fertilizer consumption. The chapter also identified periods during which yield growth rates were lower than this average due to, for example, reduced investments in R&D and reforms of farmer support policies. In regions where the yield gap is still large, temporarily accelerated yield growth is therefore attainable under favorable circumstances. In Chapter 4, this is illustrated for the province of Lublin, where yield growth rates can be increased significantly when existing agricultural practices from Western Poland or Germany could be adopted during the period 2010-2020. Table 7-2 compares the long term linear growth rates for crop yields and beef and milk production per animal as derived from the historical analysis in Chapter 2 to the projected growth rates in Lublin in the baseline and above-baseline scenarios[1]. The baseline yield growth rates in Lublin are generally comparable to ranges found for other countries in Chapter 2 and in de Wit et al. [14]. The above-baseline yield growth rates resulting from adopting agricultural practices from Western Poland or Germany are found to be significantly higher compared to the baseline.

In addition to annual crops, intensification is also possible for dedicated bioenergy crops, which in turn also contributes to higher biomass potentials. A review of yields and yield projections in Chapter 6 shows that projected yield growth rates for dedicated bioenergy crops are larger compared to annual crops. This is because various key crop varieties are still under development and management practices are still tested and further improved. As this applies to all global regions, the regional differences in intensification potentials and attainable growth rates may be smaller compared to annual crops. Yet, understanding of the influence of local agro-ecological conditions on the production of dedicated bioenergy crops is less adequate because of limited experience with (commercial) production.

1 In the baseline scenario, it is assumed that the developments in agricultural yields in Lublin from 2010 to 2020 are in line with recent historical trends in the province. In the above-baseline scenarios, it is assumed that, in 2020, yields attain levels comparable to current average (2008-2012) yields in Western Poland and Germany, which results in higher yield growth rates compared to the baseline.

TABLE 7-2 | Regional variation in historical yield growth rates (1961-2010) as derived in Chapter 2 and yield growth rates in Lublin province (2010-2020) in the case of baseline and above-baseline yield improvements (the growth rates in terms of percentage are relative to 2010).

	Chapter 2			**Chapter 4 Lublin**			
	Long term linear growth rates in selected countries across the world (% yr^{-1} relative to 2010)[a]	Projected growth rates in IMAGE (2010-2050) and MIRAGE (2008-2020) in selected countries across the world (% yr^{-1})[a]	Long term linear growth rates in EU[d] (absolute in kg ha^{-1} yr^{-1} or kg $animal^{-1}$ yr^{-1})	Baseline: linear growth rates in Lublin (% yr^{-1} relative to 2010)	Above-baseline: improved growth rates in Lublin (% yr^{-1} relative to 2010)	Baseline: linear growth rates in Lublin (absolute in kg ha^{-1} yr^{-1} or kg $animal^{-1}$ yr^{-1})	Above-baseline: improved growth rates in Lublin relative to 2010 (absolute in kg ha^{-1} yr^{-1} or kg $animal^{-1}$ yr^{-1})
Crops, average range	0.7-1.6%[b]	-	-	0.6-2.5%[e]	1.6-10.6%	-	-
-Corn	1.2-1.6%[c]	0.9%-4.9%	130	1.8%	2.3-10.6%	100	135-390
-Wheat	0.7-1.8%	0.5%-2.0%	40-110	1.3%	2.3-6.7%	45	85-385
Milk	1.0-1.5%	-	85	1.2%	1.2-5.6%	57	57-247
Beef	0.4-0.8%	-	1.1-2.8	1.2%	1.2-5.4%	2.4	2.4-11.3

Yield growth rates for crops based on crop yields in t ha^{-1}, for milk and beef yields in t $animal^{-1}$ yr^{-1}
[a] countries included: Australia, Brazil, China, India, USA, Zambia and Zimbabwe; [b] crops included: corn, wheat, rice, soybean, sugarcane; [c] excluding Zimbabwe (-1.6%); [d] also including results from de Wit et al. [14]; [e] crops included: wheat, oats, rye, barley, triticale, potato, sugarbeet, corn, rapeseed.

Driving factors and barriers for agricultural intensification

When a region has the potential for considerable intensification and high intensification rates, this does not automatically mean that this potential will also be realized. To assess how intensification can be stimulated, Chapter 2 investigated historical driving factors for intensification. The results show that agricultural intensification is primarily a direct result of technological developments. For crops these include, for example, increased fertilizer use, irrigation and mechanization and adoption of new, high-yielding crop varieties. In the cattle sector, yield improvements were often achieved through the increased use of feed crops. However, economic and institutional factors often played a vital role in the improvement of agricultural technology. First, investments in R&D enabled the development of new technologies. In most countries, these were mainly public investments. In the USA, also private investments were very important. Second, the introduction of economic liberalization often created new markets for agricultural products, e.g. in Australia, and provided an incentive for many farmers to (further) improve yields. Especially for the cattle sector, the growing importance of domestic or export markets for commercial beef and milk production was found to be a key factor for yield improvements. Third, new technologies could be adopted by farmers because of farmer support programs, e.g. in India. In some cases, yield improvements have been

attained by policies that were focused on a specific commodity: for example in Zambia, the focus of policies on corn production during a long period resulted in significantly higher yield growth rates for corn compared to other crops. In Brazil, the ProÁlcool program positively affected sugarcane yields. The import substitution policy for edible oils in India especially stimulated the increase of soybean yields. In contrast to the successful implementation of policies in the above examples, Zimbabwe also shows the impact of a lack of good governance and stimulating policies. A civil war in the 1970s and economic reforms around the 1990s disrupted agricultural production and the economy. Due to the long-term unstable situation, crop yields have much more fluctuated than in other, more stable countries.

In Lublin, high yield improvements from 2010 to 2020 are considered to be technically feasible because the yield projections are based on existing farming practices in Western Poland and Germany. Also, the results from Chapter 2 suggest that the resulting temporarily high yield growth rates can be realized under favorable conditions with regard to economics and governance that stimulate improvements in agricultural technology and management. However, Chapter 4 also identified several factors that could significantly affect the agricultural intensification potential in Lublin. As the agricultural sector in Lublin is characterized by a large number of small farms and low average management levels, scaling up, modernization and mechanization are needed. But farmers have little capital to invest, and land prices are considered by locals to be too low for selling or leasing land. To maximize the intensification potential, governance and policies could, for example, include financial support to farmers to facilitate improved production practices. Such support is already included in European and Polish agricultural and rural development policies, but should be increased to realize the full potential. When enough financial support would be available, the largest barrier for high growth rates is potentially the time needed for land reforms (selling land to enable upscaling).

Impact of intensification on agricultural land demand

The case study on ILUC prevention in Lublin shows that agricultural intensification can release approximately 100-400 thousand hectare (5-24% of the total agricultural area). This includes increased crop yields, beef and milk yields and also higher density of cattle on meadows and pastures. Meadows and pastures account for 20% of total agricultural land in Lublin (2010), but improvements in cattle production contributes to about 30% of the total reduction in land use.

Based on the regional intensification potentials for cattle found in Chapter 3, land use could be significantly reduced in all regions. The bottom-up data indicate that intensification from a pasture-based to a mixed beef production system has the potential

to reduce land occupation (in m^2 kg_{beef}^{-1}) by around 20% in the USA and by around 40% in the UK. For the same intensification strategy for milk production in Austria, a reduction potential of 20-30% is found. The disaggregated data for nine world regions studied suggested that the potentials to reduce land occupation in North America and Europe through transformational intensification are even larger. For beef, the potential is around 90% in Northern America and around 60% in Europe. For milk in Europe the potential is around 80%. For dairy and beef production in other regions, the potentials to reduce land occupation by moving from pasture-based and mixed systems are generally in the range of 60% to 95%.

Although agricultural intensification could make available significant amounts of land for biomass production, it should be taken into account that the availability of surplus agricultural land for biomass not only depends on agricultural intensification, but also on other competitive uses for the released land, such as afforestation, as well as on the development in the demand for agricultural products. In the case study on Lublin, the latter was taken into account by using top-down projections on the demand for food, feed and fibre. Based on a disaggregation of projections on a European scale, agricultural production in Lublin was projected to decline from 2010 to 2020, resulting in a significant decrease in the demand for agricultural land under the assumption of baseline yield developments. However, this projected reduction in land use was considered large compared to recent developments in Lublin and Poland. Therefore, different scenarios for land use were included in the case study. The deviation of the disaggregated projections from recent trends suggests that the use of aggregated results is not or only limited suitable to derive projections on a regional scale. In future studies, it is therefore recommended to improve the regional assessment of potential pathways for agricultural production.

Models that assess the availability of surplus agricultural land for biomass, often base their yield projections on the baseline assumption that future agricultural intensification will be in line with historical trends of linear yield growth. Chapter 2 showed that this is true generally when considering longer time periods, but that higher growth rates can be attained over shorter periods of time. In shorter term assessments of biomass potentials, it is thus important to take regional growth rate potentials into account and to analyze how different intensification rates influence land availability and biomass potentials, as was done in Chapter 4. In addition, most models assume that global future yield growth will slow down compared to the historical trend because i) the opportunities for increasing yields and exploiting existing yield gaps are becoming more and more exhausted, ii) investments in agricultural R&D have declined, and iii) considerable socio-economic constraints in many developing countries will remain a limiting factor for yield growth. Although these motivations are often justified on a global level, these are mainly

expectations about how different factors are likely to develop. There may, however, also be other possible pathways. In regions where there is still room for considerable yield improvements, stimulating policies and other factors could play a vital role in exploiting the improvement potential. Therefore, it is important to investigate different pathways that illustrate how yields may develop under different assumptions and conditions, and how this affects regional agricultural land use and biomass potentials over time. As future yield developments are region specific and depend on the development of different driving factors in that region, more detailed regional assessments of the most important driving factors are needed. Also, models now consider a limited number of endogenous drivers and should incorporate more endogenous driving factors based on regional assessments as performed in Chapter 2. In this way, the model outcomes can be used to identify what regional conditions need to be met in order to reach certain yield levels. This can then help to identify regional strategies for agricultural intensification.

Impact of intensification on biomass potentials

The analysis in Chapter 4 found large potentials for agricultural intensification and significant impacts on the availability of surplus agricultural land in Lublin. However, a limited area of 200 to 270 thousand hectare (12-15% of the total agricultural area) is considered to be legally available and biophysically suitable for miscanthus production based on criteria for protecting high conservation areas and minimum requirements for land suitability. This limitation on land use is mainly caused by the suitability criteria and especially the sensitivity of miscanthus to water stress. However, the land suitability was assessed based on a limited number of criteria like the minimum ground water level, and should be analyzed in more detail. Assuming an average miscanthus yield of 13 t ha^{-1}, 1.2 to 3.5 million tonne miscanthus could be produced in 2020 on the land made available through agricultural intensification. The related bioethanol production potential is about 8 to 23 PJ yr^{-1} (340 to 970 million liter yr^{-1}). A sensitivity analysis for the average miscanthus yield and the overall productivity of the bioethanol value chain shows that the bioethanol production potential ranges between 6 and 35 PJ yr^{-1} (260 to 1,480 million liter yr^{-1}). When additional land would be made available through increased food chain efficiencies and the use of under-utilized land, the total bioethanol production potential ranges from 12 to 35 PJ yr^{-1}, depending on the productivity of the bioethanol value chain. For comparison, the national Polish target for 2nd generation bioethanol consumption in 2020 is almost 9 PJ, and the total biofuel target in 2020 is 60 PJ. The bioethanol production potential, however, is the technical potential that accounts only for key environmental aspects such as the protection of high conservation value areas. Although the sustainable implementation potential may be lower, agricultural intensification in the province of Lublin could play a key role in achieving this target.

RESEARCH QUESTION 2:

What is the effect of different agricultural intensification pathways and biomass production on GHG emissions and the environment?

Different pathways for agricultural intensification are possible, ranging from conventional to sustainable intensification for crops and from incremental to transformational intensification for livestock. Clearly very different environmental impacts are the result of such divergent strategies. For example, when yields are improved through increased fertilizer use, this causes additional GHG emissions. When yields are improved through the adoption of precision farming, fertilizer use and GHG emissions are reduced. Given intensification is needed to allow additional production of biomass for modern energy and material purposes, these impacts need to be attributed to the biomass value chain in order to determine its net impact. This thesis addressed this issue by investigating the GHG impacts of different intensification pathways for cattle in Chapter 3 and the net environmental impacts of agricultural intensification and biomass production in Chapter 5.

