GROWING IN CIRCLES:

A CIRCULAR BIOECONOMY FOR PLASTICS

Assessing strategies to reduce the plastic sector's Greenhouse-gas emissions and resource consumption

PAUL STEGMANN

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Assessing strategies to reduce the plastic sector's Greenhouse-gas emissions and resource consumption

Groeien in cirkels:
Een circulaire bio-economie voor plastics

Beoordeling van strategieën om de broeikasgasemissies en het
grondstoffengebruik van de plasticsector te reduceren
(met een samenvatting in het Nederlands)

Proefschrift

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TABLE OF CONTENTS

TABLE OF FIGURES

ABBREVIATIONS

CBE	Circular Bioeconomy
CE	Circular Economy
CO_2	Carbon dioxide
CR	Chemical recycling
EFSA	European Food Safety Authority
EPBP	European PET Bottle Platform
EoL	End-of-life
GDP	Gross Domestic Product
GHG	Greenhouse-gas
GWP	Global warming potential
IAM	Integrated Assessment Model
IEA	International Energy Agency
IMAGE	The Integrated Model to Assess the Global Environment, developed by PBL
IPCC	Intergovernmental Panel on Climate Change
HDPE	High density polyethylene
HVC	High-Value Chemicals (outputs of steam cracking, i.e., ethylene, propylene, butadiene, aromatics)
LCA	Life Cycle Assessment
LD,LDPE	(Linear) low density polyethylene
LHV	Lower heating value
MR	Mechanical recycling
MSW	Municipal solid waste
NEDE	Non-energy demand and emissions model
PEF	Polyethylene Furanoate
PET	Polyethylene Terephthalate
PLAIA	Plastics Integrated Assessment model
PP	Polypropylene
PP&A	Polyester, polyamide, and acrylic
PS	Polystyrene
PUR	Polyurethanes
PVC	Polyvinylchloride
RCP	Representative Concentration Pathway
SSP	Shared socioeconomic pathway OR solid-state polymerisation (Chapter 5)
TIMER	The IMage Energy Regional, energy sub-model of the IMAGE framework
WTO	Waste Treatment Option


BIO

Chapter 1

Introduction

1.1 CLIMATE CHANGE & THE IMPACT OF THE PLASTICS SECTOR

Greenhouse gas (GHG) emissions caused by human activities have led to global warming of about 1.2 degrees Celsius (see Figure 1.1)[1]. Global warming leads to numerous impacts, including impacts on agricultural yields, health, sea-level rise, and biodiversity loss and also increases the severity and frequency of extreme weather events such as droughts, heat waves and storms[1]. In the Paris Agreement on climate change of 2015, the 196 parties agreed on the goal to limit global warming to well below 2°C while aiming at 1.5°C by 2100, compared to pre-industrial levels[2].

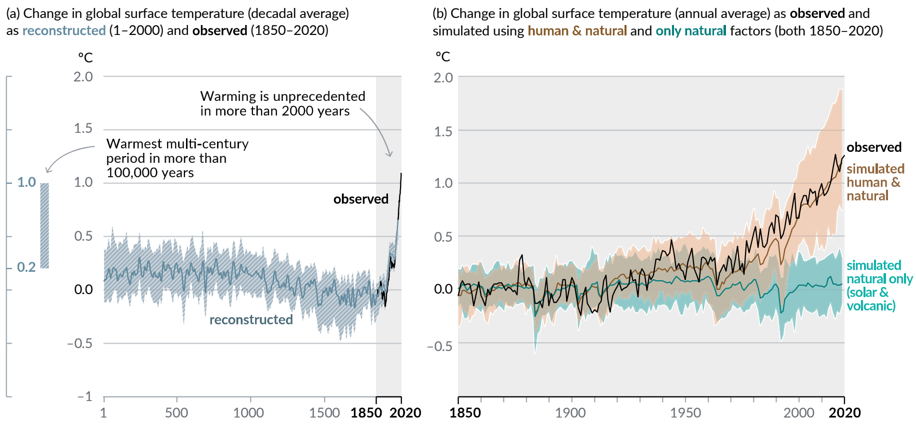

Figure 1.1: Changes in global surface temperature relative to 1850-1900 (taken from IPCC[1])

While the energy sector is responsible for the largest share of global carbon dioxide (CO_2) emissions, the industry is the second biggest contributor[3]. The chemical sector is the largest industrial energy consumer and the third-largest industrial carbon dioxide emitter, covering 18% of the global industry emissions[4]. Nevertheless, chemicals can be considered a *'blind spot'* in the energy system analysis due to the sector's complexity, lack of data, and because about half of the sector's energy inputs are used as feedstock (non-energy use)[4,5]. The latter also means that the sector could play a role in sequestering carbon: Many chemical products depend on carbon as raw material, which is provided by the - predominately – fossil fuels such as oil, natural gas, and coal. A large part of this carbon is sequestered in products, for example, plastics.

Plastics make up more than 40% of the products of the global chemical sector in weight[4,5] and have become an essential part of the global economy. Their annual production increased from 2 Mt in 1950 to 380 Mt in 2015, making plastics the bulk material with the highest production growth globally (see Figure 1.2)[4,6]. Plastics are a very versatile material that could offer environmental benefits such as substituting heavier and more GHG-intensive materials like

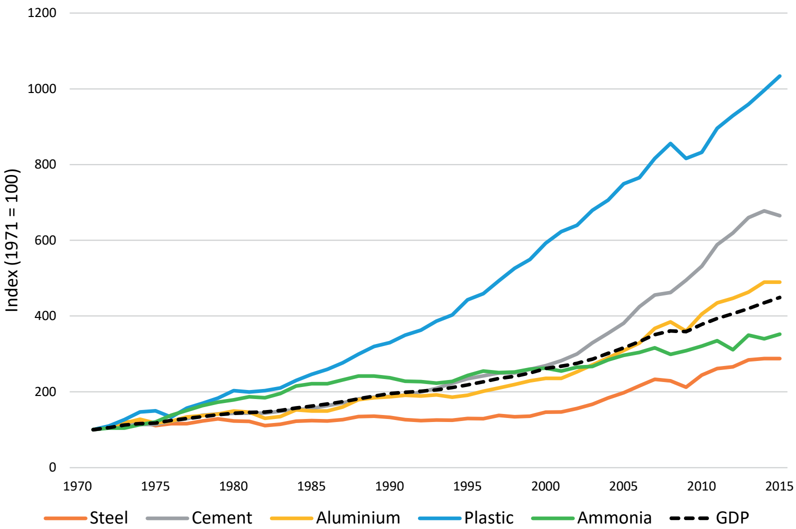

Figure 1.2: Production growth of selected bulk materials and GDP

Notes: displayed on an indexed basis referring to 1971 levels; Own illustration, adapted from IEA 2018[4]

steel and concrete[4,7]. However, their rising consumption also takes its toll on the environment. The plastics sector was responsible for 4.5% of the global GHG emissions in 2015[8]. Following current growth rates, plastic production and the corresponding GHG emissions could almost quadruple by 2050[9]. Furthermore, plastics contribute to particulate matter emissions[8] and their limited biodegradability reinforces the growing plastic pollution[10–12].

1.2 STRATEGIES FOR REDUCING GHG EMISSIONS AND RESOURCE CONSUMPTION IN THE PLASTICS AND CHEMICALS SECTOR

Reducing GHG emissions and resource consumption in the chemicals and plastics sectors requires significant efforts. Potential efficiency gains are limited as process yields are often already close to their practical limits, and reductions in process energy use have already been incentivised due to volatile fuel prices[5]. On the other hand, decarbonisation of electricity and heat supply used in plastic production promises significant GHG reductions[9], in particular, if accompanied by electrification of process energy in chemicals production, e.g. in steam crackers[13].

However, this does not solve the issue of fossil resource use as feedstock, which accounts for more than half of the total inputs in energy terms into the chemicals sector[4]. Unlike process energy, the carbon in feedstocks is not directly emitted. Nevertheless, a large share is emitted after its use in products, especially for those products that are difficult to recover (e.g., detergents, cosmetics, paints[14]) or commonly incinerated after their use (like certain plastic types and applications). However, a significant part of the carbon in feedstocks might stay sequestered, e.g., in products with a long lifetime like building & construction materials or eventually in properly managed landfills. Hence, plastics could be considered a form of carbon storage under certain circumstances.

Next to novel technologies like carbon capture & utilisation, biomass use and recycling are strategies that can significantly reduce both fossil feedstock use and the related GHG emissions of the plastics and chemicals sectors [4,15,16]. Previous studies have shown that these strategies promise significant GHG emissions reductions for the global plastics sector[9,17]. Moreover, using sustainable biomass as feedstock could potentially achieve negative CO_2 emissions by sequestering biogenic carbon in plastic products for long-term use[18]. If these plastics are kept in use, stored over the long term in landfills, or their carbon content is recycled, they could theoretically act as a medium or long-term carbon sink. So far, none of the pathways to achieve the Paris climate targets presented in the reports of the Intergovernmental Panel on Climate Change (IPCC) considers carbon sequestration in products[3].

1.3 A CIRCULAR BIOECONOMY FOR PLASTICS

Biomass is projected to play an important role in meeting the global climate targets set in the Paris Agreement[19–22]. The concept of a bio-based economy or bioeconomy has been put forward by the European Union[23] and almost 50 countries around the globe[24]. A bioeconomy can be defined as the *"production of renewable biological resources and the conversion of these resources and waste streams into value added products, such as food, feed, bio-based products and bioenergy"* [23]. Also, the potential contributions of recycling and the circular economy (CE) to climate change mitigation are increasingly highlighted[17,25–27]. The European Commission[28] defines the CE as minimising waste generation and maintaining the value of products, materials and resources for as long as possible. Merging these two concepts has led to the term 'circular bioeconomy' (CBE), which appeared around 2015 and has been increasingly used in scientific publications since 2016 (see Chapter 2). However, there have been only a few attempts to define the term and describe what the CBE concept entails.

The CBE could be a particularly powerful concept in the plastics sector to reduce its fossil resource consumption and GHG emissions. Research is just beginning to comprehensively assess the global, long-term potential of biomass use and recycling in reducing GHG emissions and resource consumption in the plastic sector; the combination of both is studied only in a few articles, such as the recent article of Meys et al.[16]. However, only by assessing both together can we understand the potential synergies and trade-offs between the goals of climate change mitigation and the circular economy targets. For example, fostering recycling might improve the circularity of the sector and reduce demand for feedstocks, but it does not change the remaining primary plastic production and might even cause more emissions than a bio-based production strategy with long-term carbon sequestration in products. At the same time, a bio-based production strategy might promise substantial emission reductions while still relying on high feedstock and energy use. Moreover, without increasing recycling, a bio-based production strategy would continue following a linear business-as-usual. Combining circular strategies with biomass use could mitigate these trade-offs and promises a compromise between the goals of climate change mitigation and the circular economy targets.

1.4 ASSESSING THE GHG EMISSIONS AND RESOURCE CONSUMPTION REDUCTION POTENTIAL OF A CIRCULAR BIOECONOMY FOR PLASTICS

Life cycle assessment (LCA) is a method to assess the impact a product has on the environment, e.g., its contribution to global warming. LCAs help identify emission hot spots in a product's life cycle and reveal trade-offs between different environmental impacts. Reviews showed that there is already a significant literature base of LCAs investigating the environmental impacts of biobased plastics[29–31]. However, the end-of-life of bio-based plastics receives limited attention in scientific literature. Assessing the end-of-life (EoL) of plastics is key to understanding the benefits of a circular economy, i.e., by comparing recycling with other waste treatment options like incineration and landfilling. A review by Spierling et al.[32] discovered that only for polylactic acid (PLA), there are eleven studies considering the EoL of plastics, followed by two studies on thermoplastic starch (TPS) and a few individual ones for other bio-based plastic types. Not to mention that this consideration of the EoL in these reviewed studies could be limited to one simplified scenario and does not have to entail a comparative analysis of different waste treatment options as it exists for fossil plastic types[33–35]. The effect on GHG emissions of recycling plastics multiple times has so far not been assessed for bio-based plastics from an LCA perspective and only a few times for fossil plastics[36,37]. The limited literature on the EoL of bio-based plastics could be explained by the fact that most bio-based plastics only have minor market shares (e.g., PLA) or are not even on the market yet (e.g., polyethylene furanoate, PEF). This makes it difficult to assess their behaviour in waste management systems.

The consideration of the EoL of plastics is key to understanding the benefits and trade-offs of the circular economy and calculating the overall GHG emissions of the produced polymers over their entire life cycle, including the emissions occurring after their production. A lack of understanding of the impact of EoL-options could hamper the transition to a circular bioeconomy in plastics value chains and lead to incomplete assessments of the overall climate benefit of bio-based compared to fossil plastics. While a bio-based plastic might have a lower GWP than a fossil competitor in production, this advantage might be partly counterbalanced by worse performance in the end-of-life, e.g., lower recyclability or more emission-intensive waste treatment.

LCA is a strong tool to analyse the environmental impact of plastic products, but it has difficulties assessing their contribution to circular economy targets[38], such as increasing a product's utility[39,40]. The Ellen MacArthur Foundation[39] defines a product's utility as a combination of the length of a product's use phase and the intensity of its use. Consequently, a conventional LCA cannot adequately assess potential synergies and trade-offs between the goals of climate change mitigation and the circular economy targets. Hence, there have been several attempts to extend the LCA methodology to include circularity indicators[38].

Furthermore, comparing plastics production and their end of life in static LCAs cannot address the aggregated impacts of plastics production and end-of-life nor the long-term dynamics, i.e., related to resource competition between sectors, technology developments and changing demand and supply. Hence, an LCA can be described as a 'snapshot' of the environmental impact of a specific product, disregarding the evolving energy and land-use systems and the developments in the related sectors.

Comprehensive, aggregate, global, and long-term assessments addressing these issues for the chemical and plastic sectors are rare. This can be explained by the fact that reliable data for these sectors is scarce, especially compared to other industry sectors like steel and aluminium[5]. A study by Zheng and Suh[9] combined LCA data to assess the global GHG impact of plastics for different extreme mitigation scenarios, including biomass use and recycling. However, this study could not cover the interaction with the broader economy and long-term dynamics. More comprehensive assessments have been conducted by the International Energy Agency (IEA)[4], assessing the current state and potential future trajectories until 2050 of the chemical sector via a TIMES-based linear optimisation model. Furthermore, Meys et al.[17] created a bottom-up model to assess pathways to net-zero emission plastics in 2050, and Cabernard et al.[8] assessed the global environmental impact of plastic production until 2030 via an enhanced multiregional input-output analysis. However, none of the climate and socioeconomic models used for the IPCC reports has included a detailed representation of the plastics sector[41].

Integrated assessment models (IAM) study the interlinkages between human and natural systems and their impact on the world's climate[42]. They consist of several sub-models covering various natural systems including land, water, biodiversity, and human systems like energy use and agriculture[43]. They have played a key role in assessing strategies to mitigate climate change[44], such as in the IPCC reports[19]. Concerning the energy system, IAMs traditionally neglect or highly stylise non-energy uses of energy resources, i.e., their use for chemicals and plastics and the associated emission. However, there are a few notable exceptions that are highly aggregate[45] or limited to one country[18]. Thus, global IAMs currently do not provide any valuable insights concerning the potential of a CBE and different waste treatment strategies for meeting strict climate goals. Instead, climate change mitigation pathways, such as those assessed by the IPCC tend to focus on supply-side mitigation action, via the decarbonisation of the primary energy supply and the use of carbon dioxide removal.

Research goals

It is impossible to fully understand the climate change mitigation potential and the trade-offs of mitigation strategies without analysing the global, long-term trends in the plastics sector and the sector's interactions with other socioeconomic and natural systems. IAMs could provide such an analysis and could thus contribute to defining pathways and overall policy goals for the plastics sector. At the same time, such an aggregate analysis needs to be complemented by case studies, e.g., based on an LCA methodology, to develop specific recommendations for implementing these goals on a product level.

Using both methodological approaches, *the overall goal of this research is to better understand to what extent a circular bioeconomy (CBE) could reduce the GHG emissions and resource consumption of the plastics sector.* To that end, this thesis investigates the following research questions:

1. What are the defining elements of a CBE, and what is its current role in regional European bioeconomy clusters*?
2. How can the impact of a CBE on GHG emissions, resource consumption, and circularity be adequately assessed?
3. What are potential developments of global plastic production, stocks, waste generation and the related resource consumption and CO_2 emissions until 2100 following current policy trends?
4. What are promising plastic production and waste management strategies to reduce the GHG emissions and resource consumption of the plastics sector, including a CBE?

Table 1.1 provides an overview of the chapters in this thesis and how they contribute to answering our research questions.

* Such bioeconomy clusters consist of interconnected stakeholders working in the bioeconomy field in a certain region, such as farmers, manufacturers, industrial associations, research institutions and governmental bodies.

Table 1.1: Overview of the thesis chapters

Chapter	Article name	Methods	Research questions addressed
2	The Circular Bioeconomy: Its elements and role in European bioeconomy clusters	Literature review & interviews	1
3	The plastics integrated assessment model (PLAIA): Assessing emission mitigation pathways and circular economy strategies for the plastics sector	Literature review, Integrated assessment modelling	2, 3, 4
4	Plastic futures and their CO2 emissions	Integrated assessment modelling	2, 3, 4
5	Message in a bottle - The global warming potential and the material utility of PET and bio-based PEF bottles over multiple recycling trips	Literature review, Life cycle assessment	2, 4
6	Summary & Conclusions	-	1-4

Chapter 2

The circular bioeconomy: Its elements and role in European bioeconomy clusters

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Resources, Conservation and Recycling: X 6 (2020).
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ABSTRACT

Biomass is projected to play a key role in meeting global climate targets. To achieve a resource-efficient biomass use, European bioeconomy strategies increasingly consider the concept of a circular bioeconomy (CBE). We define the term CBE via a literature review and analyse the concept's role in north-west European bioeconomy clusters through interviews. We identify strategies regarding the clusters' feedstock and product focus, and investigate what role biorefineries, circular solutions, recycling and cascading play. Finally, we discuss gaps in CBE literature and the potential contributions of the CBE to sustainability. The analysed bioeconomy clusters move towards a CBE by increasingly considering residues and wastes as a resource, developing integrated biorefineries and focusing more on material and high value applications of biomass. However, there is so far only little focus on the end-of-life of bio-based products, i.e. on circular product design, recycling and cascading. Key challenges for implementing circular strategies are policies and regulations, costs and the current small size of bio-based markets. Amongst the product sectors the interviewees identified as promising for the bioeconomy, plastics and construction & building materials have most recycling and cascading potential. While the CBE could contribute to improving the sustainability of the bioeconomy, the concept is not inherently sustainable and its potential trade-offs need to be addressed. Especially social aspects, cascading, circular product design, and aspects related to product use seem to be underrepresented in CBE literature, while the topics biorefinery, wastes and residues as well as waste management are significantly covered.

Graphical Abstract

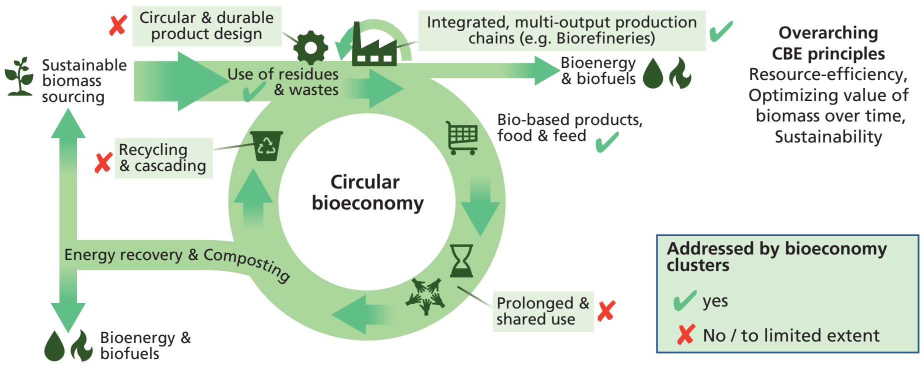

Figure 2.1: Graphical abstract - The circular bioeconomy and its role in European bioeconomy clusters

2.1 INTRODUCTION

Biomass is projected to play an important role in meeting the global climate targets set in the Paris agreement[19–22]. For the chemical industry, heavy road transport and marine and aviation sectors, biomass is one of the few options to replace their fossil feedstock with a renewable resource, thereby reducing the sectors' Greenhouse-gas (GHG) emissions[4,15,16,46,47]. Hence, the concept of a bioeconomy (BE) has been put forward by the European Union[23] and by almost 50 countries around the globe[24]. A bioeconomy can be defined as the *"production of renewable biological resources and the conversion of these resources and waste streams into value added products, such as food, feed, bio-based products and bioenergy"* [23].

A strong optimism has been observed regarding the benefits of the bioeconomy[48]; some bioeconomy strategies and scientific publications consider the bioeconomy to be inherently sustainable[49,50]. However, there are also various publications highlighting potential trade-offs and negative impacts[48,51]: They expect an increased pressure on water bodies and natural ecosystems and question the emission reduction potential. Some key issues in this debate evolve around the competition for land, i.e., direct and indirect land-use change, agricultural intensification, eutrophication and risks posed by invasive species[51]. While some authors consider the bioeconomy to be *"circular by nature"* [52,53], Hetemäki et al.[49] see the risk of following a linear business-as-usual approach if the principles of a circular economy (CE) are not considered. The CE is defined by the European Commission [28] as minimising the generation of waste and maintaining the value of products, materials and resources for as long as possible.

As a response to these critical discussions, the updated bioeconomy strategy of the European Commission announces that the *"European Bioeconomy needs to have sustainability and circularity at its heart"* [54]. Since the publication of the EU action plan for the CE[28], *"practically all of the European bioeconomy(-related) strategies"* have increasingly been linked to the CE[55].

Merging these two concepts has led to the term 'circular bioeconomy' (CBE), which appeared around 2015 and is increasingly used in scientific publications since 2016 (see section 2.2.1). However, there have been only few attempts to define the term and to describe what the CBE concept actually entails (see sections 2.2.1 and 2.3.1). Furthermore, a bottom-up perspective on the role of the CBE in regional bioeconomy clusters is missing. This perspective is particularly relevant since many key strategies towards a more resource-efficient and circular bioeconomy, e.g. integrated biorefineries and cascading use of biomass which are both defined in Chapter 2.3.1, depend on a close cooperation of local actors from agriculture, industry, research and regional public institutions, e.g. within bioeconomy clusters. While clusters greatly vary in their size and degree of organisation, they can be generally defined as a *"geographically proximate group of interconnected companies and associated institutions in a particular field, including product manufacturers, service providers, suppliers, universities, and trade associations"*[56].

The important role of regional clusters in driving the European bioeconomy is increasingly recognised[57,58].

With this paper we aim to contribute to a better understanding of the CBE concept and furthermore investigate to what extent it already plays a role in the strategies of selected regional bioeconomy clusters.

To this end, the paper addresses the following research questions (RQ):
1. What are the defining elements of a CBE?
2. What are the strategies and foci in selected bioeconomy clusters regarding (i) feedstocks; (ii) products; (iii) biorefineries; (iv) circular thinking, cascading and recycling?
3. Do the clusters implement the CBE elements defined in RQ1 in their strategies?

Moreover, we (4) discuss the potential contributions of the CBE to sustainability and (5) identify gaps in literature and cluster strategies that deserve more focus when moving towards a CBE.

The focus of this study is North-West Europe, namely the Netherlands, Belgium, the United Kingdom, Germany and France. These countries represented 51% of the turnover of the EU bioeconomy in 2015[59]. Furthermore, north-west Europe is traditionally a major hub for the European petrochemical industry, a sector that heavily relies on biomass for decarbonisation; these five countries alone are still responsible for almost 66% of the EU chemical sales in 2018[60]. This makes this region a relevant case study to explore RQ 2 and 3.

After describing this paper's materials and methods, we define the CBE concept and its elements, based on a literature review (RQ1). The second part of the results presents strategies in the selected bioeconomy clusters (RQ2) based on interviews with their representatives. Finally, we discuss (a) if the clusters implement the CBE elements defined in RQ1 and (b) the potential contributions of the CBE to sustainability.

2.2 MATERIALS AND METHODS

This research consists of two work-streams (see Figure 2.2): Work-stream A is a literature review of publications on the CBE concept to answer RQ1, and work-stream B covers interviews with bioeconomy cluster-representatives and their subsequent analysis to answer RQ2. In the discussion (C) both work streams are linked to answer RQ3.

2.2.1 CBE Literature review

We searched for the term CBE and its synonyms (Boolean string: "circular bioeconomy" OR "circular bio-based economy" OR "circular bio-economy") within titles, abstracts and

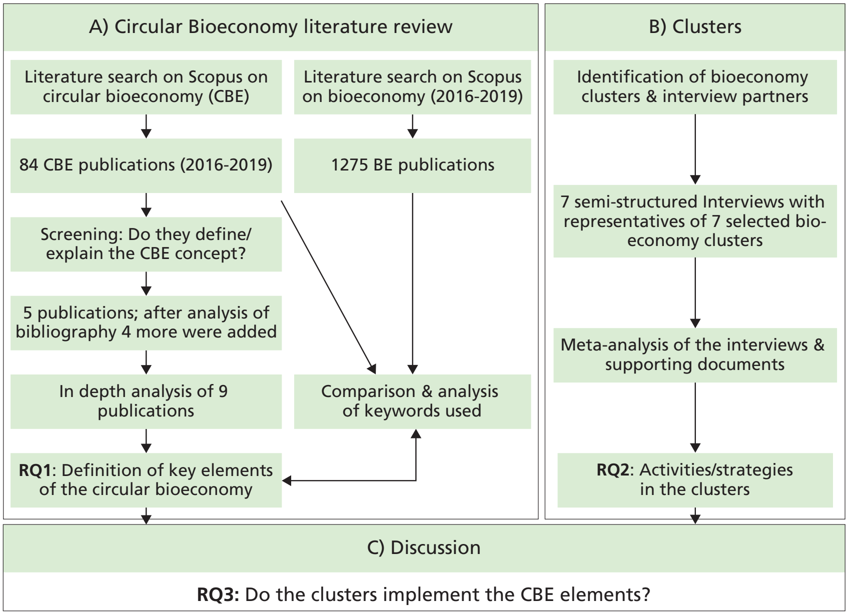

Figure 2.2: Research approach and structure

keywords on Elsevier's scientific search engine Scopus and found 84 peer-reviewed publications in English dating from 2016 to 2019.

2.2.1.1 Qualitative analysis

The titles, abstracts and - where necessary - the full texts of these 84 documents were screened to identify those publications that do not only mention the term CBE but also define or explain the concept. This limited the results to five publications. Analysing the bibliography of these five publications, e.g. identifying frequently mentioned authors, led to the inclusion of four additional documents, of which three are not peer-reviewed. We analysed these nine documents (Appendix A) in detail to identify key elements of the CBE.

2.2.1.2 Analysis of keywords

In a second step we used the software VOSviewer to compare the author and index keywords of the initial 84 CBE documents to 1275 bioeconomy (BE) publications found on Scopus for the same period 2016-2019; Boolean string: ("bioeconomy" OR "bio-based economy" OR "bio-economy") AND NOT ("circular bioeconomy" OR "circular bio-based economy" OR "circular bio-economy"). The goal was to observe differences in the occurrences of keywords, in particular on topics identified as important CBE elements in the qualitative analysis of the nine documents (see Appendix A). We clustered similar keywords and treated them as synonyms (see Appendix B). To compare the relative importance of a keyword in CBE and BE documents, the number of keyword occurrences in CBE and BE documents was divided by the total number of CBE and BE publications respectively.

Eventually, we used the CBE elements identified in this literature review to derive a CBE definition (RQ1).

2.2.1.3 Limitations

Comparing the share of keywords of the only 84 available CBE documents to keywords of the 1275 BE documents might lead to an exaggeration of some findings; the results may show high differences in relative terms, while being derived from a small number of publications. Moreover, when clustering similar keywords, a few subjective choices have been made. By attaching the list of the clustered keywords in Appendix B, these choices are made transparent.

For the detailed qualitative analysis, only nine documents were identified and selected. This sample is not very diverse as four publications are closely linked to the forest-based bioeconomy[49,61–63].

2.2.2 Interviews with representatives of bioeconomy clusters

We conducted seven semi-structured interviews with representatives of seven north-west European bioeconomy clusters (see Figure 2.3). The interviews were complemented by an analysis of overview documents and web pages of the clusters, which provided additional insights into the strategies and focus areas of the clusters (see Appendix C).

2.2.2.1 Selection of clusters and interviewees

The clusters were selected in consultation with the project stakeholders (see acknowledgments). Following the relatively open definition of clusters by Su and Hung[56] (see introduction), clusters of different size and degree of organisation were considered. According to the project focus (see

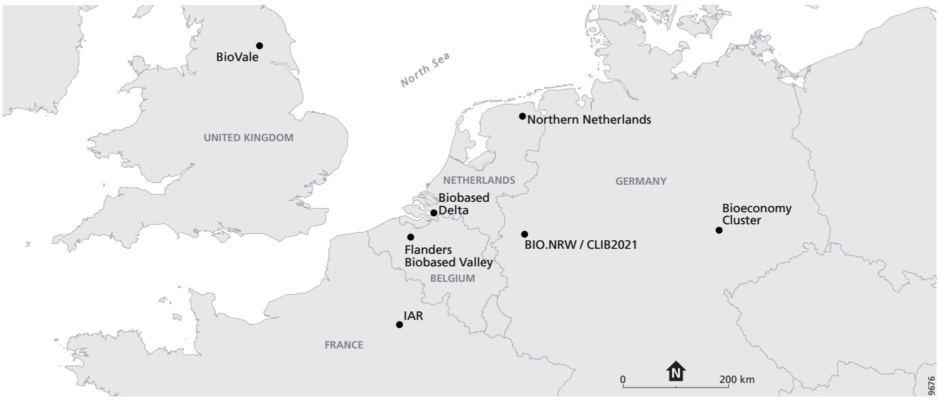

Figure 2.3: Selected bioeconomy clusters

introduction), only clusters from north-west Europe covering a wider range of bioeconomy sectors and involving both public and private actors were selected. Another important factor for consideration was international visibility and impact, signalled by their involvement in international cooperation and projects: Four of the seven selected clusters closely collaborate within the 3BI intercluster and three within the BIG-C, the Bio Innovation Growth mega Cluster, highlighting their role as key actors of the bioeconomy in north-west Europe. Within the 3BI intercluster (Brokering Bio-Based Innovation) the four clusters BioVale, Bio-based delta, Bioeconomy cluster and IAR cooperate in research, development and deployment of biomass conversion technologies[64] and within the Big-C the members Bio-based Delta, BioNRW / CLIB2021 and Flanders Biobased Valley work together on a leading cross-border bioeconomy region [65]. The bioeconomy region Northern Netherlands was added following recommendations of the project stakeholders.

For each of the clusters we interviewed a key knowledge holder with a good overview of the activities in the respective cluster and region: Managing directors (4) and in three cases other cluster representatives that were recommended by project partners or interviewees. The interviews were conducted between February and April 2018 (2 in person, 2 via Skype, 3 via phone).

2.2.2.2 Interview approach

The interview approach applied in this paper can be classified as a semi-structured interview according to Bryman[66], allowing a change of sequence in asking the prepared questions and allowing for more general, open questions as well as for further questions not defined in the interview schedule. This relatively flexible approach can help identifying new priorities and foci of the interviewees that were not yet considered by the interviewer[67] and was therefore considered suitable for the explorative nature of this study. However, there were also pre-coded questions, meaning that a range of answers was provided. But also in this case, the schedule allowed for adding additional options not foreseen by the interviewer.

The interviews focused on strategies and activities in the clusters regarding the following topics:

1. Feedstocks

Based on the interviews and supporting documents the different types of primary biomass feedstocks used in the clusters (structured in lignocellulosic crops, starch crops, sugar crops, oil crops and algae) were classified in a qualitative manner according to their importance within the cluster as main, complementary and prospective feedstock. The same method was applied for assessing the role of agricultural and forestry residues, wastes (from industry and from consumers) and CO2 as a resource. The results were then sent to the interview partners for correction and confirmation. Only the feedstocks that were at least named twice were considered in the results.

2. Product sectors

The interviewees were asked about the focus of their cluster's research and innovation programs regarding the uses of biomass, namely energetic (e.g. for biogas and biofuels) and material uses. Material biomass use is referring to all non-energetic uses of biomass, ranging from applications in the chemical industry to food & feed as well as for construction materials. Furthermore, the interviewees were asked to name bioeconomy product sectors with a high or low growth potential in the future (up to 2030).

3. Biorefineries

The questions aimed at identifying the type of biorefineries (according to Cherubini et al [68] the clusters are working on and at what stage these projects are (research, pilot and demonstration, commercial).

4. Cascading, recycling and other circular strategies

After asking the interviewees for their understanding of cascading, the further questions aimed at identifying cluster activities regarding the end of life of bio-based products, e.g. strategies or projects on cascading, recycling and circular product design.

The interviews were transcribed and the relevant answers coded according to the interview foci (as described by e.g., Gorden[69]). Together with the complementing documents providing an overview of the cluster's strategies, the coded answers were analysed and compared to identify common trends, key differences and the underlying reasons for the cluster's choices.

2.2.2.3 Limitations

The small sample size of bioeconomy clusters does not allow for general conclusions on the status of the CBE in north-west Europe. Furthermore, the clusters are not comparable in size, degree of organisation, feedstock or product focus. Nonetheless, the selected seven clusters provide interesting case studies that give valuable insights into the strategies and challenges of some of Europe's leading bioeconomy clusters. Due to the diversity of the sample, the analysis covers a wide range of bioeconomy developments in north-west Europe, by representing different local circumstances and focus areas. However, it still excludes certain bioeconomy regions, in particular the forest-rich Scandinavian countries. Moreover, the clusters are from countries that have comparably strong CE-policies and well-established waste management sectors in place. Therefore, conclusions from this study are region-specific and might not apply to regions with a different feedstock and industry focus or a less developed CE. Finally, basing the cluster assessment solely on interviews and strategy/overview documents just allows for indicative conclusions on the foci and trends in clusters, as not the whole range of projects and reports of each cluster was analysed.

2.2.3 Analysing the cluster activities and strategies in the context of the CBE

The CBE elements identified in the literature review (RQ1, Appendix A) then served as a framework for analysing the strategies of the bioeconomy clusters (RQ2) to answer RQ3: Do the clusters implement the CBE elements in their strategies. Combining both research streams in the discussion helps us to see in how far the still largely academic concept of a CBE is already a reality in practice and allows us to identify potential shortcomings in literature and industry that deserve more focus when moving towards a CBE.

2.3 RESULTS

2.3.1 Defining the CBE and its elements

In this section we identify and define key elements of the CBE based on a literature review (Appendix A) and an analysis of keywords used in scientific publications (Appendix D). From this analysis we eventually derive a definition of the term CBE and discuss its implication on biomass use.

2.3.1.1 CBE elements in literature

We identified three overarching perspectives on the CBE in relation to the bioeconomy and CE (see Figure 2.4): Before the term CBE appeared, the Ellen MacArthur Foundation [70] implied that the bioeconomy is an integral part of the CE by including the biological cycle into their CE illustration. Similarly, Temmes and Peck[61] see the CBE as a CE where *"non-renewable […] inputs to industrial systems are replaced by renewable biological resources"*. The European Commission[71] defined the CBE as the application of the CE concept to biological resources, products and materials. We analysed nine publications explaining the CBE concept. Four of them define the CBE as the intersection of BE and CE[14,62,72,73] while Hetemäki et al.[49] and Dalia D'Amato, Veijonaho, and Toppinen[63] argue for a more comprehensive vision and see the CBE as *"more than bioeconomy or circular economy alone"*.

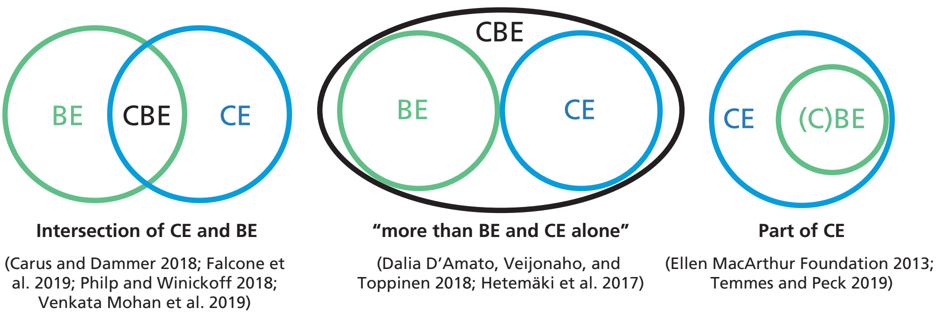

Figure 2.4: Perspectives on the circular bioeconomy (CBE) in relation bioeconomy (BE) and circular economy (CE)

While the perspectives on the term differ, the analysis of the CBE publications showed that they often refer to similar CBE elements: All of the nine analysed publications highlight the ***use of wastes and residues as a resource***. Furthermore, keywords relating to wastes & residues are used 3.5 times more in CBE compared to BE documents (see Appendix D).

The efficient use of biomass is considered part of the CBE by all nine authors, even though their definitions of efficiency vary or are not given. As Ekins et al. (2017), we argue for a definition of ***resource-efficiency*** that considers technical efficiency (material output/material input), resource productivity (economic output/material input) and emission intensity (emission output/material input). Considering these three interpretations of resource-efficiency allows for a more balanced approach; maximising for only one might negatively influence the others, e.g. maximising the technical efficiency might be very energy intensive and thus increase emissions and costs. Resource-efficiency as a keyword does only play an insignificant role (see Appendix D); although keywords referring to efficiency strategies are frequent (see the following paragraphs).

(Integrated) biorefineries are considered an important part of the CBE by seven of nine CBE publications and are seen as a measure to improve the resource-efficiency and total value of the biomass[49,61,75]. Moreover, keywords relating to biorefinery are used almost three times more in CBE documents compared to BE ones. De Jong et al.[76] define a biorefinery as *"the sustainable processing of biomass into a spectrum of marketable products (food, feed, materials, chemicals) and energy (fuels, power, heat)"*[76]. Integrated biorefineries are a combination of several biomass conversion technologies that allow for more flexibility and cost reduction[48]. They do thus ease the use of side-streams and wastes and facilitate the combined production of high value products (e.g. fine chemicals) with lower value products (e.g. bioenergy).

Maintaining the value of products, materials and resources for as long as possible and the ***waste hierarchy*** are two of the key principles of the CE concept of the European Commission[77] and therefore also apply to biological resources in a CBE[78]. The Waste hierarchy introduced by the EU Directive 2008/98/EC on waste (Waste Framework Directive) provides a priority order for waste management with waste prevention as the first priority, followed by re-use, recycling, recovery and disposal.

Five of the nine analysed CBE publications also referred to those principles. However, these principles do not necessarily result in the most economical or environmentally friendly solution (see discussion section). We therefore rather suggest the *optimisation* of the value of biomass *over time* as a key characteristic of the CBE. Such an optimisation can focus on economic (e.g. for profit), environmental (e.g. for GHG emissions) or also social aspects (e.g. for job creation) and ideally considers all three pillars of sustainability.

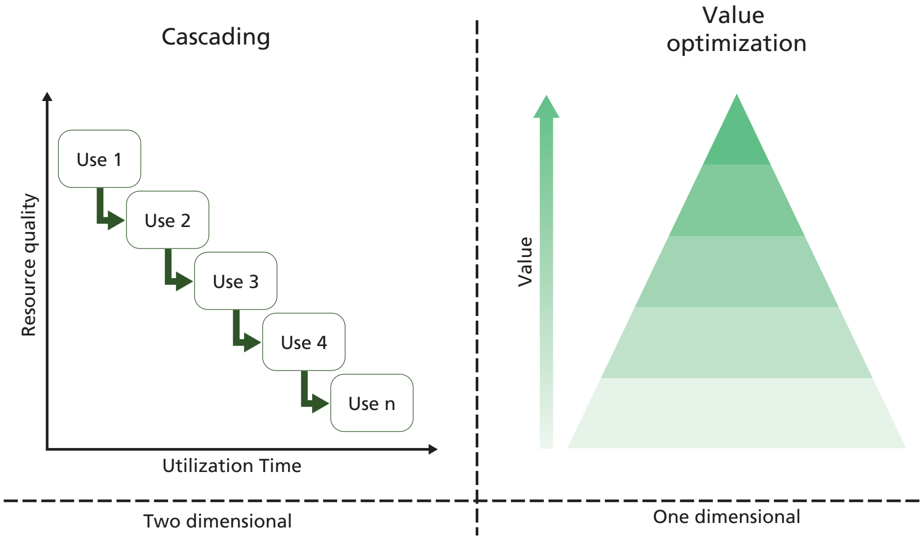

Figure 2.5: Two common interpretations of cascading (own illustration based on Sirkin and Houten (1994))

The ***cascading use of biomass*** could facilitate such an optimisation over time. Six of nine publications considered cascading as an element of the CBE, while its use in keywords is insignificant. Cascading has various definitions in literature but usually the common theme is the *"sequential use of resources for different purposes"* [79]. Within the bioeconomy literature cascading use of biomass is mostly defined similarly; e.g. Fehrenbach et al.[80] define it as the processing of biomass into a bio-based final product which is used at least once more either for material or energy purposes. We also adopt this definition, as it is in line with the European Commission[78] and further literature[81–83]. However, Olsson et al.[79] pointed out that cascading is also sometimes interpreted as an order of priority, aiming for the highest added value[75,83]. Figure 2.5 contrasts these two interpretations of cascading. Furthermore, cascading in the context of biorefineries is often used to describe co-production and factory-internal recycling and recovery loops[75,83]. Co-production can be defined as the *"production of different functional streams (e.g. protein, oil and an energy carrier) from one biomass stream"* [83]. In recent years economic aspects have been highlighted when talking about value optimisation[79], even though there are various ways of defining the resource quality or the value of cascading choices; some of them are intrinsic (e.g. chemical structure, embodied energy) and others are based on human value judgements (e.g. economic, environmental, cultural)[84].

Waste management is an important topic in CBE publications; keywords related to this theme are used 4.2 times more in CBE publications compared to BE documents. ***Recycling*** and other circular waste management strategies are considered part of the CBE by all nine analysed publications.

Circular product design was mentioned in five publications, while it only has a marginal share in keywords for both CBE and BE publications.

Four publications also advocate for an increased product utilisation in the CBE by ***sharing*** and see ***prolonged use or durability*** of bio-based products as an element of the CBE. However, the concept of the sharing economy[85] as well as prolonged use/durability are ignored by the remaining publications and are not existent within the keywords.

Looking at the keywords, ***sustainability, climate change and other environmental impacts*** seem to play a slightly more salient role in CBE publications compared to BE ones (see Appendix D). All nine analysed publications highlight sustainability issues when discussing the CBE. However, the ***social aspects*** seem to fall short in the CBE discourse, both in keywords as well as in the conceptual discussions as only three of nine highlight them.

2.3.1.2 CBE definition

Figure 2.6 illustrates the CBE and its elements and Appendix A shows their coverage in literature. Considering these elements, we suggest the following CBE definition:

The circular bioeconomy focuses on the sustainable, resource-efficient valorisation of biomass in integrated, multi-output production chains (e.g. biorefineries) while also making use of residues and wastes and optimising the value of biomass over time via cascading.

Such an optimisation can focus on economic, environmental or social aspects and ideally considers all three pillars of sustainability. The cascading steps aim at retaining the resource quality by adhering to the bio-based value pyramid and the waste hierarchy where possible and adequate.

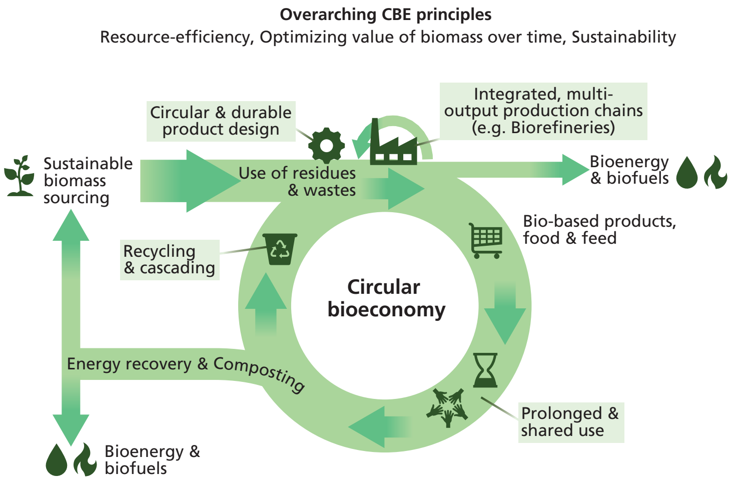

Figure 2.6: The circular bioeconomy and its elements

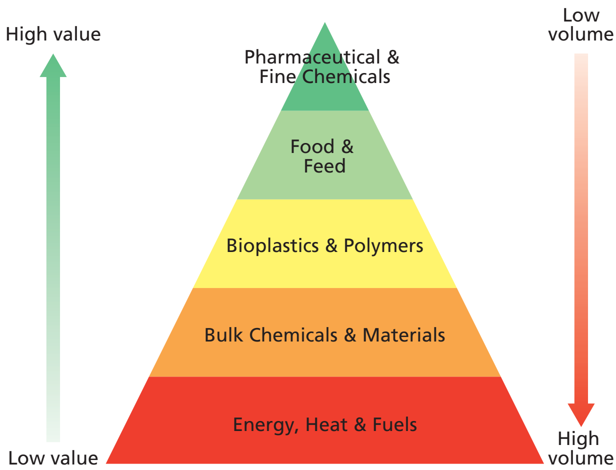

Figure 27: Bio-based value pyramid

Notes: Own illustration adapted from various sources[87–89]

The bio-based value pyramid is a commonly used way to classify biomass uses according to their value and volume, see Figure 2.7.

