NTNU
Norwegian University of
Science and Technology

Environmental benefits of household plastic and bioplastic packaging management in the municipality of Trondheim

Irmeline Eloise Astrid Frøydis Sadeleer

Master in Industrial Ecology
Submission date: June 2018
Supervisor: Helge Brattebø, EPT

Norwegian University of Science and Technology
Department of Energy and Process Engineering

NTNU
Norwegian University
of Science and Technology

Department of Energy
and Process Engineering

EPT-M-2018-77

MASTER THESIS

for

Student Irmeline de Sadeleer

Spring 2018

Environmental benefits of household plastic and bioplastic packaging management in the municipality of Trondheim

Miljøgevinster av behandling av plast- og bioplastemballasje fra husholdninger i Trondheim kommune

Background and objective

The municipality of Trondheim is in the process of developing a new waste management plan for the period 2018-2030. In this regard, the municipality is interested in gaining more knowledge about household plastic packaging, it's recyclability potentials and the composition of future plastic stream.

In its ambitious Circular Economy Package, the European Union has put focus on recycling. Among other targets, 55% of plastic packaging is aimed at being recycled within 2030. The current recycling rate in the municipality of Trondheim is of 30%, and should hence be improved. Aiming at increasing its recycling rates for improving the economics and quality of plastics recycling, the municipality considers building a central sorting facility. As plastic from households is a large waste flow in terms of volume, recycling optimization is highly relevant. In this thesis, the environmental performance of two plastic handling types will be assessed: the efficiency of the central sorting facility will be compared to the current handling, comprising a combination of manual sorting and incineration.

In addition, several challenges are related to discarded plastic products: such as large waste flows, littering problems both off shore and on land, microplastics, and use of fossil raw materials. The current emphasis on sustainable development is driving the use of alternative and more sustainable materials, which also results in changes in the waste stream composition. Bio-based plastic is for instance becoming a popular alternative to petroleum-based plastic, and will most likely become a large share of the plastic waste stream for certain product categories. This thesis will therefore further analyse the potential environmental benefits of an alternative plastic material based on a life cycle perspective, from the production to the treatment processes.

The work will be carried out in collaboration with the municipality of Trondheim, with Knut Bakkejord as co-supervisor.

The following tasks are to be considered:

1. Carry out a literature study regarding the topics of relevance to this project.
2. Develop a life cycle assessment (LCA) model to investigate the potential environmental impacts of the management of selected plastic waste types in the municipality of Trondheim. The LCA should be based on an inventory informed by using the MFA-model the student worked with in the specialization project.
3. Develop a reference scenario that reflects the current waste management practices and future similar system solutions towards 2030. Develop a set of future scenarios to assess the environmental benefits related to the implementation of a central sorting facility, and to investigate the possible effects of increasing the share of bio-based plastic packaging compared to petroleum-based plastic.
4. Analyse the differences between the given plastic waste management solutions based on the scenario results from the life cycle assessment.
5. Analyse the impacts of an increased share of bio-based plastic in the plastic waste stream.
6. Analyse strengths and weaknesses of the data, model and results.
7. Make recommendations to the municipality regarding the treatment bio-based plastic and out sorting of plastic packaging.

-- ” --

Within 14 days of receiving the written text on the master thesis, the candidate shall submit a research plan for his project to the department.

When the thesis is evaluated, emphasis is put on processing of the results, and that they are presented in tabular and/or graphic form in a clear manner, and that they are analyzed carefully.

The thesis should be formulated as a research report with summary both in English and Norwegian, conclusion, literature references, table of contents etc. During the preparation of the text, the candidate should make an effort to produce a well-structured and easily readable report. In order to ease the evaluation of the thesis, it is important that the cross-references are correct. In the making of the report, strong emphasis should be placed on both a thorough discussion of the results and an orderly presentation.

The candidate is requested to initiate and keep close contact with his/her academic supervisor(s) throughout the working period. The candidate must follow the rules and regulations of NTNU as well as passive directions given by the Department of Energy and Process Engineering.