GHG impacts of intensification pathways for cattle production

Chapter 3 investigated the impact of two intensification pathways for cattle on the GHG emissions including and excluding land use change. These pathways are i) incremental intensification within the existing production system (where it is distinguished between pasture-based, mixed and industrial production systems), and ii) transformational intensification by moving from one system to another. Based on bottom-up data for milk production in Europe, GHG emission reduction potentials <u>excluding</u> LUC ranged from 1%-14% (0.01-0.13 $kgCO_2$-eq kg_{milk}^{-1}) for intensification within one system and from 2%-26% for system transitions (0.02-0.30 $kgCO_2$-eq kg_{milk}^{-1}). The reduction potentials for both pathways are often close to one another. Therefore, which intensification pathway will be most beneficial for the GHG balance depends on the specific design of the initial and final production system. For example, when pasture-based production is characterized by extensive pasture management, improved pasture management could have a larger contribution to GHG reductions compared to increasing the share of crops or concentrates in the feed mix. But when pasture-based production is already characterized by improved pasture management, the transition to a mixed system could provide a larger potential to reduce emissions than further intensification within the pasture-based system. Based on the disaggregated global data, this also applies to other developed regions. In developing regions, there is especially a large potential to reduce emissions by intensification within pasture-based systems. For example, in Brazil and Sub-Saharan Africa, the GHG emissions per kg of milk are up to 3 or 5 times higher, i.e. 1.5 to 2.5 $kgCO_2$-eq kg_{milk}^{-1}, compared to developed regions. To reduce emissions, it is important to enhance the feed quality, for example trough improved pasture management. The additional emission reduction

potentials for moving from pasture-based to mixed milk production is generally limited. For beef production, the bottom-up analysis results and disaggregated global data show that the variation in GHG balances within and between production systems is generally larger than for milk production. This means that in both developed and developing countries, significant reduction potentials exists for intensification within pasture-based and mixed systems. For example, emission reduction of intensification within the pastures-based system in Brazil could exceed 50% (30 $kgCO_2$-eq kg_{beef}^{-1}) when changing from extensive natural grass-based pastures to improved pastures. For beef, the additional reduction potential for a transition from pasture-based to mixed production is larger compared to dairy production. Although intensification of cattle production generally results in reduced GHG emissions, a few exceptions were found in the bottom-up studies. In these cases, organic or extensive cattle production attains lower emissions compared to more intensive conventional production because of not using any synthetic fertilizers, lower fossil fuel consumption and the occurrence of soil carbon sequestration. This highlights the importance to use resources efficiently. To assess regional impacts of intensification in the cattle sector in more detail, more bottom-up studies are needed. It is especially important to include more comparisons of various production systems in same the region, to pay more attention to cattle production in developing countries, and to the performance and potential role of industrial systems in reducing emissions.

When considering GHG balances <u>including</u> LUC, intensification by moving from pasture-based systems to mixed systems is considered vital to reduce emissions. As land occupation in pasture-based systems is significantly higher compared to mixed systems for both milk and beef in all regions, the contribution of pasture-based systems to land use change and related emissions is generally higher for pasture-based systems. Only in Europe, emissions from LUC are higher for mixed systems. This is primarily due to feed imports from South America and associated LUC. For dairy production, the amount of LUC-related emissions is largest for Latin America and Sub-Saharan Africa and contributes to about 40% and 50% of the total emissions. By moving from pasture-based to mixed systems, the LUC-related emissions can be reduced by respectively 1.5 $kgCO_2$-eq kg_{milk}^{-1} (>60%) in Latin America and 3.2 $kgCO_2$-eq kg_{milk}^{-1} (>50%) in Sub-Saharan Africa. For beef, the amount of LUC-related emissions is highest in Sub-Saharan Africa. The GHG reduction potential of a system transition in this regions is 280 $kgCO_2$-eq kg_{beef}^{-1} (50%). However, compared to other developing regions, LUC-related emissions are also very high in mixed systems and these systems also need to be improved to reduce land use and GHG emissions.

Net impacts of agricultural intensification and biomass production

When agricultural intensification is combined with biomass production, the intensification pathway is found to have considerable impact on the net GHG emissions from agriculture

and biomass production. Chapter 5 investigated the impact of three ILUC mitigation scenarios, representing a low, medium and high miscanthus-based ethanol production potential, and the impact of three agricultural intensification pathways in terms of sustainability in the province of Lublin. In each ILUC mitigation scenario, this impact of the agricultural intensification pathways was found to be considerable. For example, in the moderate ILUC mitigation scenario, the net annual GHG emissions in 2020 are 2.3 $MtCO_2$-eq yr^{-1} (1.8 tCO_2-eq ha^{-1} yr^{-1}) for conventional intensification, 0.6 $MtCO_2$-eq yr^{-1} (0.4 tCO_2-eq ha^{-1} yr^{-1}) for intermediate sustainable intensification and -0.8 $MtCO_2$-eq yr^{-1} (-0.6 tCO_2-eq ha^{-1} yr^{-1}) for sustainable intensification, compared to 3.0 $MtCO_2$-eq yr^{-1} (2.3 tCO_2-eq ha^{-1} yr^{-1}) in the reference scenario that excludes ILUC mitigation and bioethanol production. The largest impact factors are found to be nutrient use efficiency, nitrogen emission factors and soil carbon stock changes due to land use management. When allocating the net GHG emissions to bioethanol, the ILUC mitigation scenarios often abate emissions compared to gasoline. However, sustainable intensification pathways are needed to attain GHG abatement potentials of 90% or higher. A qualitative assessment of other environmental impacts also emphasizes the importance of sustainable intensification. It shows that conventional intensification could pose moderate to high risks on biodiversity, water quantity and quality, soil quality and air quality. Sustainable intensification could mainly have positive impacts on each of these aspects. These findings show that future regional studies on the environmental impacts of biomass need to evaluate the influence of different pathways for agricultural intensification. This should include the effects of intensification in both crop and livestock production. The method developed in Chapter 5 provides an integrated, detailed approach that can be adopted by future studies. The approach could be further improved by making it spatially explicit. Because of the importance of sustainable intensification pathways for improving the GHG and environmental impacts of biomass value chains, sustainability criteria on, for example, nutrient leaching, water consumption and soil quality should need not only to be applied to biomass feedstock production, but to all agricultural activities in the region.

The review in Chapter 6 shows that biomass value chains for biofuels often reduce GHG emissions compared to their fossil reference. However, the reduction is generally lower for first generation biofuels, derived from annual crops (e.g. corn and rapeseed), compared to bioethanol and biodiesel based on dedicated bioenergy crops such as miscanthus. An important reason is that GHG emissions from the cultivation of annual crops have a large influence on the total GHG balance of the biofuel value chain. Sustainable intensification in the agricultural sector can contribute to an improved GHG balance for the production of feedstocks for first generation biofuels. Therefore, sustainable intensification could also increase the GHG abatement potential of biomass value chains for first generation biofuels.

As resource efficiencies, especially of nutrients, are found to have a considerable effect on the environmental impacts of agriculture and bioenergy, it is important to target the improvement of these efficiencies. The historical analysis in Chapter 2 showed that agri-environmental policies have frequently played an important role in achieving this (e.g. by setting environmental standards). These policies aimed at, for example, enhanced quality of degraded agricultural lands (Australia, China, Zambia, Zimbabwe), balanced use of inputs (China) and controlled use and management of natural resources (Australia, India, USA). However, in India, fertilizer inputs per hectare continued to increase due to weak enforcement of agri-environmental policies.

RESEARCH QUESTION 3:

Which biomass value chains are preferred today as well as in the future with regard to their economic performance and GHG balance?

Biomass value chains consist of several components, including biomass production, feedstock pretreatment, transportation and conversion. This thesis reviewed and discussed the costs of each component, the costs and the GHG balance of complete biomass value chains compared to a fossil reference, and the GHG abatement costs of these value chains for heat, power, fuel and materials (Chapter 6). In addition, it evaluated the GHG balance of miscanthus-based ethanol, and assessed how the results are changed when including the emissions from agricultural production and specifically what the influence is of different pathways for agricultural intensification (Chapter 5).

Economic performance compared to fossil reference

With regard to the current and projected (2030) costs per chain component, it was found that high yielding crops attain lower road-side costs, i.e. the costs to make the feedstock available at the roadside for transportation and storage, compared to lower yielding crops. With regard to pretreatment, several technologies exist or are being developed. Pelletization is widely applied and is generally found to be cheaper compared to the newer technology of torrefaction combined with pelletization. However, the latter could reduce transportation and subsequent final conversion costs in the future.

Currently, most biomass value chains are not cost competitive compared to their fossil reference. With regard to *heat*, large-scale systems can be cost competitive with the more expensive fossil reference systems. For example, at a wood pellet price of 9 $\$_{2009}$ GJ^{-1}, the levelized costs of heat are 18 $\$ GJ_{th}^{-1}$ for 5 MW district heating compared to 8-20 $\$ GJ_{th}^{-1}$ for fossil reference systems at 75 $\$_{2009}$ bbl_{oil}^{-1}. In 2030, these large scale systems also become competitive with cheaper fossil heating systems. For biobased *power and CHP*, none of the existing value chains are cost competitive compared to fossil-based power generation, when wood pellets are considered. The difference in costs between the bio-based and

fossil-based value chain is lowest for cofiring of wood pellets (5-100 MW), for which the costs are about 28-34 \$ GJ_e^{-1} compared to 7-20 \$ GJ_e^{-1} for the fossil reference. Considering novel technologies, gasification-based biomass value chains, which apply large-scale integrated gasification combined cycle (IGCC) and indirect cofiring technologies, are considered promising for future power generation and CHP with projected levelized costs of 11-26 \$ GJ_e^{-1} in 2030 (compared to 13-37 \$ GJ_e^{-1} for the fossil reference). With regard to *first-generation biofuels*, large-scale (1000 MW_{feed}) production of Brazilian sugarcane ethanol (in an integrated sugar mill) attains costs of 9-12 \$ GJ_{etOH}^{-1} and is therefore cost competitive to gasoline (16 \$ $GJ_{gasoline}^{-1}$ at 75 \$ bbl^{-1})[2]. The production costs of corn ethanol, soy-based biodiesel and renewable diesel are nearly competitive to gasoline and diesel (17-21 \$ $GJ_{biofuel}^{-1}$ compared to 16 \$ $GJ_{fossil\ fuel}^{-1}$). Considering *second-generation biofuels*, short-term commercial production of thermochemical hydrogen (13-15 \$ $GJ_{hydrogen}^{-1}$) and methanol (about 12 \$ $GJ_{methanol}^{-1}$) may be cost competitive to gasoline and diesel. In the longer term, all value chains of second-generation biofuels are projected to become cost competitive and attain levelized production costs from 7-8 \$ $GJ_{hydrogen}^{-1}$ for hydrogen to 12-15 \$ GJ_{diesel}^{-1} for Fischer-Tropsch (FT) diesel (22-24 \$ $GJ_{fossil\ fuel}^{-1}$ for fossil fuels at 110 \$ bbl^{-1}). The production of *biomaterials* from low cost Brazilian sugarcane (except butanol) is found to be cost competitive and current costs of ethylene, PLA, PHA and PTT are 270-865 \$ t^{-1} lower compared to their fossil reference products (sugar 141 \$ $t_{ferm.sugar}^{-1}$, oil 75 $\$_{2009}$ bbl_{oil}^{-1}). In the longer term, this difference increases to about 940-1600 \$ t^{-1} (sugar 141 \$ $t_{ferm.sugar}^{-1}$, oil 110 $\$_{2009}$ bbl_{oil}^{-1}). PLA has the best economic potential both today and in the longer term.

Avoided GHG emissions

Chapter 6 considered avoided GHG emissions by biomass value chains when excluding LUC. Most biomass value chains for heat, power, fuels and materials avoid emissions compared to their fossil reference. For domestic and district heating, the GHG emission reduction potentials of the systems assessed are all about 80 $gCO_2\text{-}eq_{avoided}$ MJ_{th}^{-1} or 85% today, with only marginal changes expected for 2030. With regard to power generation, small-scale CHP technologies with low power-heat ratios receive a significant credit for heat and have the highest GHG abatement potential (>300 $gCO_2\text{-}eq_{avoided}$ MJ_e^{-1} or >250%). For biofuels, sugarcane ethanol production is the best option amongst first-generation biofuels to abate GHG emissions (66-81 $gCO_2\text{-}eq_{avoided}$ MJ_{etOH}^{-1} or about 70%-90%). Higher abatement levels can be attained by biomass value chains of second-generation bioethanol and FT diesel (72-91 $gCO_2\text{-}eq_{avoided}$ MJ_{fuel}^{-1} or about 80%-100%).

2 Currently, fossil fuel prices are lower compared to the prices assumed in Chapter 6 for the present production of fossil-based energy and materials. For example, current oil prices are about 50 \$ bbl^{-1}, while Chapter 6 assumed an oil price of 75 \$ bbl^{-1}. How this affects the cost-competitiveness of the biomass value chains is discussed in the section on GHG abatements costs.

Sugarcane ethanol and FT diesel are also preferred in the longer term because of increasing emission abatement potentials (71-114 $gCO_2\text{-}eq_{avoided}$ MJ_{fuel}^{-1} or about 90%-150%). Of current biomaterials, sugarcane ethylene avoids most emissions (4.5-5.3 $tCO_2\text{-}eq_{avoided}$ $t_{ethylene}^{-1}$ or about 100%-120%). In the longer term, also butanol from sugarcane is projected to attain a high GHG abatement potential (4.9-6.3 $tCO_2\text{-}eq_{avoided}$ $t_{butanol}^{-1}$ or about 115%-145%). Also ethylene and FT olefins from second generation feedstocks can attain high abatement levels (3.9-8.2 $tCO_2\text{-}eq_{avoided}$ $t_{olefins}^{-1}$ or about 100%-185%).
In Chapter 5, the value chain for miscanthus-based ethanol in 2020 was found to reduce emissions compared to gasoline by more than 100% when excluding emissions from agricultural production in the GHG balance. This reduction was caused primarily by the high credits for electricity co-generation due to the high emission factor for the Polish electricity mix. Including the GHG emissions from agricultural production in the GHG balance generally resulted in increased net emissions for bioethanol. In most cases, bioethanol was still found to abate emissions compared to gasoline. However, the net GHG balance was highly influenced by the agricultural intensification pathway considered. In the scenarios with a medium and high bioethanol production potential, the sustainable intensification pathway actually had a positive effect on the net GHG balance. This means that the emission reduction compared to gasoline increased compared to excluding the effects of agricultural production. In these cases, the emission abatement potential is found to be about 145% in the medium scenario and 190% in the high scenario. The positive impact of agricultural intensification on the net GHG balance is largely explained by relatively low emissions from fertilizer use and cattle production compared to the other intensification pathways, and the high soil carbon sequestration level due to adoption of management practices that strengthen the productive capacity of the soil (e.g. reduced or no tillage).