2.3.1.3 CBE impact on biomass use

The analysis of keywords used in CBE documents showed an almost 50% stronger focus on material biomass uses (bio-based chemicals and materials, food & feed) compared to energy and fuels, while this ratio is balanced in BE documents (see Appendix D). Using biomass directly for energy or fuels makes it impossible to maintain its value via reuse or recycling[86]. Hence, the CBE could induce a reduction of direct energetic use of biomass or a reallocation of biomass resources with biomass suitable for high-value applications going to materials while lower quality biomass is used for energetic purposes. In a CBE, more biomass would ideally first be delegated to a material use before – after one or potentially multiple cascading steps – it would be delegated to a final energetic use or composting. In theory, this cascading would follow a movement down the bio-based value pyramid (Figure 2.7) and the waste hierarchy, moving from high value to lower value biomass applications. Moving from the upper part of the bio-based value pyramid and the waste hierarchy to the lower part theoretically goes along with decreasing options for further uses and cascading opportunities, due to the lowering of the resource quality. Staying on the upper part of both hierarchies would therefore theoretically be desirable in a CBE. However, in practice, applications on the lower part might still be preferable from an environmental and economic perspective (see discussion section).

Those publications advocating for a comprehensive vision of the CBE[49,63] highlight sustainable sourcing of biomass. However, the majority of the CBE documents seems to focus on how the feedstock is used and on the role of residues and waste. Consequentially, the literature review did not reveal a clear preference regarding the types of primary feedstocks or the origin of feedstocks (import or local). However, supporters of a European circular (bio) economy often highlight the argument of reducing the dependency on imports by keeping resources in the loop. Thus, one might expect a slight preference in a CBE for locally sourced feedstocks (see e.g. Bio-based Industries Consortium[90]).

2.3.2 Strategies in European bioeconomy clusters

The interviews with representatives from seven north-west European bioeconomy clusters aimed at identifying current strategies in their respective clusters and regions. The analysis of the interviews is structured in the clusters' strategies regarding a) feedstocks, b) products, c) the role of biorefineries, and d) the role of circular thinking, cascading and recycling.

2.3.2.1 Feedstock strategies

We examined the following three aspects of the cluster's feedstock strategies: the type of primary biomass used, the use of residues and wastes as a resource, and the origin of the biomass feedstocks (import or local).

Table 2.1 shows the type of primary biomass feedstocks used in the clusters and their importance. We see that lignocellulosic feedstocks (mainly wood chips but also grasses) play an important role in almost all clusters, either as main (4) or complementary feedstocks (2), while it is seen as a prospective feedstock in only one case. Starch and sugar crops are also relevant feedstocks for the majority of the clusters; for three clusters they are a main feedstock. Oil crops on the other hand play a major role in only two of the clusters. Algae were mentioned as a prospective feedstock by two interviewees but do not (yet) play a relevant role.

Table 2.1: Primary biomass feedstock types and their role in the clusters (n=7 clusters)

Feedstock role in cluster	Lignocellulosic	Starch	Sugar	Oil	Algae
Main	4	3	3	2	-
Complementary	2	2	1	-	-
Prospective	1	-	-	-	2

Even though primary feedstocks are dominant in most clusters, Table 2.2 shows that residues and wastes are already playing a relevant role as well. Especially agricultural and/or forestry residues are considered a feedstock by all clusters. Six interviewees classify them as either main (2) or complementary feedstock (4), and one considers them as a prospective feedstock. Industry and household wastes do not play such an important role. Only three clusters consider them as main or complementary feedstock and one as a prospective feedstock. Lastly, three interviewees see CO_2 as a prospective feedstock for their cluster.

Table 2.2: Role of residues and wastes in the clusters (n=7 clusters)

Feedstock role in cluster	Agricultural / forestry residues	Wastes (Industry & households)	CO_2
Main	2	2	-
Complementary	4	1	-
Prospective	1	1	3

Figure 2.8 compares the importance of residues and wastes between the clusters (Y-Axis) and provides some insights into different strategies regarding the origins of the feedstocks used in the clusters (X-Axis). Based on the discussions with the interviewees, the clusters were positioned according to their focus on imports or local feedstocks.

Only BioVale (United Kingdom) and BIO.NRW / CLIB2021 (Germany) consider residues or wastes as a key resource, while these are just complementary or prospective feedstocks for the remaining five clusters (see Y-axis of Figure 2.8). However, the clusters still seem to struggle with the practical implementation of using wastes and residues on an industrial scale. Most projects are still on the level of feasibility studies and research and development.

The results show that three of the seven clusters have a strong strategic focus on locally produced feedstocks (BIO.NRW/CLIB2021, IAR and Bioeconomy cluster) and use only very small amounts of imported biomass. For BioVale, local feedstocks play an important role (focus on local wastes and residues) but the cluster also makes use of the biomass supply (mainly wood chips) coming in from the regional Humber seaports. Biobased Delta and Northern Netherlands are much more inclined to import biomass, but they also have an interest in

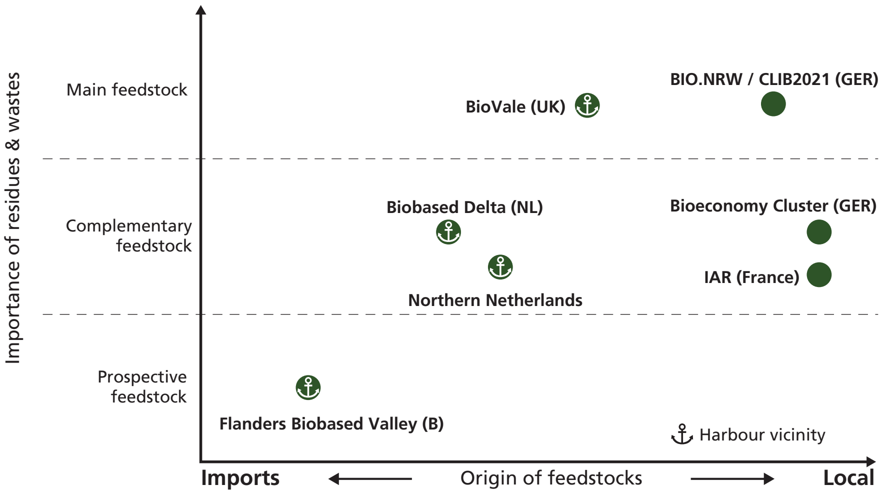

Figure 2.8: Origin of biomass in the clusters (local vs imports) and importance of wastes & residues

using locally available resources, e.g. from local sugar beet farmers (Biobased Delta). Flanders Biobased Valley has the strongest focus on imports.

Reasons for the diverging feedstock strategies between the clusters are manifold. The cluster location plays a key role. The four clusters with a stronger focus on imports are all situated close to major European harbours (see Figure 2.8). However, also the importance of certain actors within the clusters is a relevant factor. Having a strong role of farmers associations within the cluster will most likely result in a much stronger role of local feedstocks as it is e.g. the case with IAR. If the cluster evolves more around industry stakeholders, it is more likely that using local resources is less of a priority if their price and stability of supply is outperformed by imports. Harbour vicinity in combination with strong agricultural stakeholders within the cluster will likely result in dual strategies: the Biobased Delta focuses on the one hand on wood chips and pellets import (Redefinery program) and on the other hand on using locally available sugar beets (Sugar delta programme)[91].

Figure 2.8 also shows that no cluster has a strong focus on imports and at the same time considers residues and wastes as a main or complementary feedstock, while all clusters with a strong focus on local feedstocks (the French and German clusters) consider residues and waste at least as complementary feedstock. The clusters seem to not consider importing wastes and residues via their harbours but only refer to locally available wastes and residues. However, the presence of a harbour does not exclude a key role of wastes and residues, as the example of BioVale shows. The available local farmland also plays an important role in the feedstock strategy. A limited amount of agricultural areas in the region can either result in a feedstock strategy that heavily relies on imports (e.g. Flanders Biobased Valley) or in a stronger focus on making use of residues and wastes (e.g. BIO.NRW / CLIB2021).

The interviews provided insights into the diverse feedstock strategies of the clusters, which are the result of their different local circumstances. This diversity in strategies and regions is also reflected in publications providing an overview of Europe's bioeconomy regions [57,92] and makes it difficult to design a "one fits all" policy on EU level.

2.3.2.2 Product strategies: Energetic vs material biomass use and promising bioeconomy sectors

All interviewees described a clear shift in their research and innovation programs from energetic to material biomass use. A reason for this development could be concerns that the support for e.g. biofuels and bioenergy might be reduced. We can indeed notice some negative signals for certain energetic biomass sectors, e.g. that the revised Renewable Energy Directive (RED II) caps the contribution of biofuels produced from food and feed crops to the Renewable Energy goals[93]. However, the RED II also contains a target for advanced biofuels and an overall renewables target of 32% for 2030, to which bioenergy could significantly contribute if it meets the sustainability criteria. However, NGOs increasingly pressure the sector to reduce direct biomass use for energy and to focus more on cascading and material applications of biomass [94–96]. Also on national level we can observe some tendencies towards the cascading principle and material and high value-applications, such as in the discussions on the climate agreement of the Netherlands[97].

Two cluster representatives mentioned that they prefer a business model that works without subsidies and thus see a more stable business environment and better chances to compete on the market for high quality products, especially if the bio-based products offer additional features that fossil competitors do not offer, such as biodegradability. Another interviewee explained the trend to material biomass uses in research & innovation with the higher research needs in these sectors, while there are already more mature technologies for biofuels and bioenergy. Furthermore, bioenergy usually requires larger feedstock volumes to be profitable, while e.g. fine chemicals require lower biomass inputs. Densely populated bioeconomy regions without harbour access and small agricultural areas as North-Rhine Westphalia thus showed a strong focus on pharmaceuticals, fine and specialty chemicals.

The cluster's focus on materials refers to research and innovation projects and does therefore not mean that the energetic use of biomass only plays a minor role. The clusters could still have large installed capacities of e.g. biogas or biofuel plants, while their research and innovation increasingly evolves around material biomass use. For example, this is the case for the Rodenhuizedok biorefinery cluster for biofuels and bioenergy in Flanders (Biobased valley) or the Drax power station in Yorkshire, one of the largest European bioenergy plants (BioVale). Furthermore, some local policies support energetic biomass uses, such as in the Haut de France region (IAR cluster) for biogas.

Looking at the turnover of bioeconomy sectors in the countries the clusters are located in (Belgium, France, Germany, Netherlands, United Kingdom), we observe a rising turnover of bio-based chemicals, pharmaceuticals, plastics and rubber (NACE classification) by 6.4% between 2011 and 2015, while the turnover of biofuels decreased by 44% in the same period[59]. However, bio-based electricity demonstrated the by far highest percentage growth with almost 123% [59]. The International Energy Agency (IEA) projects a decline for conventional biofuels until 2023 but an increase for novel advanced biofuels from non-food crops[98]. Furthermore, bioenergy capacities are expected to increase in Europe from 41.7 GW in 2017 up to 49.9 GW in 2023, while co-firing biomass is increasingly questioned[98], e.g. the Netherlands is planning not to issue any new grants for co-firing biomass[99].

Hence, looking at commercial developments we see a mixed picture and cannot identify a clear trend towards material biomass uses. Nevertheless, the shift in the cluster's research and innovation towards material biomass use could be a relevant indicator for the future focus of the bioeconomy clusters.

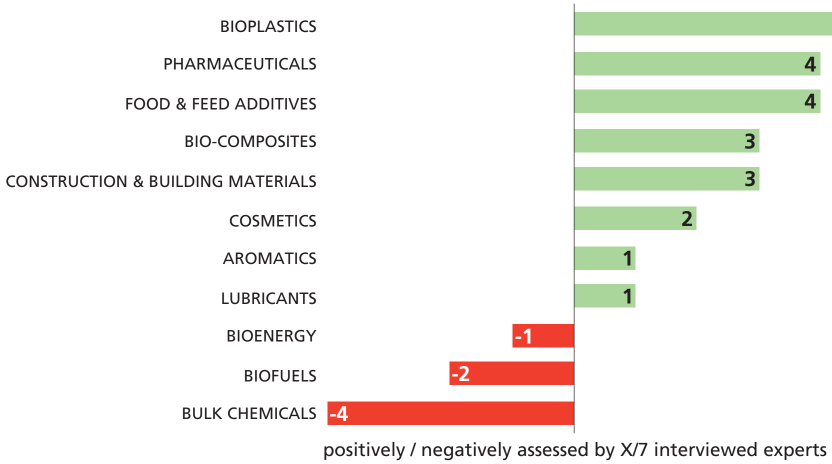

Figure 2.9: Potential of sectors for the bioeconomy according to the 7 cluster experts; 1)

Notes: A minus symbolises a negative assessment (little prospects) of the sector; The aggregation level of product sectors is not coherent, due to the open answers the interviewees could provide. For example aromatics are intermediate chemicals that could end up in a variety of products like cosmetics or pharmaceuticals.

We asked the cluster experts which product sectors they see as promising (or not) for the future bioeconomy. Not all felt comfortable in providing an encompassing answer as such an assessment is rather speculative and potentially sensitive concerning their own strategy.

Figure 2.9 shows that the experts mostly mentioned rather higher value applications of biomass as promising sectors. Especially bioplastics, pharmaceuticals and food and feed additives are considered to have a significant potential for the bioeconomy. Bio-composites and (innovative) construction and building materials were also named, which could be classified as lower to medium value on the bio-based value pyramid (see Figure 2.7). High volume but low value applications like bioenergy, biofuels and bulk chemicals were even assessed negatively by the cluster experts.

In general, most of the experts see potential mainly in highly functionalised biomass applications with high economic value such as fine and specialty chemicals. They do not see great opportunities in competing with fossil alternatives in lower value applications like bulk chemicals under current circumstances, due to their low price and lacking policy support. This suggests a change in the bioeconomy, as in particular bulk chemicals received attention in the past as a promising sector[100,101].

To conclude, the interviews revealed a product strategy of the clusters that demonstrates a shift in the clusters' research and innovation programmes from energetic to material use of biomass and an upward movement on the bio-based value pyramid towards low volume but high value biomass applications. The implications of these results for the CBE are discussed in section 24

2.3.2.3 The role of biorefineries in the clusters

IEA's biorefinery definition (see Chapter 3.1) includes more "traditional" biorefineries combining e.g. biofuel with food and feed production as well as more recent biorefinery concepts, e.g. lignocellulosic biorefineries producing a range of chemicals together with bioenergy and/or biofuels. While current biorefineries are mainly based on a single conversion technology, the goal is to move towards integrated biorefineries, i.e. a combination of several conversion technologies that allow for more flexibility and cost reduction[48]. Cherubini et al. (2009) and Gnansounou and Pandey (2017) provide an overview of the wide range of biorefinery types, which can be classified according to their feedstocks (e.g. lignocellulosic, oil or starch crops), their platforms (intermediates like syngas or sugars) and their products (e.g. fuels, chemicals).

All seven clusters are working on biorefinery concepts for their region and five of them have at least one biorefinery (pilot) plant in place or under construction. Biorefineries on commercial scale only exist in three clusters and they are producing a combination of biofuels and animal feed. One of these plants closed down shortly after the interviews took place[103]. All clusters are working towards lignocellulosic biorefineries and three of them have already pilot plants in place or under construction. Examples are listed in Appendix E.

Without going into detail on the projects, we can conclude that there are significant efforts towards biorefineries and the integrated production of bio-based energy, fuel, chemicals and materials throughout all of the considered clusters. The more advanced biorefinery projects with a high technology readiness level still evolve around bioenergy and biofuels production. However, the trend in research and development seems to go towards a more integrated co-production of a range of bio-based products with an increased importance of chemical applications such as in the pilot plants of the clusters IAR, Bioeconomy cluster and Northern Netherlands[104–106].

2.3.2.4 The role of circular solutions, cascading and recycling

When asked to define the term cascading, the interviewees revealed differences in their understanding of the concept. Six interviewees referred to the term as (1) favouring the highest value-added use of biomass and three also see (2) making use of the entire feedstocks including all by-products (i.e. via co-production) as part of cascading. Three of the interview partners defined cascading as (3) the sequential use of biomass. All three definitions are common interpretations of the term and all three approaches also have its role to play in the CBE. As discussed in Chapter 3.1, we define cascading as the sequential use of bio-based products and refer to the other two interpretations as value optimisation (1) and co-production in integrated biorefineries (2). Implementing the CBE requires a proper communication of the concept and a clarification of the term cascading across the bioeconomy sector. The same is true for other important CBE concepts as Näyhä[107] indicated.

The interviews and the analysis of publicly available cluster strategies and business plans showed that there is so far no coherent concept or strategic focus within the clusters on cascading use, recycling and generally on how to deal with bio-based products at the end of their life. Only Bio-based Delta articulated the goal of updating its strategy to incorporate the national policy goals regarding the CE[91]. Three representatives stated that they have at least one project dedicated to end-of-life options within their clusters but said that the topic is not in the focus of most projects. Two interviewees referred to rather small-scale activities of individual companies in their region in establishing closed-loop recycling schemes for their products, e.g. for bio-based carpets.

Two experts see one of the reasons for this comparably small role of the topic in their cluster in the fact that the bioeconomy is still quite nascent and that the focus is thus more on the preceding steps, i.e. developing and improving the technologies and products before thinking about potential end-of-life scenarios. Figure 2.10 shows challenges and drivers seen by the interviewed experts for implementing circular solutions. Impeding policies and regulations

Figure 2.10: Challenges, hurdles and drivers for implementing circular solutions according to the 7 interviewed experts

were mentioned most, e.g. the classification of a material as waste often limits its further use. According to the interviews, two challenges for implementing recycling schemes on a bigger scale are the comparably still small share of bio-based products on the market and the difficulty of tracing bio-based materials flows throughout value chains. Furthermore, three interviewees see higher costs of circular business models, e.g. because the low oil price facilitates the use of virgin feedstocks compared to recyclables. The interviewees also refer to technological challenges and difficulties in recycling composite materials. Moreover, inconsistent waste management practices across Europe were mentioned as a hurdle as well as the lack of waste management companies accepting and adapting to bio-based products. One interviewee also feared an unstable supply of wastes and residues as a resource and a dependency on production changes of companies supplying these wastes and residues. Lastly, also a lack of funding for circular business cases was seen as a challenge by one expert.

As a driver for circular business cases three interviewees named that investors increasingly ask for end-of-life solutions. Two experts also simply see a business case for using wastes and residues as complementary feedstock to reduce costs. Three interviewees see policies and regulations as a way to foster circular business models, e.g. by taxing GHG-emissions, fostering demand via public procurement and by introducing regulations that prevent the contamination of products hampering recycling. Fostering circular product design is seen as key, e.g. by educating product designers accordingly. Furthermore, efforts towards enhancing the cooperation along the supply chain and amongst regions have been mentioned as well as better showcasing the benefits of cascading biomass use.

2.4 DISCUSSION

2.4.1 Implementation of CBE elements in the clusters

The interviews showed that the clusters are mostly addressing elements of the CBE related to the production phase (top of Figure 2.6): They increasingly consider residues and wastes as a feedstock and work on resource-efficient biomass use in integrated biorefineries. Using residues and wastes has the potential to achieve higher GHG-emission reductions compared to primary biomass use[108] and could reduce feedstock costs (see Figure 2.10). Residues and wastes could potentially meet a significant part of global biomass demand by 2050 but sustainability constraints should be acknowledged (see e.g. Hanssen et al.[109] on biomass residues).

The clusters seem to address CBE elements aiming at the use and end-of-life of bio-based products (i.e. product design, recycling and cascading) only to a limited extent. Nonetheless, the interviews confirmed a stronger focus on material biomass uses within the clusters' research & innovation programs, which we described as a likely development in a CBE (se section 2.3.1.3). Furthermore, the interviewees see more opportunities in products with a high economic value (Figure 2.9) and thus indicate a movement up the bio-based value pyramid (Figure 2.7). This movement theoretically allows for more recycling and cascading options as a higher resource quality of biomass is maintained. However, in practice, this cascading potential is difficult to realise.

According to Braungart, McDonough, and Bollinger[110], product groups can be structured in biological nutrients (i.e. biodegradable materials) or technical nutrients, i.e. materials with the potential to stay in the technical cycle via reuse and recycling. The latter requires collection, separation and recycling schemes for bio-based products that are often not in place yet, and which are probably only profitable when implemented on a large scale; but most bio-based products still have a comparably small market share[111,112]. Moreover, many products are difficult to collect, separate and recycle [86], amongst them several named as promising in Figure 2.9: i.e. cosmetics, pharmaceuticals, lubricants, additives, bio-composites. For these materials aiming for their integration into the biological cycle, i.e. by making them biodegradable, seem like more promising approaches. From the sectors in Figure 2.9, only bioplastics and construction & building materials have a higher recycling and cascading potential because they are produced on larger scale and are easier recoverable compared to the other product groups. However, also for these sectors, this potential strongly depends on (1) the availability of suitable collection, separation and recycling systems and (2) on the design of the products, which should avoid the use of composites or incompatible and hazardous materials [113] and meet other waste management requirements. Despite being highlighted as a driver in the interviews, the analysis of CBE publications and cluster interviews/strategies showed that circular product design does not seem to play a significant role so far. Moreover, design for durability and a prolonged and shared product use did not appear to be a major topic in the analysed CBE publications.

2.4.2 Towards a sustainable, circular bioeconomy?

Implementing the overarching CBE principles, i.e. sustainability and optimising the value of biomass over time, in practice is a challenge, as it requires perfect foresight and cooperation across value chains. Furthermore, various potential trade-offs need to be considered, e.g. between sustainability dimensions or optimising product design for either prolonged use or easy repair or recycling. The benefits of CBE strategies like using wastes and residues, recycling and cascading still have to be proven in practice and are probably case specific. A paper by Daioglou et al.[45] indicated that recycling and recovery options *"do not necessarily reduce energy demand or carbon emissions"*. Case studies on cascading pathways of bioplastics, textiles, paper and wood conducted by Fehrenbach et al.[80] showed environmental benefits for intelligently designed cascading pathways, even though not overwhelmingly large for some cases. Bais-Moleman et al.[114] showed significant GHG emissions reductions for cascading wood; however, they also acknowledge short-term trade-offs with the energy sector.

By focusing on material biomass uses and aiming for high-value biomass applications, the interviewees indicated a movement up the bio-based value pyramid. While this theoretically increases the recycling and cascading potential, it might lead to a trade-off with energetic biomass use: Using biomass to displace fossil-based electricity, heat and transport fuels at a large scale might offer a higher absolute GHG mitigation potential than using biomass for material applications. For example, Daioglou et al.[115] showed that biomass use in the electricity sector promises the highest GHG-mitigation potential when replacing coal. However, as other renewables such as solar and wind are reaching higher shares in the future energy mix[116], this mitigation potential will likely diminish over time. While the electricity sector could use other renewables for decarbonisation, biomass is one of the few short term decarbonisation options for the chemical industry, heavy road transport and marine and aviation sectors[4,15,16,46,47]. The CBE could reduce the competition between biomass uses by using biomass resource-efficiently and cascade it over potentially multiple material applications before delegating it to an energetic use.

The EU bioeconomy strategy sees a sustainable and circular bioeconomy as a key contributor to a GHG-neutral Europe [117]. The importance of GHG-emissions in the CBE discourse is also reflected in the frequent use of keywords related to climate change in the analysed CBE publications. However, there are also various other sustainability aspects to be considered, which are not highlighted as much in the analysed CBE literature (see 2.3.1.1). Amongst them are social aspects, and impacts related to feedstock production like land-use change and eutrophication. By keeping biomass in the loop, the CBE has the potential to reduce primary feedstock demand and its related emissions. However, this only holds true if the CBE does not follow the paradigm of continuous economic growth [118], offsetting its potential benefits with excessive biomass use and rebound effects[63]. After all, both CE and bioeconomy are resource-focused concepts that do not address degrowth topics[119]. Criticism of the CE and bioeconomy also largely applies to the CBE, even though the ambition is to be better, to be *"more than bioeconomy or circular economy alone"* [49,63].

To achieve this goal, the CBE will need to address its critics claiming a lack of evidence regarding its environmental and social value[61,63] by following a comprehensive sustainability vision and showing its benefits case by case. However, it is a complex question which biomass uses, cascading chains and end-of-life strategies are most beneficial in terms of reducing emissions. To what extent the current cluster strategies support a sustainable, circular bioeconomy can thus not be finally answered.

2.5 CONCLUSIONS AND RECOMMENDATIONS

The concept of a CBE has increasingly found its way into European bioeconomy strategies and reports. A CBE as defined in this paper focuses on (a) the sustainable, resource-efficient valorisation of biomass in integrated production chains (e.g. biorefineries) while making use of residues and wastes and (b) on optimising the value of biomass over time via cascading steps. This will likely reduce the direct energetic use of biomass in favour of material applications to enable a prolonged and more resource-efficient biomass use (e.g. via reuse, recycling and cascading).

The analysed bioeconomy clusters seem to move towards a more circular bioeconomy, by increasingly considering wastes and residues as a resource, investigating options for integrated biorefineries and putting more focus on material and high value applications of biomass. However, there is only little focus on solutions concerning product design and the end of life of bio-based products. The interviewees highlighted as key challenges for implementing circular strategies impeding policies and regulations, costs and the small size of bio-based markets.

While the CBE has the potential to improve the sustainability of the current bioeconomy, the discussion showed that the concept is not inherently sustainable and needs to address its potential drawbacks and trade-offs. Especially social aspects, cascading, product design, and aspects related to product use (durability, sharing) seem to be underrepresented in CBE literature, while the topics biorefinery, wastes and residues as well as waste management are significantly covered.

Practitioners in bioeconomy clusters can support the development towards a CBE by (1) facilitating cooperation between stakeholders along and across supply chains; (2) fostering bio-based product design that facilitates durability, reuse, repair, recycling or biodegradability; (3) fostering the use of residues and wastes as resource; (4) intensifying the cooperation with the waste management sector to ensure that the bio-based products can be integrated in collection, separation, recycling and composting schemes.

However, to move towards a sustainable CBE clear guidance for practitioners in bioeconomy clusters is needed. This and other papers (see e.g. Näyhä[107]) revealed different understandings of key CBE concepts amongst practitioners. This calls for an alignment of CBE terminology. Secondly, all the benefits and trade-offs of the CBE have to be laid out: An integrated assessment of the CBE in the context of the wider economy is needed to analyse the aggregated impacts of different biomass uses and the potential benefits of different end-of-life strategies. Such an analysis could contribute to defining pathways and overall policy goals, but it needs to be complemented by case studies developing specific recommendations for implementing these goals. For instance, research needs to show under which circumstances multiple cascading steps are actually beneficial from an environmental, social and economic point of view but also how these cascading chains can be implemented in practice. For a successful transition to a sustainable CBE many more actors need to be involved; e.g. consumers, investors, as well as architects and engineers need guidance towards the implementation of CBE principles. Moreover, current efforts in establishing a monitoring system for the bioeconomy also need to include indicators measuring circularity.

The diversity observed in the clusters' circumstances highlights the importance of designing specific regional CBE-strategies, taking the local strengths and weaknesses into account while avoiding "one fits all" solutions. Furthermore, policies and research programs should focus more on product design and end-of-life strategies for bio-based products as there are only few initiatives addressing it within the clusters. Focusing on the recycling and cascading potential of bio-based plastics and construction & building materials is recommended, as they are produced on larger scale and are easier recoverable compared to the other named sectors. For the other sectors the interviewees considered promising, i.e. cosmetics, detergents and lubricants, efforts improving their biodegradability seem more promising. Moreover, policies increasing the CO_2 price would foster the CBE by increasing the economic competitiveness of resource-efficiency measures and using wastes & residues over primary resources. For optimising the emission mitigation potential of the CBE, clear policy incentives are needed that do not just foster the bioeconomy as a whole but focus on those biomass uses and cascading pathways that promise the highest emission mitigation potential.

Acknowledgements

We would like to thank all the interview partners for their time and valuable insights. Without them this article would not have been possible.

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2.6 APPENDIX A: PUBLICATIONS ON THE CBE CONCEPT

This table shows the nine documents chosen for the literature review and their coverage of the CBE elements presented in Chapter 3.1.1:

1 = Use of wastes and residues as resource; 2 = Resource-efficiency; 3 = biorefinery; 4 = Maintaining the value of products, materials and resources for as long as possible and/or the waste hierarchy; 5 = Cascading use of biomass; 6 = Waste management (e.g. reuse, recycling); 7 = Circular product design; 8 = Prolonged use / durability; 9 = Sharing economy; 10 = Sustainability; 11 = Social aspects.

Table 23: Publications on the CBE concept and their coverage of CBE elements

Title	Reference	CBE elements covered
Results of initial Scopus search (peer-reviewed)		
The Circular Bioeconomy - Concepts, Opportunities, and Limitations	[120]	1, 2, 3, 4, 5, 6, 8, 9, 10
Towards sustainability? Forest-based circular bioeconomy business models in Finnish SMEs	[63]	1, 2, 3, 4, 5, 6, 7, 9, 10, 11
Redesigning a bioenergy sector in EU in the transition to circular waste-based Bioeconomy - A multidisciplinary review	[75]	1, 2, 3, 4, 6, 10
Towards a sustainable forest-based bioeconomy in Italy: Findings from a SWOT analysis	[62]	1, 2, 3, 6, 7, 8, 10
Do forest biorefineries fit with working principles of a circular bioeconomy? A case of Finnish and Swedish initiatives	[61]	1, 2, 3, 5, 6, 8, 10
Added after bibliography analysis (peer-reviewed)		
Can circular bioeconomy be fueled by waste biorefineries — A closer look	[73]	1, 2, 3, 5, 6, 7, 8, 9, 10
Added after bibliography analysis (not peer-reviewed)		
The circular bioeconomy in Scandinavia	[121]	1, 2, 3, 6, 10
Leading the way to a European circular bioeconomy strategy	[49]	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11
Realising the circular bioeconomy	[72]	1, 2, 3, 4, 5, 6, 7, 10, 11

2.7 APPENDIX B: KEYWORD ALLOCATION TO TOPICS

Similar keywords used in the CBE and BE publications were clustered and treated as synonyms. The below table is the thesaurus file used for the keyword analysis in VOSviewer.

Table 2.4: Keyword allocation to topics

Keyword	Replace by	Keyword	Replace by
Resource efficiencies	resource efficiency	waste disposal	Waste Management
resource use efficiency	resource efficiency	waste incineration	Waste Management
resource-efficient	resource efficiency	waste minimization	Waste Management
Biorefineries	Biorefinery	waste recycling	Waste Management
Biorefinery Concept	Biorefinery	waste reduction	Waste Management
Biorefinery Process	Biorefinery	waste technology	Waste Management
Biorefining	Biorefinery	Waste Treatment	Waste Management
integrated biorefinery	Biorefinery	Waste Water Management	Waste Management
lignocellulosic biorefinery	Biorefinery	Wastewater treatment	Waste Management
Refining	Biorefinery	Agricultural Residues	Wastes & Residues
waste biorefinery	Biorefinery	Agricultural waste	Wastes & Residues
eco design	product design	Agricultural wastes	Wastes & Residues
ecodesign	product design	bio waste	Wastes & Residues
eco-design	product design	Biowaste	Wastes & Residues
productdesign	product design	crop residue	Wastes & Residues
Cascade	Cascading	Food Waste	Wastes & Residues
Cascading use	Cascading	forest residues	Wastes & Residues
Anaerobic digestion	Waste Management	forestry residues	Wastes & Residues
Compost	Waste Management	industrial waste	Wastes & Residues
Composting	Waste Management	lignocellulosic residues	Wastes & Residues
landfill	Waste Management	municipal solid waste	Wastes & Residues
Recovery	Waste Management	organic residues	Wastes & Residues
Recycling	Waste Management	organic waste	Wastes & Residues
resource recovery	Waste Management	organic wastes	Wastes & Residues
solid waste management	Waste Management	Residue	Wastes & Residues
residue valorization	Wastes & Residues	energy conversion	Bioenergy & Biofuels
residue valorizations	Wastes & Residues	energy market	Bioenergy & Biofuels
residues	Wastes & Residues	energy resource	Bioenergy & Biofuels
solid waste	Wastes & Residues	energy resources	Bioenergy & Biofuels
solid wastes	Wastes & Residues	energy systems	Bioenergy & Biofuels
Waste	Wastes & Residues	energy yield	Bioenergy & Biofuels
waste products	Wastes & Residues	Ethanol	Bioenergy & Biofuels

Keyword	Replace by	Keyword	Replace by
waste valorization	Wastes & Residues	Fuel	Bioenergy & Biofuels
waste valorizations	Wastes & Residues	fuel economy	Bioenergy & Biofuels
Waste Water	Wastes & Residues	Fuels	Bioenergy & Biofuels
Wastes	Wastes & Residues	lignocellulosic ethanol	Bioenergy & Biofuels
alternative energy	Bioenergy & Biofuels	Renewable Energies	Bioenergy & Biofuels
Biodiesel	Bioenergy & Biofuels	Renewable Energy	Bioenergy & Biofuels
Biodiesel	Bioenergy & Biofuels	renewable energy source	Bioenergy & Biofuels
bioelectric energy sources	Bioenergy & Biofuels	biobased chemicals	Bio-based Products
Bioenergy	Bioenergy & Biofuels	bio-based chemicals	Bio-based Products
Bio-energy	Bioenergy & Biofuels	biobased materials	Bio-based Products
bioenergy productions	Bioenergy & Biofuels	bio-based materials	Bio-based Products
bioenergy technology	Bioenergy & Biofuels	Biobased Products	Bio-based Products
Bioethanol	Bioenergy & Biofuels	Biochemicals	Bio-based Products
bio-ethanol production	Bioenergy & Biofuels	Biocomposites	Bio-based Products
Biofuel	Bioenergy & Biofuels	bio-composites	Bio-based Products
Biofuel Production	Bioenergy & Biofuels	biodegradable polymers	Bio-based Products
Biofuels	Bioenergy & Biofuels	biological materials	Bio-based Products
Biogas	Bioenergy & Biofuels	biological product	Bio-based Products
biogas production	Bioenergy & Biofuels	biological products	Bio-based Products
Biomass power	Bioenergy & Biofuels	Biomaterial	Bio-based Products
electricity	Bioenergy & Biofuels	Biomaterials	Bio-based Products
energy	Bioenergy & Biofuels	Bioplastic	Bio-based Products
bioplastics	Bio-based Products	pulp and paper	Pulp & Paper
bio-plastics	Bio-based Products	pulp and paper industry	Pulp & Paper
Biopolymer	Bio-based Products	furniture industry	wood products
Biopolymer	Bio-based Products	furniture production	wood products
Biopolymers	Bio-based Products	wood-based products	wood products
bioproduct	Bio-based Products	wooden construction	wood products
bioproducts	Bio-based Products	Animal food	Food & Feed
building materials	Bio-based Products	Food	Food & Feed
Chemicals	Bio-based Products	Food & Processing	Food & Feed
construction industry	Bio-based Products	Food industry	Food & Feed
construction materials	Bio-based Products	food processing	Food & Feed
fibers	Bio-based Products	food production	Food & Feed
industrial chemicals	Bio-based Products	food products	Food & Feed
Lactic Acid	Bio-based Products	food safety	Food & Feed
Lactid Acid	Bio-based Products	Food Supply	Food & Feed
organic chemicals	Bio-based Products	Fruit	Food & Feed
plastic	Bio-based Products	Fruits	Food & Feed
plastics	Bio-based Products	Vegetable	Food & Feed

Keyword	Replace by	Keyword	Replace by
Platform Chemicals	Bio-based Products	Vegetables	Food & Feed
Polymer	Bio-based Products	program sustainability	Sustainability
polymers	Bio-based Products	sustainability assessment	Sustainability
reinforced plastics	Bio-based Products	sustainability criteria	Sustainability
solvents	Bio-based Products	sustainability indicators	Sustainability
succinic acid	Bio-based Products	sustainability issues	Sustainability
succinic acids	Bio-based Products	sustainability transition	Sustainability
value added products	Bio-based Products	sustainability transition	Sustainability
value-added chemicals	Bio-based Products	sustainability transitiond	Sustainability
wood chemicals	Bio-based Products	sustainability transitions	Sustainability
paper and pulp industry	Pulp & Paper	sustainable agriculture	Sustainability
sustainable business	Sustainability	Ecosystem	Environmental Aspects
sustainable chemistry	Sustainability	ecosystem service	Environmental Aspects
Sustainable Development	Sustainability	ecosystem services	Environmental Aspects
sustainable development goals	Sustainability	Ecosystems	Environmental Aspects
sustainable forest management	Sustainability	emission control	Environmental Aspects
sustainable management	Sustainability	Environment	Environmental Aspects
sustainable production	Sustainability	Environmental Aspect	Environmental Aspects
Carbon	Climate Change	environmental assessment	Environmental Aspects
Carbon Dioxide	Climate Change	environmental benefits	Environmental Aspects
Carbon Footprint	Climate Change	environmental concerns	Environmental Aspects
climate	Climate Change	environmental footprints	Environmental Aspects
Climate Change Mitigation	Climate Change	environmental health	Environmental Aspects
Climate models	Climate Change	Environmental Impact	Environmental Aspects
Gas Emissions	Climate Change	Environmental Impact Assessment	Environmental Aspects
Global Warming	Climate Change	environmental impacts	Environmental Aspects
Greenhouse effect	Climate Change	environmental indicator	Environmental Aspects
Greenhouse gas	Climate Change	environmental indicators	Environmental Aspects
Greenhouse gases	Climate Change	environmental issues	Environmental Aspects
low carbon economy	Climate Change	Environmental Management	Environmental Aspects
Methane	Climate Change	environmental monitoring	Environmental Aspects
biodegradation, environmental	Environmental Aspects	environmental parameters	Environmental Aspects
biodiversity	Environmental Aspects	environmental performance	Environmental Aspects
biodiversity conservation	Environmental Aspects	Environmental Protection	Environmental Aspects
Chemical Contamination	Environmental Aspects	Environmental Sustainability	Environmental Aspects
Comparative Life cycle assessment	Environmental Aspects	environmental-friendly	Environmental Aspects
conservation of natural resources	Environmental Aspects	Eutrophication	Environmental Aspects
contaminated land	Environmental Aspects	forest ecosystem	Environmental Aspects

Keyword	Replace by	Keyword	Replace by
contamination	Environmental Aspects	hazardous waste	Environmental Aspects
ecological footprint	Environmental Aspects	land use	Environmental Aspects
land use change	Environmental Aspects	socioeconomic conditions	Social aspects
LCA	Environmental Aspects	socio-economic impacts	Social aspects
Life Cycle	Environmental Aspects		
Life Cycle Analysis	Environmental Aspects		
Life Cycle Assessment	Environmental Aspects		
Life Cycle Assessment (LCA)	Environmental Aspects		
pollution	Environmental Aspects		
soil pollution	Environmental Aspects		
water pollution	Environmental Aspects		
Agricultural worker	Social aspects		
consumption	Social aspects		
consumption behaviour	Social aspects		
economic and social effects	Social aspects		
employment	Social aspects		
environmental awareness	Social aspects		
Food security	Social aspects		
Housing	Social aspects		
rural development	Social aspects		
slca	Social aspects		
Social acceptance	Social aspects		
Social capital	Social aspects		
social life	Social aspects		
social life cycle assessment	Social aspects		
social responsibilities	Social aspects		
social responsibility	Social aspects		
social-economic	Social aspects		
society	Social aspects		
socio-economic	Social aspects		
socioeconomic condition	Social aspects		

2.8 APPENDIX C: CONSIDERED DOCUMENTS AND HOMEPAGES OF THE BIOECONOMY CLUSTERS

Table 2.5: Considered documents and homepages of the bioeconomy clusters

Name	Year	Source
Bio-based Delta, Netherlands		
Business plan Biobased Delta Foundation 2018-2020	2017	91
Homepage of Biobased Delta	2018	122
BioEconomy cluster, Germany		
Endbericht Spitzencluster BioEconomy - Zusammenfassung	2018	123
Homepage of BioEconomy Cluster	2018	124
BIO.NRW / CLIB2021, Germany		
BIO.NRW Cluster der Biotechnologie Nordrhein-Westfalen	2018	125
CLIB2021 Technologie Cluster	2018	126
RIN Stoffströme - Regionales Innovationsnetzwerk	2018	127
Bioeconomy Science Center	2018	128
BioVale, United Kingdom		
BioVale Strategy 2018-2022	2018	129
BioVale	2018	130
Flanders Biobased Valley, Belgium		
Good Practice: Ghent Bioeconomy Valley	2015	131
Flanders Biobased Valley	2018	132
IAR, France		
The Futurol project	2010	106
Recoltes des projets labellises et finances	2015	133
An Original Business Model: The Integrated Biorefinery	2015	134
Homepage of IAR	2018	135

2.9 APPENDIX D: COMPARISON OF KEYWORD USE IN CBE AND BE PUBLICATIONS

Table 2.6: Comparison of keyword use in CBE and BE publications

Production & End-of-life						Sustainability			Product focus		
	CBE	BE		CBE	BE		CBE	BE		CBE	BE
Resource-efficiency	0,01	0,01	Residues & wastes	0,25	0,07	Sustainability	0,32	0,2	Bio-based products incl. food & feed	0,48	0,24
Biorefinery	0,3	0,1	Waste management	0,3	0,07	Climate change & other environmental impacts	0,54	0,35	Bioenergy & biofuels	0,32	0,24
Product design	0,05	0,01	Cascading	0,01	0,004	Social aspects	0,05	0,08			

Indicator: Number of keyword occurrences divided by total number of CBE publications (84) and BE publications (1275) respectively

2.10 APPENDIX E: EXAMPLES OF BIOREFINERY PLANTS AND CONCEPTS IN THE CLUSTERS

Table 2.7: Examples of biorefinery plants and concepts in the clusters

Project name	Cluster	Stage	Feedstocks*	Platform*	Products*
Organosolv	Bioeconomy cluster (Leuna)	Pilot	Lignocellulosic	Sugars, Lignin,	Chemicals & building blocks, polymers & resins (plastics, binder), biomaterials
Zambezi	Northern Netherlands	Pilot	Lignocellulosic	Sugars, Lignin	Chemicals & building blocks, polymers & resins, bioethanol, electricity & heat
Bazancourt-Pomacle Biorefinery	IAR	Commercial	Sugar crops, starch crops	Sugars	Bioethanol, Chemicals & building blocks, animal feed, polymers & resins
Futurol	IAR	Demonstration	Lignocellulosic	Sugars, Lignin	Bioethanol, polymers and resins, electricity and heat
Ghent biorefinery cluster	Flanders Biobased Valley	Commercial	Oil crops, starch crops	Oils	Bioethanol, biodiesel, animal feed
Redefinery	Biobased Delta	Business plan	Lignocellulosic	Sugars, Lignin	Chemicals & building blocks, bioethanol, biomaterials, electricity and heat

* according to biorefinery classification scheme of Cherubini et al. [68]

Chapter 3

The plastics integrated assessment model (PLAIA): Assessing emission mitigation pathways and circular economy strategies for the plastics sector

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ABSTRACT

Integrated assessment models (IAM) study the interlinkages between human and natural systems and play a key role in assessing global strategies to reduce global warming. However, they largely neglect the role of materials and the circular economy. With the Plastics Integrated Assessment model (PLAIA), we included plastic production, use, and end-of-life in the IAM IMAGE. PLAIA models the global plastics sector and its impacts up to 2100 for 26 world regions, providing a long-term, dynamic perspective of the sector and its interactions with other socioeconomic and natural systems. This article summarises the model structure, mathematical formulation, assumptions, and data sources. The model links the upstream chemical production with the downstream production of plastics, their use in different sectors, and their end of life. Therefore, PLAIA can assess material use and emission mitigation strategies throughout the whole life cycle in an IAM, including the impacts of the circular economy on mitigating climate change. PLAIA projects plastics demand, production pathways and specifies the annual plastic waste generation, collection, and the impact of waste management strategies. It shows the fossil and bio-based energy and carbon flows in product stocks, landfills, and the emissions in production and at the end of life.

Highlights

- We included plastics production, use, and waste management into an Integrated Assessment Model (IAM).
- Our model PLAIA provides a long-term, dynamic perspective of the global plastics sector until 2100 and its interactions with other sectors and the environment.
- PLAIA can assess the impact of material use and emission mitigation strategies throughout the whole life cycle of plastics.