Risk assessment of the candidate's work shall be carried out according to the department's procedures. The risk assessment must be documented and included as part of the final report. Events related to the candidate's work adversely affecting the health, safety or security, must be documented and included as part of the final report. If the documentation on risk assessment represents a large number of pages, the full version is to be submitted electronically to the supervisor and an excerpt is included in the report.

Pursuant to “Regulations concerning the supplementary provisions to the technology study program/Master of Science” at NTNU §20, the Department reserves the permission to utilize all the results and data for teaching and research purposes as well as in future publications.

The final report is to be submitted digitally in DAIM. An executive summary of the thesis including title, student’s name, supervisor's name, year, department name, and NTNU's logo and name, shall be submitted to the department as a separate pdf file. Based on an agreement with the supervisor, the final report and other material and documents may be given to the supervisor in digital format.

☐ Work to be done in lab (Water power lab, Fluids engineering lab, Thermal engineering lab)
☐ Field work

Department of Energy and Process Engineering, 15. January 2018

Professor Helge Brattebø
Academic Supervisor

Research Advisors: Associate Professor Sigrun Jahren, NTNU
Fagansvarlig Knut Bakkejord, Trondheim kommune

Preface

This thesis concludes my Master of Science in Industrial Ecology at the Norwegian University of Science and Technology, Department of Energy and Process Engineering.

Plastic waste is currently at the core of the Norwegian public debate. I was asked by Knut Bakkejord working for the municipality of Trondheim to delve into this topic which is of interest for the municipality. In fact, the municipality is planning the construction of a central sorting facility for segregating plastic waste. This thesis therefore analyses the environmental impacts from such a facility compared to system currently in use. The outcomes of this thesis will hopefully be valuable when further and more detailed plans are to be developed. In addition, I was requested to look into the subject of bioplastics waste management, as little information exists on the topic. Excel files containing flowcharts, calculations, and the inventories themselves are available for the interested reader.

The following thesis deviates slightly from point 3 presented in the Master Thesis description. In consultation with my supervisor, it was judged more valuable to develop only one scenario depicting the current waste management system and rather base all future scenarios on a system including a central sorting facility. In addition, the year modelled was 2025 and not 2030 as stated in the description.

I would like to thank my supervisor Helge Brattebø for his excellent guidance throughout the semester, and for giving me confidence regarding the quality of my work. Special thanks to Carine Lausselet for her precious support and availability in the phase of the inventory development. Finally, I would like to thank my father Nicolas de Sadeleer and Thomas Rem from ROAF for proof-reading.

Abstract

Plastic waste is currently at the core of European and Norwegian public debate. On May 22, 2018, the Member States of the European Union (EU) and the European Parliament approved a set of ambitious measures as part of EU's circular economy policy. These measures were based on the Commission's proposals for implementing the circular economy package presented in December 2015, which is considered an important tool for combating climate change and resource depletion. The target of 50% plastic packaging recycling was among others adopted.

With the objective of increasing recycling rates, the municipality of Trondheim considers building a central sorting (CS) facility. However, the environmental impacts from such a CS facility has not been investigated in a holistic perspective. What are the environmental impacts of a waste management system where plastic waste is sorted out from the residual waste in a central sorting facility compared to a system where the fraction is sorted out at the household level? In addition, bioplastics have been presented as a sustainable alternative to conventional petroleum-based plastics and are increasingly becoming a part of the plastic market. Nevertheless, bioplastics have lower recyclability their conventional counterparts. How does a share of bioplastic affect the life cycle impacts of household plastic consumption? In an attempt to answer these two research questions, which both fill a knowledge gap in the literature, a life cycle assessment (LCA) based on material flow analysis (MFA) principles was developed.