GHG abatement costs

As most biomass value chains are not cost competitive at the moment but reduce GHG emissions compared to their fossil reference, generally positive GHG abatement costs are found for these value chains. However, some biomass value chains are found to be cost competitive and to avoid GHG emissions at the moment, which results in negative abatement costs. These are sugarcane ethanol (large scale production), biochemical ethylene from sugarcane and lignocellulosic feedstock, PLA from sugarcane and corn, and olefins derived from gasification of lignocellulosic feedstock. However, these value chains are cost competitive compared to their fossil reference based on an oil price of 75 $ bbl^{-1}, while the current oil price is actually around 50 $ bbl^{-1}. Chapter 6 included a sensitivity analysis for an oil price of 60 $ bbl^{-1} for a selection of biomass value chains. The results of this sensitivity analysis show that sugarcane ethanol remains cost competitive compared to gasoline, but it closely approaches the turning point. However, sugarcane-based

ethylene is no longer cost competitive compared to its fossil reference. Ethylene and olefins from lignocellulosic biomass and PLA from sugarcane and corn were not included in the sensitivity analysis. However, based on gap between the costs for the biomaterials and fossil-based materials at an oil price of 75 \$ bbl^{-1}, it is expected that PLA will remain cost competitive compared to its fossil reference at an oil price of 60 \$ bbl^{-1}, but this could potentially no longer be the case for ethylene and olefins.

All present value chains for heat and power production have positive abatement costs. The best options for heat and power are district heating (75-79 \$ $tCO_2\text{-}eq_{avoided}^{-1}$ at a fossil reference cost of 12.4 \$ GJ_{th}^{-1}), large-scale cofiring CHP (75-81 \$ $tCO_2\text{-}eq_{avoided}^{-1}$ at 100 MW_e and a reference cost of 15 \$ GJ_e^{-1}) and near-term commercial power production in a 250 MW IGCC-based CHP plant (71-115 \$ $tCO_2\text{-}eq_{avoided}^{-1}$). Based on a limited number of projections for the long term, at least the following biomass value chains are expected to achieve negative abatement costs around 2030: district heating, (indirect) cofiring, IGCC, biochemical ethanol and thermochemical FT diesel from lignocellulosic feedstock, and sugarcane butanol. At reduced feedstock prices, increased fossil fuel costs or reduced discount rates, the value chains for FT diesel production in the short term and for domestic heat and gas engine CHP in 2030 could also attain negative GHG abatement costs.

Overall

Based on the findings from Chapter 6, the currently preferred options with regard to their economic and environmental performance are the biomass value chains for sugarcane-based ethanol, ethylene and PLA, followed by wood chip combustion in district heating systems and in cofiring power plants. In the longer term, fermentation of low cost sugarcane remains attractive for the production of cost competitive ethanol and materials with high GHG abatement potentials. In addition, lignocellulosic energy crops and advanced conversion technologies, i.e. biochemical conversion of lignocellulosic biomass and gasification, can play a key role in producing bio-based products that have both a good economic and environmental performance. These are versatile technologies as biochemical conversion can produce both biofuels and biomaterials, and gasification can process various feedstocks and produce a wide range of products including heat, power, fuels and materials. However, while the results show good perspectives for second-generation feedstocks and technologies and for biomaterials, there is still uncertainty about their actual performance, especially with regard to the economics. With regard to second generation feedstocks, projections are based on assumptions about developments in the costs of biomass production and supply. Similarly, for the advanced conversion technologies, cost projections are based on assumptions about technological progress. Both technological and economical improvements are preconditions for large scale commercial application of second generation feedstocks and technologies, but experience

with these technologies is still limited. Therefore, stimulation of and investments in these technologies is very important to improve the environmental and economic performance of future biomass value chains. With regard to biomaterials, the uncertainty about costs is especially related to a lack of good data. In future research, it is thus needed to collect more data and also to further investigate how and at what speed cost reductions can be achieved.

Finally, the integration of different conversion technologies in biorefineries can maximize the use of all biomass components and enable the production of a broad product range, including food, feed, chemicals, materials, fuels, power and heat. The literature discussed in Chapter 6 suggests that biorefineries can improve the environmental and economic performance of the combined biomass value chains. More research and demonstration is needed to gain experience and gradually optimize such advanced conversion concepts.

7.4 RECOMMENDATIONS FOR FURTHER RESEARCH AND POLICY

Based on the findings, recommendations for further research and policy making are identified.

Further research

- Model assessments of biomass potentials and impacts are found to base their agricultural yield projections on historical yield trends and to include a limited number of endogenous driving factors. As a result, regional possibilities for yield improvements and temporarily increased yield growth rates are not properly accounted for. It is therefore recommended that the models include more driving factors and different scenario variables based on regional assessments as performed in Chapter 2. The models could then be used to assess how different driving factors can contribute to increased yields and yield growth rates, and what needs to be done to realize the potentials. This can then help to identify regional strategies for agricultural intensification linked to sustainable biomass production.
- It is found that the number of yield projections available for crop production is considerably larger compared to livestock. In addition, the projections for livestock are based on more aggregated and less detailed data compared to crops. As livestock intensification plays an important role in making agricultural land available for biomass production, it is recommended that biomass potential studies pay more attention to livestock production and its possibilities for intensification. In addition, studies on the GHG performance and other environmental impacts of agricultural intensification and biomass production should not only take into account the

impacts of intensification in the arable sector, but also in the livestock sector. To better include livestock intensification in biomass potential and impact studies, more case studies are needed to collect regional data and to calculate GHG balances for different livestock production systems. The data that needs to be collected includes, for example, feed consumption, animal productivity, feed conversion efficiency, and land occupation (including both feed crop production and grazing). The review of existing bottom-up studies on GHG emissions and land occupation for cattle production revealed three knowledge gaps that should specifically be addressed by future research. First, cattle production in developing countries is not or hardly covered by case studies. However, analysis of more aggregated and modeled data suggests that intensification of cattle is of high importance to reduce land occupation and GHG emissions in these regions. Second, although the coverage of cattle production in developed regions is generally better, only a few studies include different production systems and allow assessing and comparing the impacts of different intensification pathways for cattle. Third, existing case studies especially consider pasture-based and mixed systems, while the analysis of industrial livestock management is limited. This restricts the assessment of the impacts of moving from mixed to industrial management systems.

- The experience with commercial production of second generation feedstocks and advanced conversion and biorefining technologies is still limited. This leaves uncertainty about their economic performance on the short and longer term. It is therefore needed to assess technological learning potentials, the speed at which learning can take place and how production capacities should increase over time to attain these learning rates. In addition, it should be quantified what investments are needed to realize the required production capacities. Projects aiming at the demonstration and scaling-up of feedstock production and conversion technologies play an important role in providing data for these analyses.
- The review and comparison of the economic and environmental performance of biomass value chains emphasized the lack of detailed cost data for biomaterial conversion routes. This results in high uncertainty about the actual economic performance of these chains. Collecting more, better and transparent data is important for models, but also for providing a solid basis for the support of research, development, demonstration and commercialization projects. To this aim, close collaboration of industry with science is needed.
- The regional analysis of possibilities for agricultural intensification and bioenergy production in Lublin province emphasizes that there is a need to generally apply an integrated approach on land use planning for biomass feedstock production and other uses on regional level. To understand how the implementation of such

an approach could work in practice, demonstration projects are needed that focus on increasing biomass production combined with balanced modernization of conventional agriculture and integrated land use planning. In addition, the projects can also be used to verify the potentials from case studies, to develop and demonstrate monitoring methods and to identify the actual barriers for realizing the potentials.

- In order to improve the overall environmental performance of land use in regions where biomass feedstocks are (planned to) be produced, performing a regional assessment of the net GHG emissions and other environmental impacts should become a standard procedure. The assessment allows identifying go and no-go pathways for agricultural intensification and biomass production, based on predefined sustainability criteria.

Policy

- This thesis stresses the importance of i) the interlinkages between the agricultural and bioenergy sectors, and ii) sustainable intensification pathways for improving the GHG balance and environmental impacts of biomass value chains. Therefore, agricultural policies on regional, national and supranational level should have an integrated focus on both agriculture and biomass production for energy and materials. This means that, in order to avoid ILUC, goals for biomass expansion and yield improvements in crop and livestock production need to be harmonized. These goals should be adapted to the possibilities for agricultural intensification and scaling up of biomass production over time. Also, actual developments in biomass production volumes and in agricultural intensification need to be monitored, regulated and balanced according to these goals.
- Adopting integrated policies on biomass and agricultural production also means that sustainability criteria on, for example, nutrient leaching, water consumption and soil quality need not only to be applied to biomass feedstock production, but to all agricultural and forestry activities in the region. In addition to implementing sustainability criteria, policies could also stimulate sustainable biomass and agricultural production through financial incentives. This for example includes subsidizing the adoption of best available technologies for both increasing yields and improving environmental impacts.
- As the drivers for agricultural intensification are region specific, agricultural policies should account for regional drivers and barriers where possible. In addition, national and supranational policies should be flexible enough to take into account regional needs and to implement region specific measures to stimulate sustainable agriculture and biomass production.

- While second generation feedstocks and conversion technologies are considered vital for future economically feasible and sustainable production of bioenergy and biomaterials on a large scale, commercialization is still limited. Reasons for this are, for example, technological difficulties, high investment costs, a lack of clear policy incentives and therefore an unfavorable investment climate. As a result, clear and coherent long-term policies need to be formulated for increasing the share of bioenergy and biomaterials in the energy and materials markets. In addition, investments in development, demonstration and commercialization projects are needed to stimulate second generation biomass feedstocks and technologies, to increase the capacity and efficiency of biomass conversion to energy and materials, and to improve biomass infrastructure, logistics, and markets.

REFERENCES

[1] IPCC. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change: Geneva, Switzerland; 2014.

[2] Woods J, Lynd LR, Laser M, et al. Chapter 9. Land and Bioenergy. In: Souza GM, Victoria RL, Joly CA, Verdade LM, editors. Bioenergy & Sustainability: bridging the gaps. Scientific Committee on Problems of the Environment (SCOPE): Paris, France; 2015, p. 258-300.

[3] Saygın D, Gielen DJ, Draeck M, et al. Assessment of the technical and economic potentials of biomass use for the production of steam, chemicals and polymers. Renewable and Sustainable Energy Reviews 2014;40:1153-1167.

[4] Fischedick M, Schaeffer R, Adedoyin A, et al. Mitigation potential and costs. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 791-864.

[5] IEA. Energy Technology Perspectives 2012: Pathways to a Clean Energy System. International Energy Agency: Paris, France; 2012.

[6] GEA. Global Energy Assessment - Toward a Sustainable Future. Cambridge University Press and the International Institute for Applied Systems Analysis: Cambridge, UK and New York, NY, USA and Laxenburg, Austria; 2012.

[7] WWF. The Energy Report: 100% Renewable Energy by 2050. WWF – World Wide Fund For Nature: Gland, Switzerland; 2011.

[8] Greenpeace. Energy [R]evolution: A Sustainable World Energy Outlook. Greenpeace International, The Global Wind Energy Council (GWEC) and European Renewable Energy Council (EREC): Amsterdam, The Netherlands and Brussels, Belgium; 2012.

[9] Dale BE, Anderson JE, Brown RC, et al. Take a Closer Look: Biofuels Can Support Environmental, Economic and Social Goals. Environmental science & technology 2014;48(13):7200-7203.

[10] Clarke L, Jiang K, Akimoto K, et al. Assessing Transformation Pathways. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2014, p. 413-510.

[11] Smith P, Bustamante M, Ahammad H, et al. Agriculture, Forestry and Other Land Use (AFOLU). In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change . Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2014, p. 811-922.

[12] Chum H, Faaij A, Moreira J, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 209-332.

[13] Wicke B, Verweij P, van Meijl H, et al. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 2012;3(1):87-100.

[14] de Wit M, Londo M, Faaij A. Productivity developments in European agriculture: Relations to and opportunities for biomass production. Renewable and Sustainable Energy Reviews 2011;15(5):2397-2412.

Samenvatting en Conclusies

CONTEXT

Op dit moment wordt een groot deel van de mondiale primaire energie geleverd door fossiele brandstoffen (olie, kolen en gas). Deze grootschalige consumptie van fossiele brandstoffen kan echter niet worden volgehouden op de lange termijn. Redenen hiervoor zijn de eindigheid van fossiele bronnen en de hoge bijdrage van het gebruik van fossiele brandstoffen aan antropogene broeikasgasemissies [1]. Het is daarom belangrijk alternatieve grondstoffen te vinden voor de productie van energie en materialen die nu uit fossiele brandstoffen verkregen worden. Biomassa wordt gezien als een belangrijk alternatief, omdat de toepassingen veelzijdig zijn en biomassa gebruikt kan worden voor de productie van warmte, elektricititeit, vloeibare brandstoffen, biochemicaliën en biomaterialen. In 2010 bedroeg het gebruik van biomassa voor moderne toepassingen voor warmte, elektricititeit en biobrandstoffen ongeveer 22 EJ_p [2], terwijl het gebruik van biomassa voor nieuwe synthetische, organische materialen nog laag is (0.6 EJ) [3]. Om in de toekomst energie en materialen te leveren met een lage CO_2 uitstoot, moet de bijdrage van biomassa aan de totale primaire energievoorziening in 2050 naar schatting 100-200 EJ_p per jaar bedragen [3-9], toenemend naar mogelijk 300 EJ_p in de loop naar 2100 [10]. Tegelijkertijd schat het IPCC dat de mondiale technische capaciteit om biomassa te leveren in 2050 tussen 100 en 300 EJ_p zal liggen [11,12]. Dit betekent dat grote hoeveelheden biomassa moeten worden aangewend om klimaatverandering te beperken en competitie tussen toepassingen van biomassa voor voedsel, veevoer, energie en materialen te voorkomen. Daarnaast is het belangrijk om biomassawaardeketens voor de productie van warmte, elektricititeit, brandstoffen en materialen in te zetten die het meest veelbelovend zijn met betrekking tot hun technologische, economische en milieuprestaties. De recente mondiale toename in biomassaproductie voor moderne energie- en materiaaltoepassingen heeft echter ook vragen opgeroepen over de rol die biomassa werkelijk kan spelen in het beperken van klimaatverandering. Een van de belangrijjkste redenen tot bezorgdheid is ongewenste verandering in landgebruik (land use change, LUC) en met name indirecte verandering in landgebruik (indirect lands use change, ILUC). Om deze reden is het belangrijk te onderzoeken hoe ILUC kan worden beperkt of voorkomen, zodat grote hoeveelheden biomassa beschikbaar kunnen worden gemaakt terwijl broeikasgasemissies worden verminderd en klimaatverandering wordt beperkt [13].