Graphical abstract

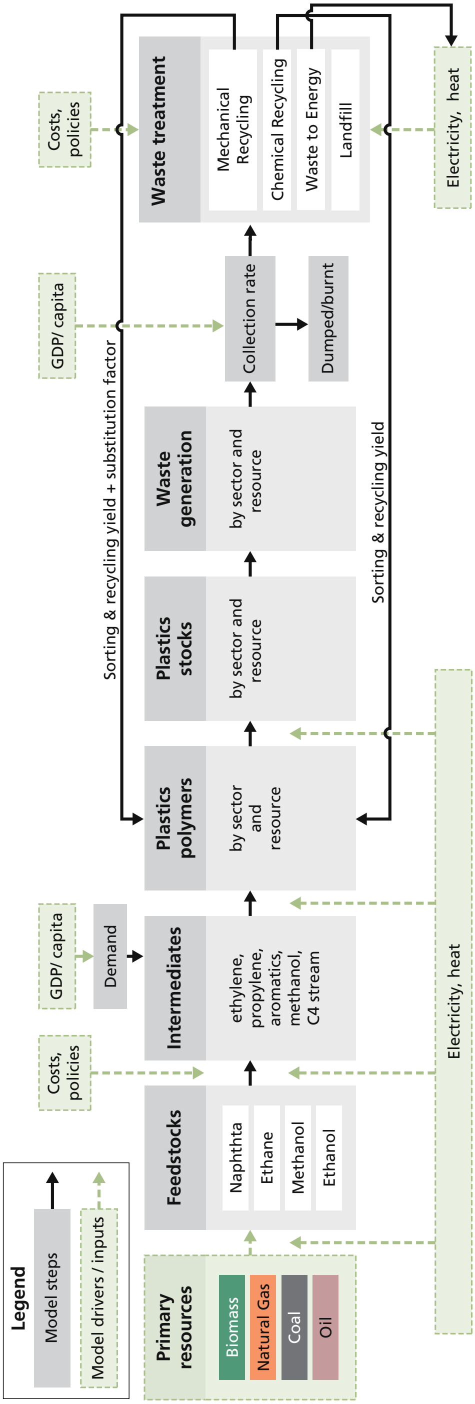

Figure 3.1: Graphical abstract - The plastics integrated assessment model (PLAIA)

3.1 OVERVIEW OF THE MODEL

3.1.1 Purpose of the model

Plastics show the fastest-growing production of all bulk materials globally - with considerable sustainability implications. The plastics sector caused 4.5% of global Greenhouse-gas (GHG) emissions in 2015[8]. GHG emissions of the sector could quadruple until 2050 with current growth rates in production[9]. Moreover, plastics pose a significant challenge once they become waste, contributing to global plastic pollution[10]. Yet, remarkably few model studies have dealt with scenarios for plastics and response strategies on a global scale until 2050 and beyond.

We are the first to add the plastics sector and circular economy strategies to one of the leading global energy, land, and emissions integrated assessment models the Intergovernmental Panel on Climate Change (IPCC) uses to analyse the pathways to achieve the Paris climate targets. Our Plastics Integrated Assessment (PLAIA) model is designed to assess the entire life cycle of plastics globally and for 26 world regions, making it one of the first models of the plastics sector that allow for a global and regional analysis. Moreover, it is the first plastics model that assesses the long-term dynamics of the plastics sector until 2100.

The PLAIA model allows the assessment of different strategies in the production, use, and waste management of plastics and how they affect the sector's material flows, energy use, and emissions. Examples of such strategies include changes in plastic demand, renewable energy use, feedstock substitution in plastic production (e.g., biomass instead of coal and oil), product lifetime extension, reuse, and recycling. As part of an integrated assessment model (IAM), PLAIA also provides insights into the interactions of the plastics sector with the energy and agricultural sectors as well as with the climate, water, and land systems (see section 3.1.1).

Key questions this model is designed to answer include:

- Quantifying the material and energy use of the plastics sector and the corresponding GHG emissions for different scenarios.
- Quantifying the impact of different GHG mitigation strategies for the plastics sector on its material and energy use and GHG emissions.
- Quantifying the impact of circular economy strategies for the plastics sector on its material and energy use and GHG emissions.
- Analyzing the impact of other economic sectors and natural systems on the plastics sector and vice versa.
- Analyzing trade-offs between sustainability targets, e.g., between climate and circular economy targets, or climate and land use.

This article describes the structure, mathematical formulation, data sources, and assumptions of the PLAIA model and discusses its key limitations. It is supposed to complement accompanying articles that discuss the model's results for different scenarios (Chapter 4). The purpose of this publication is to support other researchers in developing similar models of the plastics sector and to provide methodological details to interested readers of our other publications based on the PLAIA model.

3.1.2 The Non-energy demand and emissions (NEDE) model

The NEDE model was developed to assess trends in energy and feedstock use and explore possible mitigation strategies for the chemical sector until the year 2100[45]. NEDE is embedded in the TIMER model, a recursive dynamic simulation model of the energy system and part of the integrated assessment model IMAGE[115]. The model assesses developments for 26 world regions (see Figure 3.9 and Table 3.1 in the Appendix. The IMAGE model structure is displayed in Figure 3.2 and described in PBL and Stehfest et al.[136,137].

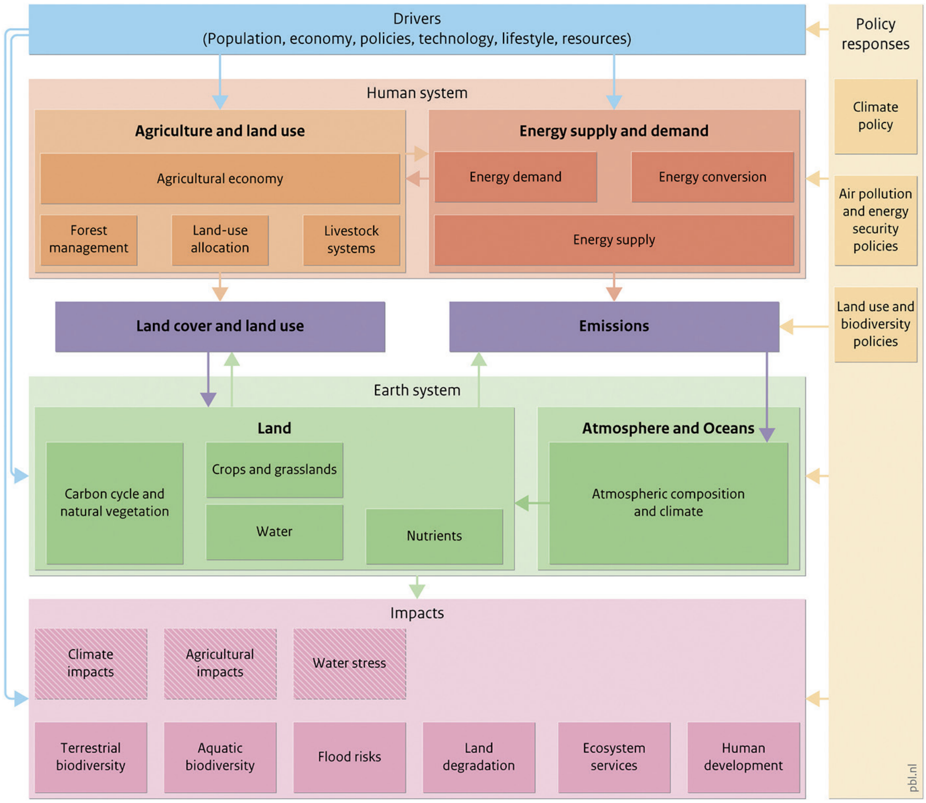

Figure 3.2: The framework of the IMAGE model. For a detailed explanation of IMAGE see PBL and Stehfest et al.[136,137].

NEDE is driven by the product demand for high value chemicals (HVC), ammonia, methanol and refinery products, which are determined by assumptions on population and economic growth (GDP/capita), e.g., based on the shared socioeconomic pathways (SSP) [138]. In NEDE, HVC are defined as the outputs of steam cracking, which are the intermediate building blocks of the organic petrochemical industry (ethylene, propylene, butadiene, aromatics)[45]. Refinery products are representing the heavier refinery products coming from the distillation of crude oil and consist of lubricants, aromatics and bitumen in NEDE[45]. This demand can be met by different technology pathways, based on the primary energy carriers coal, oil, natural gas, and biomass. The primary energy demand of each carrier (expressed in GJ) is calculated based on the conversion efficiencies of the technology pathways and their respective market shares[45].

In its original version, NEDE models the chemical sector very aggregately, and its scope is limited to the upstream production of intermediates like ethylene and aromatics, thus excluding the downstream production steps and end-products[45]. A reason for this aggregated approach with an upstream focus is the complexity of the chemical industry whose outputs can end up in a wide range of products and sectors. Furthermore, the data availability is very limited for the chemical sector, especially when compared to other sectors[5]. This is particularly true for downstream flows of the sector. Concentrating on the upstream flows of the chemical sector is sufficient when focusing on the supply side, e.g., feedstock choices and their impact on climate change. However, for investigating the impact of climate change mitigation options towards the use and end-of-life (e.g. material efficiency, waste treatment options), it is key to better integrate the downstream flows of the chemical sector[5]. This is particularly important for the chemical sector, since large parts of its carbon inputs are not directly emitted but sequestered in products. Only by covering the entire life-cycle can we get a proper grasp of the entire emissions of the chemical sector and their timing.

By linking the upstream chemical production of intermediates with the downstream production of plastics, their use in different sectors, and their end of life, the model could assess mitigation strategies throughout the whole product life. In that way also the impacts of the circular economy on mitigating climate change can be analysed.

3.1.3 Steps to enhance and further develop the NEDE model

To that end, we created the plastics integrated assessment model (PLAIA) as an extension of the NEDE model. PLAIA added the representation of plastics, waste generation & stocks, and end-of-life options to the NEDE model and updated the model's Greenhouse-gas accounting accordingly. Figure 3.3 shows the positions of the sub-modules referred to in this paper within the IAM IMAGE.

We created regional demand curves for chemicals from steam cracking by analyzing the relationship between GDP/capita and historic steam cracker capacities and by applying average

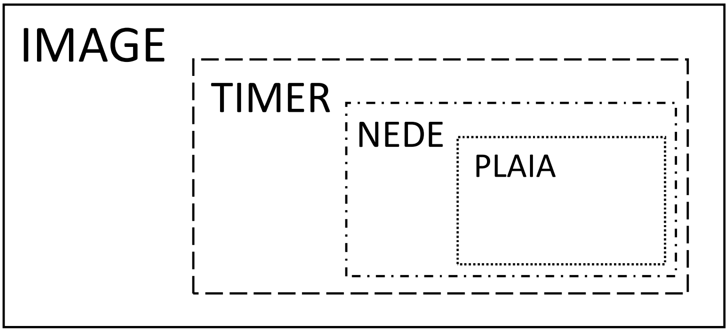

Figure 3.3: Structure within the IMAGE model: The IAM IMAGE incorporates the energy model TIMER. TIMER contains the sub-model NEDE, which covers the chemical sector. PLAIA is integrated in NEDE and models plastic production and waste management.

product yields per steam cracker feedstocks. We defined the demand for plastics as a share of the demand for steam cracking products (i.e., ethylene, propylene, C4), refinery products (aromatics, propylene, C4 stream), and methanol. NEDE already provided us with the primary energy use of the upstream monomer production, to which we added the energy use for producing plastic polymers from these monomers and the transformation of polymers to plastic products. Furthermore, we modelled the plastic stocks in use and the yearly waste generation of plastics by defining the lifetime distribution of plastics by sector. We determined regional waste collection rates based on GDP/capita development. We added waste treatment technologies and modelled their shares in waste treatment based on costs and policy interventions. We also altered the existing carbon accounting system in NEDE to cover the implications of the end-of-life scenarios and carbon sequestration in products. This methodology paper is structured along with these steps. In Figure 3.11 in the Appendix, we show the structure of the updated NEDE model, indicating the additions of the PLAIA model.

3.1.4 Overview of the Plastics Integrated Assessment model (PLAIA)

PLAIA is integrated into the NEDE model and could be considered an add-on to the original NEDE model of Daioglou et al.[45]. Figure 3.4 shows the structure of PLAIA. The model covers the plastics value chain from the extraction/production of primary resources (fossil & biomass) for the production of chemical feedstocks and intermediates to plastic polymer production, the transformation of polymers to products, their use in different sectors, and their end of life (= cradle to grave/cradle assessment).

The modelling steps and data sources behind PLAIA are described in detail in the following chapters.

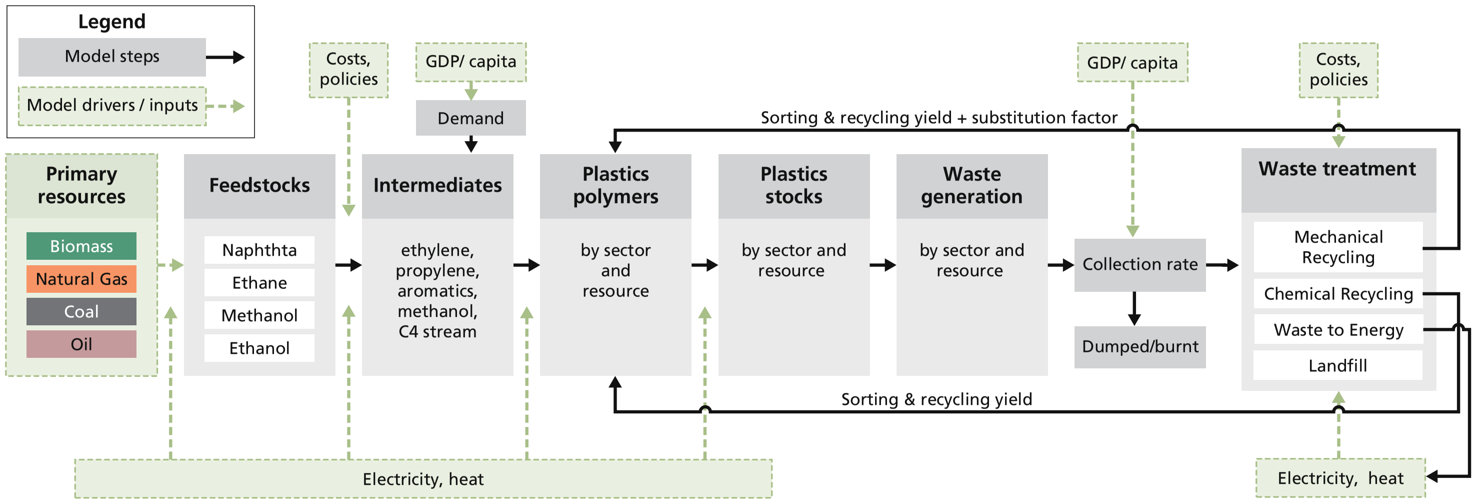

Figure 3.4: Overview of the plastics integrated assessment model (PLAIA)

3.2 CREATING DEMAND CURVES FOR CHEMICALS FROM STEAM CRACKING

The chemical sector is highly complex and has numerous and diverse technology routes and outputs. The data availability for the chemical sector is very limited, especially when compared to other industry sectors like steel and aluminium[5]. This makes it difficult to disaggregate the data used in NEDE to achieve a better representation of the outputs relevant to recycling, such as plastics.

The demand projections for high value chemicals (= steam cracking outputs) are based on data of the Oil & Gas Journal[139], which produces annual international surveys of Ethylene from Steam Crackers by country. This dataset provides the production capacity of global steam crackers in terms of feedstock (Ethane, propane, Butane, Naphtha, Gas Oil, Other) and ethylene output from 1980 to 2010. In its version of 2014[45], NEDE only considered ethylene as an output from steam crackers. We complement this representation by also accounting for other steam cracker outputs like propylene, aromatics, and C4 streams. A relevant part of these streams is used for plastics production. We need to consider the full extent of steam cracker outputs to integrate plastics into NEDE successfully.

Using average steam cracking yields from Levi & Cullen[5] and the dataset mentioned above of the Oil & Gas Journal, we estimated the historic production capacity of ethylene, propylene, aromatics, and C4 streams for each IMAGE region. Using the lower heating values (LHVs) of Table 3.2 in the Appendix, we translated the mass-based yield matrix of Levi & Cullen[5] into energy-terms (GJ product/GJ feed), see Table 3.3 in the Appendix. The resulting historic production capacity of HVC for all 26 IMAGE regions was then analyzed to determine the relationship between per capita production capacity growth and GDP per capita growth.

In the absence of actual demand data, we assumed historical production capacity data to represent demand. We model the HVC demand in NEDE as a logistic growth relationship between HVC production capacity per capita and GDP per capita (see equation 1), assuming a steam cracker utilisation rate of 90%. Equation 1 was selected as it proved to best match historical developments and expected behavior (i.e., demand growth levels off with higher GDP per capita).

Equation 1: Model for projecting HVC demand (in GJ) as a function of GDP/capita for an IMAGE Region R

$$HVC_R = \alpha_R \times e^{\frac{-\beta_R}{GDP_R}} \times Pop_R$$

The coefficients Alpha and Beta were determined with regressions of the regional historical data. The regional historical data from the Oil & Gas Journal is displayed in the Appendix (Figure 3.10).

Table 3.4 in the Appendix shows the final regression coefficients Alpha and Beta chosen as input for equation 1. We decided to limit Alpha to 13 to prevent the model from growing to unrealistically high future per capita consumption of HVC with very high economic growth (i.e., historical per capita consumption in most developed regions levels off below this value). Furthermore, choosing this limit of 13 allows for improved replication of historical production.

For the IMAGE regions USA, Western and Eastern Europe, Korea, South East Asia, and Japan, specific regression coefficients could be identified that fit the available historical data for those regions. Together, these regions covered 60% of global HVC production in 2010. For the remaining regions, the regression coefficients are based on all worldwide data points (excluding years with 0 HVC values). This is because most of the remaining regions still have limited historical data available or extremely low HVC production levels (and GDP per capita), making it nearly impossible to formulate an appropriate regression. We made exceptions for China and the Middle East, both significant producers of steam cracking products. As the data of China only reflects the early development of HVC capacity, no sound Alpha could be calculated. In the absence of better data, we fixed the Alpha for China at the value of the rest of the world (value of all global data points, see Table 3.4 in the Appendix) and calculated the Beta according to this Alpha and the existing data for China. For the Middle East, we based the regression coefficients on the global GDP/capita development instead of regional GDP/capita values. Our analysis showed that the production capacity in the Middle East does not have a strong correlation with its regional GDP development but rather the global one. This could be explained by the fact that its oil & gas industry is in large parts built for export.

3.3 INTEGRATING PLASTICS DEMAND INTO THE MODEL

3.3.1 Overview

The chemical sector can be structured in upstream production and downstream production. The upstream production produces the primary or intermediate chemicals such as light olefins (ethylene, propylene), BTX aromatics (benzene, toluene, xylene), ammonia, methanol, and a C4-stream (e.g., Butadiene, Isobutene). The production of these intermediates is responsible for around two-thirds of the energy consumption in the chemical sector (feedstock and process energy combined). The downstream production includes plastic polymers, agrochemicals (e.g., fertilisers, surfactants, pesticides), and specialty chemicals (e.g., solvents, paints, industry catalysts). These products are then further processed for final use in, e.g., packaging, agriculture, construction, or pharmaceuticals[4,5].

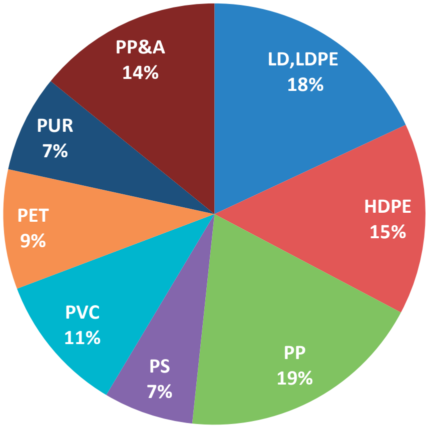

Figure 3.5: Shares of plastic types in the global market (adapted from Geyer et al [6])

Figure 3.5 shows the major plastic types on the market, which are relatively constant over time (2002 – 2014) and analysed regions (EU, USA, China, India) according to Geyer et al.[6]. In plastic resin production, the upstream production of monomers like ethylene and propylene is responsible for most energy use and emissions. For example, for PP, LDPE, LLDPE and HDPE, the energy use of the upstream monomer production is around 90% of total energy use and is responsible for about 80% of the global warming potential of plastic resin production[140,141]. After the upstream monomer production, some monomers are processed to further intermediates like ethylene glycol, styrene, terephthalic acid, or vinyl chloride before the final polymerisation step. During polymerisation, monomers are chemically bound in chains to plastic polymers[5,142] and then converted and manufactured into their final products.

The NEDE model covers the upstream chemical production from steam cracking, refinery, ammonia, and methanol production as described in Daioglou et al.[45]. To integrate plastics into the model, we analysed the shares of the upstream chemical intermediate products going to plastics. Then, we added the downstream production steps to the model (polymerisation and transformation).

3.3.2 Determining the energy content of plastics

TIMER is an energy model, and as such, it deals in energy units. Therefore we integrated plastics not in mass but energy terms. For that purpose, we define an average lower heating value (LHV) of plastics to convert the data from mass into energy units and vice versa.

We calculated average LHVs for each plastic type based on values collected from the literature (See Table 3.5 in the Appendix). We multiplied them with the global shares of these plastic types on the market (from Geyer et al[6], see Figure 3.5). This provided us with a weighted average LHV of 35 GJ/t for plastics, which is within ranges provided in literature as a plastic average[143]. Using this LHV, we converted the plastic data from mass to energy units and vice versa. We also used this LHV for calculating the benefits of incinerating plastic waste in waste to energy plants (see section 3.7.2.3). We assume this LHV to stay constant over time. This assumption is discussed further in the limitations (section 3.10).

3.3.3 Defining the upstream sources of plastic production

NEDE structures the demand for non-energy/chemical use into the aggregated categories high value chemicals (olefins coming from steam cracking), refinery products (aromatics, bitumen, lubricants), methanol, and ammonia. Most intermediate feedstocks for plastic production (see Chapter 3.3.1) originate from steam cracking, but also olefins and aromatics are sourced in significant numbers from refineries (163 million tonnes in 2013, according to Levi and Cullen[5]). Furthermore, plastic feedstocks also originate from methanol: the methanol-to-olefins (MTO) production is growing, and also other methanol-sourced products like acetic acid or formaldehyde end up in plastics[144].

Unfortunately, there is very little information on the flows within the chemical sector. Levi and Cullen[5] provide an overview of the global flows within the chemical sector for the year 2013 based on the limited available information. In the absence of other comprehensive data sources and historical material flows in the chemical sector, we used their data to define the shares of intermediate chemicals from steam cracking, refinery sourced olefins, and aromatics, and methanol going to plastics.

Using the data of Levi and Cullen[5], we defined the shares of ethylene, propylene, BTX aromatics, methanol, and C4 (upstream chemical production) going to plastics. Then we allocated these streams to the NEDE categories HVC, methanol, and refinery products, based on the share of these categories in the respective upstream intermediates. In a third step, we defined the LHVs of these intermediates and converted the mass units of Levi and Cullen[5] into energy units to adopt them to the NEDE model (million metric tonnes to Petajoule). We used those converted values to calculate the shares of steam cracking products (0,84), refinery sourced olefins & aromatics (0,59), and methanol (0,4) going to plastics (in energy terms). In the absence of historical data, we assume these shares to remain constant throughout the years.

3.3.4 Adding refinery-sourced olefins to the model

In the model version of Daioglou et al.[45], refinery products are based on historic production capacity data by country from the yearly "Worldwide Refining Survey"[145] and cover bitumen, lubricants, and aromatics. However, this data does not include olefins originating from refineries such as the C4-stream and propylene (via fluid catalytic cracking, FCC)[5,141]. Their full consideration in the model is important, as 32% of the monomers used in plastic production come from refinery-sourced aromatics, propylene, and C4[5]. In the absence of better data, we adopt a simplified representation of refinery-sourced propylene and C4 as a function of aromatics produced in refineries. The refinery-sourced aromatics are modelled based on historical data from the Oil & Gas Journal (see Daioglou et al.[45]). We express the demand for refinery sourced propylene and C4 as a fixed factor of 0.5 GJ propylene and 0.8 GJ C4 demanded per GJ of refinery sourced aromatics. These relationships between propylene and C4 to refinery-sourced aromatics are based on data of Levi and Cullen[5].

3.3.5 Determining the demand for plastics

Geyer et al.[6] provide historical data for global primary plastic production. However, regional historical production or demand data is scarce. There is (partial) plastic production data publicly available for some world regions or countries, but not consistent, and it does not cover all relevant regions. A regional representation of the chemicals and plastics sector is essential for the IMAGE model, as it provides results for 26 world regions.

The data used in NEDE for modelling the demand for chemical intermediates (see Daioglou et al.[45]) offers more granularity: The Oil & Gas Journal provides country-specific data for refineries and steam crackers and the Methanol Institute for methanol production. Therefore, we chose to represent plastic demand as a function of the upstream chemical production data. This approach allows for a regional disaggregation of plastic demand in the model.

Using the shares of upstream chemical products going to plastics (see section 3.3.3), we defined plastic demand (P) as a function of the demand for the NEDE product categories high value chemicals (HVC), refinery products (RP) and methanol (M) over time (t).

Equation 2: Plastic demand as a function of demand for upstream chemical products over time

$$P(t) = 0.84 * HVC(t) + 0.3 * RP(t) + 0.4 * M(t)$$

In section 3.10.1, we compare the results of our approach with other global projections of plastic demand.

3.4 PRIMARY PLASTIC PRODUCTION

3.4.1 Upstream monomer production

The upstream chemical production of monomers covers most of the energy use and emissions in plastic production (see section 3.3.1). These monomers and chemical intermediates used in plastic production (i.e., ethylene, propylene, aromatics, C4 stream, methanol) come in large parts from steam crackers but are also sourced from refineries and complemented by methanol. In the model, these chemical intermediates can be produced via various technology routes using coal, oil, natural gas, and biomass as feedstock. This upstream chemical production, its costs, and energy use are represented in the model as described in Daioglou et al.[45].

3.4.2 Polymer production

Plastic monomers are processed into plastic polymer resins. We calculated the weighted average electricity and heat use of plastic resin production using production data for HDPE, LDPE, LLDPE, PP, PET, PVC, and PS from PlasticsEurope[140,141,146,147] multiplied by their respective market share (see Figure 3.5). Together, these plastic polymers cover almost 80% of the plastics market[6]. Therefore, we chose them as a proxy for the entire market. The energy use covers the final polymerisation step and intermediate steps like the production of ethylene glycol, purified terephthalic acid, chlorine, and vinyl chloride production. Table 3.6 in the Appendix shows the electricity and heat use.

Monomer production is the most significant cost driver of plastic production compared to the downstream production steps, such as polymerisation: Looking at global price differences between olefin monomers (i.e., ethylene, propylene) and polymers (i.e., LDPE, HDPE, PP), we usually see that the olefin price fluctuates around 80% of the polymer price. For example, S&P Global's[148] comparison of price averages between ethylene and LDPE as well as propylene and PP from November 2017 to March 2018 showed that ethylene and propylene prices range between 0.75 and 0.85 of LDPE and PP prices. Most reports only refer to a total plastic resin costs or price and do not detail the share of monomer production and polymer production, see e.g., Hestin, Faninger, & Milios or Villanueva & Eder[143,149].

We take the costs of the process heat and electricity requirement as a proxy for the costs of producing plastic polymer resins from monomers. The costs of electricity and heat are endogenously modelled in IMAGE TIMER and are region-specific and dynamically modelled over time[136,137]. While this does not cover the entire costs of the polymerisation process, it is a reasonable proxy. Energy use is the most significant cost factor for plastic resin producers, typically around 70%[150].

3.4.3 Polymer transformation into products

Plastic resins are further transformed into semi-finished plastic products via different technologies like calendaring, blow moulding, compression moulding, extrusion, or injection moulding[151]. As it is difficult to capture the entire variety of plastic products in a long-term global-scale model, we chose to include a proxy value for the energy use of plastic transformation.

We calculated a weighted average energy use and efficiency of plastic transformation per plastic type, based on data by Keoleian et al. [151], who collected energy use and efficiency data for nine transformation technologies and data on the share of these technologies for transforming different plastic types. Combining this data of Keoleian et al.[151] with the market shares of the plastic polymer types of Geyer et al.[6], we calculated an overall weighted average energy use and efficiency for plastics transformation (see Table 3.7 in the SI, translated into the metric system).

The data of Keoleian et al.[151] did not cover all plastic types but included PP, PVC, PET, and PE (data applied for both LDPE/LLDPE & HDPE). Together, they cover 72% of the plastic market, according to Geyer et al.[6], and were thus chosen as a proxy for the entire plastic market. For simplification purposes, we assume that the entire energy use comes from electricity which is mainly in line with Keoleian et al.[151] (apart from blow molding) and Franklin Associates [152], who state that 97% of energy use in injection molding is from electricity and 95% in thermoforming.

As for the cost of polymerisation, we just account for energy use costs, using the electricity prices generated in TIMER. We also apply the plastic transformation to recycled plastic resins.

3.5 PLASTIC STOCKS & PLASTIC WASTE GENERATION

Plastics are used in various products and sectors. The model distinguishes between eight sectors of which packaging is the biggest in yearly production with around 37% (see Figure 3.6).

The duration of the use phase of plastics varies significantly between the sectors. While plastics in packaging usually become waste within a very short time (typically less than a year), plastics used in building and construction have a much longer time of use and thus often become waste decades later. We defined sector-specific lifetimes of plastic products via lognormal probability

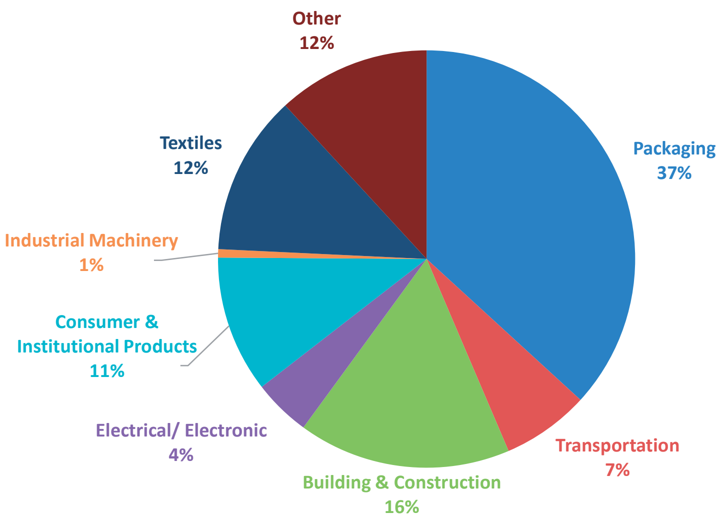

Figure 3.6: Sector shares in plastic demand

Notes: Own illustration based on data compiled by Geyer et al.[6]; data for 2002-2014 from Europe, the United States, China, and India.

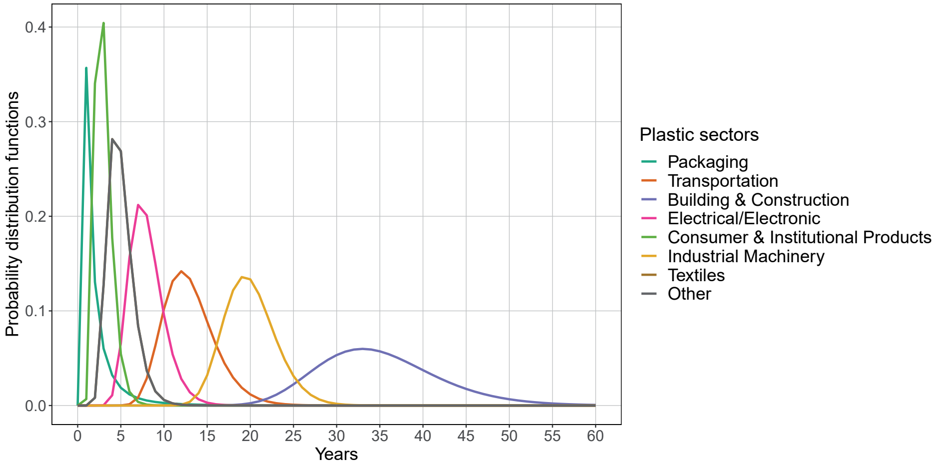

Figure 3.7: Plastic product lifetime distributions;

Notes: Own illustration based on data compiled by Geyer et al.[6]. Textiles and Other have the same lifetime distribution.

distribution functions based on data compiled by Geyer et al.[6], see Figure 3.7. Geyer et al.[6] assumed that the sector "Other" has the same distribution as textiles.

Table 3.8 in in the Appendix shows the mean use time of plastics and their standard deviation per sector as compiled by Geyer et al [6].

Based on the use time of plastic products, we calculate the plastic waste (PW) generated each year (t) per region (R) and sector (S) as the sum of the plastics (P) produced in year (t-n) becoming waste in year t. n is the number of years to look back for plastic production data, with an upper limit of 60 years (maximum lifetime of plastics in building & construction, see Figure 3.7). The distribution factor (d) defines the share of plastics produced in year t-n becoming waste each year, according to the probability distribution in Figure 3.7.

Equation 3: Calculating the annual plastic waste generation

$$PW_{R,S}(t) = \sum_{t=1}^{n} d_S(t) \times P_{R,S}(t-n)$$

Having the yearly plastic production and waste generation, we can calculate the plastics stocks (= plastics in use) in each of the eight use sectors. We define the Plastic Stock (PS) per sector (S) and region (R) in year t as the cumulative plastics produced (cP) minus the cumulative plastic waste generated (cPW):

Equation 4: Calculating the plastic stocks (= plastics in use)

$$PS_{R,S}(t) = cP_{R,S}(t) - cPW_{R,S}(t)$$

3.6 PLASTIC WASTE COLLECTION

Collection rates describe the share of generated waste that is collected and thus entering the waste management system. In the model, the collected waste is then available for the different waste treatment options like landfilling, energy recovery, mechanical and chemical recycling. The waste that is not collected is assumed to be openly burnt or dumped, potentially ending up in waterways and oceans.

We model the waste collected by region as a function of GDP per capita, using data collected by the World Bank[153]. The data of the World Bank provides collection rates for four levels of Gross National Income (GNI) ranges per capita, see Table 3.9 in the Appendix. We assume that the GNI/cap equals GDP/capita to allow for an integration of these numbers into the model. Differences between GNI and GDP are usually marginal, i.e., ranging from 0.1-2%. We smoothened the transition between income levels to avoid sudden jumps in collection rates once a region passes the World Bank thresholds.

3.7 PLASTIC WASTE TREATMENT

3.7.1 Overview

In the model, the amount of collected plastic waste can be directed to mechanical recycling, chemical recycling (via pyrolysis), waste to energy, or landfilling. The collected plastic waste is allocated to the different plastic waste treatment options (WTO) based on the WTO's relative costs, policy interventions (e.g., CO_2 tax, bans), and technological or economic constraints (e.g., maximum recyclability).

The costs of the waste treatment options consist of a fixed cost factor, endogenously modelled variable costs (for heat, electricity, and diesel use), and a CO_2 tax (if part of the modelled scenarios). These costs are reduced by the endogenously modelled benefits of replacing primary plastics (for mechanical & chemical recycling) or heat and electricity (for waste to energy). The costs exclude the collection & transportation of plastic waste. Table 3.10 in the Appendix summarises the data used for modelling the waste treatment options in PLAIA. The data and assumptions for each option and calculation steps are explained in this chapter.

3.7.2 Data & assumptions for the waste treatment options

3.7.2.1 Mechanical recycling

The process of mechanical recycling can be structured in a simplified manner in (1) collection, (2) sorting/pre-treatment, (3) transportation to a recycling plant (and transportation of sorting and recycling rejects to Energy recovery or landfilling), and (4) recycling.

Collection & transportation

The energy use and emissions of the collection and transportation steps are related to transport distances for refuse collection and distances to waste treatment options. These steps are excluded from the model as it is challenging to find representative data for all IMAGE regions on average transportation distances. Furthermore, the literature showed that these steps have a comparably small impact on the recycling process's overall energy use and GHG emissions[149,154]. We discuss this further in the limitations (section 3.10).

Sorting & recycling yields

Sorting/pre-treatment yields for plastics vary by sector. We took a weighted average of 0.75 based on data from Hestin et al.[149], who provide information on sector-specific sorting yields for Europe.

The term recycling rate is used differently through literature and reporting schemes, leading to overestimating recycling. It often refers to the number of plastics sent to recycling (i.e., the input recycling rate). It thus ignores all the process losses along the way in sorting and recycling (output recycling rate).

We use the term recycling yield to describe the process efficiency of recycling itself (excl. collection rate and sorting yields), so the amount of plastic resin produced per sorted and pre-treated waste input. Many sources describe yields for specific plastic resins and technologies which achieve relatively high rates for some resins (e.g., 0.88 for PET[35]) while other plastic resins show much lower yields. We chose a weighted average recycling yield of 0.75 for the model, based on the European industry data compiled Hestin et al.[149], covering the most relevant plastic resins. This value is also within the range of other sources, estimating the overall plastic recycling yield to be within 0.7 and 0.78[155] or 0.7 – 0.82[4].

We define the overall recycling efficiency (RE) as the product of the sorting (SY) and recycling yield (RY):

Equation 5: Recycling efficiency

$$RE = SY * RY$$

Equation 5 leads to an overall recycling efficiency of 0.56.

Treatment of rejects

A significant part of plastic waste sent to mechanical recycling will not end up in recycled plastics due to the losses in sorting and recycling processes. In the model, these rejects of mechanical recycling are sent to the other waste treatment options according to their market shares in plastic waste treatment (see section 3.7.3).

Energy use

The model covers the heat and electricity use of the sorting/pre-treatment and the recycling step. We identified in the literature that values for electricity use in sorting plastic waste range from

20 to 105 kWh/t plastic waste[35,149,156–159]. We chose the average of the electricity consumption for sorting reported in the literature as input into the model (see Table 3.10 in the Appendix).

We took data on electricity and heat requirements for recycling plastic waste from Faraca et al.[35], who provide electricity and heat use data for recycling different polymers (PP, PE, PET, PS) based on a review of 11 studies. By multiplying these energy requirements with the polymer market shares of Geyer et al.[6], see Figure 3.5, we calculated the weighted average heat and electricity use for mechanical plastic waste recycling (see Table 3.10 in the Appendix). This average does not consider all polymers, but it represents almost 70% of the plastics market.

Costs

The costs of mechanical recycling in the model cover the sorting/pre-treatment and recycling of plastics. They consist of (1) a fixed cost factor (incl. machinery and labour), (2) electricity and (3) heat costs, and (4) a CO_2 price based on the emissions of the process (for those scenarios with CO_2 price).

For the fixed cost factor, we took data from a study of Plastics Recyclers Europe[149], which provides costs data per plastic sector and an average. The data of Hestin et al.[149] is collected from European recycling plants and is in the range of other cost data reported in the literature[35,160].

Costs for electricity and heat are around 10% of total mechanical recycling costs in Denmark and the United Kingdom[35,161]. We deducted 10% of the fixed cost factor from Hestin et al.[149] and added endogenously calculated variable costs for heat and electricity from the TIMER model. We display the cost data in Table 3.10 in the Appendix.

Substitution rate

In how far recycled plastics can replace primary plastics depends on their quality. This quality is usually lower, which is also reflected in the price of recycled plastics[31,155]. This difference can be expressed by a substitution factor/rate. We set this substitution rate to 0.81 in the model, based on European Commission and Rigamonti et al.[31,156]. The use of this substitution rate in the model captures the price differences between primary and recycled plastics. The price recycled plastics achieve on the market lowers the recycling costs and thus influences the share of recycling in the model (see Equation 7).

Cost of primary plastics production as a cost driver for recycling

The cost of primary plastics is a strong driver of recycling rates. If it is high, the demand for secondary plastics rises; if it is low (e.g., due to a low oil price), there is little incentive for recycling. To cover this effect in the model, we define the total costs of mechanical recycling (MRcosts) for Region R in year t as follows:

Equation 6: Calculating the overall cost of mechanical recycling

$$MRcost_R(t) = \left(FCF_R(t) + Celec_R(t) + Cheat_R(t)\right) - SF(t) * CPP_{R(t)}$$

With FCF being the fixed cost factor, Celec being the costs of electricity, Cheat the costs of heat, SF the substitution factor, and CPP the costs of primary plastics production.

3.7.2.2 Chemical recycling via pyrolysis

The model includes a chemical recycling route via pyrolysis. The route is a three-step process that (1) transforms plastic waste into naphtha, (2) naphtha into chemical intermediates via steam-cracking (e.g., ethylene), and (3) into plastic polymers via polymerisation.

This process also has a preceding sorting step. We took over the same electricity values as for sorting for mechanical recycling (see section 3.7.2.1). However, we used a higher sorting yield of 0.85[35], as pyrolysis tolerates more impurities and mixed-plastics than mechanical recycling. Only some plastic types like PVC could cause processing problems and have to be sorted out.

Pyrolysis was already part of the original NEDE model. Therefore, the technology data (costs, efficiency, and energy-use) are described in Daioglou et al.[45]. We only added the pre-sorting step and the polymer production process as described in section 3.4.2. Table 3.10 in the Appendix summarises the yield and costs of the production of naphtha from plastic waste and the transformation of naphtha to monomers.

Plastics produced via chemical recycling can achieve the same quality as primary plastics. Therefore a substitution factor of 1 is assumed (unlike for mechanically recycled plastics, see the previous section). We deduct the financial benefit of replacing primary plastics with chemically recycled plastics from the chemical recycling costs (same as for mechanical recycling, see equation 7). As chemical recycling is a hi-tech category mainly applied in high-income economies, we activate this technology only for regions that surpass a GDP/cap of 10,000 USD. Historically, chemically recycling played a notable role only in Japan. Therefore, chemical recycling is not considered in the historical model results (before 2021) apart from Japan.

3.7.2.3 Waste to energy

Waste incineration technology is traditionally mainly used in high-income countries like Japan, the USA, and many European countries[153,162,163]. In recent years, also China drastically increased its waste to energy capacity[163]. High land prices, high technical capacity, and high financial resources favour waste to energy [162]. Furthermore, a waste composition with a high calorific value is important, which means that the share of organic waste should be low (which is usually not the case in low-income economies). To reflect this in the model, we allow waste to energy as an option only for regions that surpass a GDP/capita of 10,000 USD/capita (2005 USD).

Heat & Electricity generation efficiency

Modern incineration plants are with energy recovery and can be optimised for heat or electricity generation, resulting in a wide range of potential efficiencies. In Europe, 59 million tonnes of waste is treated in 314 waste to energy plants with an average heat and electricity efficiency of 34.6% and 14.9%, respectively[31]. At the same time, almost 36 million tonnes of waste are incinerated without energy recovery in Europe. Using the weighted average of the European Commission[31], we assume an average heat efficiency of 22% and an electricity efficiency of 9% in the model, representing a total waste incineration capacity of 94.7 million tonnes with and without energy recovery.

We assume the same technology data globally in the current model version. While we expect increases in average generation yields for Europe in the future, emerging economies might struggle with imperfect waste-to-energy processes in the initial years. Therefore, this weighted average seems more suitable for defining a global value for generation efficiencies.

Costs

In the model, we assume fixed total costs of plastic waste incineration and deduct the variable revenue from selling the generated heat and electricity. Additionally, we apply a CO_2 price on the fossil share of incinerated plastic waste (in the scenarios where a CO_2 price is present). The incineration of bio-based plastic is exempt from this CO_2 price. The costs and energy generation efficiency are shown in Table 3.10 in the Appendix.

The costs of plastic waste incineration used in the model are from a Dutch waste-to-energy plant and include operational and capital expenditure and bottom ash treatment[160]. The costs of this plant are similar to other values reported in the literature, e.g., of a Danish waste to energy plant[35] and a German plant[164]. Furthermore, it is within the range of gate fees for incinerators in the European Union[143]. However, these gate fees also include taxes and the credit from selling energy[143].

The variable benefit of selling the generated electricity and heat is based on the generation efficiencies and the regional heat and electricity price generated in TIMER.

3.7.2.4 Landfilling

Landfilling is globally the dominant end-of-life option, but the quality of landfilling and the costs can differ significantly from open dumps, controlled landfills to properly engineered sanitary landfills[153,162].

Energy use

Energy use plays a small role in operating landfills compared to the other end-of-life options. Per tonne of waste, 1.5-4 l of diesel are used for excavation works and daily operations and 5 to 8 kWh electricity[165]. We used an average of these values for PLAIA; see Table 3.10 in the Appendix.

Costs

Landfill costs strongly vary between locations, ranging from a minimum of 10 USD/t in low-income countries to 100 USD/t in high-income countries[153]. Including landfill taxes, the costs could increase further: for example, Sweden has an average landfill cost of 155 Euro/t, and individual landfills could also be more costly as an example of 219 Euro/t in Austria shows[166].

The costs depend on *"terrain, soil type, climatic factors, site restrictions and regulatory factors,"* as well as the type of waste and the volume of waste received[167]. Capital costs (incl. site development, closure & post-closure) have the highest share in total costs. Operational and Monitoring costs also vary: They account for 38% of total costs in an example of the United Kingdom[164] and 44% in an example of rural Oklahoma in the USA[167].

In the model, we differentiate between three cost factors:

1. Fixed costs, including Site Development & Construction, Equipment, personnel, monitoring, closure & post-closure, taxes
2. Electricity costs for operation
3. Diesel use for excavation works & daily on-site operations

The use of electricity and diesel is based on Manfredi et al.[165], see energy use above, and the costs are calculated via the respective energy prices generated in TIMER. They only account for a very marginal share in total landfill costs in the model, ranging from mostly less than 1% up to ca. 5% for some regions.

The dominant cost factor is the fixed costs which can vary significantly as described above. These differences are not just due to location-specific conditions (e.g. terrain, climatic factors, scale, personnel costs) but are also driven by regulations, policies, and suitable land availability. This leads to major differences also between high-income countries: While less densely populated countries like the USA or Australia show a high share of landfilling and lower costs, Japan and countries within the European Union have higher costs and lower shares going to landfills[166,168–170].

Covering these dynamics in PLAIA is a challenge. We chose to model regional and future variations in costs of landfilling based on differences in land prices, representing the GDP & population density of a region. The assumption behind our approach is that with rising GDP & population density also costs of landfilling rise, and regions increasingly switch to recycling and waste to energy. We calibrated the model by defining a baseline value Alpha for landfilling costs that keeps the model within realistic landfill cost ranges as reported in the literature [153,166,168,169]. PLAIA uses the dynamic land prices generated in IMAGE, which are based on GDP per capita, population size, and usable area in each IMAGE region. We calculate the regional costs of landfilling in PLAIA as follows:

Equation 7: Costs of landfilling in 2005 USD/GJ plastic waste

$$\text{Landfill Costs}_{\text{R}}(t) = \underbrace{\frac{LP_R(t)}{LPav_{Global}(t)}}_{\{Regional\ land\ price\ factor\}} * \underbrace{\alpha * \frac{LPav_{Global}(t)}{\beta}}_{\{global\ average\ fixed\ costs\}} + \underbrace{\text{EC}_{R,i}(t)}_{\{Energy\ costs\}}$$

With $LP_R(t)$ being the land price in year t for region R, $LPav_{Global}(t)$ being the weighted global average land price (weighted based on region size) and $EC_{R,i}$ are the costs of electricity and diesel use for landfilling per region. Alpha represents the chosen baseline fixed costs for landfilling (set as the global average in 2015) and Beta the baseline global average land price in 2015 (generated in IMAGE). This means that all future landfill costs are calculated (a) in relation to the base year 2015 and (b) changes in regional land prices and (c) regional energy prices.

3.7.2.5 Open burning & dumping

In PLAIA, plastic waste collection is driven by GDP/cap development (see section 3.6). The difference between plastic waste generated and collected is assumed to be openly burned or dumped/littered in the environment. We assume that 30% of this remaining uncollected waste will be burned and 70% dumped. This estimation is based on World Bank data, which provides the share of openly burned or dumped waste for some countries[153]. However, this estimation is uncertain as it is difficult to cover these informal waste disposal methods in national statistics.

3.7.3 Defining the market shares of the waste treatment options

The plastic waste entering the waste management system (= collected plastic waste) is allocated to the different plastic waste treatment options (WTO) based on (1) the WTO's relative costs (a described in section 3.7.2), (2) policy interventions (e.g., CO_2 price, bans) and (3) technological or economic constraints (e.g., maximum recyclability).

For defining the shares of the WTOs we use a multinomial logit function as shown in Equation 8, with C being the cost of each WTO and Region (R) and λ being the logit parameter which defines the elasticity between relative prices. This function allocates market shares amongst WTOs based on their relative costs. Thus, even non-economically optimal options get selected to small extents. Such a function allows for the representation of heterogeneity in waste management and takes into account that, in reality, decisions are not purely economic.

Equation 8: Multinomial logit function to calculate the market shares of waste treatment options

$$WTO_{Share\,R,WTO} = \frac{e^{-\lambda * C_{R,WTO}}}{\sum_{WTO} e^{-\lambda * C_{R,WTO}}}$$

The hi-tech WTOs waste to energy and chemical recycling are only active in regions that surpass a GDP/capita of 10,000 USD (2005), see sections 3.7.2.2 and 3.7.2.3.

3.8 CARBON ACCOUNTING

PLAIA accounts for the carbon flows throughout the entire life cycle of plastics: from the primary energy carrier to the products, their use, and finally, their end-of-life. Also, transformation losses and carbon emissions from heat and electricity use in production and waste management are covered. The model differentiates between the total carbon input to the plastics sector, the total carbon emissions, total carbon in use (sequestered in products), total carbon sequestered in landfills/dumps, and the total carbon recycled. Furthermore, it specifies emissions in plastic production and end of life for the different waste treatment options.

The emission accounting is in line with the 'Good Practice' methods described by the IPCC guidelines for emission inventories[171]. Only fossil carbon is accounted for in emissions as bio-based carbon is assumed to be climate neutral. We include the agricultural process emissions and land-use change emissions of biomass production (calculated in IMAGE). Indirect land-use change emissions are not relevant in IMAGE as the model adopts a food-first principle[22]. Also the production emissions of fossil energy carriers are included.

According to the International Energy Agency (IEA) (2018), emissions of the chemical sector can be distinguished in

1. Energy-related emissions: Emissions created by process energy use. They are responsible for the largest share in emissions, with around 85%.
2. Process emissions: Emission occurring when transforming the feedstock into the product. Those are expressed by the difference in the carbon content of the feedstock (e.g., methane, oil) and the carbon content of the product (e.g., Ammonia, Plastic) and are responsible for around 15% of the sector's emissions.

In PLAIA, all fossil energy carriers used for process energy are assumed to be directly emitted (= Energy-related emissions) unless carbon capture & storage technology is applied. The carbon of fossil energy carriers used as a feedstock for short-lived products (e.g., lubricants, solvents, ammonia, cosmetics, etc.) are assumed to be emitted in the same year of production. For fossil energy carriers used as feedstock in products with a lifetime of more than a year (e.g., plastics), the difference in carbon between the feedstocks and the products is assumed to be emitted, following the example of the IEA[4].

The future fate of the carbon embedded in plastic products (i.e., recycled, landfilled, incinerated) is determined by the waste management practices in the year the lifetime of the plastic product ends (see section 3.5). We assume that plastic carbon going to landfills and dumps stays sequestered for the entire analysed period (up to 2100). We went for this simplified assumption as there is very little literature on the degradation of plastics in different environments [12] and even less on its implications on Greenhouse-gas emissions[172]. Moreover, plastic buried in landfills seems to have a very slow chemical degradation rate[12,172], as discussed in section 3.10.5. For PLAIA, this means that within the model time frame until 2100, all carbon input to the plastics sector is ultimately emitted, apart from the carbon embedded in plastic products in use until 2100, and carbon stored in landfills or dumps.

We calculate the annual carbon balance as follows: Total carbon input (incl. waste generated) – Total carbon sequestered in products with a lifetime of >1 year – Total carbon additions to landfills and dumps. To calculate the annual emission balance in PLAIA, we need to define the carbon contents of plastics. We calculate the carbon content based on the shares of the feedstock energy carriers used for producing plastics, i.e., a weighted average of the carbon contents of coal, oil, natural gas, and biomass bound in plastics. Depending on the shares of these energy carriers in plastics, the carbon content of plastics differs per region and year. While this approach creates an inherently consistent carbon balance in the model, it also has disadvantages (see discussion section 3.10).

3.9 MODEL OUTPUTS

As explained in section 1.1, the purpose of the PLAIA model is to assess the plastics sector's long-term material flows, energy use, and GHG emissions for different scenarios. The results of PLAIA for different scenarios are presented in chapter 4.

PLAIA was designed to run scenarios based on the shared socioeconomic pathways (SSP) and variations thereof. The climate change research community developed the SSPs as a set of alternative futures for societal development[138,173]. The results presented in chapter 4 are based on the second shared socioeconomic pathway (SSP2), which is closely linked to historical patterns in social, economic, and technological developments[138,174]. The following list details the key outputs of the model, referring to the respective figures in chapter 4:

- PLAIA projects the annual plastic production, annual waste generation and plastics stocks based on population and economic development as provided by the SSPs (see Figure 4.1 for a SSP2 baseline).
- PLAIA provides the annual flow of plastics from production to end of life (see the sankey diagram in Figure 4.2) .
- Furthermore, PLAIA details the feedstocks used for plastics production (see Figure 4.6).
- Moreover, the model shows how much plastic waste is collected and how it is treated (see Figure 4.5).
- PLAIA summarises the final energy use of the plastics sector over the entire life cycle, detailing the types of energy carriers used (see Figure 4.4).
- A key output of the model is the overall carbon balance of the plastics sector and its emissions over the entire life-cycle (see Figure-3).

3.10 DISCUSSION OF LIMITATIONS

With PLAIA, we created one of the first global, long-term models on plastic production and waste management as part of an IAM. The model is suitable to analyse the impact of different material use and emission mitigation strategies throughout the entire life cycle of plastics. Our efforts represent the first steps to better model non-energy use, materials, and the circular economy in IAMs. Further improvements are necessary to tackle the limitations of our work.

A fundamental issue was the limited data available for the chemical sector[5]. Also, it is a challenge to find useful data for downstream plastic production and waste management, particularly when aiming to represent the 26 world regions in IMAGE. This lack of data forced us to accept several limitations in our model. We summarise the key limitations and suggestions for improvements in this section.

3.10.1 Modelling the demand for plastics

In the absence of actual demand data, we assumed historical production data to represent demand. Moreover, we chose to model plastic demand based on the upstream production of intermediate chemicals used for plastics (see section 3.5) as no regionally disaggregated plastic production data was available.

To have a benchmark for our approach of modelling plastic demand, we also created a plastic demand projection based on the global plastic production data of Geyer et al.[6]. We assumed a logistic growth relationship between plastic production per capita and GDP per capita (same as Equation 1 in section 3.2, but with an Alpha of 7.51 and a Beta of 14525). Figure 3.8 compares historical data and the projection of global plastic demand based on the data of Geyer et al.

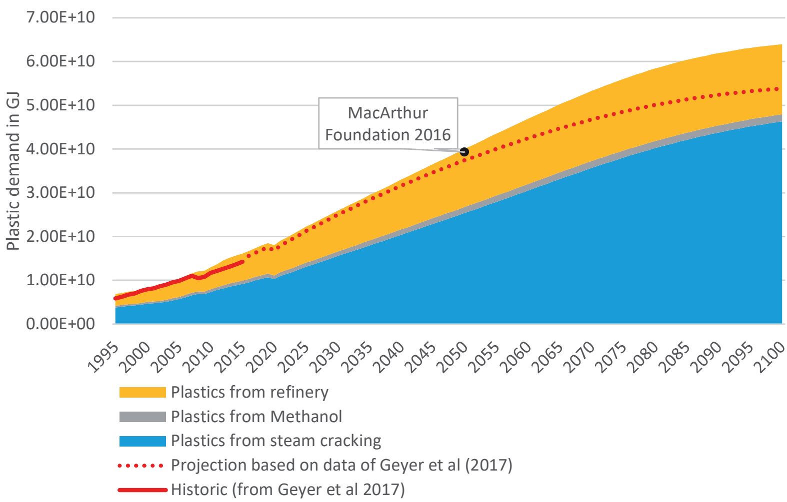

Figure 3.8: Comparison of plastic demand projections (sum of stacked lines is the PLAIA model projection)

(2017) with the demand implemented in the model (as a function of HVC, refinery products, and methanol) for an SSP2 GDP/cap projection.

PLAIA produces similar results to historical values from Geyer et al.[6], even though they differ regarding the dip in historical plastic demand after the financial crisis of 2008. The projections until 2050 are broadly similar, with differences in the range of 4-7% in this period. After 2050, the demand function implemented in PLAIA increases further before levelling off towards the end of the century, while the projections based on Geyer et al. level off at a lower value. For the year 2050, our demand projection based on upstream chemical production is almost the same as the plastic demand projection of the Ellen MacArthur Foundation [155], while the projection based on data of Geyer et al is 7% lower. Therefore, we conclude that our approach of modelling plastic demand as a function of upstream chemical production seems justifiable, especially given the advantage of allowing for a regional disaggregation of plastic demand. However, while our approach produced similar results on a global level as projections based on plastic production data, this might be different on a regional level.

The data that was used for our projections only represents primary plastic production. Therefore, the projections slightly underestimate the total plastic production as they exclude recycled plastics. Recycling of plastics was negligible until 1980[6]. Since the 1980s, recycling rates have increased significantly, but the actual amount of recycled plastics on the market is uncertain, and there is limited publicly available information. The world petrochemicals balance of 2004 from Gielen, Newman, & Patel[175] estimates that around 4% of plastics originate from recycling. A more recent study by PlasticsEurope[176] concluded that in 2018 around 4 million tonnes of plastics produced in the European Union were from recycled materials, which equals around 6% of its production by that year. This number is from a high recycling region, which sends approximately 32% of its plastic waste to recycling[176]; we expect much lower percentages of recycled plastics globally. Furthermore, recycling rates were much lower in the past, meaning that its market share in historical plastic production is likely to be marginal. Therefore, we consider demand derived from primary production data alone to be a sufficiently accurate indicator for plastic demand.

PLAIA does not include the trade of chemicals and plastic (waste). Hence, PLAIA assumes that plastics are consumed and that waste is treated in the producing region. This assumption might lead to underestimating plastic consumption in regions with historically limited chemical production, i.e., developing economies, and overestimating it in regions with high chemical production. In conclusion, our model would benefit from country-specific plastic consumption data and the inclusion of trade. The latter is a challenge because it is difficult to follow trade flows through all life cycle stages of plastics: intermediate chemicals, plastic polymers, plastic products, and finally, plastic waste.

3.10.2 Waste collection

We model waste collection based on GDP/capita development. Since our model assumes that waste is generated and collected in the producing region (see the previous section), we give more weight to the high collection rates in developed economies, probably leading to overestimating global plastic waste collection.

Moreover, modelling waste collection as a function of GDP/cap is a simplification, ignoring other factors driving collection rates such as policy developments and urbanisation. In general, reliable data for collection rates is difficult to obtain for many regions[153]. In many low to upper-middle-income countries (World Bank definition), informal waste systems play a vital role whose contributions are difficult to measure. Furthermore, the World Bank data refers to overall waste collection rates and is thus ignoring differences between specific waste streams such as plastics.

3.10.3 Data bias, omissions, and simplified assumptions

We predominately used technology and cost data from Europe and the USA due to data limitations for other regions. Only endogenously modelled inputs like the energy mix and cost as well as the land price are region-specific. The model could achieve a better regional representation of plastic production and waste management with more country-specific data sources.

Some steps in the life cycle of plastics have a simplified representation or are missing. PLAIA uses proxies and aggregated data on many occasions. While this practice reduces the granularity of the model, it is common practice for an IAM that deals with aggregated, global & long-term developments.

Limitations in the upstream chemical production

The production of chemicals from steam crackers is modelled in detail, covering various fossil and bio-based feedstocks and eight conversion routes. Steam cracking provides ca 65% of monomer inputs in plastics production [5]. The modelling of methanol production is less detailed but still includes estimates for process energy and transformation losses. The refinery is the most opaque source of olefins and aromatics used in plastics[5] and thus the most difficult to cover. Daioglou et al.[45] modelled refinery products in an aggregated, simplified way. Due to missing data, the model only provides the net energy bound in refinery products and thus ignores process energy and transformation losses.

In the absence of continuous historical data, we assume constant shares of steam cracking products, refinery sourced olefins & aromatics, and methanol as feedstock for plastics (see section 3.3.3). While this simplification ignores future changes in production pathways, it can, to a certain extent, be supported by the observation that the chemical sector seems to be relatively constant when it comes to its production pathways and end-uses. Levi & Cullen[5] compared their results for 2013 with two other studies with data of 2004 and 2006. This comparison showed that the shares between the end-use sectors were comparable to the 2013 data of Levi & Cullen[5]. However, fundamental changes in the chemical sector might change the sources of plastic feedstocks in the future.

Limitations in the downstream plastic production

We created proxies for plastic polymerisation and transformation based on weighted averages of current data. This practice ignores potential changes in plastic types and shares of technology routes in the future. Similarly, assuming a constant LHV for plastics is a simplification and ignores future changes in the market shares and the introduction of new, different plastic types. New plastics, particularly biobased ones, could have other chemical structures and thus different LHVs and carbon contents. Currently, biobased plastics represent around 1% of the global market[177]. So far, the majority of the biobased plastics on the market are drop-in plastics, meaning they have the same chemical structure and LHV as their fossil competitors (e.g., bio-PET)[177]. But there are also upcoming plastic types with different structures and lower LHVs, like Polylactic acid (PLA) or Polyethylene Furanoate (PEF). Furthermore, there are also discussions about producing plastics from captured CO_2 in the future[178]. Changes in the chemical structures of plastics could influence our model results in several ways: E.g., a lower LHV would make waste to energy a less attractive solution, while a lower carbon content would reduce the GHG emissions of incinerating plastics. However, it is highly speculative to forecast the future market shares of these new plastic types and how far they would change the average LHV of plastics. Therefore, we chose to keep the simplified assumption of a constant average LHV of plastics.

Limitations in waste management

In cases where data was unavailable or unreliable, we chose to leave it out rather than include it based on uncertain assumptions. This was the case for the collection and transportation of plastic waste, as it is difficult to find representative data for all IMAGE regions on average transportation distances. The impact of this omission on the overall GHG emissions of plastics is marginal but not negligible. For example, Hestin et al.[149] showed that in the European Union, the collection and transportation steps cause around 3-4% and 4-5 % of the GHG emissions of recycling, respectively, depending on the plastic type.

However, collection can have a significant share in total recycling costs[153]. But as all waste treatment options require waste collection & transportation, these costs have a limited impact on the relative costs of the waste treatment options to each other. However, there are differences in transportation needs between the options, depending on the region (e.g., distances to the closest recycling centre or landfill; transport of recycling rejects to landfill or incineration plant). Therefore, ignoring collection & transport is still a limitation of this model.

Next to mechanical recycling, we only include chemical recycling via pyrolysis as an alternative recycling route. There are many other ways of chemically recycling plastic waste. While pyrolysis recycles plastic back to its feedstock, other technologies allow recycling back to monomer. However, these promising routes are still in a research & development stage or are operating on a small scale, with very little public data available. Furthermore, these routes are polymer-specific and mostly require a very pure input stream. Since this model covers plastics in an aggregated way and data availability is scarce, polymer-specific chemical recycling routes are not part of the model yet. Also other plastic waste treatment methods like gasification or emerging technologies like photoreforming are not yet included[179]. With increasing data availability, the model will be updated to include further promising plastic waste treatment methods.

3.10.4 Technological learning

IMAGE and TIMER include technological improvements via learning for the production of energy carriers[136]. In particular, biomass production routes benefit from learning[45]. However, in the current model version, PLAIA does not include technological learning for waste treatment technologies, even though future gains are likely. Already today, sorting and recycling yields can reach higher values for specific sectors or plastic types. Sorting yields for packaging and construction plastic waste could be more than 80%, while other sectors like electronics only reach 50%[149]. Recycling yields for PET plastics could reach more than 90% and are thus significantly higher compared to other plastic types[35].

It is difficult to make sound projections on the development of the recycling sector. The most significant potential improvements are not technology but policy-related. Fostering circular product design and enforcing a better sorting of plastics (e.g., via deposit systems) will likely have a much more significant impact than technical improvements in sorting and recycling machinery. Moreover, we would require data on historical improvements to define technological learning rates for recycling technologies. So far, we kept the assumptions stable over time for the baseline scenario (apart from the endogenous variable costs such as energy prices). However, we will simulate developments towards a circular economy and resulting yield and cost improvements via scenarios in upcoming publications. Nevertheless, future updates regarding technological learning are desirable to improve the PLAIA model further. To achieve this, additional research on technological learning rates for plastic production, mechanical recycling, chemical recycling, and incineration technologies would be required.

3.10.5 Limitations in carbon accounting

Globally the CO_2 emissions of the chemical sector sum up to around 1.5 Gt of CO_2 emissions a year (18% of global industrial CO_2 emissions)[4]. Additionally, the sector produces globally non-CO_2 greenhouse gas emissions in the range of 350-400 million tonnes in CO_2 equivalents a year, mainly consisting of hydrofluorocarbons (HFC) and nitrous oxide[180]. Some of these emissions can be attributed to plastics; for example, the nitrous oxide emissions from adipic acid production used in nylons[180] or the HFC emissions from blowing agents used in producing extruded-polystyrene and polyurethane foams[181]. These additional non-CO_2 emissions are not accounted for in the model.

To achieve an inherently consistent carbon balance, we calculate the carbon content of plastics based on the shares of the feedstock energy carriers used for producing plastics (see section 3.8). However, this approach ignores the chemical transformations from feedstock to product: Not all feedstock carbon might end up in the product. We currently lack the necessary data to include these transformations in the model.

An alternative way would be to use a fixed carbon content that is constant over time and regions. For example, the IPCC emission factor database offers emission factors for plastics[171,182]. Using this exogenously set carbon content leads to inconsistent carbon balances since they do not relate to the feedstock inputs into the model. In extreme cases, when using high amounts of low-carbon feedstock like natural gas, this fixed carbon content could lead to a negative balance, meaning that more carbon is assumed to be sequestered than what went into the production. Furthermore, these exogenous values are averages and differ by the reporting country.

Both approaches are essentially incorrect. To guarantee consistency within the model, we chose to define the carbon content of plastics based on the shares of the feedstock energy carriers used for their production. Compared to a fixed carbon content for plastics based on the IPCC factor, we see that the endogenously modelled weighted global average carbon content is 5-15% lower, varying per year. The IPCC factor assumes a carbon weight of 75% of the plastic weight. This would result in 21,42 kg Carbon/GJ plastic (assuming an average plastic LHV of 35 GJ/t of plastics).

As described in section 3.8, we assume that plastic carbon going to landfills and dumps stays sequestered for the entire analysed period (up to 2100). The key reason for this choice was the limited data on the degradation of plastics in different environments and its implications on Greenhouse-gas emissions[12,172]. However, plastic seems to have a very slow chemical degradation rate when buried in landfills: A review of Chamas et al.[12] showed a half-life of hundreds or thousands of years for most plastics (i.e., HDPE, PVC, PET, PS) when buried. Only LDPE bags showed a short half-life of 4.6 years.

However, the degradation speeds up when the plastics are submitted to sunlight or in marine environments[12,172]. Therefore, one might consider emissions in particular of the openly dumped plastics. Unfortunately, there is not yet sufficient literature to base an emission factor on. Such an emission factor depends on plastic types, the environment they are in, their exposure to sun, heat, and oxygen, and their surface[12,172]. Initial assessments indicate that emissions of dumped plastic waste play a minor role in the global GHGeq budget[172]. Furthermore, the dumping of plastics only plays a small role in the PLAIA model due to the reasons mentioned in section 3.10.2.

3.10.6 Recommendations for further research

This section outlined some of the key limitations we faced when developing the PLAIA model. The major reason for the limitations lies in the data: The collection and provision of data on the chemical, plastic, and waste management sectors need to improve significantly to provide a better basis for research and policies. Model improvements should focus on including trade, technological learning, and a better representation of regional specifics in technologies, costs, and policies (particularly for China, a rapidly growing plastic producer). Moreover, a better link of the PLAIA model to other industry sectors would allow for a truly integrated assessment. It would allow for using wastes from other industry sectors as a resource for chemical production (e.g., black liquor from the pulp & paper industry) and for modelling the competition of plastic materials with alternatives for specific applications (e.g., packaging made of plastics or paper/cardboard). This initial version of the PLAIA model can just be seen as the first step to better integrating materials and the circular economy in IAMs and requires continuous improvement. However, despite its limitations, PLAIA can already provide valuable insights into the impact of different material use and emission mitigation strategies throughout the entire life cycle of plastics.

3.11 APPENDIX

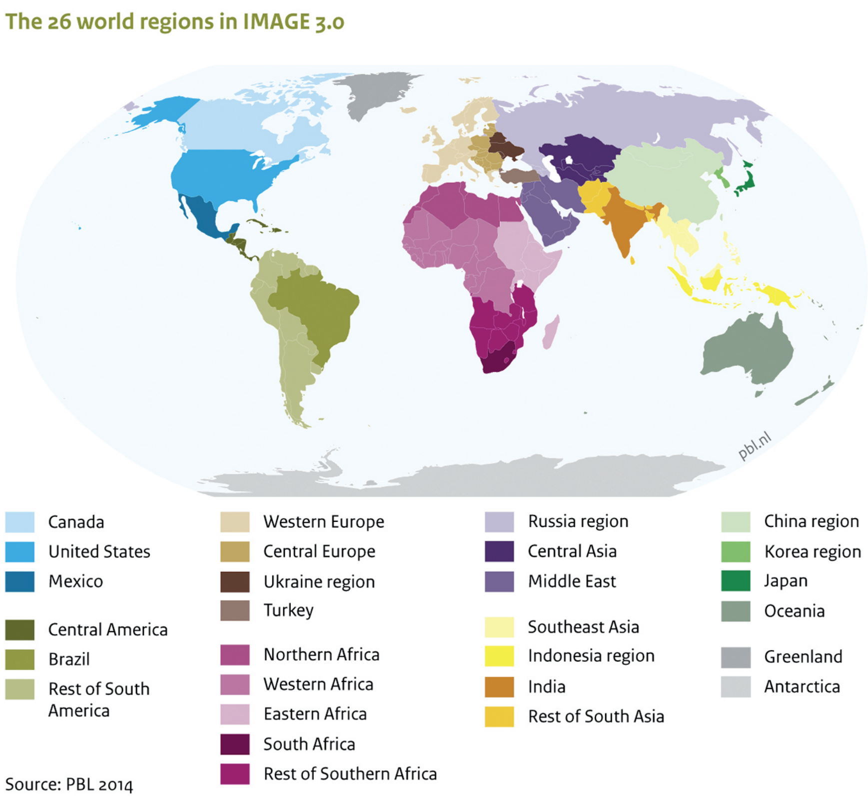

Figure 3.9: The IMAGE framework region classification[183]

Table 3.1: Classification of IMAGE regions

Region	Nr	Countries
Canada	1	Canada (124)
USA	2	St. Pierre and Miquelon (666), United States (840)
Mexico	3	Mexico (484)
Central America	4	Anguilla (660), Aruba (533), Bahamas, The (44), Barbados (52), Belize (84), Bermuda (60), Cayman Islands (136), Costa Rica (188), Dominica (212), Dominican Republic (214), El Salvador (222), Grenada (308), Guadeloupe (312), Guatemala (320), Haiti (332), Honduras (340), Jamaica (388), Martinique (474), Montserrat (500), Netherlands Antilles (530), Nicaragua (558), Panama (591), Puerto Rico (630), St. Kitts and Nevis (659), St. Lucia (662), St. Vincent and the Grenadines (670), Trinidad and Tobago (780), Turks and Caicos Isl. (796), Virgin Isl. (Br.) (92), Virgin Islands (U.S.) (850)
Brazil	5	Brazil (76)
Rest of South America	6	Argentina (32), Bolivia (68), Chile (152), Colombia (170), Ecuador (218), Falklands Isl. (238), French Guyana (254), Guyana (328), Paraguay (600), Peru (604), Suriname (740), Uruguay (858), Venezuela, RB (862)

Region	Nr	Countries
Northern Africa	7	Algeria (12), Egypt, Arab Rep. (818), Libya (434), Morocco (504), Tunisia (788), Western Sahara (732)
Western Africa	8	Benin (204), Burkina Faso (854), Cameroon (120), Cape Verde (132), Central African Republic (140), Chad (148), Congo, Dem. Rep. (180), Congo, Rep. (178), Cote d'Ivoire (384), Equatorial Guinea (226), Gabon (266), Gambia, The (270), Ghana (288), Guinea (324), Guinea-Bissau (624), Liberia (430), Mali (466), Mauritania (478), Niger (562), Nigeria (566), Sao Tome and Principe (678), Senegal (686), Sierra Leone (694), St. Helena (654), Togo (768)
Eastern Africa	9	Burundi (108), Comoros (174), Djibouti (262), Eritrea (232), Ethiopia (231), Kenya (404), Madagascar (450), Mauritius (480), Reunion (638), Rwanda (646), Seychelles (690), Somalia (706), Sudan (736), Uganda (800)
South Africa	10	South Africa (710)
Western Europe	11	Andorra (20), Austria (40), Belgium (56), Denmark (208), Faeroe Islands (234), Finland (246), France (250), Germany (276), Gibraltar (292), Greece (300), Iceland (352), Ireland (372), Italy (380), Liechtenstein (438), Luxembourg (442), Malta (470), Monaco (492), Netherlands (528), Norway (578), Portugal (620), San Marino (674), Spain (724), Sweden (752), Switzerland (756), United Kingdom (826), Vatican City State (336)
Central Europe	12	Albania (8), Bosnia and Herzegovina (70), Bulgaria (100), Croatia (191), Cyprus (196), Czech Republic (203), Estonia (233), Hungary (348), Latvia (428), Lithuania (440), Macedonia, FYR (807), Poland (616), Romania (642), Serbia and Montenegro (891), Slovak Republic (703), Slovenia (705)
Turkey	13	Turkey (792)
Ukraine region	14	Belarus (112), Moldova (498), Ukraine (804)
Central Asia	15	Kazakhstan (398), Kyrgyz Republic (417), Tajikistan (762), Turkmenistan (795), Uzbekistan (860)
Russia region	16	Armenia (51), Azerbaijan (31), Georgia (268), Russian Federation (643)
Middle East	17	Bahrain (48), Iran, Islamic Rep. (364), Iraq (368), Israel (376), Jordan (400), Kuwait (414), Lebanon (422), Oman (512), Qatar (634), Saudi Arabia (682), Syrian Arab Republic (760), United Arab Emirates (784), Yemen, Rep. (887)
India	18	India (356)
Korea region	19	Korea, Dem. Rep. (408), Korea, Rep. (410)
China region	20	China (156), Hong Kong, China (344), Macao, China (446), Mongolia (496), Taiwan (158)
Southeastern Asia	21	Brunei (96), Cambodia (116), Lao PDR (418), Malaysia (458), Myanmar (104), Philippines (608), Singapore (702), Thailand (764), Vietnam (704)
Indonesia region	22	East Timor (626), Indonesia (360), Papua New Guinea (598)
Japan	23	Japan (392)
Oceania	24	American Samoa (16), Australia (36), Cook Isl. (184), Fiji (242), French Polynesia (258), Kiribati (296), Marshall Islands (584), Micronesia, Fed. Sts. (583), Nauru (520), New Caledonia (540), New Zealand (554), Niue (570), Northern Mariana Islands (580), Palau (585), Pitcairn (612), Samoa (882), Solomon Islands (90), Tokelau (772), Tonga (776), Tuvalu (798), Vanuatu (548), Wallis ans Futuna Island (876)
Rest of South Asia	25	Afghanistan (4), Bangladesh (50), Bhutan (64), Maldives (462), Nepal (524), Pakistan (586), Sri Lanka (144)
Rest of Southern Africa	26	Angola (24), Botswana (72), Lesotho (426), Malawi (454), Mozambique (508), Namibia (516), Swaziland (748), Tanzania (834), Zambia (894), Zimbabwe (716)

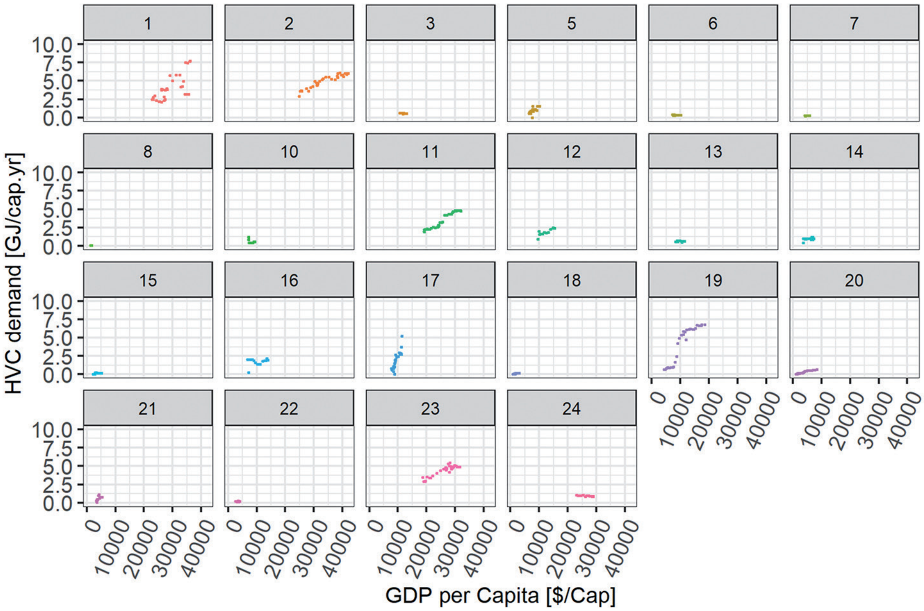

Figure 3.10: Historical data points for each IMAGE region, relating HVC demand/cap to GDP/cap (except for regions without steam crackers according to Oil & Gas Journal: 4, 9, 25, 26); for number legend and region names see Table 3.1 above

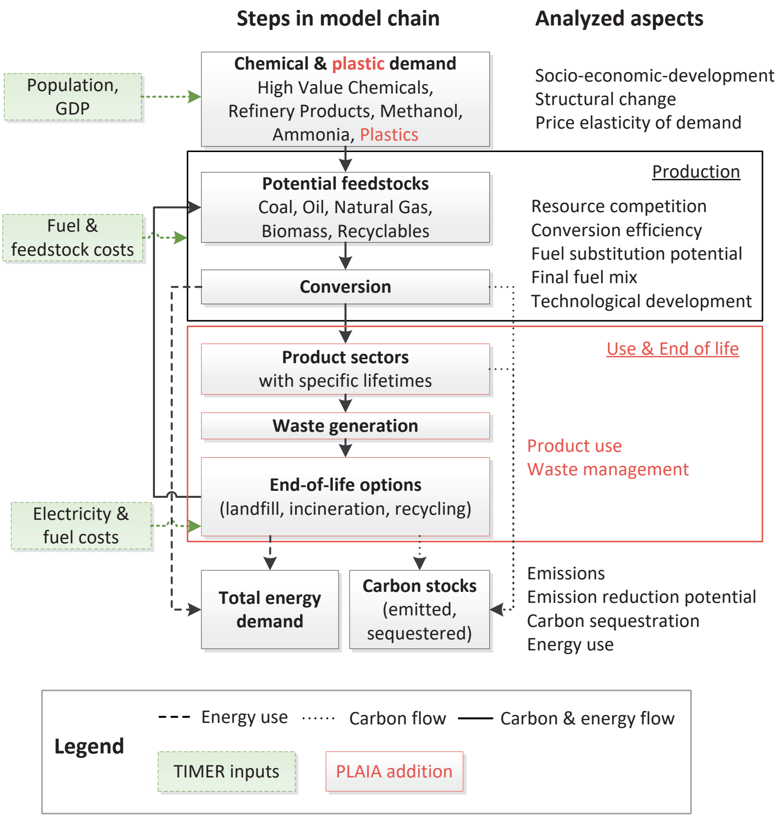

Figure 3.11: The structure of the updated NEDE model, including the additions of PLAIA. PLAIA is integrated into NEDE and models the downstream plastic production, products use, and end of life.

Table 3.2: Lower heating values of steam cracking feedstocks and products

LHV of Steam cracker feedstocks in GJ/t		LHV of Steam cracker products in GJ/t	
Ethane	47.8	Ethylene	47
Propane	46.4	Propylene	47
Butane	45.3	C4 stream	45
Naphtha	44.9	Aromatics	40
Gas Oil	42.8		
Other	45.5		

Source: Engineering ToolBox[187], assuming LHV of LPG for Other

Sources: Ren[184] for ethylene & propylene; Mozaffarian et al.[185] for Aromatics; LHV of C4 stream is a rounded assumption based on LHVs of butadiene, butene, and Isobutene[186]

Table 3.3: Steam cracker yields in GJ product / GJ feedstock

Products	Feedstocks					
	Ethane	Propane	Butane	Naptha	Gas oil	Other[a]
Ethylene	0.790	0.471	0.458	0.339	0.275	
Propylene	0.016	0.127	0.157	0.176	0.158	
Butadiene	0.022	0.047	0.044	0.050	0.053	
Other C4	0.006	0.012	0.033	0.062	0.042	
Aromatics	0	0	0	0.092	0.117	
Total HVC yield	0.833	0.656	0.691	0.719	0.644	0.708

Based on Levi & Cullen[5];

[a] Yield from feed "Other" is assumed to be the average of the other 5 yields. "Other" represents only a minor share in total feedstocks and refers to LPG, NGL, Hydrowax, Refinery gas and kerosene[139].

Table 3.4: Regression coefficients for modeling HVC demand

IMAGE region	China+	Eastern Europe	Western Europe	Japan	Korea+	Middle East	South East Asia	USA	Rest of World
Alpha	7.98	13	13	10.81	13	13	9.98	12.31	7.98
Beta	17288	28645.6	35834.8	23635.1	10419.7	13710.3	17177.6	31169.1	20055.9

Table 3.5: Average lower heating values (LHV) per plastic type

	LD,LDPE	HDPE	PP	PS	PVC	PET	PUR	PP&A
Average LHV in GJ/t	43,35	41,44	43,19	40,19	19,69	23,15	26,53	24,71 – 29,81[a]

[a] depending on polyester share;

Sources: [188–195]

Table 3.6: Energy use for plastic polymer production from monomers

Plastic polymers	HDPE	LDPE	LLDPE	PP	PET	PVC	PS	Weighted average [a]
Electricity in GJ/t polymer resin	1.56	3.43	1.27	1.27	2.26	4.93	0.59	2.13
Heat in GJ/t polymer resin	1.32	-0.25	0.69	0.84	8.54	9.03	1.29	2.84

[a] based on polymer market shares from Geyer et al.[6]

Energy use data: [140,141,146,147]

Table 3.7: Energy use & efficiency for transforming plastic resins into semi-finished plastic products

Plastic polymers	PE	PP	PET	PVC	Weighted average [a]
Energy use in GJ/t product output	12.77	7.21	7.72	5.73	9.6
Efficiency in t product output/ t resin input	1	0.91	0.94	0.97	0.96

[a] based on polymer market shares of Geyer et al.[6] and assuming PE values for HDPE, LDPE & LLDPE

Table 3.8: Mean use time of plastics by sector and their standard deviation (from Geyer et al.[6])

Market sector	Mean time of use in years	Standard deviation
Packaging [a]	0,5	0.1
Transportation	13	3
Building and Construction	35	7
Electrical/ Electronic	8	2
Consumer & Institutional Products	3	1
Industrial Machinery	20	3
Textiles	5	1.5
Other	5	1.5

[a] We changed the distribution of packaging in PLAIA as the model has a yearly resolution. In PLAIA, all packaging plastics become waste after one year.

Table 3.9: Waste collection rates for different national income levels

Income Type	**Collection rate** in %	**GNI/cap min** in 2005 USD	**GNI/cap max** in 2005 USD
High Income	96	12476	
Upper-middle income	82	4036	12475
lower-middle income	51	1026	4035
low income	39	0	1025

Source: Silpa Kaza et al[153]; translated to 2005 USD

Table 3.10: Data used for modelling the waste treatment options

Waste treatment option	Yield	Fixed cost factor [b]	Substitution rate	Electricity use	Heat use	Diesel use
	GJ plastics / GJ plastic waste	2005 USD/ GJ plastic waste	value of recycled to primary pl.	GJ/GJ plastic waste	GJ/GJ plastic waste	GJ/GJ plastic waste
Sorting	0.75 for MR[149] 0.85 for CR[35]	4[35,149]	-	0.0058[35,149,156–159]	-	-
Mechanical Recycling	0.75[149]	10[35,149,160,161]	0.81[31,156]	0.0527[6,35]	0.0064[6,35]	-
Chemical Recycling [a] (via pyrolysis)	0.315[184]	16[184]	1	-	0.59[184] (natural gas)	-
Waste to Energy	-	3.15[160]	-	-0.0926	-0.22 26	-
Landfilling	-	endogenous [162,166,168–170]	-	0.0007[165]	-	0.002[165]

[a] includes the production of naphtha from plastic waste & the transformation of naphtha to monomers. It excludes the polymerisation step, which is described elsewhere[196]

[b] excludes collection & transportation and excludes energy use (heat, electricity, diesel, which are added endogenously in the model)

For details on data choices and calculations, see separate methodology publication[196]


BIO

Chapter 4

Plastic futures and their CO_2 emissions

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ABSTRACT

Plastics show the strongest production growth of all bulk materials and are already responsible for 4.5% of global greenhouse gas (GHG) emissions. If no new policies are implemented, we project a doubling of global plastic demand by 2050 and more than a tripling by 2100, with an almost equivalent increase in CO_2 emissions. Here we analyse three alternative CO_2 emission mitigation pathways for the global plastics sector until 2100, covering the entire life-cycle from production to waste management. Our results show that through bio-based carbon sequestration in plastic products, a combination of biomass use and landfilling can achieve negative emissions in the long term; however, this involves continued reliance on primary feedstock. A circular economy approach without an additional bioeconomy push reduces resource consumption by 30% and achieves 10% greater emission reductions before 2050 while reducing the potential of negative emissions in the long term. A circular bioeconomy approach combining recycling with higher biomass use could ultimately turn the sector into a net carbon sink while phasing out landfilling and reducing resource consumption. Our work improves the representation of material flows and the circular economy in global energy & emission models and provides insight into the long-term dynamics in the plastics sector.

4.1 INTRODUCTION

Plastics have become an essential part of our economy. Their production increased from 2 Mt in 1950 to 380 Mt in 2015, making plastics the bulk material with the strongest production growth globally[4,6]. While plastics could offer environmental benefits such as reducing fuel consumption by making vehicles more light-weight[4,7], their rising consumption takes its toll on the environment. In 2015, the plastics sector was responsible for 4.5% of global GHG emissions[8]. Following current growth rates, plastic production and the corresponding GHG emissions could almost quadruple by 2050[9]. Furthermore, plastics contribute to particulate matter emissions[8] and growing pollution[10].

Using biomass as feedstock and circular economy (CE) measures such as recycling are two options that may significantly reduce both fossil feedstock use and the related GHG emissions of the plastics sector[4,9,15–17]. Together, they could contribute to a circular bioeconomy (CBE)[197] for plastics, potentially even achieving negative CO_2 emissions by sequestering biogenic carbon in plastic products for long-term use[18]. If these plastics are then kept in use via recycling or sequestered in landfills, they could theoretically be a medium or a long-term carbon sink. Renewable energy use in plastic production and waste management could further reduce the GHG emissions of the plastics sector[9]. It is impossible to fully understand the climate change mitigation potential and the trade-offs of these mitigation strategies without analysing the global, long-term trends in the plastics sector and the sector's interactions with other socio-economic and natural systems. However, none of the climate and socioeconomic models used for the Intergovernmental Panel on Climate Change (IPCC) reports has included a detailed representation of the plastics sector[41].

Here, we present the Plastics Integrated Assessment model (PLAIA)[196], which covers the entire life-cycle of plastics, from the upstream chemical production to the downstream production of plastic polymers, their transformation into plastic products, their use in different sectors, and their end of life. As part of the integrated assessment model IMAGE[136], PLAIA interacts with the energy and agricultural sectors and with the climate, water, and land systems. Using PLAIA, we compare different climate change mitigation pathways for the global plastics sector until 2100, based on feedstock substitution (e.g., biomass use), renewable energy use, recycling, and biogenic carbon sequestration in products and landfills. Regional results and key input variables are presented in the supporting information.

4.2 PLASTIC PRODUCTION, WASTE, AND STOCKS

Using socioeconomic projections from a *middle-of-the-road* development scenario (Shared Socio-economic Pathway-2, SSP2)[138], we project more than a doubling of 2020 plastic production by 2050 and more than a tripling by 2100 (Figure 4.1a). In this baseline scenario, waste generation increases accordingly and is dominated by plastics for packaging and other products with a short lifetime (Figure 4.1b).

Products with a long lifetime dominate the plastic stocks (see Plastics in use, Figure 4.1c); here, building & construction materials alone make up more than half of the plastics in use, despite only having a share of around 17% in annual production. Using plastic product lifetime distributions[6], we estimate the total plastic stocks in use in 2020 at almost 3.2 billion metric tonnes (Gt), and this could rise to around 7.7 Gt in 2050 and to almost 15 Gt in 2100.

4.3 SCENARIOS OF FEEDSTOCK USE AND EMISSIONS

The growth in plastic demand shown in Figure 4.1a implies a further increase in the GHG emissions of plastic production if there are no significant changes in feedstock and process energy use. The PLAIA model determines the use of coal, oil, natural gas, biomass, and plastic waste as feedstock or process energy based on the endogenously modelled economic competitiveness of these resources and the respective plastic production pathways[196]. Furthermore, the model includes the secondary energy carriers electricity and heat, mainly used in plastic polymerisation, transformation into products, and recycling. Hence, PLAIA analyses the impact of feedstock substitution (e.g., biomass for oil) and renewable process energy use on the CO_2 emissions of the plastics sector. Additionally, PLAIA assesses different waste management strategies for plastics, covering mechanical recycling and chemical recycling (via pyrolysis), landfilling, and waste-to-energy (electricity & heat), see Figure 4.2.

Estimated emissions of the global plastics sector in 2020 are 2.2 Gt CO_2, which represent around 7% of the global energy-related CO_2-emissions[198]. In Figure 4.3, we compare four scenarios for the future plastic industry. The baseline follows an SSP2 *middle-of-the-road* socio-economic path which results in a continued focus on fossil resources and only a small uptake of biomass as feedstock (see Figure 4.2 and Figure 4.3). In this scenario, coal use continues to increase until 2030, driven by China, which is currently investing in coal-based chemical technologies to reduce its dependency on oil and gas imports[199,200]. Compared to 2020, this scenario almost doubles its emissions up until 2050, reaching its peak in 2090 with 5.7 Gt CO_2, which would nearly equal the total net US GHG emissions of 2019[201]. In the baseline scenario, a key emission driver is the transition to waste-to-energy as dominating waste treatment technology (see Figure 4.2 and Figure 4.5 in the Appendix). This scenario has the highest emissions and final energy use (see Figure 4.4) due to the small share of recycling.