Intensivering in de agrarische sector (verhoging van de productie op hetzelfde landoppervlak) wordt gezien als een belangrijke manier om overtollige landbouwgrond beschikbaar te maken voor de productie van bioenergiegewassen, waardoor ILUC wordt beperkt en daarmee de broeikasgasbalans van biomassawaardeketens wordt verbeterd. Daarnaast zou intensivering in the akkerbouw en veeteelt ook kunnen

bijdragen aan het verbeteren van de milieuprestaties van de agrarische sector zelf. Veel studies hebben de effecten onderzocht van landbouwintensivering op de beschikbaarheid van overtollige landbouwgrond en op het biomassapotentieel. Diverse studies hebben ook bestudeerd wat de broeikasgasbalans en andere milieu-effecten van landbouwintensivering en biomassaproductie zijn en/of wat de milieu- en economische prestaties van biomassawaardeketens zijn. Meer onderzoek is echter nodig op het gebied van regionale mogelijkheden voor landbouwintensivering, de invloed van verschillende intensiveringsroutes en de comparatieve prestatie van biomassawaardeketens over de tijd. Regionale mogelijkheden voor landbouwintensivering behelzen het tempo waarin intensivering plaats kan vinden en de invloed van technologische, economische en institionele factoren op dit tempo. Veeteelt speelt een belangrijke rol in agrarisch landgebruik, maar is relatief weinig onderzocht. Het is daarom in het bijzonder van belang om de mogelijkheden te onderzoeken voor intensivering in de veehouderij en de invloed daarvan op de regionale beschikbaarheid van overtollige landbouwgrond en op het biomassapotentieel. Daarnaast is het belangrijk om beter te begrijpen wat de invloed van verschillende intensiveringsroutes is op de broeikasgasbalans en milieuprestaties van zowel de agrarische sector als de biomassa sector. Tenslotte zou een vergelijking van biomassawaardeketens zowel de economische als de milieuprestaties van deze ketens moeten omvatten.

Doel en onderzoeksvragen

Dit proefschrift onderzoekt de regionale potenties, de milieu-effecten en economische prestatie van verschillende routes voor landbouwintensivering, biomassaproductie en biomassagebruik. Daartoe zijn de volgende onderzoeksvragen geformuleerd:

1. Wat zijn de regionale mogelijkheden voor landbouwintensivering en hoe beïnvloeden deze de vraag naar landbouwgrond en het biomassapotentieel?
2. Wat is het effect van verschillende routes voor landbouwintensivering en van biomassaproductie op broeikasgasemissies en het milieu?
3. Welke biomassawaardeketens worden geprefereerd met betrekking tot hun economische prestatie en broeikasgasbalans, zowel in het heden als in de toekomst?

Deze onderzoeksvragen worden behandeld in de Hoofdstukken 2 tot en met 6, zie Tabel 1. De onderzoeksvragen worden beantwoord in de volgende paragraaf.

TABEL 1 | Overzicht van de hoofdstukken in dit proefschrift en de onderzoeksvragen behandeld per hoofdstuk.

Hoofdstuk	Onderwerp	Regio(s)	Onderzoeksvraag		
			1	2	3
2	Analyse van de invloed van verschillende factoren op de ontwikkeling van opbrengsten en productiviteit in akkerbouw en veeteelt	Australië, Brazilië, China, India, VS, Zambia, Zimbabwe	x		
3	Analyse van de invloed van verschillende intensiveringsroutes voor melk- en rundveeproductie op broeikasgasemissies, grondgebruik en landgebruikverandering	Europa, Brazilië, Noord Amerika (gedetaileerde bottom-up data) 9 wereld regio's (gedisaggregeerde mondiale data en modelresultaten)	x	x	
4	Productiepotentieel van bioethanol uit miscanthus met een laag ILUC risico in de provincie Lublin, Polen.	Lublin (Polen)	x		
5	Broeikasgasemissies en andere milieu-effecten van ILUC mitigatie	Lublin (Polen)		x	x
6	Analyse en vergelijking van de economische en milieuprestaties van biomassawaardeketens voor warmte, elektriciteit, brandstoffen en materialen	Mondiaal		x	x

HOOFDBEVINDINGEN EN CONCLUSIES

ONDERZOEKSVRAAG 1:

Wat zijn de regionale mogelijkheden voor landbouwintensivering en hoe beïnvloeden deze de vraag naar landbouwgrond en het biomassapotentieel?

Dit proefschrift beschouwt drie aspecten van regionale mogelijkheden voor landbouwintensivering: ten eerste de mate waarin opbrengsten (bijvoorbeeld de productie per hectare) van akkerbouw en veeteelt verhoogd kunnen worden. Het tweede aspect is de snelheid waarmee intensivering plaats kan vinden. Het derde aspect heeft betrekking op factoren die landbouwintensivering en modernisering stimuleren of hinderen.

Potenties om opbrengsten te verhogen

Vanwege de verschillen in productiemethoden en grondgebruik, moet er een onderscheid gemaakt worden tussen akkerbouw en veeteelt om te onderzoeken wat de mogelijkheden zijn voor het verbeteren van de opbrengsten. In de akkerbouw wordt de mogelijkheid voor intensivering met name bepaald door locale factoren zoals bodemeigenschappen en het klimaat. In de afwezigheid van beperkende factoren zoals ziektes, wordt het verschil tussen de werkelijke opbrengst en de maximale agro-ecologische opbrengst

bepaald door de toegepaste technologiën en productiemethodes. Vergelijking met omringende regio's met een vergelijkbare potentiële agro-ecologische opbrengst laat zien in welke mate opbrengsten verbeterd kunnen worden door bepaalde technologiën en productiemethodes toe te passen. Deze aanpak is gebruikt in Hoofdstuk 4. In dit hoofdstuk zijn drie verschillende scenario's gedefineerd voor de mate van intensivering in de provincie Lublin in het oosten van Polen, gebaseerd op huidige opbrengsten in West-Polen en Duitsland. Deze scenario's laten zien dat de opbrengsten van verschillende gewassen in Lublin met 15% tot 100% verhoogd kunnen worden wanneer productiemethoden uit West-Polen en Duitsland worden overgenomen (bijvoorbeeld door mechanisatie, meer gebruik van kunstmest en schaalvergroting).

Voor veeteelt is het niet gebruikelijk en lastiger om een maximale opbrengst te defineren. Terwijl de opbrengsten van gewassen direct gerelateerd zijn aan grondgebruik en daarom direct gebruikt kunnen worden als graadmeter voor mogelijke intensivering, wordt de opbrengst van veeteelt meestal aangegeven als de melk- of vleesproductie per dier (bijv. kg_{melk} koe^{-1} jr^{-1}) of per eenheid veevoer (kg_{melk} kg_{voer}^{-1}, oftwel de efficiëntie waarmee veevoer wordt omgezet in melkproductie of de groei van dieren). Omdat deze indicatoren niet direct zijn gerelateerd aan grondgebruik, moeten ook de consumptie van veevoer en het grondgebruik voor de productie van dit voer meegenomen worden in de analyse van regionale intensiveringsmogelijkheden. Dit is gedaan in Hoofdstuk 3 om de mogelijkheden voor intensivering van melk- en rundvleesproductie te onderzoeken. Intensivering kan plaatsvinden binnen het huidige productiesysteem of door een overstap te maken van het ene systeem naar een ander systeem. Gebaseerd op bottom-up data heeft de overstap of transformatie van een op gras gebaseerd productiesysteem naar een gemengd productiesysteem (gebaseerd op gras en mengvoer) voor rundvlees de potentie om opbrengsten te verhogen met 0.02 $t_{rundvlees}$ ha^{-1} (ongeveer 125%) in de VS en met 0.18 $t_{rundvlees}$ ha^{-1} (±170%) in het Verenigd Koninkrijk. Voor dezelfde intensiveringsstrategie voor melkproductie in Oostenrijk is de potentie ongeveer 1.7 t_{melk} ha^{-1} (125%-140%). Gebaseerd op gedisaggregeerde data voor negen wereldregio's is de potentie om opbrengsten in Noord Amerika en Europa te verhogen door transformationele intensivering nog hoger. De opbrengsten van rundvleesproductie kunnen worden verhoogd met gemiddeld 0.3 $t_{rundvlees}$ ha^{-1} (780%) in Noord Amerika en 0.2 $t_{rundvlees}$ ha^{-1} (270%) in Europa. Voor melkproductie in Europa is een verbetering van gemiddeld 2.0 t_{melk} ha^{-1} (550%) mogelijk. In andere regio's zijn de relatieve verschillen in opbrengsten tussen gras-gebaseerde en gemengde systemen vaak hoger. Daarnaast bestaan er tussen regio's grote verschillen in de opbrengsten in een bepaald systeem. Dit suggereert dat in veel regio's de opbrengsten kunnen worden verhoogd door intensivering binnen het huidige productiesysteem. Vergelijkbaar met de akkerbouw is het dan mogelijk om de potenties voor intensivering van veeteelt te onderzoeken aan de hand van informatie over grondgebruik en

productiesystemen in andere regio's met hogere opbrengsten. In Hoofdstuk 4 is de potentiële mate van intensivering in de provincie Lublin dan ook gebaseerd op de huidige veeteelt in West-Polen en Duitsland. De analyse in dit hoofdstuk laat zien dat de runvlees- en melkproductie per dier in Lublin kan worden verhoogd met ongeveer 40%-55%. Daarnaast kan de dichtheid van rundvee op graslanden verhoogd kan worden met 15%-160%. Deze verbeteringen kunnen worden gerealiseerd door bijvoorbeeld de productiviteit van graslanden te verbeteren, de hoeveelheid mengvoer in het dieet te verhogen en door genetische selectie van vee.

Tempo van intensivering

De snelheid waarmee intensivering gerealiseerd kan worden is afhankelijk van de mate van intensivering die mogelijk is en de benodigde tijd om nieuwe technologiën aan te schaffen of te ontwikkelen. Om meer inzicht te krijgen in het mogelijke tempo van intensivering is het nuttig om naar historische ontwikkelingen te kijken, zoals is gedaan in Hoofdstuk 2. De bevindingen in dit hoofdstuk laten zien dat de snelheid waarmee opbrengsten in de akkerbouw en veeteelt in de afgelopen vijf decennia (1961-2010) werden verbeterd sterk verschilde tussen regio's. Over deze vijf decennia is meestal een lineaire toename in opbrengsten gevonden. Het hoofdstuk laat echter zien dat er gedurende kortere perioden verbeteringen plaatsvonden in een tempo dat hoger lag dan het gemiddelde. Ook waren er perioden waarin het tempo onder het gemiddelde lag. In regio's waar het verschil tussen de huidige en de maximale agro-ecologische opbrengsten groot is, is het daarom mogelijk om de opbrengsten tijdelijk versneld te verhogen. Hoofdstuk 4 laat zien dat het tempo van intensivering in de provincie Lublin significant versneld kan worden als bestaande productiemethoden uit West-Polen en Duitsland worden overgenomen tussen 2010 en 2020. Tabel 2 vergelijkt het tempo van opbrengstverbeteringen over de lange termijn, zoals gevonden in Hoofdstuk 2, met de geprojecteerde snelheid van verbeteringen in Lublin in de baseline and above-baseline scenario's[1]. De groeisnelheid van de opbrengsten in Lublin in het baseline scenario is over het algemeen in overeenstemming met de resultaten uit Hoofdstuk 2 en uit de Wit et al. [14]. Het groeitempo in de above-baseline scenario's is significant hoger in vergelijking met het baseline scenario.

1 In het baseline scenario wordt aangenomen dat de ontwikkelingen in landbouwopbrengsten tussen 2010 en 2020 overeenkomen met recente historische trends in de provincie. In de above-baseline scenario's wordt aangenomen dat in 2020 de opbrengsten gelijk zijn aan huidige, gemiddelde (2008-2012) opbrengsten in West-Polen en Duitsland, wat resulteert in een snellere toename in opbrengsten vergeleken met de baseline.

TABEL 2 | Regionale variatie in de historische groei van landbouwopbrengsten zoals gevonden in Hoofdstuk 2 (1961-2010) en geprojecteerde groei in de provincie Lublin (2010-2020) in de baseline en above-baseline scenario's (de percentuele groei is gegeven ten opzichte van 2010).