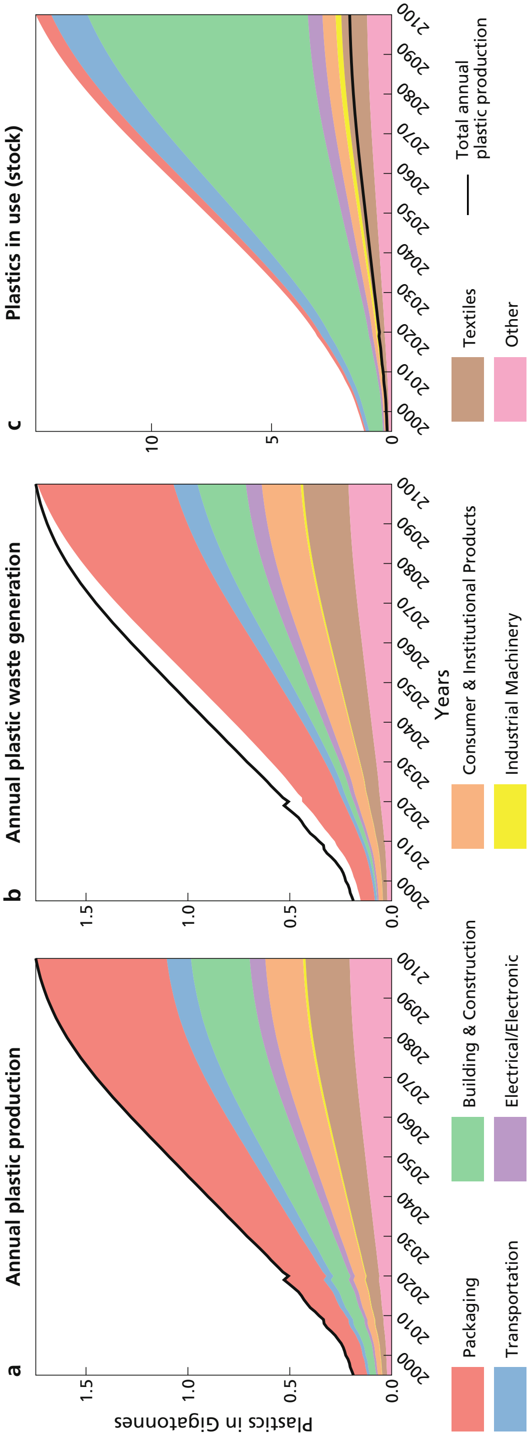

Figure 4.1: Projections of global plastic production, waste generation, and plastic stocks in use by sector

Notes: different scale for plastics in use

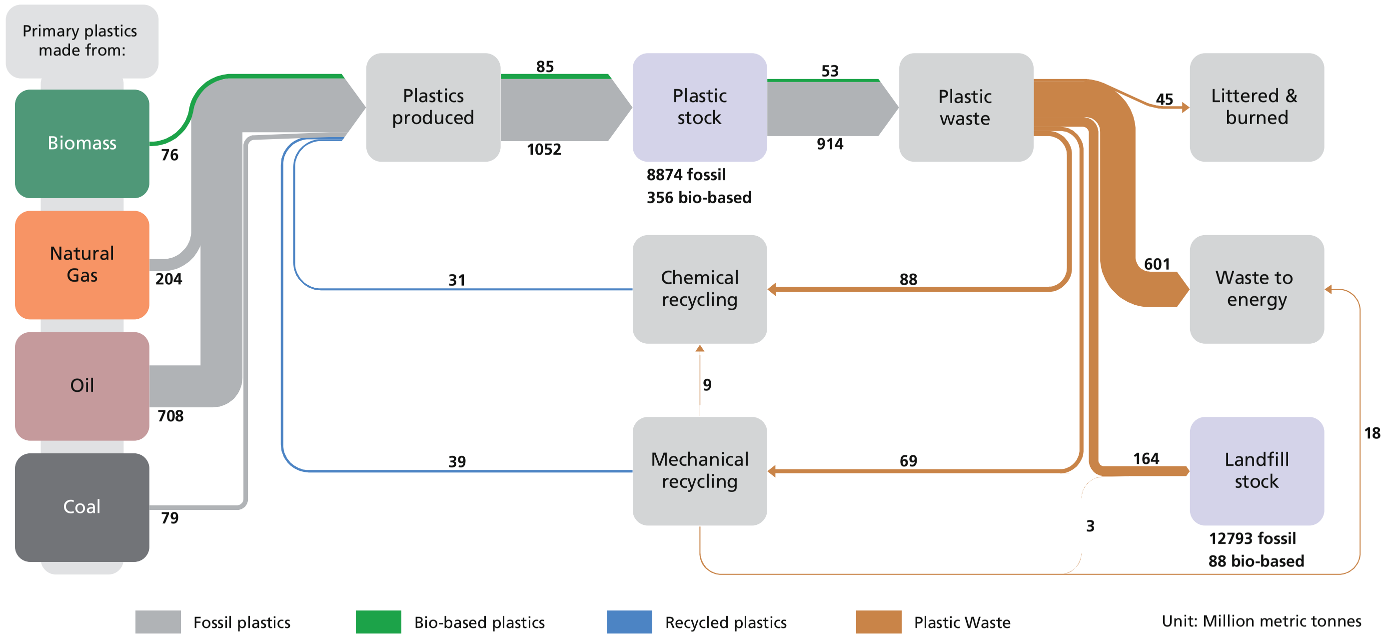

Figure 4.2: Global plastic flows for the SSP2 baseline scenario in the year 2050

Notes: Sankey diagrams for the other scenarios can be found in the Appendix (Figure 4.8, Figure 4.9, Figure 4.10)

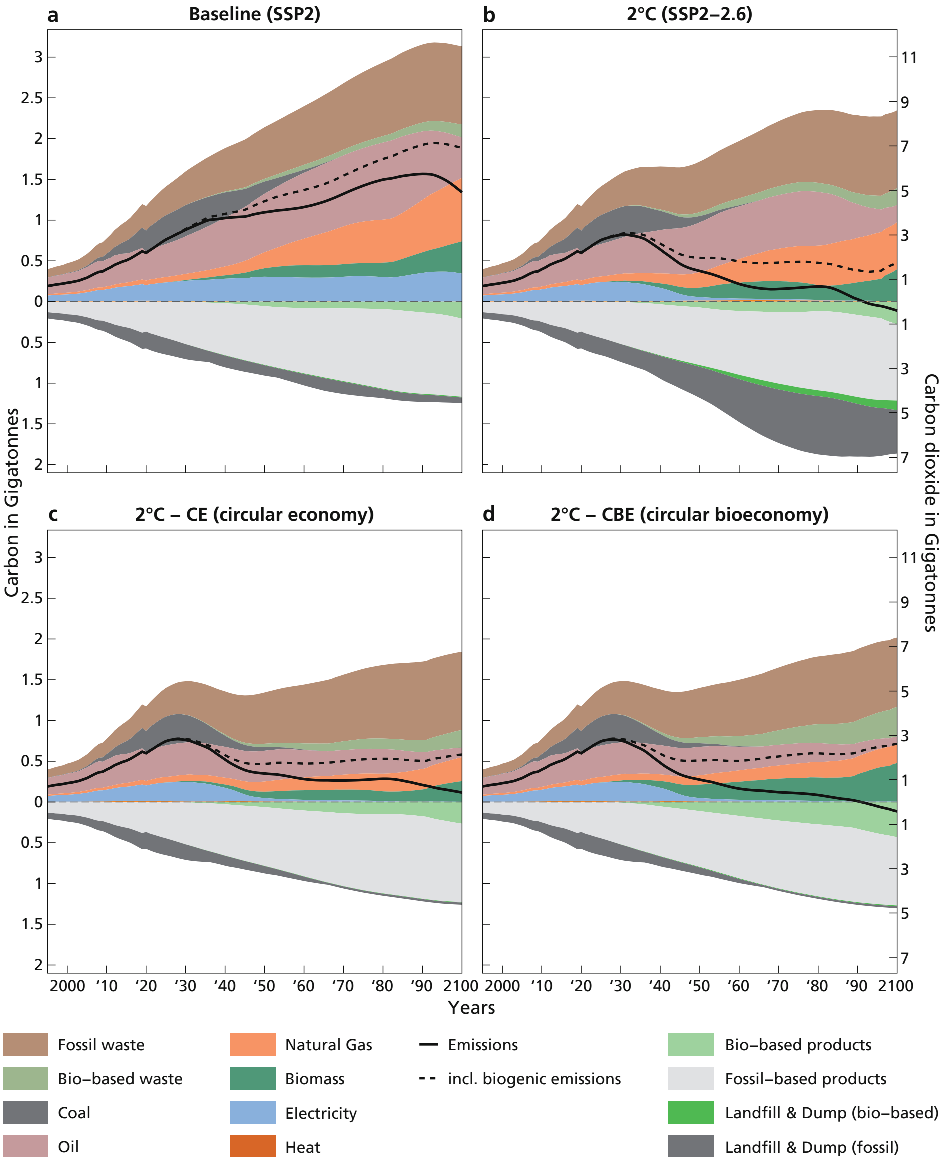

Figure 4.3: Carbon balance of the plastics sector over the entire life cycle

Notes: Above the horizontal axis each graph shows the carbon used for plastic production and the carbon in the generated plastic waste. This carbon in plastic waste is either used as a resource for plastic production via recycling, or ends up incinerated or in landfills and dumps. Below the horizontal axis the graphs present all carbon additions to plastic product stocks, landfills, and dumps. The black lines in each graph display the net emission balance, with and without biogenic emissions; biogenic emissions are assumed to be renewable and therefore have no net contribution to climate change[171].

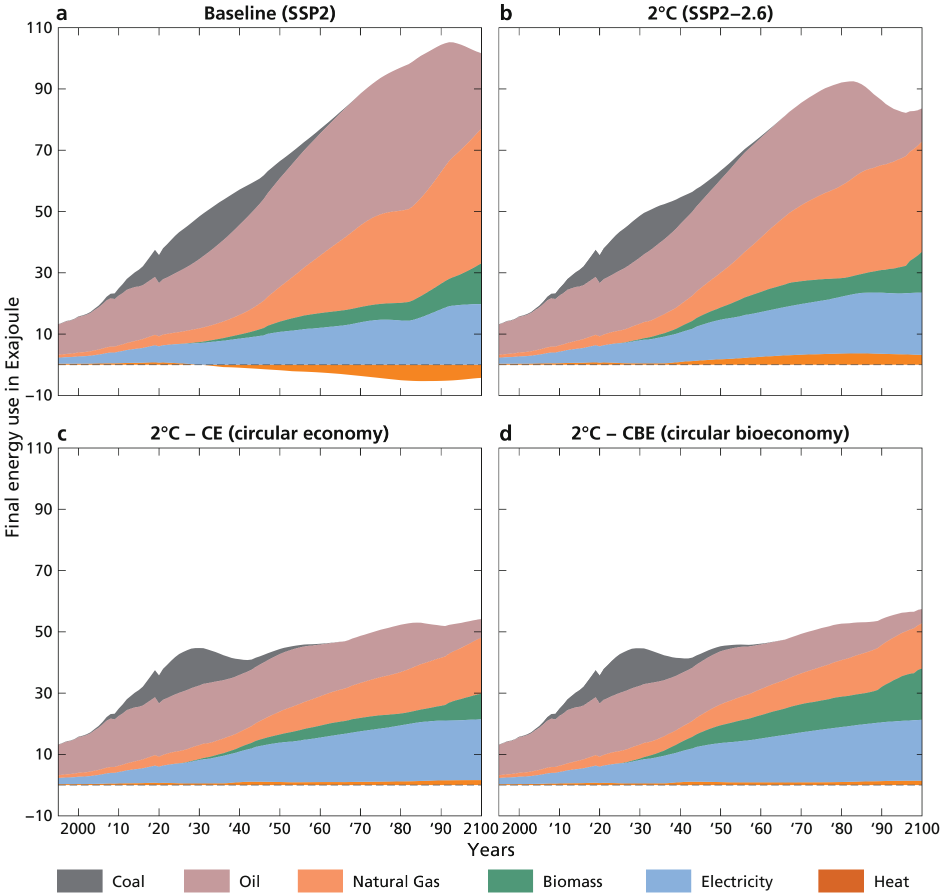

Figure 4.4: Global final energy use in the plastics sector over the entire life cycle

Notes: Heat use can become negative if more heat is produced via waste-to-energy than consumed by the sector

The three mitigation scenarios include an increasing price for GHG emissions, leading to energy and land-use system changes consistent with a 2°C global mean temperature change target by 2100 (SSP2-2.6, see chapter 4.7.6). The 2°C-CE and 2°C-CBE scenarios include additional circular economy strategies; for the CBE scenario, we also subsidise the use of biomass in the plastics sector (see chapter 4.7.6). All three mitigation scenarios reach their emission peak of 2.8-3 Gt CO_2 around 2030 (see Figure 4.3). A sensitivity analysis of the model (see chapter 4.7.7) showed that a lower oil price would significantly increase the GHG emissions of the plastics sector. This highlights the importance of regulating fossil fuel prices via carbon pricing to facilitate GHG emission mitigation. While the GHG emission results are also sensitive to the assumed chemical production efficiencies, conventional production pathways are largely operating close to their theoretical maximum[5]. Only the novel pathways (e.g., bio-based routes) could expect significant improvements, potentially further reducing the sector's GHG emissions.

In all three mitigation scenarios, the rising CO_2 price leads to the decarbonisation of electricity production, which has a significant impact on the emissions of the plastics sector. Moreover, the CO_2 price leads to a shift toward biomass and natural gas in upstream chemical production, thus phasing out coal and reducing the use of oil. Furthermore, it drastically reduces the use of waste-to-energy (see Figure 4.5, 4.8, 4.9 and 4.10 in the Appendix), whose emissions are penalised while being replaced by an increasingly greener heat and electricity mix. Nevertheless, when aiming at phasing out fossil fuels from the plastics sector, more is needed than only an increased CO_2 price. Unlike in the energy system, large parts of the carbon input in the plastics sector are not directly emitted but sequestered in products and thus not exposed to the CO_2 price.

The CO_2 price alone (2°C scenario) leads to a moderate increase in recycling while the use of primary feedstocks still dominates plastic production (see Figure 4.5, Figure 4.6 and 4.8 in the Appendix). Chemical recycling via pyrolysis actually reduces compared to the baseline scenario due to its high energy requirements and the corresponding penalties resulting from the CO_2 price (see Figure 4.5 in the Appendix). Furthermore, the CO_2 price leads to a drastic increase in cumulative landfilled plastics, ranging from an estimated 6.4 Gt landfilled plastic products in 2020 to 17.5 Gt in 2050 and almost 66 Gt in 2100. With high CO_2 prices, landfilling plastic waste becomes an attractive alternative, as it sequesters most plastics and their carbon for centuries[12] and is cheaper than other waste treatment technologies[153].

4.4 CARBON STORAGE AND NEGATIVE EMISSIONS

Since circa 75% of the weight of conventional plastics is made up of carbon[171], their stocks (in use and in landfills) form a type of carbon storage. Therefore, by using renewable biomass as feedstock plastics may potentially achieve negative emissions. We project that between 2020 and 2100 about 100 Gt of plastics will be cumulatively produced (see Figure 4.1a). If all of these were non-biodegradable biobased plastics, then 75 Gt of biogenic carbon could hypothetically be sequestered, equal to 275 Gt of negative CO_2 emissions (almost 9 times the current global annual energy-related CO_2 emissions[198]). This is also a significant amount compared to the total Bioenergy Carbon Capture and Storage (BECCS) potential reported by the IPCC for scenarios meeting the 1.5˚C temperature target (a maximum of 1191 Gt biogenic CO_2 stored cumulatively by 2100)[202]. However, the long-term sequestration potential of this carbon in plastics depends on the product lifetimes and the waste management strategies.

As a consequence of the negative emissions achieved by the sequestration of bio-based plastics in products and landfills, the 2°C scenario could significantly reduce the plastics sector's CO_2

emissions, even turning the sector into a carbon sink by the end of the century. However, due to its focus on primary plastic production and landfilling, the 2°C scenario maintains a high input of energy and materials (see Figure 4.4). Moreover, it could exacerbate other negative environmental impacts caused by the extraction and production of these resources (e.g., land-use change, biodiversity loss, nitrogen emissions from biomass production), chemical and plastic production (e.g., particular matter emissions[8]), and landfilling (e.g., increased land use and microplastics in leachates[203]).

4.5 BENEFITS OF CIRCULAR STRATEGIES

Only with a circular economy does the model show significantly reduced final energy input into the plastics sector (see Figure 4.4). By phasing out landfilling and promoting recycling pathways (see chapter 4.7.6), the circular economy scenario (2°C-CE) reaches a recycling rate of above 70% by 2050 (see Figure 4.5 and 4.9 in the Appendix), further increasing until 2100. In 2050, this results in a 60% market share of recycled plastics in yearly plastic production (see Figure 4.6 in the Appendix), leading to about 30% lower final energy use by 2050 compared to the baseline and the 2°C scenario. Not all plastic types and products can be mechanically recycled, and the quality of plastics declines with use and mechanical recycling[204,205]. Therefore, complementary chemical recycling via pyrolysis plays a growing role in the circular economy scenario (see Figure 4.5 in the Appendix), even though also pyrolysis is not suitable for all types of plastic waste[196,205].

However, it is impossible to achieve full circularity for plastics as the available waste feedstock cannot keep up with the assumed growing demand for plastics (see Figure 4.1b), even when ignoring processing losses in recycling. In our 2°C-CE scenario, the maximum market share that recyclates achieve is around 80% by the end of the century (see Figure 4.6 in the Appendix). Full circularity of the sector could only be achieved by stabilising or reducing final demand.

In the first half of the century, the CE scenario has cumulatively circa 10% lower CO_2 emissions than the 2°C scenario (see Figure 4.7 in the Appendix). However, reducing the CO_2 emissions of plastic production over the decades leads to a lower marginal GHG benefit of recycling. Eventually, the 2°C scenario has lower net emissions in the second half of the century, as it benefits from a growing amount of bio-based carbon in product and landfill stocks.

By combining circular economy measures and increasing biomass use, the CBE strategy (2°C-CBE) achieves the greatest cumulative emission reductions of all analysed scenarios while phasing out landfilling (see Figure 4.5 in the Appendix) and reducing the final energy demand of the plastics sector (see Figure 4.4). However, the high biomass use in this scenario leads to higher final energy consumption than in the 2°C-CE scenario (see Figure 4.4). The projected biomass use in the plastics sector ranges from 2.9 Exajoule in a CE scenario to 5.9 Exajoule in the CBE scenario in 2050, which would be equivalent to about 13% of the total current global bioenergy use[206]. This increases to 8.5 (CE) and 18.7 Exajoule (CBE) until 2100. Ensuring high sustainability standards in biomass production is key to this strategy to avoid negative impacts of biomass production (e.g., land and water use and nitrogen emissions).

4.6 TOWARDS A SUSTAINABLE PLASTICS SECTOR

The analysis presented here is a first step towards a better consideration of plastics and the circular economy in global energy and emission models. Clearly, the PLAIA model can still be improved in terms of technology representation (see chapter 4.7.7 and chapter 3). Also, the trade-offs with other environmental impacts could be analysed for a more integrated assessment of strategies toward a sustainable plastics sector. The development of regional and more technology-specific plastic models would allow for a better representation of the diversity of local challenges and technologies. In this article, we explored potential long-term dynamics of the global plastic sector and revealed the benefits and trade-offs of different climate change mitigation strategies. We showed that a uniform CO_2 price would decarbonise the electricity and heat supply of the plastic sector. This would also drastically reduce waste incineration unless the application of carbon capture to waste-to-energy becomes an economical option[207].

However, a CO_2 price alone is unlikely to lead to a net-zero emission plastics sector by 2050 nor would it be sufficient to achieve a circular economy for plastics. Still, additional policy measures are necessary to speed up biomass deployment in the plastic sector, such as subsidies for biomass use. In addition, carbon capture (and utilisation) during plastic production should be considered, to further reduce the sector's GHG impact[9,29].

We showed that a circular economy could significantly reduce the final energy demand of the plastics sector and achieve substantial CO_2-emission reductions until 2050. Achieving the high recycling rates of the 2°C-CE and 2°C-CBE scenarios requires a paradigm shift that not only improves the collection and sorting of plastic waste (e.g., closed-loop recycling via deposit systems) but also phases out landfilling and includes fundamental changes in product design[204] (see also chapter 4.7.6).

Furthermore, mechanical recycling needs to be complemented by chemical recycling, to improve the quality of recyclates and thus increase the number of recycling trips[208]. Besides polymer-specific CR technologies, pyrolysis is also an important technology as it accepts a wider range of mixed polymers that would otherwise not be recycled. Nevertheless, a fully circular plastic sector will be impossible as long as plastic demand keeps growing. Accordingly, future work and policy measures should look into potential behavioural and societal changes that could reduce the fast-growing demand for plastics.

Moreover, we showed that focusing on CE targets alone might lead to trade-offs with GHG emission mitigation, as a CE could reduce the potential for negative emissions in the long term. Hence, developing a circular bioeconomy strategy presents a synergy between climate and circular economy targets and could turn the plastics sector eventually into a net carbon sink while reducing the need for feedstocks.

4.7 METHODS

This section provides an overview of the methodology behind the Plastics Integrated Assessment model (PLAIA) used for this article. We describe the model in much more detail in a separate publication[196].

4.7.1 The model framework

PLAIA is embedded in the integrated assessment model IMAGE[209]. IMAGE is an ecological-environmental model framework that explores the long-term dynamics (until 2100) between society, the climate system, and the biosphere. It can analyse the impacts of socio-economic activities on issues like climate change, land use, and biodiversity for 26 world regions[136,137]. Figure 3.2 in Chapter 3 provides a graphical overview of the model framework. TIMER is a recursive dynamic simulation model of the energy system and part of the IMAGE framework[209]. TIMER projects the supply and demand of energy carriers and their associated emissions[210]. It also includes biomass as a resource, whose supply is linked to agricultural production and land-use dynamics[22,115]. TIMER includes the Non-energy Demand and Emission model (NEDE), which was developed to assess trends in primary feedstock use for the chemical industry and explore possible climate change mitigation strategies in the sector[45].

With PLAIA, we added a detailed representation of the plastics sector to NEDE. Figure 3.4 in Chapter 3 shows the structure of PLAIA. The model follows the plastics sector's material, energy, and emission flows for 26 world regions until 2100 from the cradle to the grave. It differentiates between eight plastic sectors (see Figure 4.1), six types of resources (oil, coal, natural gas, biomass, fossil- and bio-based plastic waste), and fossil and biogenic emissions. PLAIA relies on inputs from the TIMER model, which provides the availability and costs of resources and the carbon prices necessary to reach a given climate target.

4.7.2 Modelling plastic demand

Plastics are produced from intermediate chemicals like ethylene, propylene, aromatics, methanol, and C4 streams (e.g., Butadiene, Isobutene), which are sourced from steam crackers, refineries, and methanol producers[5]. In the absence of country-specific plastic demand data, we defined the demand for plastics as a share of the demand for these upstream chemical products, using material flow analysis data of the chemical sector[5]. The demand for the upstream chemical products was already defined in the NEDE model[45]: it is based on historical, country-specific production capacity data[139,145,211] in relation to GDP/cap development, assuming a utilisation rate of 90%[45]. We improved the representation of steam cracker outputs and refinery products to represent the full range of chemical intermediates used in plastic production, using average steam cracker yields and material flow analysis data[5,196]. The future plastic and chemical demand is driven by projections on GDP and population development based on the second shared socioeconomic pathway (SSP2)[138].

4.7.3 Plastic production

Different technology pathways can meet the product demand, using coal, oil, natural gas, or biomass, based on their endogenously modelled economic competitiveness[196]. The final energy demand is calculated based on the conversion efficiencies of the technology pathways and their respective market shares[45]. The upstream production (resources to feedstocks, feedstocks to intermediates, see Figure 3.4 in Chapter 3) are modelled as described in the initial NEDE model version[45].

To integrate plastics into the model, we added the energy use of downstream production processes, namely plastic polymerisation to granulates and their transformation into plastic products. We used energy use data from Life Cycle Assessments[140,141,146,147,151] and polymer market share data[6] to create a weighted average energy use for plastic polymerisation and transformation, assuming constant shares of plastic types. The production mix of this energy use and its costs are endogenously modelled for each world region in TIMER[136]. We take the costs of the process heat and electricity use in polymerisation and transformation as a proxy for the total costs of those two processes.

4.7.4 Waste treatment

PLAIA calculates the yearly plastic waste generation and the plastic stocks in use based on the lifetime of plastic products per sector. These sector-specific lifetimes are defined via lognormal probability distributions, using data compiled by Geyer et al.[6]. Reliable current and estimated future region-specific plastic waste collection rates are not available in the literature. Therefore, we based the collection rate on the economic and population development in a region, using general waste collection data from the World Bank[153]. The remaining, uncollected plastic waste is assumed to be burnt in the open air (30%) or dumped in the environment (70%), based on World Bank data on those informal waste disposal methods[153].

In the model, the collected plastic waste can either be directed to mechanical recycling, chemical recycling (via pyrolysis), incineration with energy recovery, or landfilling. The collected plastic waste is allocated to the different plastic waste treatment options (WTO) based on (1) the WTO's relative costs, (2) policy interventions (e.g., CO_2 price, bans), and (3) technological or economic constraints. The CO_2 price is applied to all fossil emissions in waste treatment, including process emissions (electricity, heat, and diesel use) and incineration. For waste-to-energy, we subtract the fossil carbon content of displaced heat and electricity (which varies between the scenarios, regions, and over time) from the fossil carbon emissions of the incinerated plastic waste.

For defining the market shares of each WTO, we use a multinomial logit function as shown below, with C being the cost of each WTO and Region (R) and λ being the logit parameter which defines the elasticity between relative prices.

This allocates market shares based on relative prices, with the cheapest WTO option getting the largest market share while more expensive options still get a share, albeit a smaller one. This method avoids “penny-switching”, where entire system configurations shift the moment the cheapest technology changes. This method aims to simulate the heterogeneity in waste management, where decisions are not made on cost considerations alone. By smoothening the results over several years, we also account for technology lock-ins and the fact that waste management practices do not completely change from one year to the other.

The costs of the WTOs consist of a fixed cost factor (i.e., capital costs and non-energy related operational costs), endogenously modelled variable costs (for heat, electricity, and diesel use), and a CO_2 price. These costs are reduced by the endogenously modelled benefits of replacing primary plastics (for mechanical & chemical recycling) or heat and electricity (for waste to energy). The modelled energy use and costs exclude the collection and transportation of plastic as worldwide data for region-specific collection methods, and transport distances are difficult to find. Furthermore, studies showed that these steps have a comparably small impact on the overall energy use and GHG emissions of the recycling process[149,154]. Next to the sorting and recycling efficiency, we also apply a substitution factor to mechanically recycled plastics. This substitution factor represents the quality losses of recycled material compared to virgin plastics and leads to a lower price recycled plastics receive on the market[31,156].

Table 3.10 in Chapter 3 shows the chosen data for process efficiencies, the substitution factor, the energy use, and the costs of the different WTOs. Chemical recycling and waste-to-energy are only made available to regions that reached a GDP/cap higher than 10,000 USD_{2005}. Next to fixed and variable costs, landfilling costs also include a dynamic factor that changes the regional costs of landfilling based on GDP per capita, population size, and usable area[196].

4.7.5 Carbon accounting

The model accounts for the carbon in- and outflows from the primary resource production to the production of chemicals and plastics and their end-of-life. According to standard IPCC guidelines, we only account for fossil carbon in emission accounting and treat the biogenic carbon as climate neutral. Nevertheless, we still specify biogenic carbon emissions (see Figure 4.3). Additionally, we include land-use change emissions and agricultural process emissions of biomass production. We assume that all carbon in fossil resources used for process energy is directly emitted as CO_2 unless carbon capture & storage technology is applied. The release of carbon embedded in plastic products depends on the product lifetime and their fate at the end of life (recycling, incineration, landfilling). Plastic-embedded carbon ending up in landfills and dumps is assumed to stay sequestered for the analysed period (up to 2100). The sensitivity analysis showed that this simplification has a limited impact on cumulative CO_2 emissions of the plastics sector (see section 4.7.7). Research indicates that the chemical degradation rate of plastics buried in landfills is very low and that most plastic types stay sequestered for hundreds or thousands of years when buried[12]. Even in other environments, the impact of plastic degradation on GHG emissions seems to be limited[172]. To achieve an inherently consistent carbon balance in PLAIA, we calculate the carbon content of plastics as a weighted average of the carbon contents of the feedstocks used for plastics production.

4.7.6 Scenarios

The scenarios of this study build upon the IMAGE implementation of the second shared socioeconomic pathway (SSP2), which describes middle-of-the-road long-term developments in demographic, economic, technological, and behavioural characteristics[138,174,210,212]. Building upon this SSP2 baseline, the 2˚C-scenario also includes a globally homogenous price on Greenhouse-gas emissions which leads to energy system changes consistent with a 2°C global mean temperature change target by 2100 [212].

The circular economy scenario and the circular bioeconomy scenario are sensitivities of the 2°C-scenario. For both of them, we assume a global paradigm shift toward a circular economy, involving all relevant actors. Next to gradually phasing out landfilling, we assume that policies incentivising circular product design, standardised plastic types, and avoiding additives, opaque colours, and multi-material plastic products will increase sorting and recycling efficiencies [204]. Along with technological innovations, the introduction of material markers, streamlined collection & sorting systems, and fostering deposit systems will further increase the sorting and recycling yields[204]. We assume that these measures will increase sorting and recycling yields of mechanical and chemical recycling by 20 % (linear increase between 2020 and 2030) and reduce their costs by 30% (linear decrease between 2020 and 2030). Moreover, these changes will also contribute to the improved quality of recycled plastics, leading to a higher substitution factor (from 0.81 to 0.9 between 2020 and 2030). Additionally, the circular bioeconomy scenario includes 30% subsidies on biomass use for the chemicals and plastics sector (implemented linearly between 2020 and 2030).

4.7.7 Discussion of limitations and sensitivities

Our results have to be used with caution as they only explore potential CO_2 emission reduction pathways and do not necessarily represent a realistic forecast of the developments in the plastic sector. Hence, our results only allow for generic conclusions on the relative performance of the analysed mitigation strategies.

We assume a largely uniform global carbon pricing to identify the optimal CO_2 mitigation pathway. The choice of a uniform carbon price does not intend to present a realistic forecast of climate policy and its impact on the plastics sector. Instead, it acts as a tool, together with other normative choices, to explore the impact of emission mitigation options in this sector. In reality, carbon pricing is currently fragmented across global regions. This fragmentation is likely to continue - at least in the short term – given the lack of global agreements. Furthermore, the application of homogenous carbon pricing across all emitting sectors, the inclusion of negative emissions in pricing mechanisms, and the treatment of biomass and associated land-use change emissions pose significant difficulties in reality.

Similarly, technology data for plastic production and waste management are mostly homogenous throughout regions in the model, apart from variable energy and land costs, ignoring the geographical differences in reality. This and other key model limitations regarding data, technological learning, and carbon accounting are discussed further in chapter 3.

Additionally, Figure 4.11 in the Appendix shows a sensitivity analysis of the cumulative net CO_2 emissions (2020-2100) for selected key assumptions, compared to the 2-degree scenario (SSP2-2.6). The analysed variables affecting the upstream chemicals and plastics production have the largest impact. While a 25% change in biomass yields leads to variations of 4-8% compared to the baseline, the oil price has a significant impact on the results, with variations up to 32%. This highlights the necessity to regulate fossil fuel prices via carbon pricing to facilitate GHG emission mitigation. The efficiencies in the chemical production, i.e., for transforming feedstocks such as naphtha and ethanol into intermediates such as ethylene and aromatics, show the highest sensitivity, ranging from 39 to 64%. However, the 25% efficiency alteration assessed here is far beyond the potential efficiency changes of the mature, conventional chemical production pathways, which are close to their theoretical maximum already[5]. Only for novel chemical production pathways (e.g. the bio-based routes) could we still expect significant changes in efficiencies.

All sensitivities of the end-of-life assumptions are below 13%, with the waste collection rate being the most impactful. This highlights the importance of increasing global waste collection rates, not only to fight plastic pollution but also to reduce GHG emissions. Moreover, the mechanical recycling rate has a significant impact of up to 11%. As an upcoming waste treatment technology, our assumptions for chemical recycling via pyrolysis have high uncertainty. For the sensitivity analysis, we changed the costs, efficiencies, and energy use of the pyrolysis process by 25%. This had only a limited impact (up to 5%). However, combining all pyrolysis sensitivities could lead to a higher impact than the sum of its parts, as it could significantly increase the market share of pyrolysis. A 25% change in the mean product lifetime only had a small impact for the analysed period.

For our main results, we assumed that carbon sequestered in plastics in landfills and dumps will stay sequestered for the analysed period up to 2100. For the sensitivity analysis, we assumed the highest GHG emission rate of degrading plastics reported by Royers et al.[172], which was for aged LDPE under direct solar radiation. Even assuming this value for all plastics, the impact of degrading plastics in landfills and dumps on the cumulative net plastic sector emissions (2020 -2100) would be below 4%. In reality, tests showed that most plastic types have a half-life of hundreds or thousands of years when buried[12].

Furthermore, we analysed the impact the shared socioeconomic pathways (SSPs) have on the net CO_2 emissions. While the relative performance of the SSPs changes throughout the years, cumulatively (2020-2100), SSP1 and SSP3 show 4.3% and 3.1% higher net CO_2 emissions compared to the SSP2 baseline (Figure 4.12 in the Appendix). This can partly be explained by the differences in GDP and population development affecting plastic demand (Figure 4.13 in the Appendix). However, most of the impact is linked to feedstock use in plastic production. While SSP1 has less emission-intensive electricity production, it has more restrictions regarding the available land for biomass production, thus reducing the bio-based carbon sequestration potential. SSP3 uses cumulatively more biomass for plastic production than SSP2, but its growing coal use eventually leads to higher emissions than the SSP2 baseline.

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Data availability

Model documentation and data of the IMAGE model can be found online[137]. A detailed description of the PLAIA model and its data sources is published[196].

Code availability

The code of the PLAIA model is published[262].

Author contributions

M.J. and M.L. developed the idea. P.S. and V.D. developed the method. P.S. calculated and compiled the results and wrote the article, with inputs from all authors. All authors discussed the results and contributed to the manuscript.

4.8 APPENDIX: SUPPLEMENTARY FIGURES

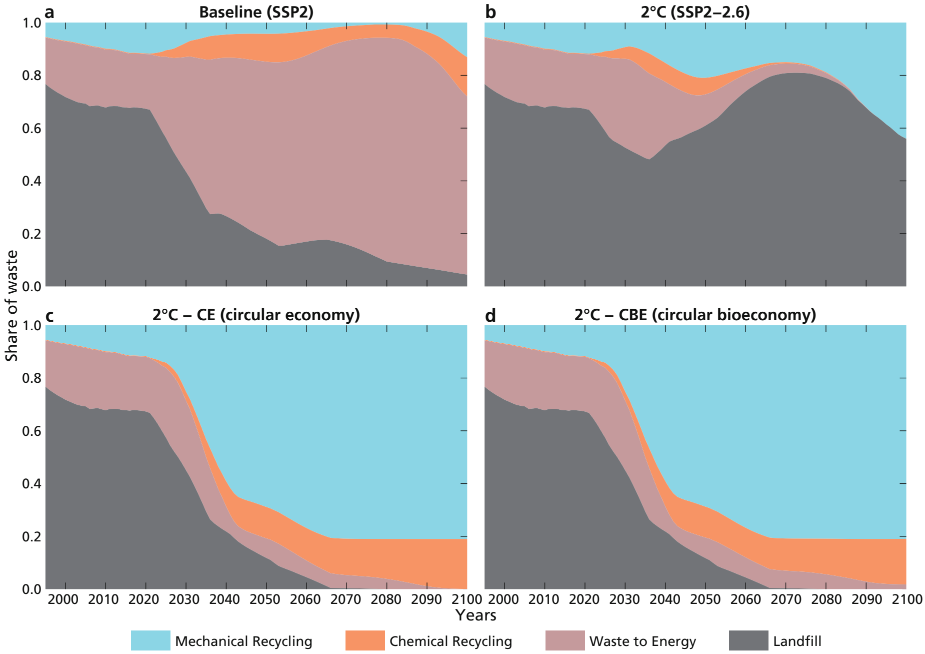

Figure 4.5: The global shares of waste treatment technologies.

Notes: This graph represents the fate of collected plastic waste; Sorting and recycling losses of mechanical recycling were allocated to the remaining waste treatment options; The chemical recycling share represents the plastic waste sent to pyrolysis.

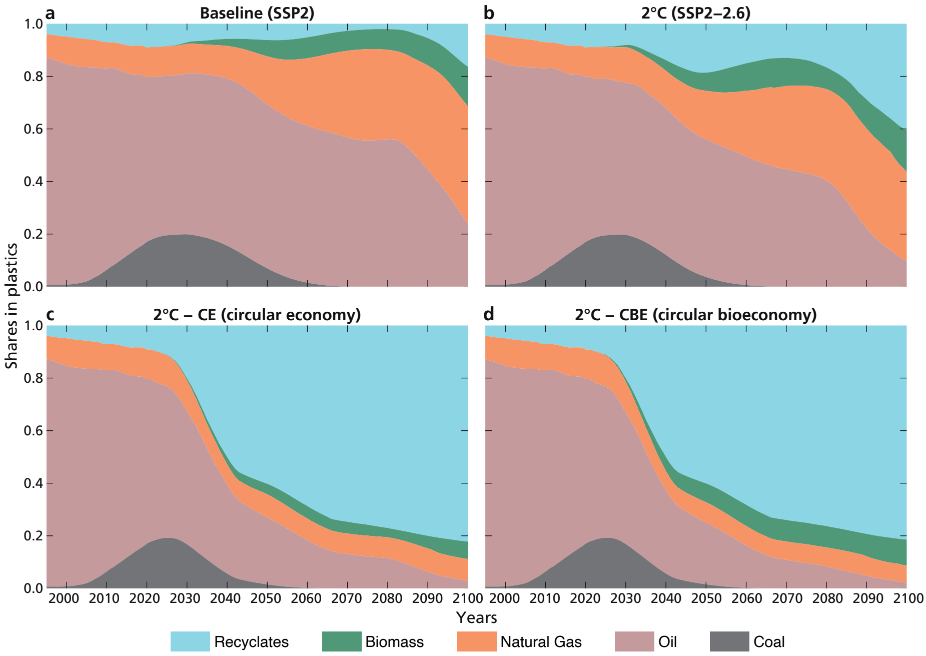

Figure 4.6: Feedstock shares in global plastic production.

Notes: This figure shows the final shares of resources in the annually produced plastic products (not the primary resource use for plastics).

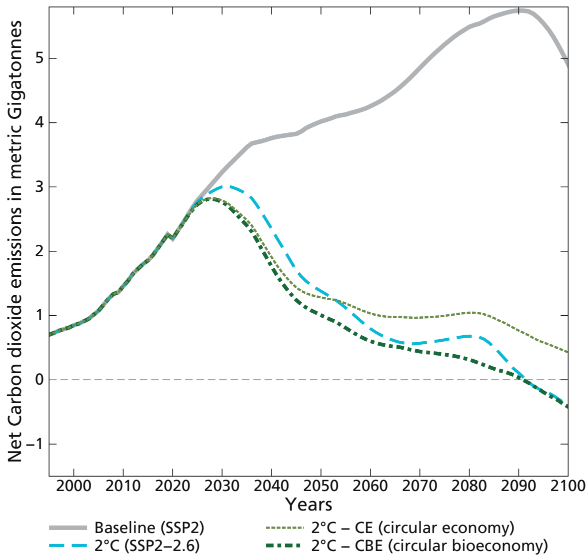

Figure 4.7: Comparing the global CO_2-emissions of the four scenarios

Notes: These emission lines are the same as the solid net emission lines of Figure 4.3; biogenic emissions are assumed to be renewable and therefore have no net contribution to climate change.

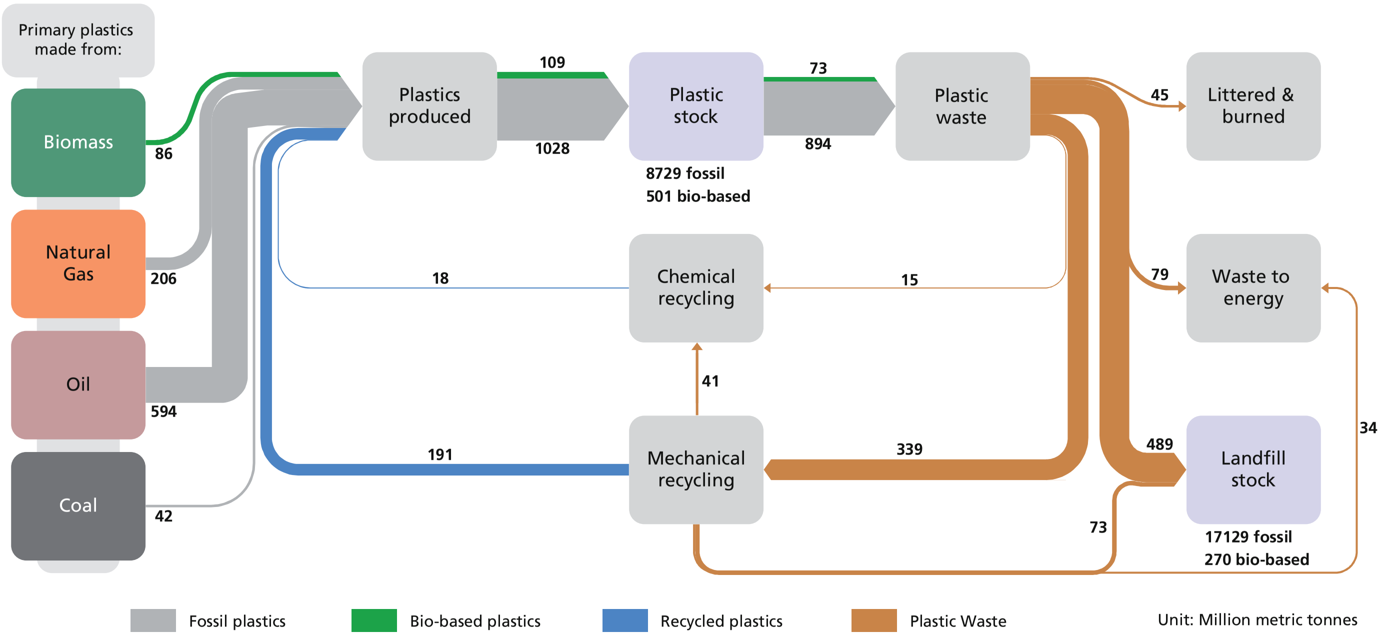

Figure 4.8: Global plastic flows for the 2°C scenario (SSP2-2.6) in the year 2050

Notes: Processing losses in sorting and mechanical recycling are allocated to other waste treatment options. Chemical recycling refers to pyrolysis and its processing losses are assumed to be emitted.

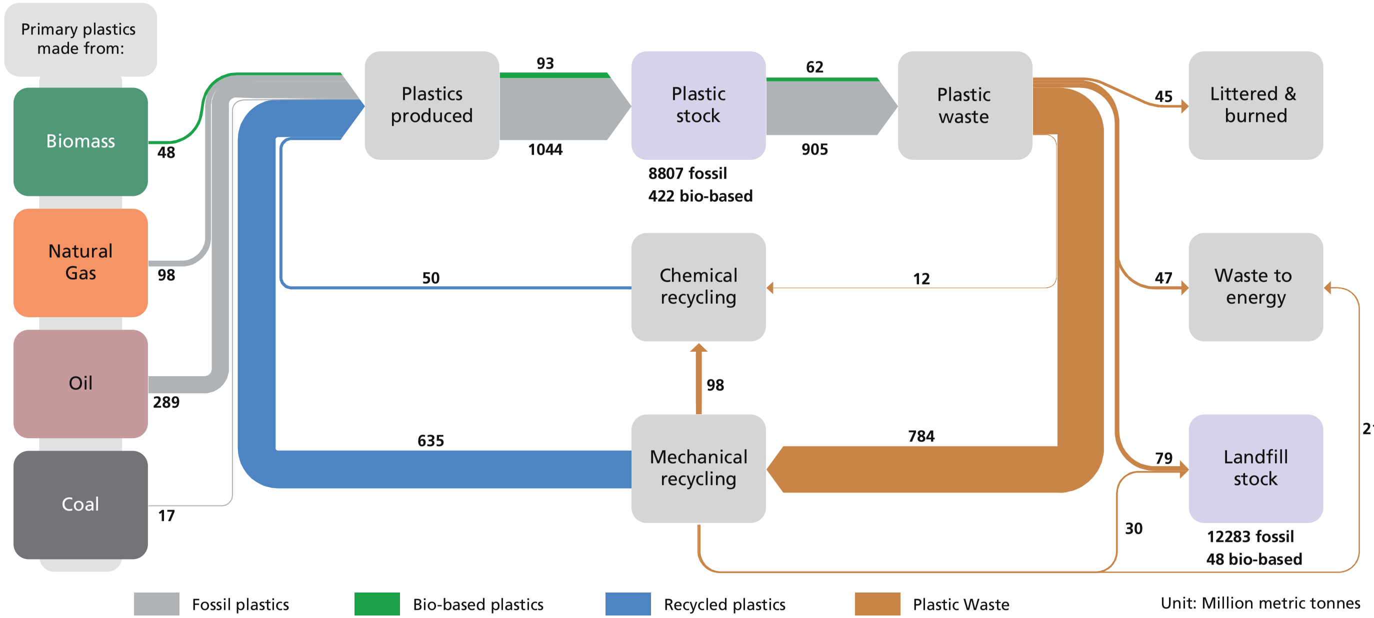

Figure 4.9: Global plastic flows for the 2°C-CE scenario in the year 2050

Notes: Processing losses in sorting and mechanical recycling are allocated to other waste treatment options. Chemical recycling refers to pyrolysis and its processing losses are assumed to be emitted.