	Hoofdstuk 2			**Hoofdstuk 4 Lublin**			
	Lange termijn lineaire groei in geselecteerde landen in verschillende werelddelen (% jr^{-1} ten opzichte van 2010)[a]	Geprojecteerde groei in IMAGE (2010-2050) en MIRAGE (2008-2020) in geselecteerde landen in verschillende werelddelen (% jr^{-1})[a]	Lange termijn groei in de EU[d] (absoluut in kg ha^{-1} jr^{-1} of kg $dier^{-1}$ jr^{-1})	Baseline: lineaire groei in Lublin (% jr^{-1} ten opzichte van 2010)	Above-baseline: versnelde groei in Lublin (% jr^{-1} ten opzichte van 2010)	Baseline: lineaire groei in Lublin (absoluut in kg ha^{-1} jr^{-1} of kg $dier^{-1}$ jr^{-1})	Above-baseline: versnelde groei in Lublin ten opzichte van 2010 (absoluut in kg ha^{-1} jr^{-1} of kg $dier^{-1}$ jr^{-1})
Gewassen, gemiddelde variatie	0.7-1.6%[b]	-	-	0.6-2.5%[e]	1.6-10.6%	-	-
-Mais	1.2-1.6%[c]	0.9%-4.9%	130	1.8%	2.3-10.6%	100	135-390
-Tarwe	0.7-1.8%	0.5%-2.0%	40-110	1.3%	2.3-6.7%	45	85-385
Melk	1.0-1.5%	-	85	1.2%	1.2-5.6%	57	57-247
Rundvlees	0.4-0.8%	-	1.1-2.8	1.2%	1.2-5.4%	2.4	2.4-11.3

De groei in de opbrengst van gewassen is gebaseerd op de opbrengst in t ha^{-1}, de groei in de opbrengst van veeteelt is gebaseerd op de melk- en runvleesproductie in t $dier^{-1}$ yr^{-1}
[a] geselecteerde landen: Australië, Brazilië, China, India, VS, Zambia en Zimbabwe; [b] geselecteerde gewassen: mais, tarwe, rijst, soja, suikerriet; [c] uitgezonderd Zimbabwe (-1.6%); [d] hierin zijn ook resultaten uit de Wit et al. [14] meegenomen; [e] geselecteerde gewassen: tarwe, haver, rogge, gerst, triticale, aardappel, suikerbiet, mais, koolzaad.

Intensivering is niet alleen mogelijk voor traditionele landbouwgewassen, maar ook voor gewassen die specifiek voor bioenergie verbouwd worden. Ook intensivering van deze gewassen draagt bij aan een hoger biomassaproductiepotentieel. Hoofdstuk 6 laat zien dat de verwachte groei in de opbrengst van specifieke bioenergiegewassen hoger is in vegelijking met traditionele landbouwgewassen.Omdat in alle wereldregio's nog ervaring wordt opgedaan met het verbouwen van deze nieuwe gewassen, wordt verwacht dat de regionale verschillen in intensiveringsmogelijkheden kleiner zijn in vergelijking met traditionele gewassen. Kennis over de invloed van lokale agro-ecologische omstandigheden op de productie van specifieke bioenergiegewassen is echter nog beperkt.

Stimulansen en hindernissen voor landbouwintensivering

Wanneer intensivering en versnelde groei van opbrengsten mogelijk is, betekent dit niet automatisch dat deze mogelijkheden ook worden gerealiseerd. Hoofstuk 2 laat zien dat landbouwintensivering in het verleden primair een direct resultaat is geweest van technologische ontwikkelingen. Voor gewassen hadden zulke ontwikkelingen bijvoorbeeld betrekking op de toename in kunstmestgebruik, irrigatie en mechanisatie en het gebruik van nieuwe rassen met een hogere opbrengst. In de veeteelt werden hogere opbrengsten vaak gerealiseerd door een toename in de consumptie van mengvoer. Economische

en institutionele factoren hebben echter vaak een belangrijke rol gespeeld in de verbetering van landbouwtechnologiën. Ten eerste maakten investeringen in onderzoek en ontwikkeling (R&D) de ontwikkeling van nieuwe technologiën mogelijk. Ten tweede creëerde de openstelling van de economie nieuwe markten voor landbouwproducten, zoals bijvoorbeeld in Australië, wat boeren stimuleerde om hun opbrengsten (verder) te verhogen. Een groeiende binnenlandse of exportmarkt blijkt met name belangrijk te zijn geweest voor het verbeteren van opbrengsten in de melk- en rundvleessector. Ten derde konden nieuwe technologiën door boeren worden overgenomen dankzij ondersteuningsprogramma's, zoals in bijvoorbeeld India. In sommige gevallen werden hogere opbrengsten gerealiseerd door beleid dat op een specifiek gewas of product was gericht: bijvoorbeeld mais in Zambia en bioethanol in Brazilië. Het voorbeeld van Zimbabwe laat echter ook het gevolg zien van gebrek aan goed bestuur en beleid. Een burgeroorlog in de jaren '70 en economische hervormingen in de jaren '90 verstoorden de economie en de landbouwproductie. Dit had tot gevolg dat de opbrengsten van gewassen sterk fluctueerden in vergelijking met andere, stabielere landen.

Het wordt aangenomen dat het technisch mogelijk is om tussen 2010 en 2020 in Lublin grote verbeteringen in landbouwopbrengsten te realiseren, omdat de above-baseline projecties zijn gebaseerd op bestaande landbouwmethoden in West-Polen en Duitsland. Ook laat Hoofdstuk 2 zien dat er tijdelijk versnelde groei in opbrengsten mogelijk is onder gunstige economische en bestuurlijke omstandigheden die technologische ontwikkelingen stimuleren. In Hoofdstuk 4 werden echter ook verschillende factoren geïdentificeerd die intensivering in Lublin zouden kunnen hinderen. De agrarische sector in Lublin wordt gekarakteriseerd door een groot aantal kleine boerenbedrijven en beperkt gebruik van moderne productiemethoden. Opschaling, modernisering en mechanisatie zijn daarom nodig, maar boeren hebben weinig kapitaal om te investeren. Ook zijn de prijzen van landbouwgrond te laag om de verkoop of pacht hiervan te stimuleren. Om significante intensivering te realiseren, zou bestuur en beleid zich bijvoorbeeld moeten richten op het financieel steunen van boeren om productiemethoden te verbeteren. In Europees en Pools landbouw- en plattelandsbeleid bestaat financiële steun aan boeren al, maar deze steun zou vergroot moeten worden om de potentie voor intensivering optimaal te bewerkstelligen. Wanneer voldoende finaciële steun gegeven zou worden, zal de snelheid waarmee opbrengsten verhoogd kunnen worden waarschijnlijk met name afhangen van de tijd die nodig is voor landhervormingen.

Invloed van intensivering op de vraag naar landbouwgrond

De case studie naar ILUC preventie in Lublin (Hoofdstuk 4) laat zien dat door intensivering in de akkerbouw en veeteelt ongeveer 100-400 duizend hectare landbouwgrond vrij kan worden gemaakt (5-24% van het totale landbouwareaal). Hoofdstuk 3 laat zien

dat, gebaseerd op de regionale potenties voor intensivering van veeteelt, grondgebruik significant kan worden verminderd in alle wereldregio's. Gebaseerd op de bottom-up data kan grondgebruik (in m^2 $kg_{runvlees}^{-1}$ of m^2 kg_{melk}^{-1}) in Europa en de VS met 20%-40% verminderd worden wanneer wordt overgestapt van een op gras gebaseerd systeem naar een gemengd systeem. Gebaseerd op de gedisaggregeerde data voor negen wereldregio's is de potentiële reductie in grondgebruik voor deze intensiveringsroute nog hoger: voor rundvleesproductie ongeveer 90% in de VS en 60% in Europa en voor melkproductie in Europa ongeveer 80%. Voor rundvlees- en melkproductie in andere regio's varieert de potentie tussen de 60% en 95% voor dezelfde intensiveringsroute.
Hoewel landbouwintensivering een grote hoeveelheid land beschikbaar kan maken voor biomassaproductie, moet in aanmerking worden genomen dat de hoeveelheid landbouwgrond die beschikbar is voor biomassa niet alleen afhangt van landbouwintensivering. Andere factoren die van invloed zijn, zijn onder andere concurrerende gebruiken van de grond, bijvoorbeeld bebossing, en ook de ontwikkeling in de vraag naar landbouwproducten. In de case studie voor Lublin wordt verwacht dat de vraag naar landbouwproducten afneemt tussen 2010 en 2020, gebaseerd op projecties op Europese schaal. Wanneer daarbij wordt aangenomen dat landbouwopbrengsten zullen toenemen, in lijn met recente historische ontwikkelingen (baseline scenario), dan betekent dit dat de vraag naar landbouwgrond in Lublin sterk zal afnemen. Deze afname is echter niet in lijn met recente ontwikkelingen in Lublin en Polen. Om deze reden werden verschillende scenario's voor grondgebruik toegepast in de case studie. Het wordt aanbevolen om te onderzoeken hoe in toekomstige studies de regionale projecties voor landbouwproductie kunnen worden verbeterd.

In modelanalyses van de beschikbaarheid van overtollige landbouwgrond voor biomassaproductie blijken de projecties van landbouwopbrengsten vaak te zijn gebaseerd op de basisaanname dat toekomstige intensivering in lijn zal zijn met historische trends. Hoofdstuk 2 laat zien dat dit over de lange termijn vaak klopt, maar dat snellere groei van opbrengsten mogelijk is over kortere perioden. In de analyse van kortere termijn potenties voor biomassaproductie is het dus van belang om regionale mogelijkheden voor de groei van opbrengsten mee te nemen. Ook is het belangrijk om te onderzoeken hoe het tempo van intensivering de beschikbaarheid van land en het biomassapotentieel beïnvloed, zoals gedaan in Hoofdstuk 4. Naast de basisaanname veronderstellen de meeste modellen dat de mondiale groei in landbouwopbrengsten in de toekomst zal afnemen. Redenen voor deze verwachting zijn dat i) de mogelijkheden om opbrengsten verder te verhogen steeds beperkter worden, ii) investeringen in R&D op het gebied van landbouw zijn afgenomen, en iii) socio-economische beperkingen voor het verbeteren van opbrengsten in veel ontwikkelingslanden zullen blijven bestaan. Hoewel deze redenen goed onderbouwd kunnen worden, zijn het slechts verwachtingen van hoe verschillende factoren zich zullen

ontwikkelen. Er bestaan mogelijk ook andere routes. In regio's waar er nog steeds veel mogelijkheden zijn om landbouwopbrengsten te verhogen kunnen stimulerend beleid en andere factoren bijdragen aan het realiseren van de potenties. Het is daarom van belang verschillende routes te onderzoeken die laten zien hoe de ontwikkeling van opbrengsten afhangt van verschillende aannames en omstandigheden en wat de invloed daarvan is op toekomstig regionaal grondgebruik en het biomassaproductiepotentiaal. Omdat de ontwikkeling van landbouwopbrengsten afhangt van regionale factoren zijn gedetailleerde, regionale analyses van deze factoren nodig. De modellen gebruiken nu een gelimiteerd aantal endogene factoren en zouden moeten worden aangevuld met meer endogene factoren die uit de regionale analyses naar voren komen. Op deze manier kunnen de modelresultaten gebruikt worden om te identificeren aan welke regionale voorwaarden voldaan moet worden om een bepaalde verbetering in landbouwopbrengsten te behalen. Deze bevinden kunnen vervolgens gebruikt worden om regionale strategiën voor landbouwintensivering te ontwikkelen.

Invloed van intensivering op het biomassapotentieel

De analyse in Hoofdstuk 4 liet zien dat de potentie voor landbouwintensivering in Lublin groot is en dat op deze manier een grote hoeveelheid landbouwgrond vrijgemaakt kan worden voor biomassaproductie. Echter, gebaseerd op criteria voor natuurbescherming en eisen aan de geschiktheid van land voor het verbouwen van miscanthus kan maar een beperkte hoeveelheid land van 200 tot 270 duizend hectare (12-15% van het totale lanbouwareaal) gebruikt worden voor miscanthusproductie. De hoeveelheid land wordt met name beperkt door de eisen aan de geschiktheid van land voor miscanthus en in het bijzonder door de gevoeligheid van miscanthus voor waterstress. De geschiktheid van het land was echter gebaseerd op een beperkt aantal criteria, zoals bijvoorbeeld de minimale grondwaterstand, en zou in meer detail moeten worden beoordeeld. Op het land dat beschikbaar komt door intensivering en dat geschikt is voor miscanthus kan 1.2 tot 3.5 miljoen ton miscanthus worden geproduceerd in 2020 (bij een gemiddelde opbrengst voor miscanthus van 13 t ha^{-1}). Daaruit kan op jaarbasis 8 tot 23 PJ bioethanol (340 tot 970 miljoen liter jr^{-1}) gemaakt worden. Een gevoeligheidanalyse voor de gemiddelde miscanthusopbrengst en de totale efficiënctie van de ethanolketen laat zien dat het bioethanolproductiepotentieel tussen de 6 en 35 PJ jr^{-1} (260 to 1,480 miljoen liter jr^{-1}) ligt. Wanneer ook landbouwgrond wordt meegenomen dat beschikbaar wordt gemaakt door verbeterde efficiëntie van voedselketens en gebruik van onderbenut land, dan ligt het totale potentieel tussen 12 en 35 PJ jr^{-1}, afhankelijk van de efficiënctie van de ethanolketen. Ter vergelijking, Polen heeft zich tot doel gesteld om in 2020 bijna 9 PJ bioethanol uit tweede generatie biomassa te gebruiken. Daarnaast streeft het land ernaar om in hetzelfde jaar in totaal 60 PJ biobrandstoffen te gebruiken. Het is wel belangrijk om te noemen dat het bioethanolpotentieel dat hier berekend is voor Lublin het technisch

potentieel is. Het duurzame potentieel zou lager kunnen liggen. De resultaten laten echter duidelijk zien dat landbouwintensivering een belangrijke rol kan spelen in het realiseren van de nationale doelen voor biobrandstofgebruik.

ONDERZOEKSVRAAG 2:

Wat is het effect van verschillende routes voor landbouwintensivering en van biomassaproductie op broeikasgasemissies en het milieu?

Voor landbouwintensivering zijn verschillende routes mogelijk, variërend van conventionele tot duurzame intensivering voor gewassen en van incrementele tot transformationele intensivering voor veeteelt. De milieueffecten van deze verschillende strategiën lopen behoorlijk uiteen. Wanneer bijvoorbeeld de opbrengsten van gewassen worden verhoogd door het gebruik van meer kunstmest, veroorzaakt dit extra broeikasgasemissies. Wannneer de opbrengsten worden verhoogd door het toepassen van precisielandbouw, kunnen kunstmestgebruik en broeikasgasemissies afnemen. Omdat intensivering nodig is om het verbouwen van gewassen voor bioenergie en biomaterialen mogelijk te maken, moeten de effecten van intensivering aan de biomassawaardeketens worden toegeschreven en mee worden genomen in de netto effecten van deze ketens. Dit proefschrift heeft dit punt behandeld door in Hoofstuk 3 te analyseren wat de broeikasgaseffecten zijn van verschillende intensiveringsroutes voor melk- en rundvleesproductie en in Hoofdstuk 5 te onderzoeken wat de de netto milieueffecten zijn van landbouwintensivering en biomassaproductie.

Broeikasgaseffecten van intensiveringsroutes voor melk- en rundvleesproductie

In Hoofdstuk 3 is onderzocht wat de invloed is van twee intensiveringsroutes voor melk- en rundvleesproductie op de broeikasgasemissies, zowel inclusief als exclusief de effecten van landgebruikverandering. Deze routes zijn i) incrementele intensivering binnen het bestaande productiesysteem (waarbij een onderscheid wordt gemaakt tussen gras-gebaseerde, gemengde en industriële productiesystemen), en ii) transformationele intensivering waarbij een overstap wordt gemaakt van één productiesysteem naar een ander productiesysteem. Gebaseerd op de bottom-up data voor melkproductie in Europa varieert de potentie om broeikasgasemissies exclusief LUC te verminderen van 1%-14% (0.01-0.13 $kgCO_2$-eq kg_{melk}^{-1}) voor incrementele intensivering en van 2%-26% voor systeem transities (0.02-0.30 $kgCO_2$-eq kg_{melk}^{-1}). De potenties voor beide routes liggen vaak dicht bij elkaar. Welke intensiveringsroute de meeste invloed zal hebben op de broeikasgasbalans hangt af van het specifieke ontwerp van het huidige en het toekomstige productiesysteem. Wanneer bijvoorbeeld het gras-gebaseerde systeem wordt gekarakteriseerd door extensief graslandmanagement kan een grotere reductie van broeikasgasemissies behaald worden door dit graslandmanagement te verbeteren dan door het aandeel

van mengvoer in het dieet te verhogen. Maar wanneer het gras-gebaseerde systeem wordt gekarakteriseerd door verbeterd graslandmanagement heeft de transitie naar een gemengd system mogelijk een gotere reductiepotentie dan verdere intensivering binnen het gras-gebaseerde systeem. Gebaseerd op gedisaggregeerde, mondiale data geldt dit ook voor melkproductie in andere ontwikkelde regio's. In ontwikkelingregio's is de potentie om emissies te verminderen vooral groot voor intensivering binnen het gras-gebaseerde productiesysteem. In bijvoorbeeld gras-gebaseerde systemen in Brazilië en Sub-Sahara-Afrika zijn de broeikasgasemissies per kg melk tot 3 of 5 keer zo hoog (1.5 tot 2.5 $kgCO_2$-eq kg_{melk}^{-1}) in vergelijking met ontwikkelde regio's. Om deze broeikasgasemissies te reduceren is het belangrijk om de kwaliteit van het veevoer te verhogen, bijvoorbeeld door verbetering van graslandmanagement. De potentie om emissies verder te verminderen door over te stappen op een gemengd systeem is in ontwikkelingsregio's vaak beperkt.

Met betrekking tot rundvleespoductie laat een analyse van de bottom-up data en van de gedisaggregeerde mondiale data zien dat de variatie in broeikasgasemissies binnen en tussen productiesystemen vaak groter is dan voor melkproductie. Dit betekent dat in zowel ontwikkelde als ontwikkelingslanden er een significante potentie bestaat voor het verminderen van broeikasgasemissies door incrementele intensivering binnen gras-gebaseerde en gemengde systemen. De reductiepotentie van intensivering binnen het gras-gebaseerde systeem kan in Bazilië bijvoorbeeld oplopen tot meer dan 50% (30 $kgCO_2$-eq $kg_{rundvlees}^{-1}$) wanneer wordt overgegaan van extensief, natuurlijk grasland naar verbeterd grasland. Daarnaast is de potentie om emissies verder te verminderen door over te stappen op een gemengd systeem groter voor runvleesproductie dan voor melkproductie. Hoewel intensivering van melk- en runvleesproductie meestal resulteert in vermindering van broeikasgasemissies zijn er ook een aantal uitzonderingen gevonden in de bottom-up studies. In deze gevallen zijn de emissies voor biologische of extensieve productie lager dan voor meer intensieve, conventionele productie. Redenen hiervoor zijn dat er in het biologische of extensieve productiesysteem geen kunstmest gebruikt wordt, het gebruik van fossiele brandstoffen lager is en er koolstof wordt opgeslagen in de bodem. Dit onderstreept het belang van efficient grondstoffengebruik.

Met betrekking tot broeikasgasbalansen <u>inclusief LUC</u> is transformationele intensivering van gras-gebaseerde naar gemengde productiesystemen van groot belang om emissies te verminderen. In gras-gebaseerde systemen is het grondgebruik aanzienlijk hoger dan in gemengde systemen, waardoor de bijdrage aan landgebruikverandering en gerelateerde emissies in het algemeen groter is voor gras-gebaseerde systemen. Alleen in Europa zijn de LUC-gerelateerde emissies hoger voor gemengde systemen dan voor gras-gebaseerde systemen. De reden hiervoor is de import van veevoer uit Zuid-Amerika en

de landgebruikverandering die hiermee in verband wordt gebracht. Voor melkproductie is de hoeveelheid LUC-gerelateerde emissies het hoogst in Latijns-Amerika en Sub-Sahara-Afrika, waar deze emissies een bijdrage leveren van 40% en 50% aan de totale emissies. Door over te stappen van gras-gebaseerde naar gemengde systemen kunnen de LUC-gerelateerde emissies verminderd worden met respectievelijk 1.5 $kgCO_2$-eq kg_{melk}^{-1} (>60%) in Latijns-Amerika en 3.2 $kgCO_2$-eq kg_{melk}^{-1} (>50%) in Sub-Sahara-Afrika. Voor rundvleesproductie is de hoeveelheid LUC-gerelateerde emissies het hoogst in Sub-Sahara-Afrika. De emissiereductiepotentie voor een systeemtransitie in deze region is 280 $kgCO_2$-eq $kg_{rundvlees}^{-1}$ (50%). Echter, in vergelijking met andere ontwikkelingsregio's zijn de LUC-gerelateerde emissies van gemengde systemen erg hoog en ook deze systemen moeten verbeterd worden om grondgebruik en broeikasgasemissies te verminderen.

Netto effecten van landbouwintensivering en biomassaproductie

Wanneer lanbouwintensivering wordt gecombineerd met biomassaproductie blijkt de intensiveringsroute een grote invloed te hebben op de netto broeikasgasemissies van landbouw en biomassaproductie. In Hoofdstuk 5 is het effect van drie ILUC mitigatiescenario's en drie intensiveringsroutes in de provincie Lublin onderzocht. De drie ILUC mitigatiescenario's representeren een laag, modaal en hoog productiepotentieel voor bioethanol uit miscanthus, de drie intensiveringsroutes representeren verschillende maten van duurzaamheid. De invloed van de intensiveringsroutes blijkt aanzienlijk te zijn voor elk ILUC mitigatiescenario. In bijvoorbeeld het modale ILUC mitigatiescenario is de netto broeikasgasbalans in 2020 gelijk aan 2.3 $MtCO_2$-eq jr^{-1} (1.8 tCO_2-eq ha^{-1} jr^{-1}) voor conventionele intensivering, 0.6 $MtCO_2$-eq jr^{-1} (0.4 tCO_2-eq ha^{-1} jr^{-1}) voor beperkt duurzame intensivering en -0.8 $MtCO_2$-eq jr^{-1} (-0.6 tCO_2-eq ha^{-1} jr^{-1}) voor duurzame intensivering. Ter vergelijking, in het referentiescenario dat geen ILUC mitigatie en biomassaproductie toepast is de netto emissiebalans in 2020 gelijk aan 3.0 $MtCO_2$-eq jr^{-1} (2.3 tCO_2-eq ha^{-1} jr^{-1}). De belangrijkste factoren voor deze verschillen zijn de efficiëntie waarmee voedingsstoffen uit kunstmest worden gebruikt voor de groei van gewassen, de emissiefactoren voor stikstof en de verandering van de hoeveelheid koolstof in de bodem door de manier waarop het land wordt beheerd. Wanneer de netto broeikasgasemissies toegeschreven worden aan bioethanol blijken de ILUC mitigatiescenario's in veel gevallen de uitstoot van emissies te verminderen ten opzichte van benzine. Om emissiereducties van meer dan 90% te behalen zijn duurzame intensiveringsroutes echter noodzakelijk. Een kwalitatieve analyse van andere milieu-effecten onderstreept ook het belang van duurzame intensivering. De analyse laat zien dat conventionele intensivering matige tot hoge risico's kan vormen voor biodiversiteit, waterkwaliteit en -kwantiteit, bodemkwaliteit en luchtkwaliteit. Duurzame intensivering kan grotendeels positieve effecten hebben op deze aspecten. Deze bevindingen tonen het belang om de invloed van verschillende intensiveringsroutes te onderzoeken in toekomstige studies die

betrekking hebben op de milieu-effecten van biomassa. Bovendien zouden de effecten van intensivering in zowel de akkerbouw als veeteelt moeten worden meegenomen. De methode die ontwikkeld is in Hoofdstuk 5 biedt een geïntegreerde en gedetailleerde aanpak die ook toegepast kan worden in toekomstige studies. De methode zou verder kunnen worden verbeterd door deze ruimtelijk expliciet te maken. Vanwege het belang van duurzame intensiveringsroutes voor het verbeteren van de broeikasgas- en andere milieu-effecten van biomassawaardeketens zouden duurzaamheidscriteria (bijvoorbeeld met betrekking op watergebruik en bodemkwaliteit) niet alleen van toepassing moeten zijn op biomassaproductie, maar op alle agrarische activiteiten in de regio.

De bespreking van biomassawaardeketens in Hoofdstuk 6 laat zien dat biobrandstoffen in veel gevallen broeikasgasemissies verminderen ten opzichte van hun fossiele referentie. De reductie is echter over het algemeen lager voor eerste generatie biobrandstoffen (geproduceerd uit conventionele landbouwgewassen zoals mais en koolzaad) dan voor bioethanol en biodiesel geproduceerd uit gewassen die specifiek voor bioenergie verbouwd worden (bijvoorbeeld miscanthus). Een belangrijke reden hiervoor is dat het verbouwen van conventionele landbouwgewassen een grote invloed heeft op de totale broeikasgasbalans van de biomassawaardeketen. Duurzame landbouwintensivering kan daarom ook bijdragen aan het verbeteren van de broeikasgasbalans en het emissiereductiepotentieel van eerste generatie biobrandstoffen.

Omdat de efficiëntie van het gebruik van grondstoffen, met name van voedingsstoffen, een aanzienlijke invloed heeft op de milieu-effecten van landbouw en bioenergie is het van belang om verbetering van deze efficiënties na te streven. De historische analyse in Hoofdstuk 2 laat zien dat agro-milieubeleid hierin vaak een prominente rol heeft gespeeld. Dit beleid streefde bijvoorbeeld naar verbeterde kwaliteit van gedegradeerde gronden (Australië, China, Zambia, Zimbabwe), gebalanceerd gebruik van grondstoffen (China) en gecontroleerd gebruik en beheer van natuurlijke bronnen (Australië, India, VS). In India bleef het gebruik van kunstmest per hectare echter toenemen als gevolg van zwakke handhaving van agro-milieubeleid.

ONDERZOEKSVRAAG 3:

Welke biomasssawaardeketens worden geprefereerd met betrekking tot hun economische prestatie en broeikasgasbalans, zowel in het heden als in de toekomst?

Biomassawaardeketens bestaan uit verschillende componenten, waaronder productie, voorbehandeling, transport en conversie van biomassa. De kosten van elk component, de kosten en de broeikasgasbalans van complete waardeketens in vergelijking met een fossiele referentie en de broeikasgasreductiekosten zijn bestudeerd en besproken in

Hoofdstuk 6 van dit proefschrift. Daarnaast is in Hoofdstuk 5 de broeikasgasbalans van ethanol uit miscanthus geanalyseerd en is onderzocht hoe deze balans verandert als de emissies gerelateerd aan landbouw worden toegeschreven aan de biomassawaardeketen en wat de invloed is van verschillende landbouwintensiveringsroutes.