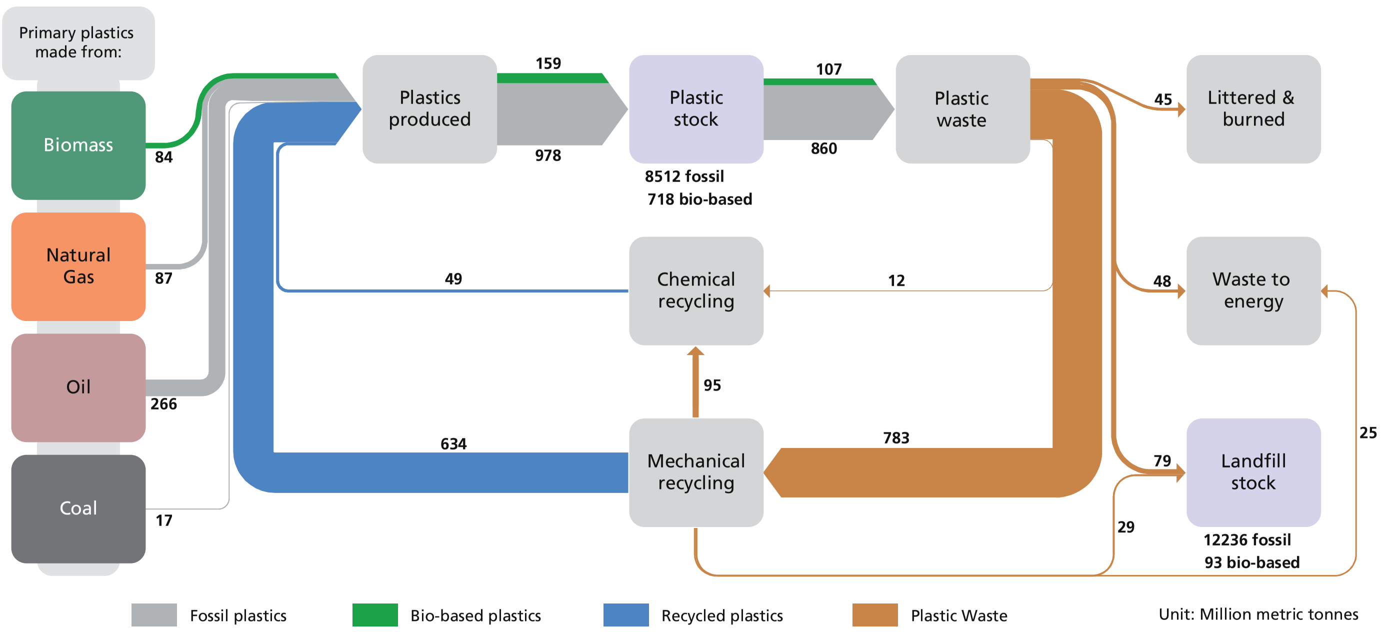

Figure 4.10: Global plastic flows for the 2°C-CBE scenario in the year 2050

Notes: Processing losses in sorting and mechanical recycling are allocated to other waste treatment options. Chemical recycling refers to pyrolysis and its processing losses are assumed to be emitted.

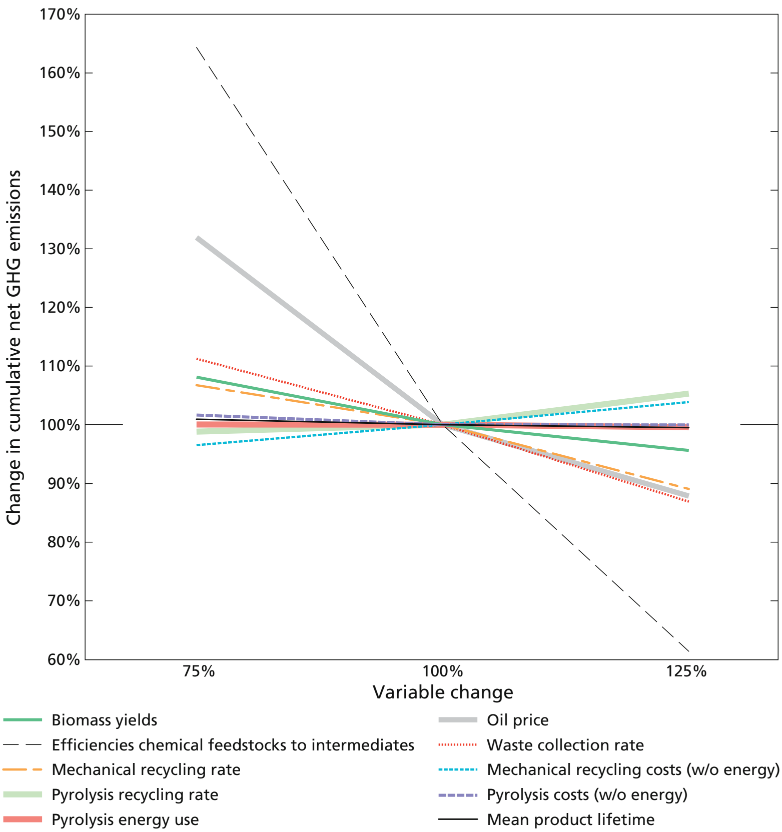

Figure 4.11: Sensitivity analysis of the global cumulative net CO_2-Emissions (2020-2100)

Notes: This figure shows how changes in model variables affect the cumulative net CO_2-Emissions (2020-2100) of the global plastic sector over its entire life cycle.

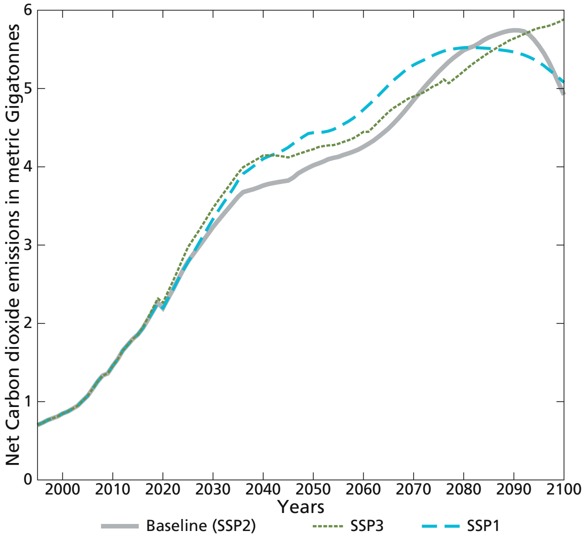

Figure 4.12: Global net CO_2 emissions of the shared socioeconomic pathways (SSP)

Notes: biogenic emissions are assumed to be renewable and therefore have no net contribution to climate change; The narratives behind the shared socioeconomic pathways are described in O'Neill et al. (2017)[138].

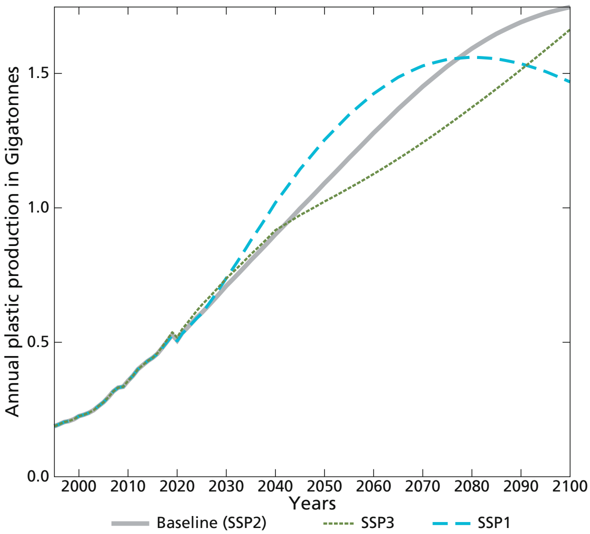

Figure 4.13: Global annual plastic production of the shared socioeconomic pathways (SSP)

Notes: The narratives behind the shared socioeconomic pathways are described in O'Neill et al. (2017)[138].


BIO

Chapter 5

Message in a bottle - The global warming potential and the material utility of PET and bio-based PEF bottles over multiple recycling trips

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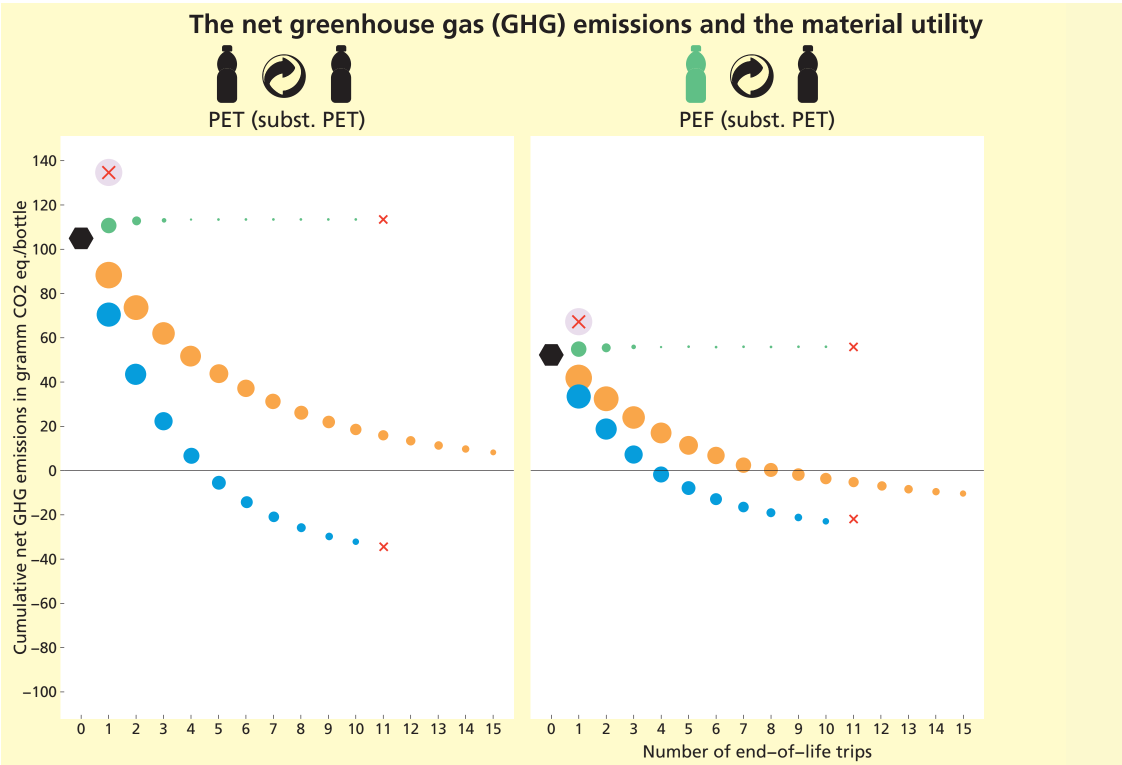
Submitted.

ABSTRACT

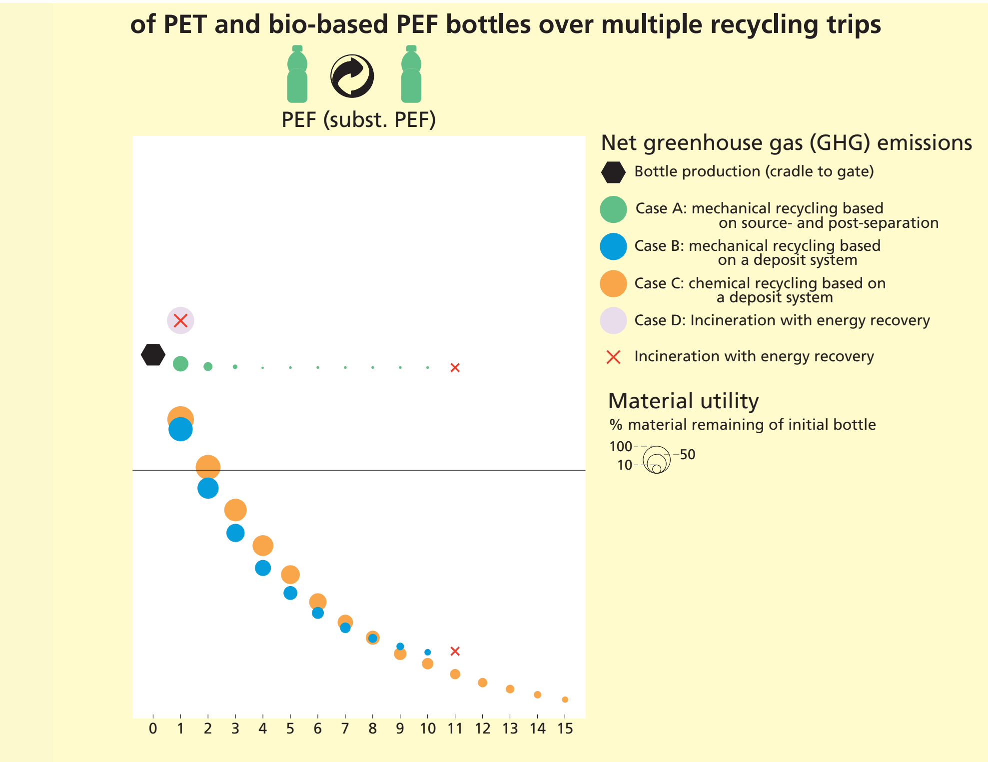
Biomass use and recycling are among the few options to reduce the greenhouse gas (GHG) emissions of the growing plastics sector. The bio-based plastic polyethylene furanoate (PEF) is a promising alternative to polyethylene terephthalate (PET), in particular for small bottle applications. We assessed the life-cycle global warming potential (GWP) and the material utility (MU) for 250 mL PET and PEF bottles over multiple recycling trips based on mechanical (MR) and chemical recycling (CR) in the Netherlands. We found that bio-based PEF would offer 50-74% lower life-cycle GHG emission after one end-of-life trip compared to PET, depending on the waste management case. Our results also show that deposit-based recycling systems significantly reduce the cumulative cradle-to-grave net GHG emissions for both bottle types, especially when multiple recycling trips are applied. We reveal trade-offs between GWP and MU: While deposit-based CR shows the best performance in terms of MU, it falls behind deposit-based MR when it comes to net GHG emissions due to the energy intensity of chemical recycling. Hence, combining mechanical and chemical recycling could contribute to achieving the goals of the circular economy and climate change mitigation alike.



Highlights

- A circular bioeconomy could reduce the growing environmental impact of plastics
- Bio-based PEF offers 50-74% lower life-cycle GHG emission than PET in 250 mL bottles
- Deposit-based recycling systems lower GHG emissions and increase material utility
- Multiple recycling trips improve the performance of deposit-based recycling systems
- We show trade-offs between mechanical and chemical recycling systems

Figure 5.1: The net GHG emissions and the material utility of PET and bio-based PEF bottles over multiple recycling trips
(Graphical abstract)



5.1 INTRODUCTION

5.1.1 Background

The production volume of plastics has grown faster than any other bulk material since 1971[4]. The plastics sector was estimated to be responsible for 4.5% of the global GHG emissions in 2015[8]. With a contribution of almost 45%, the packaging sector poses the largest demand for plastic polymer resins [6]. Among them, polyethylene terephthalate (PET) covers 22.5% of the global plastic packaging market, making it the second most used polymer resin in plastic packaging after LDPE [6]. Moreover, PET is the most recycled polymer in Europe [213,214].

Biomass use and recycling are among the few options to lower the plastic sector's growing Greenhouse-gas emissions (GHG) emissions and reduce dependence on virgin fossil feedstocks[4,9,17]. Together, biomass use and recycling are an integral part of a circular bioeconomy; a concept increasingly brought forward within the European Union [197].

A potential alternative to PET could be 100% bio-based polyethylene furanoate (PEF). PEF was developed by the Dutch company Avantium and is expected to be commercialised as of 2024 [215]. PEF is formed by polymerising sugar-based furandicarboxylic acid (FDCA) with bio-based mono-ethylene glycol (MEG). PEF has superior gas barrier properties compared to PET, especially for O_2 and CO_2, which means that less material is required to achieve the same shelf life as conventional PET [216–219]. PEF also has a higher modulus than PET, making it possible to produce containers of equivalent mechanical strength with less material [219]. This makes PEF particularly suited for food packaging applications that require a long shelf life while keeping the packaging lightweight. The applicability of PEF is especially attractive in small packaging applications as these have a relatively high material footprint per unit of packaged product volume. Hence, one of Avantium's initial focus areas for the use of PEF are small bottles for carbonated or oxygen-sensitive products.

5.1.2 Research gap & goal

A recent Life Cycle Assessment (LCA) conducted by the nova-Institut showed a GHG emission reduction potential for a clear, 250 mL monolayer PEF bottle of 33% when compared to an equivalent PET bottle over their life cycle [220]. An assessment by Eerhart, Faaij, and Patel [221] estimated cradle-to-grave GHG emissions savings in the range of 45 – 55% when comparing PEF and PET polymers, but disregarding any application.

Regarding the end-of-life (EoL), Eerhart, Faaij, and Patel[221] only assessed incineration. Also for other bio-based plastics, the end-of-life phase has so far received limited or no attention in scientific literature. A review by Spierling et al. [32] discovered that only for polylactid acid (PLA) there is a significant amount of eleven LCA studies also addressing end-of-life options, followed by two studies on thermoplastic starch (TPS) and a few individual ones for other

plastic types. While Puente and Stratmann[220] included a simplified end-of-life scenario for PEF bottles based on incineration and open-loop mechanical recycling, the study did not analyse alternative scenarios or the impact over multiple recycling trips.
The effect of multiple recycling trips on GHG emissions has so far not been assessed for bio-based plastics from a LCA perspective and only twice for PET[36,37]. Analysing multiple recycling trips would allow us to calculate the overall GHG emissions of the produced polymers over their entire life cycle, including the emissions occurring after the first EoL phase.

Multiple recycling trips would contribute to circular economy goals by increasing the product's utility[39,40]. The Ellen MacArthur Foundation[39] defines a product's utility as a combination of the length of a product's use phase and the intensity of its use.

Next to mechanical recycling (MR), chemical recycling (CR) is increasingly considered an alternative solution for treating plastic waste[208]. MR refers to recovering plastic waste via mechanical processes, like shredding, washing and re-granulating, while CR breaks down the polymer structures of plastics. For PET, depolymerisation via glycolysis is seen as one of the most promising CR options[222] and has already been implemented in industrial pilots in the Netherlands and Italy[208]. Also for PEF, CR via glycolysis has already been demonstrated to be feasible[223]. Avantium is also investigating technologies to chemically recycle PEF, amongst them glycolysis[219,224]. There are already initial LCAs for the glycolysis of PET[225,226] , but these do not assess multiple recycling trips even though an analysis of multiple recycling trips could highlight the advantages of CR technologies in terms of higher recycling yields and better quality of recyclates.

A lack of understanding of the impact of end-of-life options could hamper the transition to a circular (bio)economy in plastics value chains and lead to incomplete (life cycle) assessments of the overall climate benefit of bio-based compared to fossil plastics. While a bio-based plastic might have a lower GWP than a fossil competitor in production, this advantage might be (partly) counterbalanced by worse performance in the end-of-life. Technical barriers or contaminations caused by bio-based plastics could hamper their integration into existing recycling systems[227]. Simultaneously, a separate collection & treatment of bio-based plastics is economically challenging due to their current small market shares[228]. These issues could prevent the recycling of bio-based plastics or allow fewer or lower quality recycling loops compared to their fossil competitors.

We want to address these challenges and identify the cradle-to-grave climate impact of different waste management cases in the Netherlands for a small (250 mL) plastic bottle made from bio-based PEF compared to fossil-based PET, including the effects of multiple recycling trips. By complementing this with an analysis of the material utility for each waste management scenario and bottle type, we want to identify and discuss potential trade-offs between circular economy and climate change mitigation goals.

We focus on the Netherlands as this is one of the potential initial target markets of Avantium's PEF bottle applications, after signing bottle offtake agreements with Refresco, a bottling company located in the Netherlands, and Resilux, a Belgian preform and bottle producer[229]. Furthermore, waste management data is well available for this country. The Netherlands recently introduced a deposit system for the more than 900 million small plastic bottles sold every year[230], which we compare to the previous collection systems. This study is complementary to an LCA conducted by Puente and Stratmann[220], which provides a detailed assessment of PEF bottles compared to PET bottles, including only one simplified end-of-life scenario. This study adds a more thorough analysis of the end-of-life by analysing the impact of different Dutch waste management scenarios over multiple recycling trips.

5.2 MATERIALS & METHODS

5.2.1 LCA Goal & Scope definition

We assessed the global warming potential (GWP) of our analysed systems, following the LCA methodology laid out in the ISO standards 14040 and 14044[231,232] using the LCA software SimaPro (version 9.1.0.11) and the Ecoinvent database version 3.7 for background data.

We aim to quantify the potential global warming impacts of 250 mL fossil-based PET and bio-based PEF bottles including four different waste management cases for the Netherlands (see also Figure 5.2), being:

A. the waste management system for small plastic bottles in the Netherlands until 2021, based on post-separation, source-separation, mechanical recycling and incineration with energy recovery.
B. a waste collection predominantly based on a deposit system combined with mechanical recycling and incineration with energy recovery.
C. a waste collection predominantly based on a deposit system combined with chemical recycling and incineration with energy recovery.
D. a non-circular scenario, assuming the complete incineration of the bottles with energy recovery.

The GWP over the life cycle of PET and PEF bottles including the above mentioned waste cases were assessed using the impact assessment method 'IPCC 2013 GWP100a'.

The functional unit of this study is *a 250 mL monolayer plastic bottle designed for single-use, providing minimum shelf life of at least 12 weeks for carbonated soft drinks*. The monolayer PET bottle fulfilling this function should weigh 24 grams, and the monolayer PEF bottle weighs 13 grams, according to calculations of Avantium, and substantiated by literature review, and feedback of industry experts[220,233]. The weights were calculated based on the gas permeability

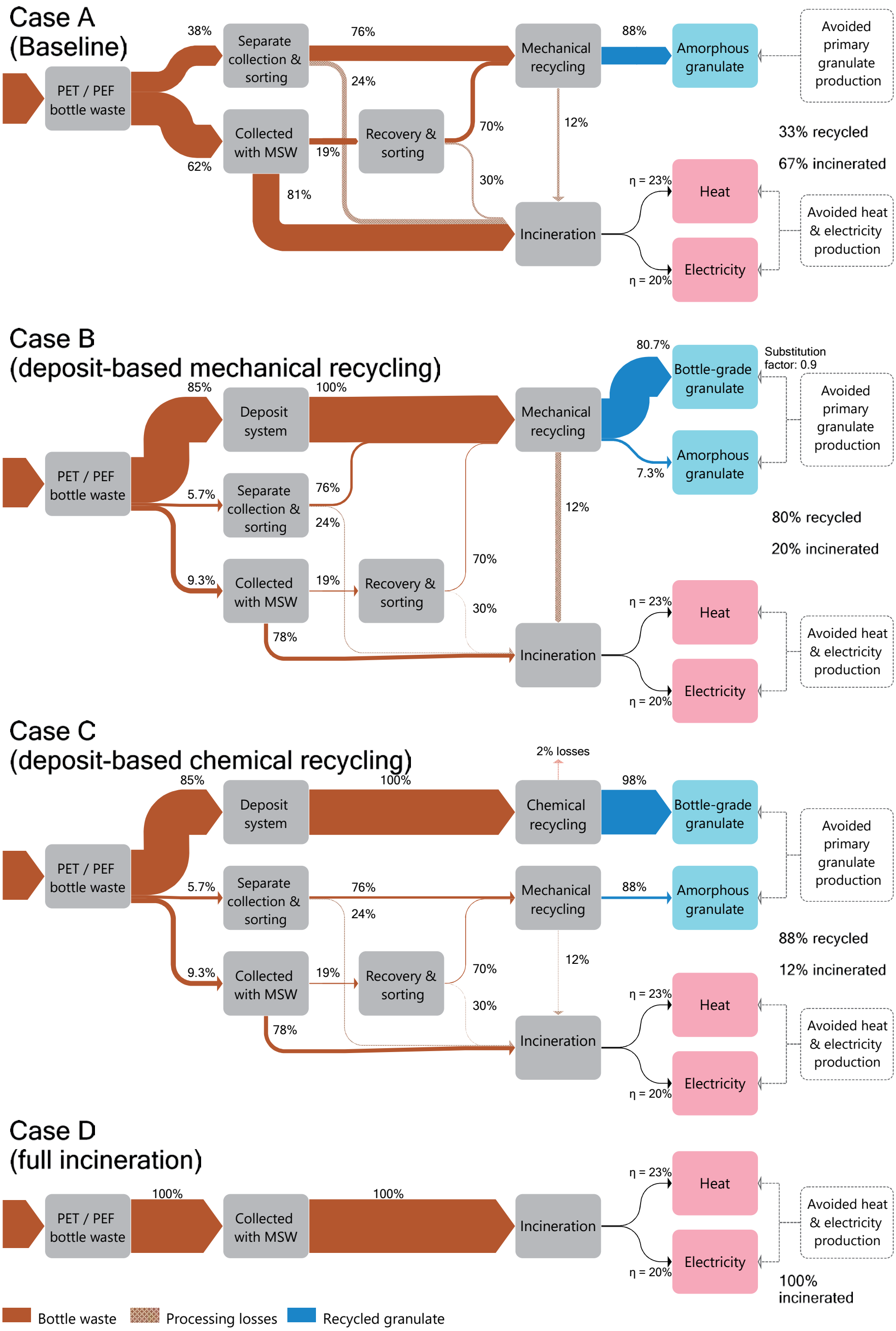


Figure 5.2: The four analysed waste management cases for small PET & PEF bottles

values and material strength of the polymers, assuming no barrier-enhancing additives were used, and that ideal stretch ratios were gained during bottle blowing. The sensitivity of the results to the bottle weights was assessed in Appendix B. Due to superior barrier properties, the shelf life of the PEF bottle extends to more than 20 weeks[220,233]. This functional unit is in line with the LCA of Puente and Stratmann[220] and represents one of the potential initial target markets of Avantium's PEF bottles.

The LCA has a scope from cradle-to-grave with a strong focus on end-of-life, following the goal of the study. The waste treatment cases are in the foreground analysis. The impacts of bottle production (cradle-to-gate), assessed in an LCA of nova-institute[220,233], were taken as the background system in our analysis. Due to our focus on the material flows of PEF and PET, we exclude the bottle's caps, neck rings, and labels since we assume they can be identical in PET and PEF bottles.

5.2.1.1 Product systems

We cover the production of PET bottles from petrochemical feedstock and PEF bottles from bio-based feedstocks from cradle-to-gate, using results of existing assessments[146,220,233]. Appendix A provides an overview of bottle production (Figure 5.7 and Figure 5.8). Potential emissions from the use phase are excluded because the impacts are considered negligible and comparable between PET and PEF bottles. However, the shelf-life difference between both bottles is addressed when discussing the material utility of both bottle types for multiple recycling trips.

We cover the end of life (EoL) of the plastic bottles, including their collection, sorting, and waste treatment. Transport activities within and between the EoL-stages are also included.

We distinguish between four waste management cases (see Figure 5.2) for PEF and PET bottles. These cases differ in collection & sorting methods (post-consumer separation from municipal solid waste (MSW), source separation, and deposit system), recycling technologies (mechanical and chemical recycling), and the corresponding differences in the amount and quality of the recycled material. We assume that all bottles are eventually collected and ignore the impacts of littering plastic bottles.

In the Netherlands, post-consumer plastic packaging waste collection differs by municipality. An assessment by Brouwer et al.[234] estimates that in 2017 38% of Dutch post-consumer plastic packaging waste was collected separately at the source, and 62% ended up in MSW. 19% of the MSW fraction is sent to material recovery facilities for sorting, with the rest being sent to incineration plants [234]. We assume the same collection rates for the small PET and PEF bottles in our baseline case A, see Figure 5.2. The mass flows of all processes are described in section 5.2.2.3. Overall, baseline case A leads to a high share of incineration with energy recovery (67%) and a mechanical recycling rate of 33%. The term recycling rate is used differently in literature, often referring to the plastics sent to recycling. We define the recycling rate as the net weight of recycled material divided by the net weight of collected material. Our recycling rate is thus the product of the sorting efficiency and the efficiency of the recycling process.

In July 2021, the Netherlands introduced a deposit system for plastic bottles smaller than 0.5L[230]. We assume that 85% of the small PET and PEF bottles will be collected via a deposit system in waste management cases B and C, based on a prognosis for the Netherlands[235] and a study of the relationship between the collection rate and the deposit amount[236]. The remaining 15% are assumed to be collected in the same ratio as in case A. Case B continues with the mechanical recycling of the plastic bottles, now achieving a higher recycling rate and higher plastic quality compared to Case A due to the introduced deposit system. In contrast, Case C uses chemical recycling for the plastic bottles collected via the deposit system. The remaining bottles in the chemical recycling case are mechanically recycled or incinerated in the same ratio as in case A. Case D assumes that all plastic bottles are collected along with the mixed municipal solid waste and then directly sent to incineration with energy recovery. The mass and energy balances for the waste management cases in Figure 5.2 are described in the inventory, section 5.2.2.

5.2.1.2 Assessing the waste treatment of PEF

Given the novelty of PEF plastic, there is no data available for PEF waste treatment. Due to similarities between PET and PEF, Avantium claims that PEF can be recycled using existing PET mechanical recycling assets like dryers, extruders, crystallizers and SSP equipment[219]. Moreover, PEF could be sorted by commercial near-infrared sorting equipment[219,237]. Furthermore, tests showed that small shares of PEF would not have a negative impact on recycled PET if mixed[237,238]. An assessment by Avantium even claims that a 5% fraction of PEF in the PET stream would improve the quality of recycled PET and lead to a better crystallinity and a longer shelf life [239]. Hence, the European PET Bottle Platform (EPBP)[237] provided an interim approval for up to 2% market penetration of PEF. In the absence of PEF specific data, we used the same waste treatment data for PEF as PET in this study after adjusting for differences in heating value and carbon content. With small market shares in the short term, Avantium expects that PEF will be integrated into PET recycling processes (open loop), but closed-loop PEF to PEF recycling systems are preferred, certainly at higher market shares[219].

5.2.1.3 Geographical, temporal, and technological scope

The feedstock supply for Avantium's PEF production is based on starch from wheat cultivated in France used for fructose production and on ethanol-based bio-MEG produced from sugarcane in India. Avantium's FDCA plant will be located in Delfzijl, Netherlands, and the polymerisation of FDCA will mainly happen in European facilities. PET production data represents average European production. We assume that polymerisation, bottle manufacturing and waste treatment occur in the Netherlands.

Data for Avantium's YXY technology (see Figure 5.7 in Appendix A) was taken from Avantium's 5kt/a flagship plant design[219], scaled up to 100kt/a, to represent the first commercialisation phase of PEF[220]. The datasets are based on Aspen Plus process simulations and experimentally-derived data from Avantium's pilot plant in Geleen.

Background energy inputs are based on the energy mix as presented in Ecoinvent (IEA data from 2017, extrapolated to 2020). Regarding the EoL, we combine state-of-the-art data for sorting, mechanical recycling and incineration (from Ecoinvent) with small-scale production data for chemical recycling[226]. All end-of-life data was adapted to the Netherlands.

5.2.2 Life Cycle Inventory Analysis

Table 5.1 in Appendix A summarises all data used and assumptions made for the inventory of the LCA.

5.2.2.1 Allocation

In PEF production, partitioning the environmental burdens is required for by-products in wheat cultivation, wet milling, Avantium's YXY technology, and the sugarcane refinery, see Figure 5.7 in Appendix A[233]. We use the cradle-to-gate PEF and PET polymer production results from Puente and Stratmann[220,233] as input to our study, who applied economic allocation to allocate the environmental burdens. This is in line with the PAS 2050 recommendation on allocation choices when assessing bio-based products[240].

For modelling the end-of-life, the avoided burden approach was applied, following the recommendation of the ISO standards[231,232]. This approach enables us to capture the environmental consequences of the analysed waste management cases. Some scenarios achieve better quality in recycled output (e.g., higher viscosity, clean streams suitable for bottle-grade applications), and PET achieves a higher energy recovery in incineration, which we want the results to reflect. Accordingly, we provide credits to our production systems for the recycled material and the generated energy, see section 5.2.2.3 on substituted product systems.

5.2.2.2 PET and PEF bottle production

PEF production consists of wheat cultivation, wet milling, and fructose production, followed by FDCA production via Avantium's YXY technology. The FDCA is then copolymerised with bio-based MEG to PEF granulate (see Figure 5.7 in Appendix A). Data on bottle-grade PEF granulate production was taken from the LCA conducted by nova-Institute[220,233].

PET production consists of oil refining, MEG and PTA production and their copolymerisation into PET. Puente and Stratmann[220,233] used ecoinvent data to model PET granulate production[146]. Their results for PEF and PET production were calculated using the ecoinvent 3.6 database, while our assessment uses the ecoinvent 3.7 database for background data. To have consistent results in the granulate production, we used the results for both PEF and PET granulate production from Puente and Stratmann[220,233] as input to our study.

Bottle-grade polymers are stretch blown into bottles. For stretch-blow moulding, we used Eco-Invent data (see Table 5.1 in Appendix A). The downstream processing steps (polymerisation and bottle production) are adjusted for the Dutch energy mix as provided by Ecoinvent (IEA data of 2017, extrapolated to 2020).

5.2.2.3 Post-consumer waste management of PET and PEF bottles

We gathered the data for the waste management systems for PET and PEF bottles via literature review and interviews. As waste treatment is assumed to occur in the Netherlands, all waste management processes described below are adjusted with the Dutch heat and electricity resources. All processes are assumed to be the same for PEF unless a difference is explicitly mentioned.

Table A.1 in the Appendix summarises all data and assumptions related to the waste management of small plastic bottles used for this study. Table 5.2 in Appendix A details the datasets and transport distances used for all transportation steps in the waste management steps.

Collection & transportation

We chose Swiss Ecoinvent data to assess plastic waste collection with a 21-ton lorry (see Table 5.2 in Appendix A). The differences in transportation distances between the collection systems are considered, using the average transportation distances of waste collection systems in the Netherlands[241]. For the deposit system, we did not consider the transport of the consumers to the bottle collection points, assuming that it is part of regular consumer movements for groceries. We also used ecoinvent data to model the plastic waste transportation emissions between the different waste treatment facilities for each analysed waste management case, adjusted for average Dutch transport distances (see Table 5.2 in Appendix A).

Sorting

We assume no sorting losses for the bottles collected by the *deposit system*. They are directly sent to PET and PEF recyclers.

The plastics collected by *source separation* are sorted into fractions based on plastic type. A standard sorting process consists of the removal of impurities like metals, followed by sorting into different materials and colours, using sink-float separation, electronic sorting via laser- and near-infrared sensors, and finally manual sorting[242,243]. In the Netherlands, sorted fractions are distinguished based on the DKR (Der Gruener Punkt) standards adopted from Germany, with DKR 328-1 containing PET bottles[244]. Brouwer et al[234] report that 76% of small PET bottles (<= 0.5l) are sorted for recycling. The remaining fraction is assumed to be incinerated with energy [234]. See also the mass flow presented in Figure 5.2. We used Swiss data on sorting PET waste into sorted PET bales[245], and adjusted it for the Dutch energy mix (see Table A.1 in appendix A).

In a *post-separation* system, roughly 19% of packaging plastics collected with MSW are sent to material recovery & sorting facilities (MRF), where 70% of the small clear PET bottles (<= 0.5l) entering these facilities are recovered [234]. In a pre-treatment step, hard plastics are recovered from the remaining MSW before being sorted into specific plastic fractions. According to industrial data obtained from Dutch MRF's [241], the energy requirements for this pre-treatment step are approximately four times higher than those for sorting into plastic fractions. We adapted the Ecoinvent processes for the recovery and sorting from MSW with energy use including pre-treatment (see Table 5.1 in Appendix A). The processing losses from the post-separation system are assumed to be sent to incineration plants, along with the remaining MSW.

Mechanical recycling (MR)

For MR, the sorted PET bales are opened, purified (removing labels, caps and contaminants), and shredded, before the purified PET flakes are washed and dried[245]. Recycled PET and PEF directed at bottle-grade applications also goes through a solid-state polymerisation process (SSP) to improve crystallinity.

The technology data of MR is the same in all three recycling cases A-C. We modelled the production of recycled PET bottle-grade granulate (rPET) by adapting the ecoinvent process *"Polyethylene terephthalate, granulate, bottle-grade, recycled {CH}"* [245], with the Dutch energy mix as provided by ecoinvent. We derived the production of recycled amorphous PET from the same dataset by subtracting the requirements for SSP. Following our system boundaries, we excluded the waste treatment of bottle accessories (caps, labels). We assume a recycling efficiency from sorted, baled PET bottles to recycled PET pellets of 88% based on assessments of the Dutch and Danish waste management systems for PET plastic packaging [35,234].

We assume the same data for PEF bottles. Some processes in PEF recycling are expected to require less and others more energy than PET recycling. Therefore, in the absence of sufficient data, we assume that the overall energy requirement of mechanical recycling would be similar between PET and PEF.

Chemical recycling (CR)

Our waste management case C focuses on CR based on depolymerisation via glycolysis. This CR technology is one of the most advanced for PET[222,246] and has also been proven to work with PEF[223].

We assume that only the deposit fraction is directed to glycolysis since so far the EU regulations on the guaranteed 95% food packaging origins also apply to chemically recycled material [247,248]. Data on glycolysis is hard to come by in publicly available literature. Shen et al.[226] present data from a Taiwanese small-scale production plant, covering the glycolysis of PET waste to the oligomer bis-hydroxyl ethylene terephthalate (BHET), which is then filtered and repolymerised to PET. We adapted the data of Shen et al.[226] to our system boundaries (see see Table 5.1 in Appendix A). In the absence of data on the chemical recycling of PEF, we assumed the same process requirements as for the chemical recycling of PET.

Incineration with energy recovery

Sorting and recycling losses from waste management cases A-C are assumed to be incinerated with energy recovery. Case D assumes full incineration of PET and PEF waste with energy recovery. In the Netherlands, incineration plants' average electricity and heat generation efficiencies are 20% and 23%, respectively[249]. The Swiss ecoinvent process of waste incineration was adjusted accordingly (see see Table 5.1 in Appendix A).

The CO_2 emissions from incineration are calculated based on the carbon content of PET and PEF, which are 62.2% and 52.7%, respectively (calculated based on the molar masses). Also, the generated energy through incineration differs between PET and PEF, as PET has a lower calorific value of 22.1 MJ/kg (ecoinvent) whereas the lower caloric value of PEF is 16.7 MJ/kg (based on experimental calorimetric calculations by Avantium).

Substituted product systems

We follow the "avoided burden approach" and provide credits to our production systems for the provision of recycled material and recovery of energy in the end-of-life (see section 5.2.2.1).

Sorted PET bottle fractions from the source separation and post-separation of MSW have a 17-24% share of non-food flasks in the Netherlands[250], while 5% is the legal limit set by the European Food Safety Authority (EFSA) for food-grade recycling[248]. Hence, PET and PEF bottles collected in MSW and via source separation can legally not be recycled into new bottle applications used for drinks. Therefore, we assume that all recycled PET and PEF bottles from these collection methods will be "downcycled". As a consequence, the recycled polymers are assumed to substitute virgin *amorphous PET and PEF*.

We assume that only bottles collected via a deposit system are recycled for bottle-grade applications, as those conform with the EFSA regulation[250]. However, due to quality losses in mechanical recycling (e.g. decrease in crystallinity), we apply a substitution factor of 0.9 to account for the quality loss for recycled bottle-grade PET and PEF when substituting virgin *bottle-grade PET and PEF*. This factor reflects the decrease of intrinsic viscosity when mechanically recycling blue post-consumer PET bottles compared to virgin PET[251] and is also recommended by the *"Product Environmental Footprint"* guide (Annex C) of the European

Commission[252]. Moreover, Brouwer, Chacon, and van Velzen[253] showed that recycled PET from mono-collection systems could meet industry standards of bottles at a recycled content ratio of around 90%.

The differences in intrinsic viscosity are also present in the recycled amorphous granulate. However, we do not apply a substitution factor to amorphous granulate, assuming that a decreasing viscosity does not affect their use for amorphous applications (e.g. fibres). Chemical recycling does not require a substitution factor since it achieves the same polymer quality as primary production.

For recycled PEF, we analyse the substitution of both PET (open-loop recycling) and PEF (closed-loop recycling). We do so as initially the PEF bottles are planned to be recycled together with PET, hence, replacing primary PET production. Once larger amounts of PEF are on the market, closed-loop recycling of PEF substituting primary PEF bottles is assumed. For open-loop recycling of PEF, we assume that one gram of PEF will substitute one gram of PET. In theory, PEF could substitute a higher weight of PET, due to its better material properties (see functional unit). However, in practice, we do not expect this to influence the weight of PET bottles blended with PEF as long as the PEF market shares stay as low as the 5 % limit set by the EPBP[237].

We also credit the energy generated from incinerating the bottle waste, assuming a substitution of the average Dutch electricity and heat production mix according to ecoinvent (IEA data of 2017, extrapolated to 2020).

5.2.3 Biogenic emissions

Following IPCC guidelines[171], we treat biogenic CO_2 emissions as climate neutral. Our results for bio-based PEF production and waste treatment include biogenic emissions, but we also account for the net biogenic carbon removals (i.e. carbon embedded in the product) during the growth of the bio-based feedstock. We do not credit the biogenic carbon removal for the substituted product system (PEF recyclates substituting primary PEF production).

We do not apply any credits for the delayed emission of carbon because of the relatively short carbon cycles of the bottle application. However, we specify the overall storage time of carbon per waste management case as complementary information (see section 5.2.4.2 and Figure 5.6).

5.2.4 Assessing multiple recycling trips

PET has a higher recyclability than other packaging plastics, as it absorbs fewer post-consumer contaminations than, e.g., polyolefins[254]. Nevertheless, there is limited information on how often PET could be mechanical recycled without losing its critical properties, like its intrinsic viscosity, colouring and the presence of contaminants. Pinter et al.[254] assessed eleven recycling trips for PET in a closed-loop system, showing that the quality of the mechanically recycled bottles was not negatively affected when mixed with 25% virgin PET. Brouwer et al.[253] assessed the accumulation of contaminants over ten recycling trips, showing that recycled bottles from mono-collection systems could meet acceptable standards even when only mixed with around 10% of virgin PET. Lab-scale assessments by Avantium showed that recycled PEF resins could keep their mechanical properties over 12 loops at a 70% and 90% recycled content ratio (Personal communication of Roy Visser from Avantium, 11.5.2022).

We assess the cumulative net GHG emissions and material utility achieved by the waste management cases A and B for PET and PEF bottles over 10 recycling trips. After that, we assume that the remaining material will be incinerated. CR does not cause material quality deterioration and could therefore achieve more recycling trips. We chose to assess 15 recycling trips for case C to make this advantage of CR visible.

5.2.4.1 Cumulative net GHG-emissions

As shown in Equation 1, we calculate the cumulative, cradle-to-grave net GHG emissions 'CE' for each waste management case 'i' and bottle type 'k' by adding the cumulative net end-of-life emissions over all end-of-life trips 't' to the cradle-to-gate bottle production emissions 'PE'. The maximum number of end-of-life trips 'n' differs per waste management case and is ten for cases A and B, 15 for C, and one for D. The net GHG emissions are the sum of direct GHG emissions, the GHG credit received for the substituted virgin plastic granulate and the substituted energy, and the biogenic carbon uptake. The cumulative end-of-life net GHG emission is calculated based on the mass 'm' of the PET and PEF material entering the waste treatment at each recycling trip and the net GHG emissions 'b' of one recycling trip (in g CO_2 eq./g polymer waste). All equation variables are further explained in Appendix A.

Equation 1: Cumulative cradle to grave net GHG emission over multiple recycling trips (variables are explained further in Appendix A)

$$CE_{k,i} = PE_k + \sum_{t=1}^{n} m_{k,i}(t-1) * b_{k,i}$$

5.2.4.2 Material utility

A product's utility is defined by the length and the intensity of the product's use [39]. We propose the concept of material utility, inspired by the product utility defined by Ellen MacArthur Foundation[39]. The *material utility* consists of the *material use intensity* and the *length of the material's use*, which we assess separately.

Material use intensity

This study defines the cumulative material use intensity (MI) as the percentage of additional material use achieved out of the initial virgin material, as shown in Equation 2. We calculate the weight 'm' of the cumulatively recycled polymers for a maximum of n = 15 recycling trips 't', using the overall recycling rate 'r' (sorting yield times recycling yield), for each waste management case 'i'. To calculate the material intensity, we then divide this by the weight of the virgin PET or PEF bottle type 'k'.