Economische prestatie vergeleken met de fossiele referentie

De huidige en geprojecteerde (2030) kosten voor het produceren en beschikbaar maken van biomassa voor transport en opslag zijn lager voor gewassen met een hoge opbrengst dan voor gewassen met een lagere opbrengst. Voor de voorbehandeling van biomassa bestaan verschillende technologiën en er worden ook nog steeds nieuwe technologën ontwikkeld. Pelletisering wordt veel toegepast en is in het algemeen goedkoper dan het nieuwere torrefactie gecombineerd met pelletisering. Torrefactie kan in de toekomst echter de kosten van transport en conversie reduceren.

Op het moment zijn de kosten van de meeste biomassawaardeketens niet concurrerend met de kosten van hun fossiele referentie. Grootschalige biomassasystemen voor *warmte* kunnen alleen concurreren met duurdere fossiele systemen. De gemiddelde kosten voor 5 MW stadsverwarming zijn bijvoorbeeld 18 $ GJ_{th}^{-1} (voor een houtpelletprijs van 9 $\$_{2009}$ GJ^{-1}) vergeleken met 8-20 $ GJ_{th}^{-1} voor fossiele systemen. Het wordt verwacht dat grootschalige biomassasystemen in 2030 ook kunnen concurreren met goedkopere fossiele systemen. Met een houtpelletprijs van 9 $\$_{2009}$ GJ^{-1} zijn geen van de biomassawaardeketens voor *elektriciteitsproductie* en warmtekrachtkoppeling (WKK) concurrerend met fossiele electriciteitsproductie. Het verschil in kosten is het laagst voor het meestoken van houtpellets. In 2030 zouden biomassawaardeketens gebaseerd op geavanceerde vergassingstechnologiën (bijvoorbeeld indirecte bijstook van biomassa) concurrerend kunnen worden met fossiele electriciteitsproductie. Met betrekking tot eerste generatie biobrandstoffen is grootschalige (1000 MW_{feed}) productie van ethanol uit suikerriet in Brazilië (9-12 $ GJ_{EtOH}^{-1}) het meest concurrerend met benzine (16 $ $GJ_{gasoline}^{-1}$ bij een olieprijs van 75 $ bbl^{-1})[2]. De productiekosten van ethanol uit mais en diesel uit soja liggen net iets hoger dan de kosten van fossiele benzine en diesel (17-21 $ $GJ_{biobrandstof}^{-1}$ vs. 16 $ $GJ_{fossiele\ brandstof}^{-1}$). Met betrekking tot tweede generatie biobrandstoffen zou de commerciële productie van thermochemische waterstof en methanol op de korte termijn concurrerend kunnen worden met benzine en diesel. Op de langere termijn wordt verwacht dat alle waardeketens voor tweede generatie biobrandstoffen concurrerend worden. De productie van biomaterialen uit goedkope suikerriet uit Brazilië is op dit moment al concurrerend en huidige kosten van ethylene, PLA, PHA en PTT zijn 270-865 $ t^{-1} lager in vergelijking met hun fossiele referentieproducten (suiker 141 $ $t_{ferm.suiker}^{-1}$, olie

2 Op het moment zijn fossiele brandstofprijzen lager dan wordt aangenomen in Hoofdstuk 6. Wat hiervan de invloed is op de resultaten zal worden besproken in de sectie over broeikasgasreductiekosten.

75 $\$_{2009}$ bbl_{olie}^{-1}). Op de lange termijn neemt dit verschil toe tot ongeveer 940-1600 $\$ t^{-1}$ (suiker 141 $\$ t_{ferm.suiker}^{-1}$, olie 110 $\$_{2009}$ bbl_{olie}^{-1}).

Avoided GHG emissions

Hoofdstuk 6 onderzocht de broeikasgasreductie door biomassawaardeketens wanneer LUC buiten beschouwing wordt gelaten. De meeste waardeketens verminderen emissies ten opzichte van hun fossiele referentie. Voor huidige en toekomstige (2030) huis- en stadsverwarming is het emissiereductiepotentieel ongeveer 80 gCO_2-$eq_{avoided}$ MJ_{th}^{-1} of 85%. Met betrekking tot elektriciteitsproductie hebben kleinschalige warmtekrachttechnologiën een lage elektriciteit-warmte ratio, waardoor deze toepassingen een hoge credit voor warmte krijgen. Daardoor hebben deze technologiën het hoogste emissiereductiepotentieel (>300 gCO_2-$eq_{reductie}$ MJ_e^{-1} of >250%). Voor eerste generatie biobrandstoffen is de productie van ethanol uit suikerriet de beste optie om broeikasgasemissies te verminderen (66-81 gCO_2-$eq_{reductie}$ MJ_{EtOH}^{-1} of ongeveer 70%-90%). Hogere reducties kunnen worden gerealiseerd door tweede generatie bioethanol en FT diesel (72-91 gCO_2-$eq_{reductie}$ $MJ_{brandstof}^{-1}$ of ongeveer 80%-100%). Ethanol uit suikerriet en FT diesel hebben ook op de langere termijn de voorkeur omdat de emissiereductiepotenties zullen toenemen (71-114 gCO_2-$eq_{reductie}$ $MJ_{brandstof}^{-1}$ of ongeveer 90%-150%). Van de huidige biomaterialen is de emissiereductie het grootst voor ethyleen uit suikerriet (4.5-5.3 tCO_2-$eq_{reductie}$ $t_{ethyleen}^{-1}$ of ongeveer 100%-120%). Op de langere termijn wordt verwacht dat ook butanol uit suikerriet en ethyleen en FT olefinen uit tweede generatie biomassa een hoog emissiereductiepotentieel kunnen behalen (ongeveer 100%-185%).

Hoofstuk 5 laat zien dat, wanneer de emissies uit de landbouw buiten beschouwing worden gelaten, ethanolproductie uit miscanthus de broeikasgasemissies in 2020 met meer dan 100% reduceert ten opzichte van benzine. Deze reductie is primair het gevolg van de hoge credits voor de elektriciteit die wordt opgewekt tijdens de productie van ethanol. Deze credits zijn met name hoog vanwege de hoge emissiefactor voor de Poolse elektriciteitsproductie. Wanneer de broeikasgasssen uit de landbouw wel in de analyse worden meegenomen resulteerd dit over het algemeen in een toename in de netto emissies voor bioethanol. Toch verminderd bioethanol in de meeste gevallen nog steeds emissies ten opzichte van benzine. De netto broeikasgasbalans wordt echter aanzienlijk beïnvloed door de route voor landbouwintensivering. In de scenario's met een modaal en hoog potentieel voor bioethanolproductie heeft de duurzame intensiveringsroute zelfs een positief effect op de netto broeikasgasbalans. Dit betekent dat de reductie van emissies ten opzichte van benzine toeneemt wanneer de effecten van landbouwintensivering worden meegenomen in de broeikasgasbalans. In deze gevallen is de emissiereductiepotentie ongeveer 145% in het modale scenario en 190% in het hoge scenario. De postieve invloed van landbouwintensivering op de de netto broeikasgasbalans is met name het resultaat

van de relatief lage emissies gerelateerd aan kunstmestgebruik en veeteelt in vergelijking met de andere intensiveringsroutes en van de opslag van koolstof in de bodem door de toepassing van methoden die de bodemkwaliteit verbeteren (bijvoorbeeld alternatieve grondbewerkingssystemen).

Broeikasgasreductiekosten

De meeste huidige biomassawaardeketens kunnen op het gebied van kosten niet concurreren met fossiele waardeketens, maar verminderen vaak wel emissies ten opzichte van deze fossiele referenties. Dit resulteert in positieve broeikasgasreductiekosten voor de biomassawaardeketens. Sommige huidige biomassawaardeketens zijn wel concurrerend en in combinatie met emissiereducties resulteert dit in negatieve broeikasgasreductiekosten. Dit zijn de ketens voor grootschalige ethanolproductie uit suikerriet, biochemische ethyleenproductie uit suikerriet en uit lignocellulose-houdende (tweede generatie) biomassa, PLA-productie uit suikerriet en mais en olefineproductie op basis van vergassing van lignocellulose-houdende biomassa. Deze waardeketens zijn echter concurrerend met hun fossiele referentie bij een olieprijs van 75 $ bbl^{-1}, terwijl de huidige olieprijs rond de 50 $ bbl^{-1} ligt. In Hoofdstuk 6 is een gevoeligheidsanalyse voor een olieprijs van 60 $ bbl^{-1} uitgevoerd voor een geselecteerd aantal biomassawaardeketens. De resultaten van deze analyse laten zien dat ethanol uit suikerriet concurrerend blijft met benzine, maar het omslagpunt komt wel dichtbij. Ethyleen uit suikerriet kan niet langer concurreren met zijn fossiele referentie. De andere ketens waren niet meegenomen in de gevoeligheidsanalyse. Gebaseerd op het huidige verschil in productiekosten voor een olieprijs van 75 $ bbl^{-1} wordt verwacht dat PLA concurrerend kan blijven bij een olieprijs van 60 $ bbl^{-1}, maar dat dit niet het geval zal zijn voor ethyleen en olefinen uit lignocellulose-houdende biomassa.

Gebaseerd op een gelimiteerd aantal projecties voor de lange termijn, wordt verwacht dat tenminste voor de volgende biomassawaardeketens negatieve broeikasgasreductiekosten kunnen worden gerealiseerd rond 2030: stadsverwarming, bij- en meestook van biomassa, IGCC, biochemische ethanol en thermochemische FT diesel uit lignocellulose-houdende biomassa en butanol uit suikerriet. Bij lagere biomassaprijzen, hogere fossiele brandstofkosten of lagere rentepercentages zouden ook de waardeketens voor FT dieselproductie op korte termijn en voor huisverwarming en gasmotor-WKK in 2030 negatieve broeikasgasreductiekosten kunnen behalen.

Slotoverwegingen

Gebaseerd op de resultaten uit Hoofdstuk 6 worden de biomassawaardeketens voor ethanol, ethyleen en PLA op het moment geprefereerd met betrekking op hun economische en milieuprestatie. Andere goede opties zijn stadsverwarming en de meestook van

biomassa in bestaande elektriciteitscentrales. Op de langere termijn blijft fermentatie van goedkoop suikerriet aantrekkelijk voor de productie van ethanol en materialen met hoge broeikasgasreductiepotenties. Daarnaast kunnen lignocellulose-houdende gewassen en geavanceerde conversietechnologiën een belangrijke rol gaan spelen in de productie van energie en materialen met goede economische en milieuprestaties. Deze technologiën zijn veelzijdig: biochemische conversie kan worden ingezet voor de productie van zowel biobrandstoffen als biomaterialen en bij vergassing kunnen veel verschillende soorten biomassa worden verwerkt tot een breed scala aan producten waaronder warmte, elektriciteit, brandstoffen en materialen. Maar hoewel de resultaten goede perspectieven tonen voor zowel tweede generatie biomassa en technologiën en voor biomaterialen, is er nog veel onzekerheid over hun werkelijke prestaties, met name met betrekking tot de kosten. De kostenprojecties voor tweede generatie biomassa en conversietechnologiën zijn gebaseerd op aannamen over de ontwikkeling van kosten voor biomassaproductie en –levering en over technolgische ontwikkelingen. Zowel technologische als economische ontwikkelingen zijn voorwaarde voor grootschalige commerciële toepassing van tweede generatie biomassa en technologiën, maar de ervaring met deze technologiën is nog beperkt. Om de economische en milieuprestatie van toekomstige biomassawaardeketens te verbeteren is het daarom is het erg belangrijk om deze technologiën te stimuleren en erin te investeren. Wat betreft biomaterialen is de onzekerheid over kosten vooral gerelateerd aan een gebrek aan goede data. In toekomstig onderzoek is het daarom van belang om meer data te verzamelen en verder te onderzoeken hoe en in welke snelheid kostenreducties gerealiseerd kunnen worden.

Tenslotte kan de integratie van verschillende conversietechnologiën in bioraffinaderijen het gebruik van alle biomassacomponenten optimaliseren en de productie van een breed palet van producten mogelijk maken, waaronder voedsel, veevoer, chemicaliën, materialen, brandstoffen, elektriciteit en warmte. Gebaseerd op de literatuur die in Hoofdstuk 6 is besproken zouden bioraffinaderijen ook de economische en milieuprestatie van de biomassawaardeketens kunnen verbeteren. Verder onderzoek en demonstratie van bioraffinageconcepten is nodig om meer ervaring op te doen en deze concepten geleidelijk te optimaliseren.

AANBEVELINGEN VOOR TOEKOMSTIG ONDERZOEK EN VOOR BELEIDSMAKERS

Gebaseerd op de bevindingen van dit proefschrift kunnen de volgende aanbevelingen worden gegeven.