Equation 2: Material use intensity of a bottle per waste management case (variables are explained further in Appendix A)

$$MI_{i,k} = \left(\sum_{t=1}^{n} m_{k,i}(t-1) * r_i \right) \Big/ m_k(t=0)$$

Length of material use expressed in carbon sequestration time

The duration of a material's use is considered part of the material's utility. By assessing the overall amount and duration of carbon sequestration of the initial virgin bottle material, we combine an assessment of the use time of the PET and PEF material (expressed in bottle shelf life) with an evaluation of the total embedded CO_2 emissions. The effect of delayed emissions is not part of our LCA results but merely presented as complementary information.

We use the initial bottle weight 'm' per bottle type 'k' (24 g for PET, and 13 g for PEF bottle) and the carbon content 'CC' (62.5% for PET, 52.7% for PEF) as input. To calculate the amount of sequestered carbon 'C' remaining after each recycling trip 't', we multiply the overall recycling rate 'r' of each waste management case 'i' with the remaining carbon from the previous recycling-trip (t-1).

Equation 3: Sequestered carbon after each recycling trip

$$C_{k,i}(t) = CC_k * m_{k,i}(t-1) * r_i$$

The molecular weight ratio of carbon dioxide to carbon (44/12) is used to report the embedded CO_2 emissions.

We put the sequestered carbon over multiple recycling trips in relation to the use time of the bottle material (see Figure 5.6). As a proxy for the length of use, we chose the bottle's shelf life, which is 12 weeks for a PET bottle and 20 weeks for the PEF bottle in our product system, due to the superior barrier properties of PEF.

5.3 RESULTS & DISCUSSION

5.3.1 The global warming potential of the bottles assuming one recycling trip

5.3.1.1 Comparing the waste management scenarios

For both PEF and PET bottles, the order of waste management cases in terms of GWP is the same: Case B performs the best, followed by Case C and Case A. The complete incineration with energy recovery (Case D) shows the largest emissions.

Incineration of bottle waste is the major contributor to the end of life emissions. Also, the benefit of substituting primary plastic production has a crucial impact on the results (see Figure 5.9 in Appendix B). Hence, assuming a closed-loop PEF recycling system (green triangle) shows lower net GHG emissions compared to an open-loop system for PEF, substituting PET (black dot). This is because we assume PEF substitutes PET one to one, despite its advantages in material properties (see methods), and because one gram of PEF granulate is more emission-intensive than 1 gram of PET if we ignore the biogenic carbon uptake.

The overall recycling rate is a key driver of the emissions advantages achieved by the waste management cases B and C because it defines how many bottles are recycled to substitute primary plastics and how many are incinerated. Hence, the deposit system greatly influences

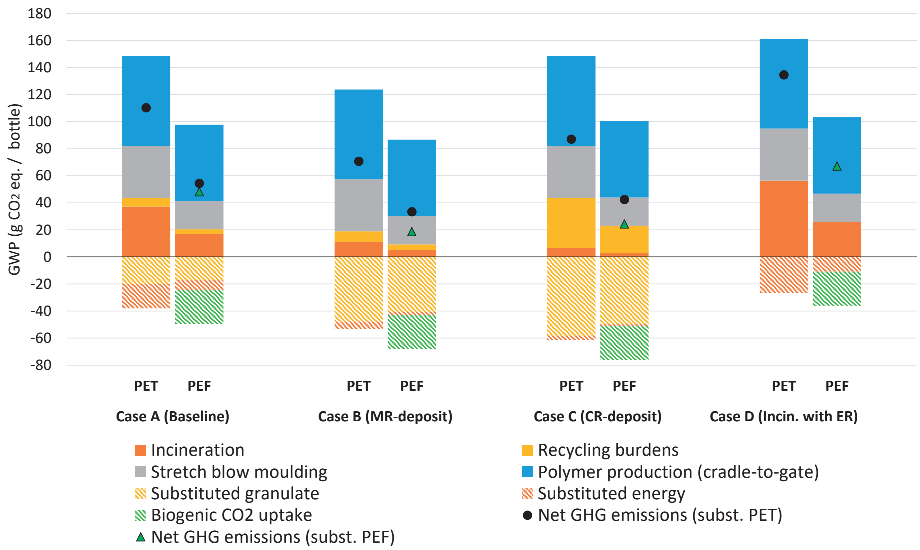


Figure 5.3: The cradle-to-grave net GHG emissions for one bottle after one recycling trip

Notes: Recycling burdens include collection, sorting, and transportation; The columns display closed-loop recycling (PEF subst. PEF, PET subst. PET); The net GHG emissions differentiate between open- and closed-loop recycling for PEF the overall net emissions since it avoids sorting losses: Case C for PET bottles has 21% lower net GHG emissions than the baseline (case A) and case B even 36%. For closed-loop PEF recycling, these savings compared to Case A could reach up to 49% and 61% respectively. The net GHG emission savings of case B compared to case D could even reach 72% for PEF when substituting PEF. Also the substitution rate plays a significant role (see discussion of sensitivities in Appendix C).

However, despite the fact that chemical recycling achieves the highest recycling rate and substitution factor, case C performs worse than case B. This is because the high energy requirements of the chemical recycling process undermine its advantages regarding the amount and the quality of recycled plastics produced. To perform better than mechanical recycling after one recycling trip, chemical recycling would need to reduce its process emissions by 44% for PET bottle recycling and by 28% for closed-loop PEF bottle recycling.

5.3.1.2 Comparing PET & PEF bottles

Overall, in terms of cradle-to-grave net GHG emissions, a PEF bottle performs better than a PET bottle if we assume the same waste management case and one recycling trip. However, it makes a difference if an open-loop recycling system (PEF substituting PET) or a closed-loop system (PEF substituting PEF) is in place. If PEF is assumed to substitute PEF, it receives more credits for recycling, as the impact of the displaced primary PEF granulate production is higher when ignoring the biogenic carbon uptake. For such a closed-loop system, the cradle-to-grave net GHG emissions of a PEF bottle could be 56%-74% lower than for the PET bottle after one recycling trip (depending on the waste management case). When substituting PET, the relative GHG emission savings only range from 51-53%, as the substituted primary PET granulate is less emission-intensive when substituted one to one. Hence, recycling PEF becomes even more beneficial in terms of net GHG savings when PEF is recycled separately in a closed-loop system. The relative emission savings of a PEF bottle are 50% when comparing the full incineration of bottles (Case D). The net GHG emission advantage of PEF bottles diminishes if PEF would be recycled less than PET or even incinerated (e.g., comparing case B for PET with case A or D for PEF). Hence, it is important to properly integrate PEF into the recycling system to maintain its advantage in net GHG emissions.

When ignoring the biogenic carbon uptake during biomass cultivation, the cradle-to-gate GHG emissions of the PEF bottle production are 26% smaller than those of the PET bottle. A key driver for this is the superior barrier qualities of PEF, which could reduce the polymer use in bottle production by almost 46 %. This difference in the cradle-to-gate GHG emissions increases to 50% when we also account for the sequestration of biogenic carbon (25 g/bottle) which is taken up during biomass cultivation.

Without the biogenic carbon uptake and the credits from the avoided impacts, the contribution of EoL waste treatment in total cradle-to-grave gross GHG emissions of the PEF bottle is ranging from 11% (Case B) to 25% (Case D) (see Figure 5.10 in Appendix B). For the PET bottles, this range is 15-35%. The EoL phase of a PEF bottle causes fewer GHG emissions than the EoL of a PET bottle, as a PEF bottle has a lower weight and carbon content.

5.3.2 Cumulative net GHG emissions for multiple recycling trips

When looking at the development of the cumulative net GHG emissions over multiple recycling trips, the relative performance of the waste management cases changes significantly. While the results for case A are barely affected by assuming multiple recycling trips, the net GHG emission saving benefits of both deposit-based cases (B and C) increase with each recycling trip compared to the baseline case A and the non-recycling case D. For example, for PET bottles, the cumulative net GHG emission reductions compared to case D increase from 64 g CO_2 eq. (case B) and 48 g CO_2 eq. (case C) after one recycling trip to, respectively, 165 and 130 g CO_2 eq. after 10 (case B) and 15 recycling trips (case C).

Case B benefits less from increasing the number of recycling trips than case C. After five trips, case B already achieved 85% of total cumulative net GHG emission reductions through recycling, while case C achieves around 72% of its total cumulative reductions (in 15 trips) until the fifth recycling trip.

Moreover, the difference between cases B and C gets reduced after the 10th recycling trip, as we assume that the polymers cannot be mechanically recycled more than ten times, while CR enables additional recycling trips as its recyclates are equivalent to primary plastics. Case C

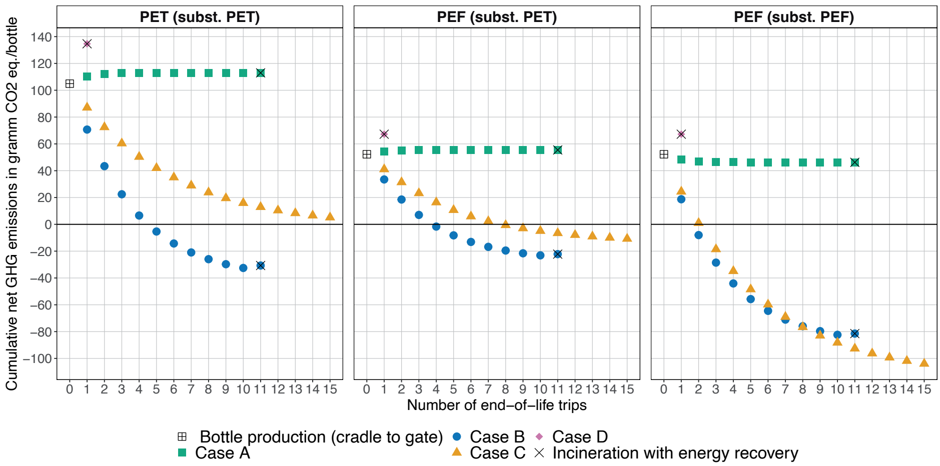


Figure 5.4: Cumulative cradle to grave net GHG emissions over 10 recycling trips for mechanical recycling (for cases A and B) and 15 trips for chemical recycling (case C)

performs better for PEF bottles, as their lower weight significantly reduces the high energy use of chemical recycling. For closed-loop PEF recycling, case C even becomes the best option starting with the 8th recycling trip. CR retains more bottle material due to the higher recycling and substitution rate and thus benefits most from the higher carbon benefit achieved by displacing primary PEF production.

However, case B stays the cumulatively best performing option for the entire number of analysed recycling trips for both analysed systems substituting PET. Moreover, also for closed-loop PEF recycling it stays superior until the 7th recycling trip. Case B even achieves cumulative net negative emissions with the second recycling trip for closed-loop PEF recycling, with the fourth trip for open-loop PEF recycling, and with the 5th trip for PET recycling, when considering the carbon credits achieved from displacing primary plastics.

5.3.3 Material utility

Figure 5.5 expresses the material use intensity as the % of additional material use achieved from the initial, virgin plastic material over 10-15 recycling trips. The baseline case A only achieves 48% of additional material use and only for lower-grade amorphous applications. The deposit-based cases achieve a significantly higher material intensity with over 300% (case B) and almost 500% (case C). Moreover, their outputs are to a large extent usable in new bottle applications. The cumulatively produced and recycled bottle-grade material would equal a total of 3.8 and 5.7 bottles respectively (incl. the initial virgin bottle).

So in terms of material use intensity, CR is clearly the preferable option. Next to the higher efficiency of CR, this can be mainly attributed to the fact that CR allows for many more recycling trips than MR, while MR reaches its recyclability limit earlier due to quality losses.

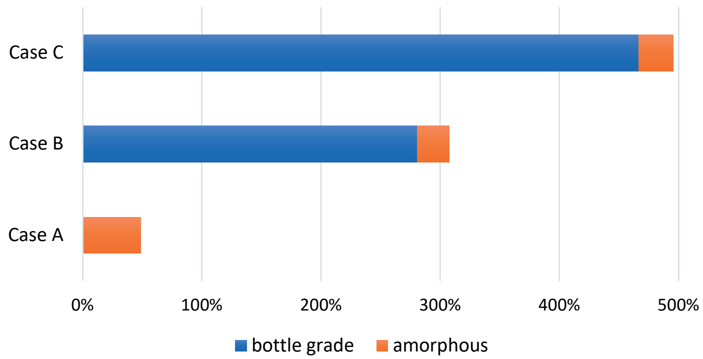


Figure 5.5: Material use intensity achieved by the three recycling cases over 10 (MR) and 15 (CR) recycling trips

Notes: expressed as the percentage of additional material use achieved out of the initial virgin material over multiple recycling trips, differentiating between bottle and amorphous applications; It is the same for PET and PEF bottles since we assume the same waste management cases.

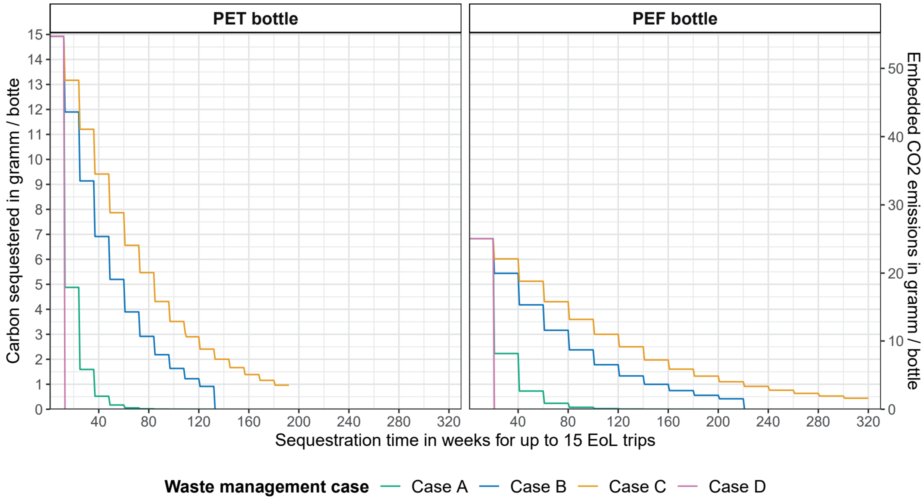


Figure 5.6: Carbon sequestration over time per bottle type and waste management case, showing how long the carbon of the initial bottle is sequestered and how much of it (in new bottles or other applications).

This advantage in the material use intensity of CR would even be higher if we would have analysed more than 15 recycling trips.

Next to the material use intensity, the overall length of the material use is another part of a material's utility. By showing the carbon sequestration over time per bottle and waste management case, Figure 5.6 combines an assessment of the overall use time of the bottle material with the corresponding embedded carbon emissions. Also here chemical recycling (case C) clearly outperforms the other analysed waste management cases as it sequesters more of the material (see material use intensity) over a longer time.

The PEF bottle could theoretically achieve a significantly longer carbon sequestration time over multiple recycling trips than a PET bottle (a maximum of 320 weeks compared to 192 weeks for a PET bottle for case C over 15 recycling trips), due to the longer shelf-life a PEF bottle provides. Moreover, the PEF bottle acts as a short-term carbon sink due to the biogenic nature of its carbon content. However, this effect is negligible due to the short lifetime of bottles. Only in long-term applications like in the building & construction sector, the effect of bio-based carbon sequestration could be considerable[18].

Overall, CR (case C) is superior to the other waste management cases in terms of material utility. However, the high energy use of CR largely offsets its advantages in material utility when assessing the GWP.

5.4 CONCLUSIONS

We analysed four waste management scenarios for PET and PEF bottles over 10-15 recycling trips, which clearly showed the superiority of deposit-based recycling systems over the baseline based on the 2017 Dutch mix of post-separation and source separation.

Furthermore, we observe a trade-off between circular economy goals (expressed in material utility) which favour CR (case C) and climate change mitigation, which favours MR (case B). The high recycling yield and substitution factor of CR (case C) make it the best option in terms of material utility, by keeping more of the plastics in the technosphere for a longer period of time. However, the high energy requirements of CR hinder its performance when looking at the GWP impact. Deposit-based MR (case B) shows the lowest net GHG emissions.

Changing the analysis from one recycling trip to multiple trips changes the relative performance of the waste management cases. The cumulative net emissions savings of the deposit-based cases B and C increase with each recycling trip compared to case D (incineration with energy recovery), while case A (baseline) barely changes. Furthermore, CR is catching up with deposit-based MR in terms of cumulative net GHG emissions after ten recycling trips. However, for PET recycling and open-loop PEF recycling (substituting PET) deposit-based MR (case B) remains the option with lowest net GHG emissions. Only for closed-loop PEF recycling (substituting PEF), do we see a preference for CR (case C) compared to MR (case B), but only after 8 recycling trips.

Increasing the number of recycling trips also clearly improves the material utility achieved by the deposit-based waste management cases B and C, with CR (case C) providing the best result. Hence, combining MR and CR could be a promising compromise between the material utility and the GWP. CR could upgrade polymers that have been degraded through MR. This would allow for further recycling trips and thus contribute to keeping the polymers longer in use and avoiding primary plastic production.

The cradle-to-grave net GHG emissions of a 250 mL PEF bottle are 50-74% lower than the ones of an equivalent PET bottle after one end-of-life trip, depending on the waste management case. The combined effect of biomass use and material savings (46%) is the key reason behind the 50% lower cradle-to-gate GHG emissions of the 250 mL bio-based PEF bottle compared to its PET equivalent (26% when ignoring the biogenic carbon uptake during biomass cultivation). The lower weight and carbon content of the PEF bottle reduces its emissions at the EoL, as less process energy is required and less carbon emitted during incineration. Moreover, a PEF bottle offers a longer shelf-life, which could increase its material utility.

All analysed recycling cases are clearly improving the net GHG emissions when compared to the full incineration of bottle waste. Recycled PEF is expected to replace primary PET production in the initial years. With higher PEF market shares, closed-loop PEF to PEF recycling could be established, further increasing the net GHG benefit of PEF recycling.

Our results showed the importance of the EoL in the overall cradle-to-grave gross emissions of PET and PEF bottles. Depending on the case, the EoL has a share of 15-35% in gross cradle-to-grave emissions of PET bottles and a share of 11-25% for PEF bottles. For the recycling cases (A-C), these shares increase further the more recycling trips we assume. However, also the benefits of recycling could increase the more recycling trips are achieved. This could even result in cumulative net negative GHG emissions after 2-8 recycling trips for some cases when accounting for the replaced primary polymer production.

This also highlights the importance of enabling recycling for new bio-based polymers like PEF. If PEF would be recycled less than PET or even incinerated, the cumulative net GHG emission advantage of PEF compared to PET diminishes or could even turn in favour of PET after multiple recycling trips. Only if we achieve the same recycling performance for PEF as for PET do we maintain the full advantage of PEF compared to PET in terms of GHG emissions.

While our work provides valuable insights into the net GHG emissions and material utility of different waste management cases for small PET and PEF bottles over multiple recycling trips, our results have to be used with caution, due to data limitations (see discussion in Appendix C), modelling choices (e.g., allocation), and the specificity of the analysed product systems. Updated assessments would be worthwhile, once industrial data on the upcoming chemical recycling of PET in the Netherlands is available, the impact of the recently introduced deposit systems in the Netherlands is known, and once there is more evidence on the behaviour of PEF in mechanical and chemical recycling systems.

Nevertheless, there are several key lessons from this assessment. The LCA of nova institute comparing PEF and PET bottle types already showed that application matters when comparing different plastics: PEF performs best in applications that require high barrier properties and light weight. Overall, switching from PET to PEF is a robust strategy to reduce the GHG emissions of small plastic bottles. Our analysis concludes that also end of life and the amount of recycling trips matter. Only a *circular* bioeconomy, i.e., integrating PEF bottles into the recycling systems, ensures that we can sustain the GWP advantage of PEF bottles. Moreover, our results show that extending the use of deposit systems could significantly reduce the GHG impact of both PET and PEF bottles and increase their material utility, particularly over multiple recycling trips. Especially CR benefits from increasing the number of recycling trips, while MR already achieves 85% of its cumulative net GHG emissions savings after 5 trips (case B) or even close to 100% for case A. Also, it is important that new plastic materials maintain their quality also over multiple recycling trips, otherwise, their overall cradle to grave performance compared to existing materials worsens. And lastly, policy goals matter: When aiming for material utility alone, CR prove to be the superior option in our study. If your goal is climate change mitigation, our results favour deposit-based MR in most cases. These trade-offs between MR and CR could be overcome by combining the two waste treatment options to achieve the goals of both the circular economy and climate change mitigation alike.

Acknowledgements

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5.5 APPENDIX A: SUPPORTING INFORMATION FOR THE METHOD SECTION

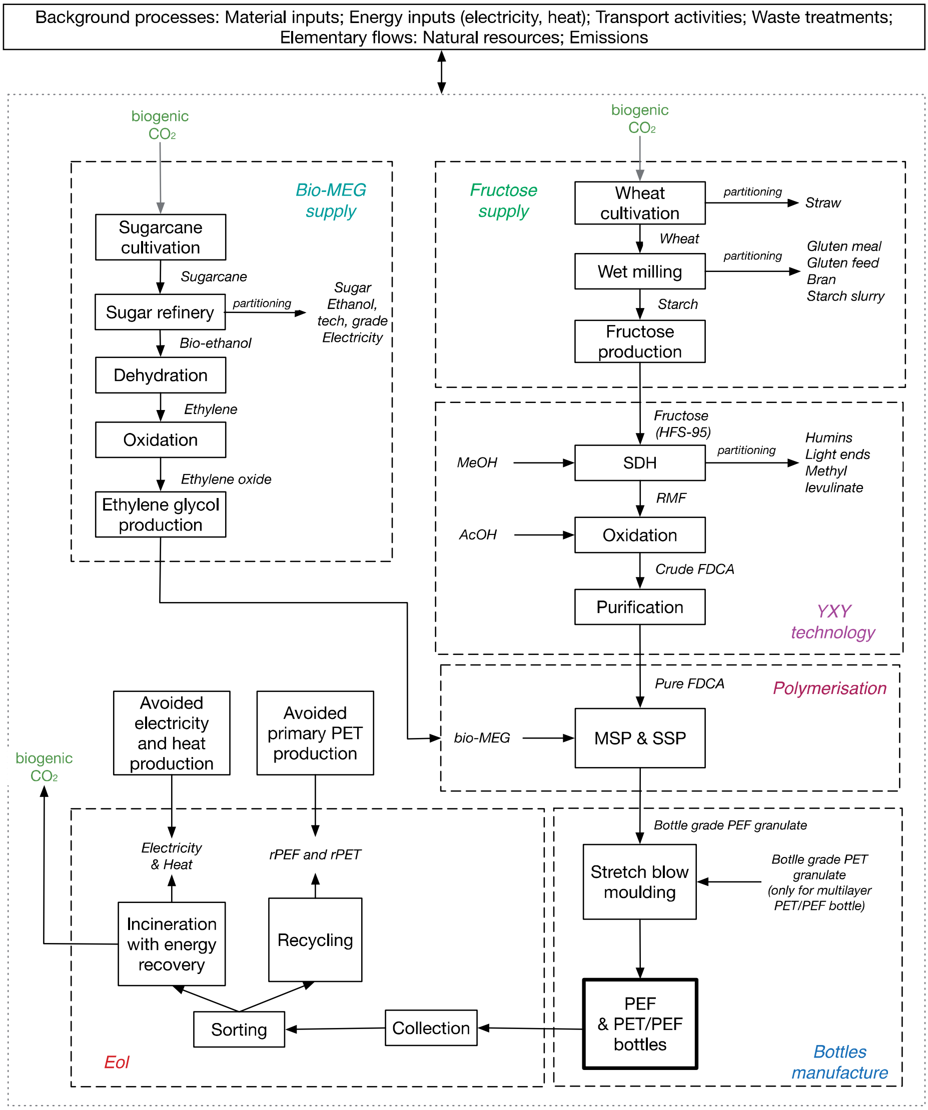

Figure 5.7: Product system of PEF bottles (taken from Puente and Stratmann 2021)

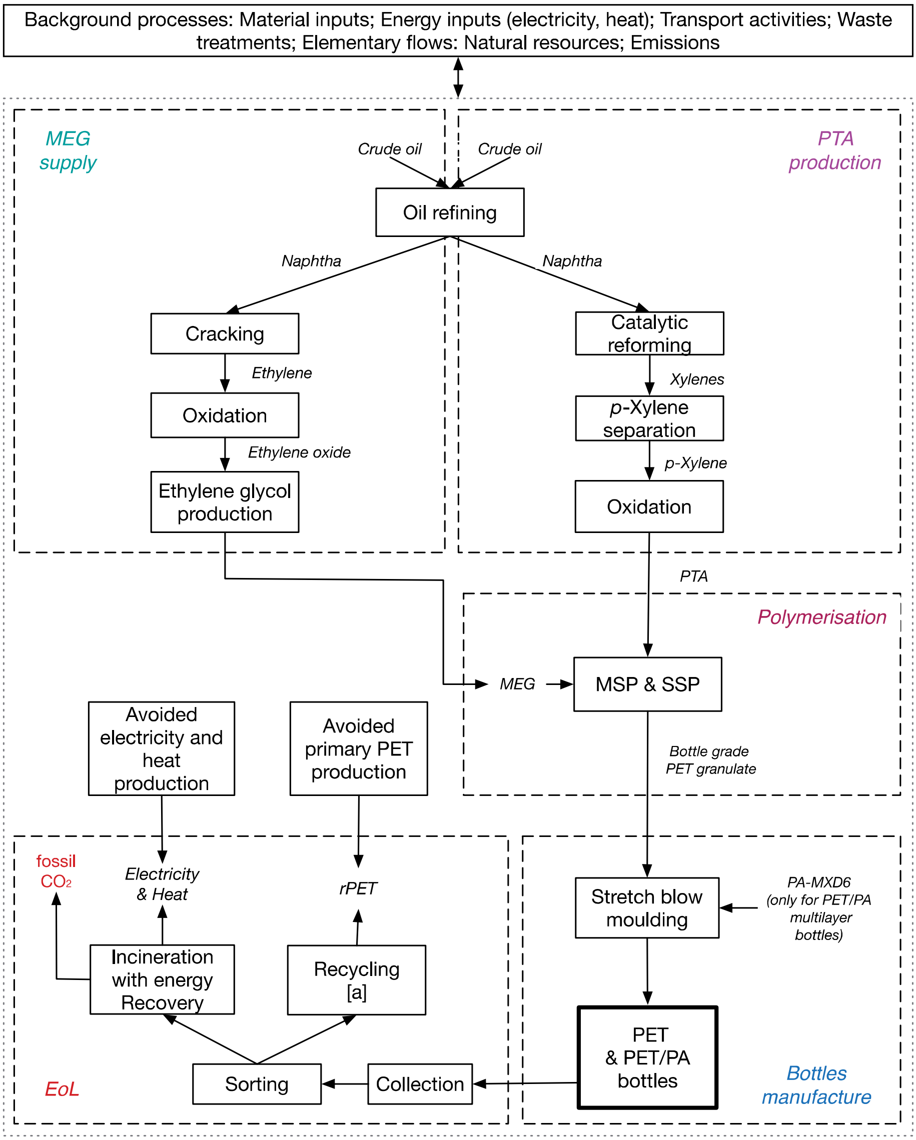

Figure 5.8: Product system of PET bottles (taken from Puente and Stratmann 2021);

Notes: for simplicity, links to background processes and co-production were removed from the flowchart

Table 5.1: Data and assumptions for modelling PET and PEF bottle production and waste treatment

Process	Mass or energy Efficiency	Data	Adjustments
Bottle production			
Virgin PET bottle-grade granulate production	-	LCA results from nova-Institute [220,233], which are based on 'PET, bottle-grade, at plant/RER' (Ecoinvent 3.6)	The polymerisation step is adjusted for the Dutch electricity and heat mix.
Solid State Poly-condensation (SSP)	-	Based on CPME (2017)	Improves amorphous granulate with intrinsic viscosity (IV) of ≈0.6 into bottle- grade granulate quality (IV≈0.82). Energy requirements for SSP were taken from CPME[146].
Virgin PET amorphous granulate production	-	Based on 'PET, bottle-grade, at plant/RER' (Ecoinvent 3.7)	We subtracted the SSP step from the bottle-grade granulate production dataset to model amorphous granulate production.
Virgin PEF bottle-grade granulate production	-	LCA results from nova-Institute [220,233]	-
Virgin PEF amorphous granulate production	-	LCA results from nova-Institute [220,233]	We subtracted the SSP step from the bottle-grade granulate production data to model amorphous granulate production.
Stretch-blow moulding		Based on 'Stretch-blow moulding {RER}| production | Cut-off, U' (Ecoinvent 3.7)	Dutch electricity production mix and heat source is used.
Treatment of bottle waste			
Sorting of source separated PET/PEF plastic waste	76% [234]	Based on 'Polyethylene terephthalate, for recycling, sorted {CH}| treatment of waste polyethylene terephthalate, for recycling, unsorted, sorting | Cut-off U' (Ecoinvent 3.7)	Dutch electricity production mix and heat source is used. Treatment of process losses is removed from this process and modelled separately. Sorting efficiency is taken from Brouwer et al.[234].
Sorting of post-separated PET/PEF plastic waste	70% [234]	Based on 'Polyethylene terephthalate, for recycling, sorted {CH}| treatment of waste polyethylene terephthalate, for recycling, unsorted, sorting | Cut-off U' (Ecoinvent 3.7)	Dutch electricity production mix and heat source is used. Treatment of process losses is removed from this process and modelled separately. We added a pre-treatment recovery step, which was assumed to have a four times higher energy requirement compared to the standard sorting process of ecoinvent (based on Bergsma et al. 2011). Sorting efficiency is taken from Brouwer et al.[234].

Process	Mass or energy Efficiency	Data	Adjustments
Processing of baled PET/PEF bottle waste into bottle-grade PET/ PEF granulate at factory gate	88% [234]	Based on 'Polyethylene terephthalate, granulate, bottle-grade, recycled {CH}| polyethylene terephthalate production, granulate, bottle-grade, recycled | Cut-off, U' (Ecoinvent 3.7)	Dutch electricity production mix and heat source is used. Treatment of process losses is removed from this process and modelled separately. Recycling efficiency is taken from Brouwer et al.[234].
Processing of baled PET/PEF bottle waste into amorphous PET/ PEF granulate at factory gate	88% [234]	Based on 'Polyethylene terephthalate, granulate, bottle-grade, recycled {CH}| polyethylene terephthalate production, granulate, bottle-grade, recycled | Cut-off, U' (Ecoinvent 3.7)	We subtracted the SSP step from bottle-grade recycled PET data to model amorphous recycled granulate. Dutch electricity production mix and heat source is used. Treatment of process losses is removed from this process and modelled separately. Recycling efficiency is taken from Brouwer et al.[234].
Processing of baled PET/PEF bottle waste into bottle-grade PET/PEF granulate at factory gate via glycolysis and repolymerisation	98 % [226]	Based on Shen et al.[226].	Non renewable energy use (NREU) of glycolysis and repolymerisation process was taken from Shen et al.[226]. Adjusted to our system boundaries by subtracting energy requirements for fibre spinning and finishing and adding SSP. The remaining NREU use was fully allocated to heat from natural gas with an assumed efficiency of 85%.
Incineration of PET/ PEF bottle waste	20% electricity and 23% heat[249]	Based on 'Waste polyethylene terephthalate {CH}| treatment of, municipal incineration with fly ash extraction | Cut-off, U' (Ecoinvent 3.7)	Dutch electricity and heat generation efficiencies were used[249]. Considers the differences in carbon content and heating value between PET and PEF.

Table 5.2: Details on transport distances within the waste management cases

Route	Distance (km)	Mean of transport	Database used	Adjustments
Municipal waste to mechanical recovery facility	35	Collection vehicle	Municipal waste collection service by 21 metric ton lorry {CH}| market for municipal waste collection service by 21 metric ton lorry | Cut-off, U (*Ecoinvent 3.7*)	Average distance calculated on multiple municipalities – transhipment station distances [241]
Source separated packaging plastics to transhipment station	35	Collection vehicle	Municipal waste collection service by 21 metric ton lorry {CH}| market for municipal waste collection service by 21 metric ton lorry | Cut-off, U (*Ecoinvent 3.7*)	Average distance calculated on 19 transhipment stations from SITA and 6 other remaining collection companies [241]
Deposit packaging via a distribution center to the PET processer	75	Truck	Transport, freight, lorry 16-32 metric ton, EURO5 {GLO}| market for | Cut-off, U (*Ecoinvent 3.7*)	Based on distance to 3 to 4 deposit PET processors within the Netherlands [241]
Deposit packaging to chemical recycling plant	225	Truck	Transport, freight, lorry 16-32 metric ton, EURO5 {GLO}| market for | Cut-off, U (*Ecoinvent 3.7*)	Based on average distances from the middle of the Netherlands (Amersfoort) to the locations of the current pilot plants by Ioniqua (Geleen) and CURE (Emmen) (Own calculation)
Material from facility to AEC	40	Truck	Transport, freight, lorry 16-32 metric ton, EURO5 {GLO}| market for | Cut-off, U (*Ecoinvent 3.7*)	Transportstion distance based on Corsten et al.[255]
Source separated packaging plastics to the sorting facility	170	Truck	Transport, freight, lorry 16-32 metric ton, EURO5 {GLO}| market for | Cut-off, U (*Ecoinvent 3.7*)	Weighted average distance in the Netherlands. Weighting done based on material amounts [241]
Post separated packaging plastics to the sorting facility	230	Truck	Transport, freight, lorry 16-32 metric ton, EURO5 {GLO}| market for | Cut-off, U (*Ecoinvent 3.7*)	Average distance from MRF to processors [241]
Sorted plastics to the processor and producer	200	Truck	Transport, freight, lorry 16-32 metric ton, EURO5 {GLO}| market for | Cut-off, U (*Ecoinvent 3.7*)	Average distance of source separated and post separated plastics back to the Netherlands. It is assumed here that the recycled plastic are processed and used in the Netherlands [241]

Explanation of equation variables

The following variables are used in the equations 1-3.

k	referring to the bottle types (PEF or PET)
i	referring to the waste management cases A-C
t	referring to the number of recycling trip (1-15). t=0 refers to the initial bottle production.
n	referring to the maximum number of recycling trips per waste management case
$m_{k,i}$	weight in grammes of the bottle (t=0) or the remaining bottle material (t= 1 to 15) per bottle type (k) and waste management case (i)
r_i	Overall recycling rate in % per waste management case (i). It is the product of the sorting and recycling efficiencies.
$b_{k,i}$	net GHG emissions in gramm of one recycling trip per bottle type (k) and waste management case (i)
CC_k	Carbon content per bottle type (k) in %
PE_k	net GHG emissions in gramm of cradle-to-gate bottle production per bottle type (k)

5.6 APPENDIX B: SUPPORTING FIGURES FOR THE RESULTS SECTION

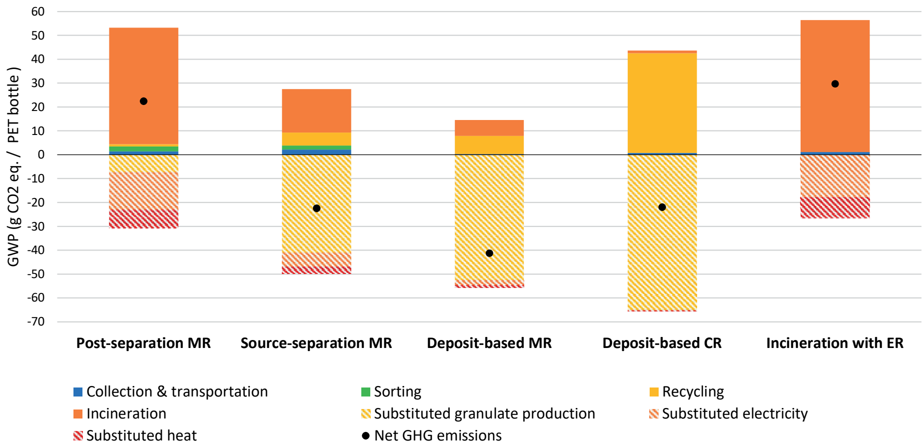

Figure 5.9: GHG emissions of the waste treatment trajectories for one PET bottle

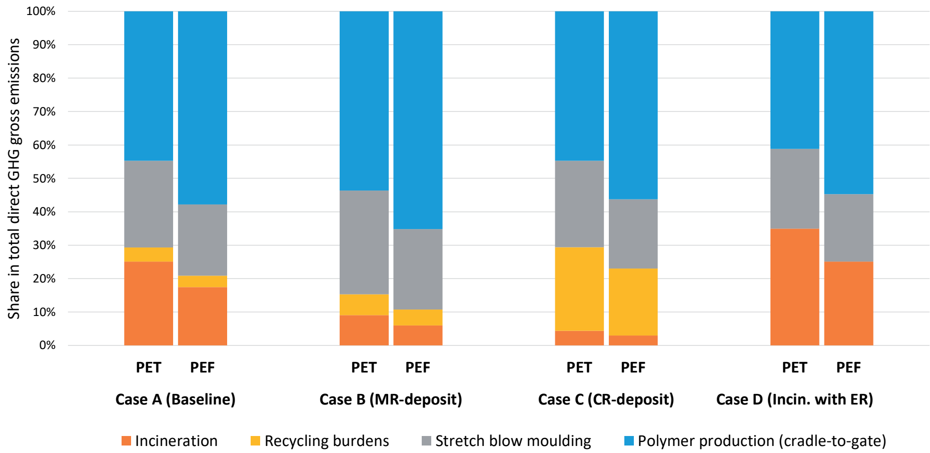

Figure 5.10: Contribution analysis of direct cradle-to-grave gross GHG emissions

Notes: excl. biogenic carbon uptake, excl. credits for substituted polymers and energy

5.7 APPENDIX C: DISCUSSION OF SENSITIVITIES

A key driver for the results is the overall recycling rates achieved by the waste management cases. Since the recycling efficiencies are already high (88% for MR, 98% for CR), the collection and sorting system will play a decisive role in the future performance of the PET and PEF recycling system in the Netherlands. The shift to a deposit system for small plastic bottles will likely have a substantial impact as our results indicated. However, it is still uncertain how efficient the deposit system in the Netherlands will be, given the initial problems observed since its start [256]. Furthermore, Dutch municipalities are increasingly shifting from source separation to post-separation [257,258] and the impact of this change on the recycling rate still has to be analysed.

Moreover, the substitution factor plays a significant role, as it affects the amounts of credits received for replacing primary plastic granulates. Defining the exact substitution factor of mechanically recycled plastics compared to primary plastics is difficult and there are different values reported in literature. Reducing the substitution factor of MR from 0.9 to 0.75 worsens the performance of Case B and would just tip the scale in favour of chemical recycling for a closed-loop system of PEF after one recycling trip. However, for PET recycling the order stays the same and for open-loop PEF recycling it only shifts after 11 recycling trips.

We chose an analysis scope of 15 recycling trips to showcase the benefits of CR. Choosing more recycling trips would further benefit the results for CR. Similarly, if PET or PEF bottles could be mechanically recycled more than ten times, it would improve the results of MR compared to CR. However, both changes would not impact the overall conclusions of our study.

The largest uncertainties regarding the analysed technologies relate to CR. The small-scale production data from Taiwan we used might differ from the technologies being developed in the Netherlands. In general, we expect an improved performance of CR compared to mechanical recycling in the future, as the potential for technological learning is higher compared to the already mature MR technologies.

The Dutch electricity production mix is shifting increasingly to renewables[259]. This will reduce the GHG emission advantage of PEF compared to PET, as PET requires more electricity for bottle production due to its higher material use. Nevertheless, we do not expect that this would substantially change the conclusions of this study as also PEF benefits from a greener electricity mix and the key difference in net GHG emissions relate to the feedstocks used.

A greener electricity and heat mix would further worsen the net GHG emissions of the cases largely depending on incineration (cases A and D), as the benefits from the corresponding electricity production would be lower. More renewables in electricity production would mainly benefit MR, as CR relies mostly on heat. However, a switch from natural gas as process energy for CR to alternative heat sources like green hydrogen could significantly reduce the GHG impact of CR.

Changing the allocation method for the end-of-life from the avoided burden approach to economic allocation, increases the net GHG emissions results throughout all waste management cases but does not change their order (see Figure 5.11). However, over multiple recycling trips the cumulative net GHG emissions of the deposit-based scenarios would no longer decrease when applying economic allocation.

In reality, there are varying PET bottle weights on the market, depending on bottle design. Decreasing our chosen PET bottle weight of 24 g would reduce the GHG emission advantage of the PEF bottle. But only when the PET bottle weight would be reduced by more than half while the weight of the PEF bottle stays the same, the net GHG emissions would speak in favour of the PET bottle after one recycling trip. However, such a low weight would no longer meet the requirements of our functional unit.

Lastly, it still remains to be seen how well PEF can be integrated into the current recycling system. Following the approval of EPBP[237], we assume that PEF could initially be recycled together with PET, but there might be unforeseen practical hurdles that hamper PEF sorting and recycling. Also, the behaviour of PEF in chemical recycling still needs to be assessed, which could either lower or increase emissions of PEF recycling compared to PET.

The economic allocation of incineration and recycling emissions was based on data from Gradus et al.[160] and on prices of PET recyclates on Ali Baba (22.04.2022).

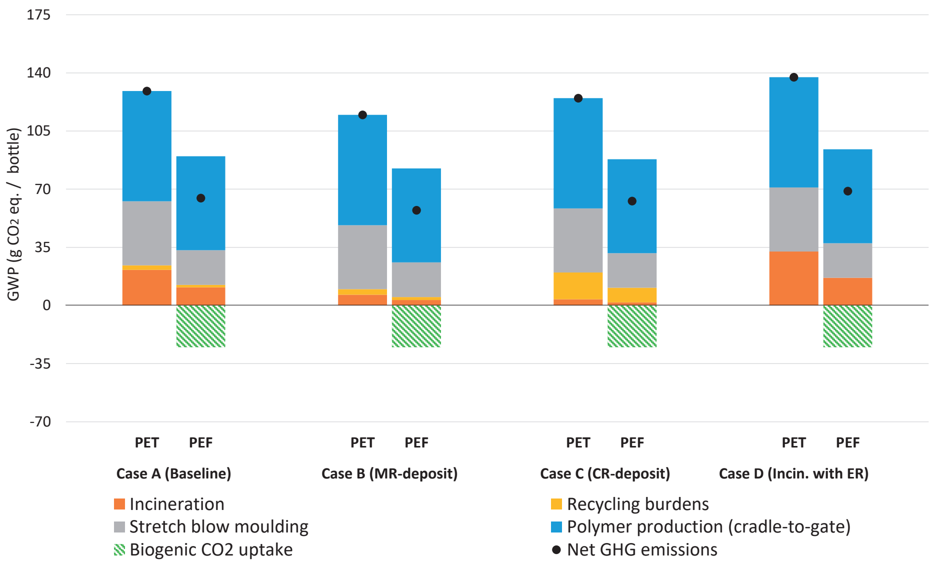

Figure 5.11: The cradle-to-grave net GHG emission for one bottle after one recycling trip with economic allocation

Chapter 6

Summary & Conclusions

6.1 SUMMARY

Plastics have become an essential part of our economy. Their production increased from 2 Mt in 1950 to 380 Mt in 2015, making plastics the bulk material with the highest production growth globally [4,6]. In 2015, the plastics sector was responsible for 4.5% of the global GHG emissions[8]. Biomass use and recycling are two options that can significantly reduce both fossil feedstock use and the GHG emissions of the plastics sector[4,9,15–17]. Together, biomass use and recycling could contribute to a circular bioeconomy (CBE) for plastics, potentially even achieving negative CO_2 emissions by sequestering biogenic carbon in plastic products. The circular bioeconomy evolved from an increasing link between the concepts of a circular economy (CE) and a bioeconomy since the EU action plan for the CE of 2015[28] and the updated European bioeconomy strategy of 2018[54].

The overall goal of this research was to understand to what extent a CBE could reduce the GHG emissions and resource consumption of the plastics sector. We approached this issue by combining several methods and perspectives:

- A quantitative and qualitative literature review combined with interviews to provide insights into industry developments and scientific literature regarding the CBE concept. (Chapter 2)
- Integrated assessment modelling, investigating the long term supply and demand dynamics in the global plastics sector, and their implications on GHG emissions across different scenarios. (Chapters 3 and 4)
- A life-cycle-based assessment of small plastic bottles. This product-focused case study provided a bottom-up perspective of a CBE for plastics. (Chapter 5)

Together, these methods and perspectives help highlight the benefits, trade-offs, and challenges of moving towards a circular bioeconomy for plastics. In particular, they provide answers to our four research questions:

1. What are the defining characteristics of a CBE and what is its current role in European bioeconomy clusters?
2. How can the impact of a CBE on GHG emissions, resource consumption and circularity be adequately assessed?
3. What are potential developments of global plastic production, product stocks, waste generation and the related resource consumption and CO_2 emissions until 2100 following current policy trends?
4. What are promising plastic production and waste management strategies to reduce the GHG emissions and resource consumption of the plastics sector, including a CBE?