TOEKOMSTIG ONDERZOEK

- In modelanalyses van de potenties en effecten van biomassaproductie blijken de projecties van landbouwopbrengsten vaak te zijn gebaseerd op historische trends. Er wordt slechts een beperkt aantal endogene factoren meegenomen in deze projecties. De consequentie is dat regionale mogelijkheden om landbouwopbrengsten te verbeteren en om de groei van deze opbrengsten tijdelijk te versnellen onvoldoende worden meegenomen in de analyses. Het wordt daarom aanbevolen om meer endogene factoren en verschillende variabelen voor scenario's te implementeren in de modellen. Deze factoren en variabelen zouden gebaseerd kunnen worden op regionale analyses zoals uitgevoerd in Hoofdstuk 2. Aan de hand van deze aanpak kunnen de modellen gebruikt worden om te onderzoeken hoe verschillende factoren bij kunnen dragen aan hogere opbrengsten en versnelde verbetering van de opbrengsten. Ook kan op deze manier onderzocht worden wat er gedaan moet worden om de potenties te realiseren. De uitkomsten van deze analyses kunnen vervolgens gebruikt worden om regionale strategiën te bepalen voor landbouwintensivering gecombineerd met duurzame biomassaproductie.
- Het aantal projecties voor de opbrengsten van gewassen ligt aanzienlijk hoger dan het aantal projecties voor veeteelt. Daarnaast zijn de projecties voor veeteelt gebaseerd op meer geaggregeerde en minder gedetaileerde data in vergelijking met gewassen. Intensivering van veeteelt speelt echter een belangrijke rol in het beschikbaar maken van landbouwgrond voor biomassaproductie. Het wordt daarom aanbevolen om in studies op het gebied van biomassapotenties meer aandacht te besteden aan veeteelt en de mogelijkheden voor intensivering in deze sector. Daarnaast zouden studies op het gebied van broeikasgasemissies en andere milieu-effecten gerelateerd aan intensivering en biomassaproductie niet alleen aandacht moeten besteden aan de effecten van intensivering in de akkerbouw, maar ook in de veehouderij. Om de effecten van intensivering in de veehouderij goed te kunnen analyseren moeten er meer case studies worden uitgevoerd voor het verzamelen van regionale data en het berekenen van broeikasgasbalansen voor verschillende productiesystemen voor vee. Aspecten waarover meer data verzameld moet worden zijn bijvoorbeeld de consumptie van veevoer, productiviteit van vee, de

efficiëntie waarmee veevoer wordt omgezet in groei of melkproductie van het dier en grondgebruik (van zowel akkerland voor de productie van veevoer als grasland voor het laten grazen van de dieren). Bestudering van bestaande bottom-up studies op het gebied van broeikasgasemissies en grondgebruik gerelateerd aan veeteelt bracht drie hiaten aan het licht die met name zouden moeten worden geadresseerd in toekomstig onderzoek. Ten eerste, huidige case studies zijn niet of nauwelijks gericht op rund- en melkveeproductie in ontwikkelingslanden. Een analyse van meer geaggregeerde en gemodeleerde data suggereert echter dat intensivering van rund- en melkvee bijzonder belangrijk is om grondgebruik en broeikasgasemissies in deze regio's te verminderen. Ten tweede, hoewel er voor ontwikkelde regio's in het algemeen meer case studies beschikbaar zijn, is er maar een beperkt aantal studies dat verschillende productie systemen in beschouwing neemt en het mogelijk maakt om de effecten van verschillende intensiveringsroutes te analyseren en vergelijken. Ten derde nemen bestaande case studies met name op gras gebaseerde en gemengde productiesystemen in beschouwing, terwijl de analyse van industriële veehouderij gelimiteerd is. Dit beperkt de mogelijkheid om de effecten te onderzoeken van de overstap van gemengde productiesystemen naar industriële systemen.

- De huidige ervaring met commerciële productie van tweede generatie biomassa en geavanceerde conversie- en bioraffinagetechnologiën is beperkt. Dit veroorzaakt onzekerheid over the economische prestatie van deze grondstoffen en technologiën op de korte en langere termijn. Het is daarom van belang onderzoek te doen naar de potenties van technologische vooruitgang, de snelheid waarmee deze vooruitgang plaats kan vinden en de benodigde toename in productiecapaciteit over tijd om deze snelheid te realiseren. Daarnaast moet ook gekwantificeerd worden hoe groot de benodigde investeringen zijn om de geraamde productiecapaciteit te halen. Een belangrijke rol in het leveren van data is daarbij weggelegd voor demonstratie- en opschalingsprojecten op het gebied van biomassaproductie en conversietechnologiën.
- De bespreking en vergelijking van de economische en milieuprestaties van biomassawaardeketens onderstreept het gebrek aan data met betrekking tot conversieroutes voor biomaterialen. Dit resulteert in een grote onzekerheid over de economische prestaties van deze ketens. Het is daarom belangrijk om meer, betere en transparante data te verzamelen voor het gebruik in modellen, maar ook om een solide onderbouwing te leveren voor het subsidiëren van onderzoeks-, ontwikkelings-, demonstratie- en commercialiseringsprojecten. Voor het verzamelen van deze data is een hechte samenwerking tussen industrie en wetenschap noodzakelijk.

- De regionale analyse van mogelijkheden voor landbouwintensivering en bioenergieproductie in de provincie Lublin laat zien dat het belangrijk is om standaard op regionaal niveau een geïntegreerde aanpak toe te passen met betrekking tot landgebruiksplanning voor biomassaproductie en andere landgebruiken. Om te begrijpen hoe de implementatie van een dergelijke aanpak in de praktijk kan werken, zijn demonstratieprojecten nodig die zich richten op het verhogen van biomassaproductie gecombineerd met gebalanceerde modernisering van conventionele landbouw en geïntegreerde landgebruiksplanning. Deze projecten kunnen daarnaast ook gebruik worden om de resultaten met betrekking tot intensiverings- en biomassapotenties uit case studies te verifiëren, om methoden voor monitoring te ontwikkelen en demonstreren en om belemmeringen voor het realiseren van de potenties te identificeren.
- Voor het verbeteren van de totale milieuprestatie van landgebruik in regio's waar biomassa wordt of zal worden geproduceerd, zou standaard een regionale analyse moeten worden uitgevoerd van de netto broeikasgasemissies en andere milieueffecten. Een dergelijke analyse maakt het mogelijk om go and no-go routes voor landbouwintensivering en biomassaproductie te identificeren, gebaseerd op voorgedefineerde duurzaamheidscriteria.

Beleid

- Dit proefschrift onderstreept het belang van i) de onderlinge verbondenheid van de agrarische en bioenergie sectoren en ii) duurzame intensiveringsroutes om de broeikasgasbalans en milieueffecten van biomasawaardeketens te verbeteren. Om deze reden zou landbouwbeleid op regionaal, landelijk en bovennationaal niveau een geïntegreerde visie moeten hebben op landbouw en biomassproductie voor energie en materialen. Dit betekent dat, met het doel om ILUC te vermijden, doelen voor biomassa-expansie en het verbeteren van opbengsten in de akkerbouw en veeteelt met elkaar in overeenstemming gebracht moeten worden. Deze doelen moeten ook worden aangepast aan de mogelijkheden voor landbouwintensivering en opschaling van biomassaproductie en de snelheid waarmee deze gerealiseerd kunnen worden. Tenslotte moeten de werkelijke ontwikkelingen in de groei van biomassaproductie en in intensivering worden gemonitord en worden bijgestuurd zodat deze in overeenstemming zijn met de gestelde doelen.
- Het aannemen van geïntegreerd beleid voor biomassaproductie en landbouw betekent ook dat duurzaamheidscriteria niet alleen worden toegepast op biomassaproductie, maar op alle landbouw- en bosbouwactiviteiten in de regio. Naast het implementeren van duurzaamheidscriteria zouden duurzame

biomassaproductie en landbouw ook financieel gestimuleerd kunnen worden, bijvoorbeeld door het verstrekken van subsidies voor het overstappen op de beste technologiën die zowel bijdragen aan het verhogen van de opbrengsten en het verbeteren van de millieueffecten.

- Omdat de factoren die landbouwintensivering stimuleren of hinderen regiospecifiek zijn zouden deze regionale factoren zoveel mogelijk mee moeten worden genomen in landbouwbeleid. Daarnaast zou nationaal en bovennationaal beleid flexibel genoeg moeten zijn om ruimte te geven aan regionale behoeften en de implementatie van regiospecieke maatregelen mogelijk te maken die duurzame landbouw en biomassaproductie stimuleren.
- Tweede generatie biomassa en conversietechnologiën zijn van groot belang zijn voor toekomstige grootschalige, economisch rendabele en duurzame productie van bioenergie en biomaterialen. De commercialisering van deze technologiën is echter nog beperkt. Redenen zijn onder andere technologische moeilijkheden, hoge investeringskosten, gebrek aan duidelijk beleid en daarmee een ongunstig investeringsklimaat. Het is daarom belangrijk duidelijk en coherent beleid voor de lange termijn te formuleren om het aandeel bioenergie en biomaterialen in de energie- en materialenmarkt te vergroten. Daarnaast zijn investeringen nodig in ontwikkelings-, demonstratie- en commercialiseringsprojecten om tweede generatie biomassa en technologiën te stimuleren, de capaciteit en efficiënctie van biomassaconversie te verhogen en de infrastructuur, logistiek en markten voor biomassa te verbeteren.

REFERENTIES

[1] IPCC. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change: Geneva, Switzerland; 2014.

[2] Woods J, Lynd LR, Laser M, et al. Chapter 9. Land and Bioenergy. In: Souza GM, Victoria RL, Joly CA, Verdade LM, editors. Bioenergy & Sustainability: bridging the gaps. Scientific Committee on Problems of the Environment (SCOPE): Paris, France; 2015, p. 258-300.

[3] Saygın D, Gielen DJ, Draeck M, et al. Assessment of the technical and economic potentials of biomass use for the production of steam, chemicals and polymers. Renewable and Sustainable Energy Reviews 2014;40:1153-1167.

[4] Fischedick M, Schaeffer R, Adedoyin A, et al. Mitigation potential and costs. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 791-864.

[5] IEA. Energy Technology Perspectives 2012: Pathways to a Clean Energy System. International Energy Agency: Paris, France; 2012.

[6] GEA. Global Energy Assessment - Toward a Sustainable Future. Cambridge University Press and the International Institute for Applied Systems Analysis: Cambridge, UK and New York, NY, USA and Laxenburg, Austria; 2012.

[7] WWF. The Energy Report: 100% Renewable Energy by 2050. WWF – World Wide Fund For Nature: Gland, Switzerland; 2011.

[8] Greenpeace. Energy [R]evolution: A Sustainable World Energy Outlook. Greenpeace International, The Global Wind Energy Council (GWEC) and European Renewable Energy Council (EREC): Amsterdam, The Netherlands and Brussels, Belgium; 2012.

[9] Dale BE, Anderson JE, Brown RC, et al. Take a Closer Look: Biofuels Can Support Environmental, Economic and Social Goals. Environmental science & technology 2014;48(13):7200-7203.

[10] Clarke L, Jiang K, Akimoto K, et al. Assessing Transformation Pathways. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2014, p. 413-510.

[11] Smith P, Bustamante M, Ahammad H, et al. Agriculture, Forestry and Other Land Use (AFOLU). In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2014, p. 811-922.

[12] Chum H, Faaij A, Moreira J, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, et al, editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA; 2011, p. 209-332.

[13] Wicke B, Verweij P, van Meijl H, et al. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 2012;3(1):87-100.

[14] de Wit M, Londo M, Faaij A. Productivity developments in European agriculture: Relations to and opportunities for biomass production. Renewable and Sustainable Energy Reviews 2011;15(5):2397-2412.

Dankwoord
Curriculum vitae
Publications

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CURRICULUM VITAE

Sarah Gerssen-Gondelach (1985) completed the BSc program Mechanical Engineering at Delft University of Technology (2008) and the MSc program Sustainable Development at Utrecht University (2010). During her master studies she specialized in Energy and Resources and graduated on an assessment of the performance of batteries for electric cars. In the beginning of 2011, Sarah started her PhD research and joined the Science, Technology & Society (now Energy and Resources) group of the Copernicus Institute of Sustainable Development at Utrecht University. Her research was part of the KIS project which aimed at identifying strategies, options, preconditions and pathways to optimize the environmental and socio-economic impacts of biomass production and utilization over time and to develop the potentials sustainably. In addition, she also contributed to the ILUC prevention project which aimed at providing insights into how ILUC risks can be mitigated, how this can be quantified and how ILUC may be governed. Sarah's research focused on the bottom-up assessment of improvement potentials in biomass value chains and agricultural and livestock production systems, and the impacts on GHG emissions and other environmental impacts. Sarah presented her work at several scientific conferences in the Netherlands, Italy and Austria. During her PhD she contributed to several MSc courses related to energy and sustainable development and supervised MSc thesis projects.

PUBLICATIONS

Peer reviewed journal articles by the author:

[1] **Gerssen-Gondelach SJ**, Wicke B, Borzęcka-Walker M, Pudelko R, Faaij APC. Bioethanol potential from miscanthus with low ILUC risk in the province of Lublin, Poland. GCB Bioenergy 2015.

[2] **Gerssen-Gondelach SJ**, Wicke B, Faaij APC. Assessment of driving factors for yield and productivity developments in crop and cattle production as key to increasing sustainable biomass potentials. Food and Energy Security 2015;4(1):36-75.

[3] **Gerssen-Gondelach SJ**, Saygin D, Wicke B, Patel MK, Faaij APC. Competing uses of biomass: Assessment and comparison of the performance of bio-based heat, power, fuels and materials. Renewable and Sustainable Energy Reviews 2014;40:964-998.

[4] **Gerssen-Gondelach SJ** and Faaij APC. Performance of batteries for electric vehicles on short and longer term. Journal of Power Sources 2012;212:111-129.

[5] Wicke B, van der Hilst F, Daioglou V, Banse M, Beringer T, **Gerssen-Gondelach SJ**, Heijnen S, Karssenberg D, Laborde D, Lippe M, van Meijl H, Nassar A, Powell J, Prins AG, Rose SNK, Smeets EMW, Stehfest E, Tyner WE, Verstegen JA, Valin H, van Vuuren DP, Yeh S, Faaij APC. Model collaboration for the improved assessment of biomass supply, demand, and impacts. GCB Bioenergy 2015;7(3):422-437.