Chapter 2, defines the term CBE via a literature review and analysed the concept's role in northwest European bioeconomy clusters through interviews. It identifies strategies regarding the clusters' feedstock and product focus and investigated what role biorefineries, circular solutions, recycling and cascading play. Finally, the chapter discusses gaps in CBE literature and the potential contributions of the CBE to sustainability. The analysed bioeconomy clusters move towards a CBE by increasingly considering residues and wastes as a resource, developing integrated biorefineries and focusing more on material and high-value applications of biomass. However, there is so far only little focus on the end-of-life of bio-based products, i.e., on circular product design, recycling and cascading. Key challenges for implementing circular strategies are policies and regulations, costs and the current small size of bio-based markets. Amongst the product sectors the interviewees identified as promising for the bioeconomy, plastics and construction & building materials have the most recycling and cascading potential. While the CBE could contribute to improving the sustainability of the bioeconomy, the concept is not inherently sustainable and its potential trade-offs need to be addressed. Especially social aspects, cascading, circular product design, and aspects related to product use seem to be underrepresented in CBE literature, while the topics of biorefinery, wastes and residues as well as waste management are significantly covered.

Chapter 3 tackles the problem that Integrated Assessment Models (IAM), which study the interlinkages between human and natural systems and play a key role in assessing global strategies to reduce global warming, largely neglect the role of plastics and the circular economy. This chapter presents the Plastics Integrated Assessment model (PLAIA), which adds plastic production, use, and end-of-life to the IMAGE IAM. PLAIA models the global plastics sector and its impacts up to 2100 for 26 world regions, providing a long-term, dynamic perspective of the sector and its interactions with other socioeconomic and natural systems. Chapter 3 summarises the model structure, mathematical formulation, assumptions, and data sources. The model links the upstream chemical production with the downstream production of plastics, their use in different sectors, and their end of life. Therefore, PLAIA can assess material use and emission mitigation strategies throughout the whole life cycle in an IAM, including the impacts of the circular economy on mitigating climate change. PLAIA projects plastics demand, and production pathways and specifies the annual plastic waste generation, collection, and the impact of waste management strategies. It shows the fossil and bio-based energy and carbon flows in product stocks, landfills, and the emissions in production and at the end of life.

Chapter 4 uses the PLAIA model to analyse three alternative CO_2 emission mitigation pathways for the global plastics sector until 2100, covering the entire life-cycle from production to waste management. If no new policies are implemented, PLAIA projects a doubling of global plastic demand by 2050 and more than a tripling by 2100, with an almost equivalent increase in CO_2 emissions in the baseline scenario. The results show that through bio-based carbon sequestration in plastic products, a combination of biomass use and landfilling can achieve negative emissions in the long term; however, this involves continued reliance on primary feedstock and ignores other ecological and social implications of expanding landfilling multiple times over today's capacity. A circular economy approach without an additional bioeconomy push reduces resource consumption by 30% and achieves 10% greater emission reductions before 2050 while reducing the potential of negative emissions in the long term. A circular bioeconomy approach combining recycling with higher biomass use could ultimately turn the sector into a net carbon sink while phasing out landfilling and reducing resource consumption. This work improves the representation of material flows and the circular economy in global energy & emission models and provides insight into the long-term dynamics in the plastics sector.

After looking at potential long-term developments of the global plastics sector in Chapters 3 and 4, **Chapter 5** dives into a case study on small plastic bottles. This chapter explores the GHG emissions and material utility of different waste management strategies for the bottles and discusses trade-offs between climate change mitigation and circular economy targets. The chapter assesses the life-cycle global warming potential (GWP) and the material utility (MU) for 250 mL PET (polyethylene terephthalate) and bio-based PEF (polyethylene furanoate) bottles over multiple recycling trips based on mechanical (MR) and chemical recycling (CR) in the Netherlands. The results show that bio-based PEF would offer 50-74% lower life-cycle GHG emission compared to PET after one end-of-life trip, depending on the waste management case. Furthermore, this chapter concludes that deposit-based recycling systems significantly reduce the cumulative cradle-to-grave net GHG emissions for both bottle types, especially when multiple recycling trips are applied. It could be seen that only a *circular* bioeconomy strategy, i.e., integrating the bio-based PEF bottles into the recycling systems, ensures that we can sustain the GWP advantage of PEF bottles. Moreover, Chapter 5 reveals trade-offs between GWP and MU: While deposit-based CR shows the best performance in terms of MU, it falls behind deposit-based MR when it comes to net GHG emissions due to the energy intensity of chemical recycling. Hence, combining mechanical and chemical recycling could contribute to achieving the goals of the circular economy and climate change mitigation alike.

6.2 THE DEFINING ELEMENTS OF A CBE AND ITS ROLE IN EUROPEAN BIOECONOMY CLUSTERS (RESEARCH QUESTION 1)

Despite the rising attention to the concept of a circular bioeconomy (CBE) since 2015, there is a limited understanding of what the CBE entails. Chapter 2 identified the defining characteristics of a CBE based on a literature review and a quantitative analysis of keywords used in scientific publications.

There were three perspectives on a CBE observed in literature: firstly, the CBE as part of a circular economy (CE); secondly, the CBE as an intersection of the CE and the bioeconomy; and lastly, a perspective that sees the CBE as more than the CE and the bioeconomy alone. While the perspectives differ, the literature largely refers to the same CBE elements, among them the *use of wastes and residues as a resource*, *sustainability*, *resource efficiency* and the use of *integrated biorefineries*. Moreover, they see the *cascading use of biomass* as an element of the CBE, which can be defined as the processing of biomass into a bio-based final product which is used at least once more for material or energy purposes[80]. Also, all publications consider *recycling and other circular waste management strategies* as part of the CBE. Other circular strategies like *circular or durable product design* and *shared and prolonged product use* were less prominent in the analysed literature. Furthermore, *social aspects* seem to fall short in the CBE discourse. Based on these identified CBE characteristics, Chapter 2 defined the CBE as follows:

The circular bioeconomy focuses on the sustainable, resource-efficient valorisation of biomass in integrated, multi-output production chains (e.g. biorefineries) while also using residues and wastes and optimising the value of biomass over time via cascading.

Such optimisation can focus on economic, environmental or social aspects and ideally considers all three pillars of sustainability, i.e., economic, environmental, and social aspects. The cascading steps aim at retaining the resource quality by adhering to the bio-based value pyramid and the waste hierarchy where possible and adequate. This implies shifting to more material uses of biomass instead of direct energy use, a trend also observed in the analysed literature and industry clusters.

Regional industry clusters are an important driving force of the European bioeconomy[57,58]. Such bioeconomy clusters consist of interconnected stakeholders working in the bioeconomy field in a particular region, such as farmers, manufacturers, industrial associations, research institutions and governmental bodies. Chapter 2 investigated how far the CBE concept already plays a role in Northwest European bioeconomy clusters via interviews and a literature review.

The analysed bioeconomy clusters move towards a CBE by increasingly considering residues and wastes as a resource, developing integrated biorefineries and focusing more on material and high-value applications of biomass. However, there is so far only little focus on the end-of-life of bio-based products, i.e., on circular product design, recycling and cascading.

As key challenges for implementing circular strategies the interviewed cluster representatives mentioned impeding policies and regulations, costs and the current small size of bio-based markets. Amongst the product sectors that the interviewees identified as promising for the bioeconomy, plastics and construction & building materials have the most recycling and cascading potential.

6.3 METHODS TO ASSESS THE IMPACT OF A CBE ON GHG EMISSIONS AND RESOURCE CONSUMPTION (RESEARCH QUESTION 2)

LCA can be a valuable tool to assess the GHG emissions of plastic products in a circular bioeconomy, but only when the end-of-life is extensively considered.

A comprehensive assessment of the end-of-life of bio-based plastics is currently missing in most LCAs of these materials. The results in Chapter 5 showed the substantial impact the end-of-life has on the overall net GHG emissions of the analysed plastic bottles and revealed significant differences between varying waste treatment strategies, particularly in scenarios where the plastics are recycled multiple times. This highlights the need to consider various waste treatment strategies in LCAs and more than one recycling trip (if these can occur in practice). Only then can we provide a complete picture of the net GHG emissions of a product over its entire life cycle.

An LCA should be complemented by an additional analysis focusing on a product's circularity and waste management strategies. Chapter 5 proposes the concept of material utility for this purpose.

The complimentary assessment of the material utility of a product revealed synergies and trade-offs between climate change mitigation and circular economy targets for different waste treatment strategies (e.g., between mechanical and chemical recycling) that conventional LCA indicators like fossil fuel depletion could not provide. The material utility consists of the material use intensity, which assesses how much material is cumulatively recycled and used after multiple end-of-life trips and the length of a material's use, showing how long the material is kept in the technosphere. These indicators could be used to assess how much primary production and related GHG emissions and resource consumption could be avoided.

In principle, these circularity indicators could be combined with LCA indicators. But to avoid subjective weighing when combining them, we suggest to simply reporting them as complementary information.

But even when extending LCAs as done in Chapter 5, the results only provide a snapshot of the product's performance under specific circumstances. Hence, conclusions stay specific to the analysed product, technology, time and geography and cannot provide guidance on overall, long-term strategies for the plastics sector.

It is impossible to fully understand the climate change mitigation potential and the trade-offs of a circular bioeconomy for the plastic sector without analysing the global, long-term trends and the sector's interactions with other socioeconomic and natural systems. However, none of the integrated assessment models, which represent climate and socioeconomic interactions, used for the IPCC reports has included a detailed representation of the plastics sector.

Integrated Assessment Modelling is a powerful tool to assess long-term developments of the plastics sector and its aggregated impacts. IAMs can provide guidance on promising mitigation pathways for the plastics sector, such as a CBE.

Chapter 3 presented the Plastics Integrated Assessment model (PLAIA), which covers the whole world in 26 regions and the entire life-cycle of plastics, from the upstream chemical production to the downstream production of plastic polymers, their transformation into plastic products, their use in different sectors, and their end of life. As part of the integrated assessment model IMAGE, PLAIA interacts with the energy and agricultural sectors and with the climate, water, and land systems. This model can assess global, long-term dynamics and the impact of different mitigation strategies on the sector's global GHG emissions and resource consumption.

However, modelling plastics in a global IAM until 2100 requires simplifications; a detailed representation of the global plastics sector is not feasible nor desirable for such a long-term analysis. Simplifications such as technology aggregation and the use of regional data reduce the granularity of the results. Hence, PLAIA cannot provide country- and technology-specific recommendations.

Moreover, region-specific data on the plastics sector and waste management is not consistently available for all world regions, forcing modellers to use proxy values and assumptions. In particular, the flows in the upstream chemical sector are poorly understood. Furthermore, waste management data is very limited in most world regions, which can partly be explained by the large role of informal waste management and by inconsistent definitions of key terms like the recycling rate. While the resulting inaccuracies make precise assessments difficult, IAMs can still provide valuable insights into the plastics sectors' overall dynamics and the relative performance of different future pathways, consistent with global energy and land-use systems geared towards different climate targets. Thus, using scenario-analysis, they can help answer *what-if* type questions, investigating the resource demand and consequent emissions of different technology, emission pricing, CE and CBE strategies in the plastics sector.

6.4 POTENTIAL DEVELOPMENTS IN THE PLASTICS SECTOR UNTIL 2100 FOLLOWING CURRENT POLICY TRENDS (RESEARCH QUESTION 3)

Using PLAIA, this thesis assessed potential future developments of plastic production, waste generation and plastic stocks in use in Chapter 4, differentiating between eight plastic sectors.

Without the implementation of new policies, PLAIA projects a doubling of global plastic production by 2050, reaching almost 1.1 billion metric tonnes (Gt), and more than a tripling by 2100 (1.75 Gt). In total, PLAIA projects that between 2020 and 2100 about 100 Gt of plastics will be produced.

The rising plastic production is driven by population and economic growth, with the highest growth happening in emerging and developing economies. While production patterns differ across the *shared socioeconomic pathways* (SSPs), the sensitivity analysis in Chapter 4 showed that cumulative plastic production between 2020 and 2100 was consistently high across the SSPs 1-3, staying within a range of 8% from the baseline SSP2. Most of the plastics are produced for packaging (37%), followed by building & construction materials (16%) and Textiles (11%)[6].

The annual waste generation follows the growth in plastic production, reaching almost 1 Gt in 2050 and 1.74 Gt by 2100. Waste generation is dominated by plastics from packaging and other product sectors with a short lifetime. In total, PLAIA projects that 92 Gt of plastic waste will be generated cumulatively until 2100. This highlights the importance of proper waste management to avoid littering and further plastic pollution.

PLAIA estimates the total plastic stocks in use in 2020 at almost 3.2 Gt, and this could rise to around 7.7 Gt in 2050 and to almost 15 Gt in 2100. Building & construction materials make up more than half of the plastics in use, due to their long product lifetime. Hence, this plastic sector seems most promising for biogenic carbon sequestration to achieve negative emissions.

PLAIA estimates the final energy use of the global plastics sector in 2020 at almost 36 Exajoule and the emissions at 2.2 Gt CO_2, which would equal around 7% of the global energy-related CO_2 emissions.

Following current trends, the plastics sector could almost double its emissions up until 2050, reaching its peak in 2090 with 5.7 Gt CO_2, which would nearly equal the total net US GHG emissions of 2019. The final energy use reaches 100 Exajoule in the 2090s.

A key reason for the rise in emissions in this baseline scenario is the sector's continued reliance on oil and the ongoing rise in coal use until 2030, mostly driven by China. Moreover, waste-to-energy becomes the dominating waste treatment technology in this baseline, which drastically increases the end-of-life emissions in the plastics sector.

6.5 STRATEGIES TO REDUCE THE GHG EMISSIONS AND RESOURCE CONSUMPTION OF THE PLASTICS SECTOR, INCLUDING A CBE (RESEARCH QUESTION 4)

Biomass use, recycling, and decarbonising the electricity supply promise significant GHG emissions reductions for the plastics sector[9,17]. Together, biomass use and recycling contribute to a CBE for plastics. Moreover, using sustainable biomass as feedstock could potentially achieve negative CO_2 emissions by sequestering biogenic carbon in plastic products[18].

This thesis used two perspectives and methods to understand better how these strategies could reduce GHG emissions and resource consumption in the growing plastics sector. Firstly, the PLAIA model provided insights into the strategies' global and long-term impacts (Chapters 3 and 4). Secondly, a life-cycle-based assessment of small plastic bottles made from bio-based polyethylene furanoate (PEF) compared to fossil polyethylene terephthalate (PET) was conducted, providing a case study of the CBE's impact on GHG emissions and circularity.

6.5.1 The impact of a global CO_2 price on the plastics sector

Firstly, Chapter 4 tested how a global CO_2 price in line with the Paris 2-degree climate target would affect the CO_2 emissions and resource consumption of the global energy & plastics sector. This exercise provided valuable insights into promising emission mitigation strategies for plastics but also revealed the limitations of a CO_2 price for the plastics sector.

The decarbonisation of the electricity supply would significantly reduce the GHG emissions of the plastics sector. A global CO_2 price leads to the decarbonisation of electricity production, reducing the emissions of the plastics sector by around 65 Gt CO_2 eq. or 18% cumulatively between 2020 and 2100 compared to the baseline. Hence, chemical and plastic producers could significantly lower their carbon footprint by procuring renewable electricity or producing it onsite.

The role of waste-to-energy will likely diminish on the way to a climate-neutral plastics sector unless it would be combined with carbon capture technologies. Introducing a price to all emitted CO_2 drastically reduces the share of waste-to-energy, as incineration causes emissions while replacing an increasingly greener heat and electricity mix (Chapter 4). Also, the case study in Chapter 5 showed that incinerating plastic bottles with energy recovery performs significantly worse in terms of GHG emissions than recycling bottle waste.

Negative emissions achieved by the sequestration of bio-based plastics in product stocks and landfills could significantly reduce the plastics sector's CO_2 emissions, even turning the sector into a net carbon sink by the end of the century. A CO_2 price for incineration emissions could lead to an increase in the landfilling of plastics. With rising CO_2 prices, landfilling plastic waste would become an attractive alternative, as it sequesters most plastics for centuries and is cheaper than other waste treatment technologies. At the same time, the CO_2 price is expected to induce increased biomass use in plastic production. Sequestering biogenic carbon in plastic products and landfills could lead to net CO_2 sequestration. Chapter 4 showed that such a scenario would reach more than 20% biomass share in plastic feedstocks towards the end of the century, allowing for more than 1.5 Gt negative CO_2 emissions annually towards 2100. At the same time, landfilling also prevents significant fossil emissions by storing fossil-based carbon in plastic products. Combined with a decarbonised energy supply, this scenario could turn the plastics sector into a net carbon sink by the end of the century.

However, such a strategy, with a strong focus on primary plastic production and landfilling, would face trade-offs with other sustainability goals. Due to its focus on primary plastic production, such a strategy maintains a high input of energy and materials. Moreover, it could exacerbate other negative environmental impacts caused by the extraction and production of these resources (e.g., land-use change, biodiversity loss, nitrogen emissions from biomass production), chemical and plastic production (e.g., particulate matter emissions), and landfilling (e.g., increased land use and microplastics in leachates).

Introducing a CO_2 price alone will not be enough to substantially reduce fossil feedstock use and resource consumption in the plastic sector. Unlike in the energy system, large parts of the carbon input in the plastics sector are not directly emitted but sequestered in products and thus not exposed to the CO_2 price. Even though a CO_2 price would phase out coal and reduce the use of oil by substituting them with biomass and natural gas, additional efforts are required for a more substantial change in the sector's feedstock use.

According to the PLAIA model, the CO_2 price alone only leads to a moderate increase in recycling while primary plastic production still dominates. Consequentially, the CO_2 price had almost no impact on the overall energy and resource use in the plastic sector when compared to the baseline scenario (see Chapter 4). Furthermore, the CO_2 price could cause some unwanted side-effects, by drastically increasing landfilling of plastics (see above).

6.5.2 The benefits of circular strategies

A circular economy for plastics could significantly lower the resource consumption in the plastics sector and reduce CO_2 emissions, particularly in the first half of the 21st century.

Chapter 4 showed that in combination with a CO_2 price, circular economy measures could reduce the final energy use (incl. for feedstocks) by almost a third until 2050 compared to a business-as-usual scenario and a scenario with a CO_2 price alone. Moreover, it would reduce the global net CO_2 emissions of the plastics sector by more than two-thirds compared to the baseline scenario until 2050 while also performing better than the scenario with a CO_2 price alone and its focus on primary production and carbon sequestration in landfills.

However, such a circular economy would require a paradigm shift that not only improves the collection and sorting of plastic waste but also phases out landfilling and includes fundamental changes in product design. Chapter 5 showed that shifting to a deposit system for plastic bottles would drastically improve the cradle-to-grave net GHG emissions of plastic bottles, particularly when recycled multiple times. Similarly, it also enhances the utility of the plastic by retaining more and higher quality plastic material over multiple recycling trips, thus replacing significant amounts of primary plastic production. Furthermore, the bio-based PEF bottle also highlights another important strategy to reduce plastic impact: making packaging more lightweight. Due to better barrier properties and a higher modulus, PEF bottles require less material to fulfil the same function as PET bottles.

Combining mechanical and chemical recycling could contribute to achieving the goals of the circular economy and climate change mitigation alike.

While mechanical recycling (MR) mostly showed better net GHG emissions results in the plastic bottle case study (Chapter 5), chemical recycling (CR, depolymerisation via glycolysis) retained more and higher-quality plastic material, thus better fulfilling the goals of the circular economy. MR degrades the plastic material with each recycling trip. CR could upgrade polymers that have been degraded through MR. This would allow for additional recycling trips and thus contribute to keeping the polymers longer in use and avoiding the need for primary plastic production.

Not all plastics can be economically mechanically recycled. Pyrolysis accepts a broader range of polymers and could thus complement mechanical and polymer-specific chemical recycling (e.g., glycolysis, as mentioned above). Hence, chemical recycling via pyrolysis played an increasing role in the circular economy scenario in Chapter 4. However, mechanical recycling will most likely stay the dominant recycling technology in a world with a growing CO_2 price unless chemical recycling significantly reduces its emissions from its high process energy use.

6.5.3 A circular bioeconomy for plastics

This thesis showed that both a bioeconomy and a circular economy strategy can reduce GHG emissions in the plastics sector, but that they also have trade-offs.

The long-term and global analysis with the PLAIA model, and the case study on plastic bottles showed that biomass use could significantly reduce GHG emissions in the plastics sector. However, both approaches also showed that a bioeconomy has limitations when not complemented by circular economy policies. While some researchers argue that a bioeconomy is "circular by nature" (see Chapter 2), it does not change the linear business as usual at the products' end-of-life. Moreover, substantial amounts of land might be required to produce biomass.

Only if the same recycling performance is achieved for bio-based products like PEF as for their fossil equivalents do we maintain the full advantage of bio-based products compared to their fossil equivalents in terms of GHG emissions. Chapter 2 showed that so far bioeconomy clusters had only a limited focus on the end-of-life of bio-based products. But without ensuring circular solutions at the end-of-life, bio-based products might risk losing their GHG emission benefit compared to their fossil equivalents. Chapter 5 revealed that if PEF bottles were recycled less than PET bottles or even incinerated, the cumulative net GHG emission advantage of PEF bottles compared to PET bottles diminishes or could even turn in favour of PET after multiple recycling trips. Moreover, the results showed that deposit-based recycling systems for PET bottles could achieve higher emission reductions than a mostly linear PEF bottle system. Furthermore, Chapter 4 showed that without additional circular economy measures, biomass use barely decreases the final energy use in the plastic sector.

A circular economy for plastics could reduce the potential of negative emissions through biogenic carbon sequestration and still relies on fossil fuels. Even in an optimised CE, the plastic sector still requires fossil fuels for primary plastic production if plastic demand keeps growing (see Chapter 4). This results in only minor reductions in total fossil fuel consumption in the plastics sector compared to 2020. Moreover, Chapter 4 showed that a CE could reduce the potential of negative emissions through biogenic carbon sequestration in the long term.

By combining circular economy measures and increasing biomass use, a circular bioeconomy strategy promises a compromise between the climate and circular economy targets and could turn the plastics sector eventually into a net carbon sink.

Chapter 4 showed that combining a CO_2 price with increasing recycling and biomass use could reduce the global net CO_2 emissions of the plastic sector by 75% until 2050, compared to the baseline, and ultimately turn the sector into a net carbon sink. The analysed CBE scenario increases the share of bio-based plastics to more than a third towards 2100 and thus cumulatively achieves around 55 Gt negative CO_2 emissions between 2020 and 2100, an amount that could be increased significantly by fostering biomass use in plastics even more. This is a notable amount of CO_2 sequestration compared to the range of 30–780 Gt negative CO_2 emissions through bioenergy with carbon dioxide capture and storage (BECCS) reported by IPCC mitigation pathways for reaching the 1.5°C Paris climate target. At the same time, a CBE strategy could help phase out landfilling and reduce the sector's final energy use by 30% until 2050 and 40% until 2100 compared to the baseline.

Similarly, Chapter 5 showed that a closed-loop recycling system for bio-based PEF bottles achieves significantly higher emission reduction than an equivalent system for fossil PET bottles or a more linear scenario for PEF bottles. At the same time, the deposit-based recycling systems could increase the utility of the plastic material by 300-500%, avoiding significant amounts of primary plastic production.

Chapter 4 showed that a circular bioeconomy strategy could increase biomass use for plastics up to 5.9 Exajoule (EJ) in 2050, equivalent to about 13% of the total current global bioenergy use[206], and increases further to 18.7 EJ by 2100 (Chapter 4). Using wastes and residues is prioritised in a CBE (Chapter 2). Biomass residues have significant potential and could provide 55 EJ/year by 2050, but they will not be sufficient to cover all biomass supply for plastics when considering bioenergy demand[109]. Cascading biomass use (Chapter 2) could mitigate the competition between biomass use for materials and energy, but significant demand for primary biomass will remain.

When ensuring sustainable biomass supply, the circular bioeconomy could help achieve both the goals of the Paris climate agreement and the circular economy. Nevertheless, a CBE alone will most likely not be enough to achieve a net-zero plastics sector within the first half of the 21st century. Chapter 4 already showed that the decarbonisation of the electricity and heat sector will have an important role to play. Additional measures could further reduce the GHG impact of the plastics sector, such as carbon capture and utilisation and the electrification of chemical production.

6.6 KEY LIMITATIONS

A fundamental limitation of this research is a European bias. Starting with Chapter 2, the analysis relied on a small and primarily European literature base and interviews with representatives of seven European bioeconomy clusters. Moreover, Chapter 5 focused on a particular Dutch product system. Even the global plastics modelling relied largely on European technology data. Hence, our conclusions should be interpreted within their context and not be generalised.

Another limitation that was present throughout all chapters is the limited data availability. Being a relatively new concept, there was a limited literature base on the CBE available for our research in Chapter 2. The plastics modelling suffered from the fact that the plastics and chemicals sectors are complex and that data availability for these sectors and waste management is insufficient and globally inconsistent. As explained in Chapter 3, this required some improvisation and use of proxy values when creating PLAIA and is also responsible for the European or sometimes American technology bias. Moreover, the assessment of non-CO_2 GHG emissions was not included for chemical processing in Chapter 4. In Chapter 5, an ex-ante LCA was conducted, relying on a range of proxy assumptions since PEF bottles are not on the market yet and their actual performance in waste management systems is unknown. Furthermore, chemical recycling technologies are still developing, and open-source data is barely available.

Nevertheless, the sensitivity analyses in the respective chapters showed that the conclusions prove to be relatively robust for most changes in input data and assumptions.

This research was limited to assessing GHG emissions, resource consumption and circularity. For a more comprehensive assessment of strategies towards a sustainable plastics sector, other environmental impacts would need to be considered, e.g., the effect of littering plastics or critical impacts related to biomass production such as biodiversity loss and eutrophication. As part of IMAGE, PLAIA includes the interaction and competition with the energy and agricultural sectors for resources but still lacks a more dynamic link with other relevant sectors such as buildings or the pulp & paper industry.

Lastly, each chapter's conclusions must be interpreted while keeping the capabilities of the respective assessment methods in mind. While the global plastics model (Chapters 3 and 4) has strengths in providing overarching guidance on promising mitigation pathways for the plastics sector, it cannot tell us the most suitable solution for a specific country or product. At the same time, the conclusions of the case study on plastic bottles in Chapter 5 are limited by its specific product, technology and regional focus and ignore the costs of the analysed scenarios. None of the results in this thesis necessarily present a realistic forecast of the plastics sector. However, they provide valuable insights into the relative performances of the analysed strategies to reduce GHG emissions and resource consumption.

6.7 RECOMMENDATIONS FOR FURTHER RESEARCH

It remains a crucial challenge to collect data in the complex chemicals and plastics sectors with all their intermediate production steps and many relevant actors along the life cycle. For modelling, the representation needs to sufficiently transparent and simple. With continued efforts toward the comprehensive digitalisation of industrial production, the collection and sharing of data should become more accessible. Having a transparent supply chain will also help with the acceptance of the controversially debated bio-based products.

To have consistent data across regions, it is essential to establish standard definitions and terminology, e.g., what is understood as a recycling rate. Chapter 2 revealed different understandings of key CBE concepts amongst practitioners. This calls for an alignment of CBE terminology. Chapter 2 also showed that future CBE research should focus more on social aspects, cascading, product design, and aspects related to product use (durability, sharing), which all seem to be underrepresented in current CBE literature.

The analysis presented in Chapters 3 and 4 is the first step towards better considering plastics and the circular economy in global energy and emission models. Clearly, the PLAIA model can still be improved regarding technology representation (see Chapter 3). Model improvements should focus on including trade, technological learning, and a better representation of regional specifics in technologies, costs, and policies (particularly for China, a rapidly growing plastic producer). Developing regional and more technology-specific plastic models would allow for a better representation of the diversity of local challenges and technologies.

Furthermore, improving the links of plastic models with other industry sectors would allow for an improved integrated assessment, including the competition of plastic materials with alternatives for specific applications (e.g., packaging made of plastics or paper/cardboard) and the use of wastes from other sectors (e.g., black liquor from pulp & paper production). For this purpose, IAMs are ideally suited if they extend their focus beyond energy use to include material flows. Also, combining sector-specific models could be beneficial and probably allow for a more technology-specific analysis.

Moreover, the trade-offs with other environmental impacts could be analysed for a more integrated assessment of strategies toward a sustainable plastics sector, particularly those relevant to biomass production like water use, eutrophication, and biodiversity loss. Furthermore, PLAIA could also be extended to assess plastic pollution, which could reveal synergies and trade-offs with a CBE strategy. However, for this purpose, trade needs to be incorporated into the analysis to understand better the plastic waste flows worldwide.

In general, further analysis of long-term dynamics in the plastic sector and its interactions with other industries and the environment is important to understand the broader impact and the effectiveness of specific mitigation measures for plastics. This should also include mitigation strategies not assessed here, like carbon capture and utilisation and the electrification of the chemical sector. Moreover, the impact of behavioural changes in reducing plastic demand deserves more attention. Future scenarios, e.g., SSPs, should better consider material flows in the economy and shifts in demand independent from population and economic growth (e.g., behavioural changes, de-growth movement). Also, the costs of the analysed mitigation strategies deserve more attention.

Sector-specific integrated assessment models like PLAIA could contribute to defining pathways and overall policy goals. However, they need to be complemented by more technology- and region-specific models and case studies developing specific recommendations for implementing these goals on a country and product level.

Hence, product-specific life cycle assessments are essential to understand the benefits and challenges of novel bio-based or circular products. Such assessments should also include a detailed analysis of the potential end-of-life scenarios to understand the cradle-to-grave emissions better. This work showed that the end of life could enormously impact the overall emissions, particularly if multiple recycling trips are considered. Moreover, further work should be dedicated to developing circular economy indicators that could be linked to life cycle assessments.

While an LCA cannot compete with IAMs in assessing long-term dynamics, there is still much room for improvement to make LCAs more dynamic. For example, by considering potential future developments in the sector and related systems affecting efficiency and energy mix. A promising pathway could be the combination of LCA databases with IAMs, which is already taken up in recent research[260].

6.8 RECOMMENDATIONS FOR POLICYMAKERS AND INDUSTRY

This work showed that fostering a circular bioeconomy for plastics could substantially contribute to lowering the GHG emissions and the resource consumption of the plastics sector. When implementing a CBE for plastics, both policy and industry must comprehensively consider climate and circular economy goals together to avoid potential trade-offs. The diversity observed in the bioeconomy clusters' circumstances (Chapter 2) highlights the importance of designing specific regional CBE strategies, considering the local strengths and weaknesses while avoiding "one size fits all" solutions.

Since a CO_2 price alone has a limited effect on the plastic sector, policymakers need to provide additional incentives for increasing biomass use and recycling in the plastics sector. This could be in the form of subsidies and quotas but would most likely also require a more targeted support of specific key steps and technologies needed for the transition towards a CBE for plastics. Moreover, the CO_2 price should also apply to waste incineration, which is currently excluded from the European Emissions Trading System (ETS)[261]. Chapter 4 showed that a rising CO_2 price could lead to an increase in landfilling. To prevent that, recycling needs to be fostered and landfilling discouraged via a ban or increased taxes.

Ensuring high sustainability standards in biomass production is key to a CBE strategy. Chapter 4 projected that the plastics sector will consume up to 5.9 Exajoule (EJ) biomass in a CBE scenario in 2050, which will increase further to 18.7 EJ by 2100. Following the CBE concept, policymakers and industry should prioritise the use of wastes and residues and foster cascading biomass use (Chapter 2) to meet this demand and mitigate the competition between biomass use for materials and energy. Nevertheless, due to the limited potential of residues and growing bioenergy demand[109], a significant demand for primary biomass will remain. To ensure a sustainable biomass supply, reliable certification systems must be adopted by all relevant countries to avoid a situation in which unsustainable biomass is entering an economy from countries with lower sustainability standards.

Furthermore, policies and research programs should focus more on product design and end-of-life strategies for bio-based products, as only a few initiatives address them within the analysed bioeconomy clusters (Chapter 2). Relatively low-hanging fruits are improvements in the collection and sorting of plastic waste. The sensitivity analysis in Chapter 4 showed that high collection rates are not just essential to fight plastic pollution but also to reduce GHG emissions. High-quality recycling requires clean and homogenous input streams. Chapter 5 showed that deposit systems would significantly reduce emissions and resource consumption in the plastic sector. Incentivising deposit systems and more streamlined collection and sorting systems across regions would provide high-quality input streams to recycling in the quantity needed for an economical recycling system.

Many plastic products are inherently not recyclable[204]. To achieve the high recycling rates of the circular bioeconomy scenario (Chapter 4), policy and industry thus also have to focus on product design to increase the overall recyclability of products. The interviewed bioeconomy cluster representatives (Chapter 2) suggested a better education of product designers and the introduction of regulations that prevent the use of materials in products that hamper recycling. Also, public procurement was seen as an important tool to foster the demand for circular, bio-based products.

Moreover, the development and implementation of chemical recycling technologies should be supported to complement mechanical recycling. They are important to upgrade deteriorated plastics and allow for more recycling trips. Technologies like pyrolysis could focus on plastics that are not suitable for MR.

Chapter 4 showed that waste-to-energy would not play a significant role in a CBE for plastics, with an increasingly decarbonised electricity and heat production. Consequently, policymakers should avoid a lock-in situation in which costly, long-term incineration capacity is built up that will not be favourable in the future.

Industry must consider the EoL of a product from the start. Chapter 5 showed that a bad performance at EoL could drastically reduce the relative GHG benefit achieved in production. This requires more cooperation along a product's life cycle, e.g. involving actors from the waste management sector already at the product design stage to ensure that the products are suitable for existing recycling systems. The importance of enhancing the cooperation along the supply chain and amongst regions has also been highlighted by the interviewed industry representatives in Chapter 2.

Industry can support the development of a CBE by (1) facilitating cooperation between stakeholders along and across supply chains; (2) fostering bio-based product design that facilitates durability, reuse, repair, recycling or biodegradability; (3) fostering the use of residues and wastes as resources; (4) intensifying the cooperation with the waste management sector to ensure that the bio-based products can be integrated into the collection, separation, recycling and composting schemes. By supporting such circular strategies and increasing the use of residues and waste, the industry could also reduce its dependency on global resource markets.

Lastly, better reporting of the flows in plastic production and waste management should be incentivised or mandated. Currently, the chemicals and plastics sector is amongst the most opaque compared to others like the steel industry[5]. Current efforts in establishing a monitoring system for the European bioeconomy also need to include indicators measuring circularity. Only with good data can we measure the success of policies and improve the research in the field.

To conclude, this thesis showed that reducing resource consumption and GHG emissions in a rapidly growing plastics sector requires decisive and comprehensive action, combining various mitigation strategies. This should not be limited to establishing a circular bioeconomy for plastics but also include further mitigation options, such as carbon capture and utilisation, electrification of production, and, first and foremost, reducing the demand for plastics where possible.

SAMENVATTING

Plastics zijn een essentieel onderdeel van onze economie geworden. De productie van plastic is toegenomen van 2 Mt in 1950 tot 380 Mt in 2015, waardoor het van alle bulkmaterialen wereldwijd de grootste groei in productie heeft doorgemaakt[4,6]. In 2015 was de sector verantwoordelijk voor 4,5% van de wereldwijde broeikasgasemissies[8]. Het gebruik van biomassa en recycling zijn twee opties die significant kunnen bijdragen aan de reductie van zowel het gebruik van fossiele grondstoffen in de plasticsector, als de bijbehorende broeikasgasemissies[4,9,15-17]. Samen kunnen beide opties bijdragen aan een circulaire bio-economie (CBE) voor plastic en in potentie zelfs tot negatieve CO_2-emissies door het vastleggen van biogene koolstof in plastic producten. Sinds het EU-actieplan voor de circulaire economie uit 2015[28] en de nieuwe strategie voor de bio-economie voor een duurzaam Europa uit 2018[54], is de CBE als concept ontstaan vanuit een toenemende link tussen de concepten van de circulaire economie (CE) en de bio-economie.

Het hoofddoel van dit onderzoek is te onderzoeken in welke mate een CBE bij kan dragen aan het reduceren van broeikasgasemissies en van het gebruik van grondstoffen in de plasticsector. We hebben dit probleem benaderd door verschillende methodes en perspectieven te combineren:

- Een kwantitatief en kwalitatief literatuuronderzoek aangevuld met interviews om inzichten op te doen rondom ontwikkelingen in de industrie en wetenschappelijke literatuur over het concept van een CBE. (Hoofdstuk 2)
- Geïntegreerde modellering voor een beoordeling van de dynamiek van vraag en aanbod op de lange termijn binnen de wereldwijde plasticsector en tevens van de gevolgen hiervan op broeikasgasemissies in verschillende scenario's. (Hoofdstukken 3 en 4)
- Een levenscyclusanalyse van kleine plastic flesjes. Deze productgeoriënteerde casestudy geeft een bottom-up perspectief van een CBE voor plastic. (Hoofdstuk 5)

Gezamenlijk bieden deze methodes en perspectieven inzicht in voordelen, afwegingen en uitdagingen wanneer we naar een circulaire bio-economie voor plastics gaan. Specifiek geven ze antwoord op de vier onderzoeksvragen:

1. Wat zijn de bepalende karakteristieken van een CBE en wat is de huidige rol ervan in Europese bio-economieclusters?
2. Hoe kan de impact van een CBE op broeikasgasemissies, materiaalgebruik en circulariteit goed beoordeeld worden?
3. Wat zijn de potentiële ontwikkelingen in de wereldwijde productie van plastic, bestaande producten, afval en gerelateerd materiaalgebruik en CO_2-emissies tot 2100, uitgaande van huidige beleidstrends.
4. Wat zijn veelbelovende strategieën, inclusief CBE, voor productie van plastic en afvalmanagement om broeikasgasemissies en materiaalgebruik van de plasticsector te reduceren?

Hoofdstuk 2 gaat in op de definitie van het begrip CBE middels een literatuuronderzoek. De rol van CBE in Noordwest-Europese bio-economieclusters is geanalyseerd middels interviews. We identificeren strategieën voor grondstof- en productfocus van de clusters, en onderzoeken de rol van bio-raffinaderijen, circulaire oplossingen, recycling en cascadering. Tot slot gaan we in op de hiaten in CBE-literatuur en de potentiële bijdrage van CBE aan duurzaamheid. De geanalyseerde bio-economieclusters bewegen naar een CBE door in toenemende mate rest- en afvalstromen als grondstof te beschouwen, hiervoor geïntegreerde bio-raffinaderijen te ontwikkelen en zich meer op hoogwaardige toepassingen van biomassa te richten. Echter, tot nu toe is er weinig focus op de end-of-life van biobased producten, bijvoorbeeld door circulair productontwerp, recycling en cascadering. Belangrijke uitdagingen voor het implementeren van circulaire strategieën liggen in het domein van beleid en regulering, kosten en de momenteel kleine biobased markten. Van de door de geïnterviewden genoemde veelbelovende productsectoren voor de bio-economie hebben plastics en bouwmaterialen de grootste potentie voor recycling en cascadering. Hoewel het mogelijk is dat de CBE helpt om de duurzaamheid van de bio-economie te verbeteren, is CBE niet inherent duurzaam en moeten mogelijke afwegingen in ogenschouw worden genomen. In het bijzonder lijken sociale aspecten, cascadering, circulair productontwerp en productgebruik minder vertegenwoordigd in de literatuur over de CBE. Bio-raffinaderijen, afval, reststromen en afvalmanagement worden daarentegen wel goed gedekt.

Hoofdstuk 3 behandelt het probleem dat Integrated Assessment Modellen (IAM), die ingaan op de relatie tussen menselijke en natuurlijke systemen en een belangrijke rol spelen in strategieën om klimaatverandering tegen te gaan, de rol van plastics en de circulaire economie grotendeels negeren. In dit hoofdstuk wordt het Plastics Integrated Assessment Model (PLAIA) gepresenteerd, waarmee productie, gebruik en end-of-life van plastic wordt toegevoegd aan het IMAGE IAM. PLAIA modelleert de wereldwijde plasticsector en de impact van de sector tot 2100 voor 26 regio's in de wereld. Dit levert een dynamisch langetermijnperspectief voor de sector en de interacties met andere socio-economische en natuurlijke systemen. De structuur van het model, de wiskundige formulering, aannames en databronnen worden allen in dit hoofdstuk samengevat. Het model verbindt de upstream chemische productie met de downstream productie van plastics, het gebruik in verschillende sectoren, en hun end-of-life. Daarom kan PLAIA materiaalgebruik en emissiereductiestrategieën over de gehele levenscyclus beoordelen in een IAM, inclusief de impact van een circulaire economie op mitigatie van klimaatverandering. PLAIA projecteert de vraag naar plastics en productieroutes en specificeert zo de jaarlijkse afvalproductie, afvalinzameling en de impact van afvalmanagement. Het laat de fossiele en biobased energie- en koolstofstromen zien in productvoorraden, stortplaatsen en de emissies tijdens productie en end-of-life.

Hoofdstuk 4 gebruikt het PLAIA-model om voor de plasticsector drie alternatieve routes voor mitigatie van CO2-emissies richting 2100 te analyseren, bezien over de gehele levenscyclus van productie tot afvalmanagement. Uitgaande van bestaand beleid voorziet PLAIA in het baseline-scenario een verdubbeling van de vraag naar plastics in 2050 en meer dan een verdriedubbeling in 2100, met een bijna equivalente stijging van CO_2-emissies. De resultaten laten zien dat door het vastleggen van biobased koolstof in plastic producten, de combinatie van biomassa-inzet en het opslaan ervan in stortplaatsen kan leiden tot een significante reductie van emissies op lange termijn. Dit gaat echter uit van een continue afhankelijkheid van primaire grondstoffen en daarnaast negeert het andere ecologische en sociale gevolgen van de hiervoor benodigde uitbreiding van stortplaatsen tot ver boven de huidige capaciteit. Een aanpak volgens het concept van circulaire economie zónder een toegevoegde bio-economie reduceert het grondstofgebruik en zorgt voor meer emissiereducties tot 2050, maar beperkt het potentieel van negatieve emissies op de lange termijn. Een aanpak volgens het concept van circulaire bio-economie, waarin recycling gecombineerd wordt met een groter gebruik van biomassa, kan de sector uiteindelijk doen veranderen in een netto opslag van koolstof ('carbon sink'). Ook kan afvalstort hierbij uitgefaseerd worden en grondstofgebruik gereduceerd. Het werk in dit onderzoek heeft de representatie van materiaalstromen en de circulaire economie in wereldwijde energie- en emissiemodellen verbeterd en heeft inzicht in de langetermijndynamiek van de plasticsector opgeleverd.

Nadat in hoofdstuk 3 en 4 is gekeken naar potentiële langetermijnontwikkelingen van de plasticsector, wordt met **hoofdstuk 5** een casestudy over kleine plastic flessen uitgelicht, waarbij op broeikasgasemissiesen materiaalgebruik van verschillende afvalmanagementstrategieën wordt ingegaan. Daarbij bespreken we ook de afwegingen tussen doelen voor mitigatie van klimaatverandering enerzijds en circulaire economie anderzijds. Het hoofdstuk beoordeelt over de gehele levenscyclus de bijdrage aan opwarming van de aarde (global warming potential, GWP) en aan uitputting van abiotische middelen (material utility, MU) van kleine plastic flessen (250 ml) gemaakt uit PET (polyethyleentereftalaat) dan wel biobased PEF (polyethyleenfuranoaat). Daarbij wordt gerekend over meerdere recyclingtrips met mechanische (MR) dan wel chemische recycling (CR) in Nederland. De resultaten laten zien dat biobased PEF een 50% tot 74% lagere broeikasgasemissie dan PET heeft over de levenscyclus, na één end-of-life trip, afhankelijk van de wijze van afvalmanagement. Ook concludeert dit onderzoek dat recyclingsystemen gebaseerd op statiegeld de cumulatieve cradle-to-grave broeikasgasemissies significant doen afnemen voor beide typen flessen, vooral wanneer meerdere recyclingtrips worden toegepast. Het kan worden gezegd dat alleen een strategie volgens een circulaire bio-economie (door het integreren van biobased PEF-flessen in de recyclingsystemen) kan verzekeren dat het GWP-voordeel van PEF-flessen behouden blijft. Afsluitend gaat dit hoofdstuk in op de afwegingen tussen GWP en MU: op statiegeld gebaseerde CR laat de beste prestaties zien op MU, maar op statiegeld gebaseerde MR laat de beste prestaties zien wanneer het aankomt op broeikasgasemissies, vanwege de hoge energie-intensiteit van chemische recycling. Daarom kan een combinatie van mechanische en chemische recycling bijdragen aan het behalen van de doelen voor zowel de circulaire economie als het tegengaan van klimaatverandering.

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ABOUT THE AUTHOR

Paul Stegmann was born in 1987 in Munich. His multi- and interdisciplinary interests inspired him to do a Bachelor in European Studies in Magdeburg (Otto-von-Guericke University) and Barcelona (University Pompeu Fabra), followed by a Master in Sustainable Development at Utrecht University and Leipzig University. Afterwards, he worked on projects in the fields of circular economy, waste management and development cooperation for international organizations, namely the Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ), and International Solid Waste Association (ISWA).

He joined the Copernicus Institute of Sustainable Development at Utrecht University as PhD candidate in September 2017. During his PhD, he was working on the interlinkages between the bio-based and the circular economy, namely the "circular bioeconomy". His focus was the investigation of climate change mitigation pathways for the plastics sector. Moreover, he was teaching in the courses Sustainable Resource Use, Bio-based economy, and supervised consultancy projects and a Master thesis for the M.Sc. in Energy Science.

During his PhD he worked as guest researcher for PBL, the Netherlands Environmental Assessment Agency, and had a research stay at the International Institute for Applied Systems Analysis (IIASA) for the Young Scientists Summer programme.

Currently he is working as Consultant at TNO (Netherlands Organisation for Applied Scientific Research), in the team for circular plastics modelling, which is part of TNO's unit for Climate, Air and Sustainability.