The environmental burdens related to global warming potential (GWP), fossil depletion potential (FDP), freshwater ecotoxicity potential (FETP), human toxicity potential (HTP), natural land transformation potential (LTP) and terrestrial acidification potential (TAP) were found to be lower when plastics are sorted out in a CS facility than when sorted out in households. This primarily occurs with the elimination of individual packing and sorting processes and with decreased amounts of incinerated plastics. However, this conclusion becomes less evident when the amounts of recycled materials increase with higher out-sorting rates. In fact, impacts in the categories FETP, HTP and LTP are increased given the influence of the recycling process on these impact categories.

Higher recycling rates hence lead to diminished impacts in regard to GWP, FDP and TAP but induce higher environmental stress in regard to HTP, FETP and LTP. The plastic recycling rates were found to double when sorting the fraction out in a CS facility, but the target set by the EU was only reached in an ideal scenario.

Further, this analysis disclosed that the life cycle impacts of household plastic consumption are reduced importantly for freshwater ecotoxicity and slightly reduced for global warming and fossil depletion potentials when bioplastics are introduced. Nonetheless, all other impact categories experience increases in impacts, mirroring the high environmental stress caused by the bioplastic production process.

For mitigating climate change and resource depletion, it was found to be more effective to improve the plastic waste out-sorting system than to promote the production and use of bioplastics as alternative to petroleum-based plastics.

Sammendrag

Plastavfallproblematikk er en viktig pågående debatt i Europa og Norge. EUs medlemsland vedtok 22. mai 2018 ambisiøse regler som en del av politikken om sirkulær økomomi. Reglene er basert på Kommissionens forslag til sirkulærøkonmipakke som ble fremmet i desember 2015, og som er ansett som et viktig verktøy for å hindre klimaendringer og ressursutarming. Et av målene som ble vedtatt, er å resirkulere 50% av all plastemballasje.

Trondheim kommune ønsker å bygge et ettersorteringsanlegg med mål om å øke resirkuleringsratene. Miljøpåvirkningene av et slikt anlegg har ikke blitt analysert i et helhetlig perspektiv. Hva er miljøpåvirkningene av å sortere ut plast i et ettersorteringsanlegg sammenlignet med å sortere fraksjonen ut på husholdningsnivå? I tillegg har bioplast blitt presentert som et bærekraftig alternativ til fossil-basert plast. Hvordan vil en andel bioplast påvirke miljøeffekten av plastforbruket i husholdninger fra et livssyklusperspektiv? En livssyklusanalyse basert på materialflytsprinsipper ble utviklet for å svare på disse forskningsspørsmålene, som begge fyller et kunnskapshull i litteraturen.

Når plastavfallet blir sortert ut i et ettersorteringsanlegg fremfor i husholdninger, reduseres påvirkningene på global oppvarming, utarming av fossile kilder, ferskvannsøkotoksisitet, menneskelig toksisitet, endring av naturlige landarealer og landforsuring. Dette skyldes hovedsakelig elimineringen av individuel pakke- og sorteringsprosess samt reduserte mengder forbrent plast. Når mengdene resirkulert plast øker grunnet høyere utsorteringsrater, blir derimot konklusjonen nevnt over en annen. Påvirkninger av toksisitet- og landarealendringer vil nemlig øke grunnet en høy innvirkning av resirkuleringsprosessen på disse kategoriene.

Høyrere resirkuleringsrater minker dermed miljøbelastning for global oppvarming, utarming av fossile kilder og landforsuring, men øker samtidig miljøbelastning for toksisitet- og landarealendringer. Resirkuleringsratene vil dobles når plast er utsortert i et ettersorteringsanlegg, men målet satt av EU blir bare nådd i en ideell situasjon.

Videre viser denne analysen at et økt forbruk av bioplast fører til store miljøgevinster for ferskvannsøkotoksisitet og minimale miljøgevinster vedrørende global oppvarming, utarming av fossile kilder i et livsløpsperspektiv, sammenlignet med en situasjon hvor kun fossil-basert plast blir produsert og forbrukt. De resterende miljøkategoriene vil derimot erfare høyrere miljøpåvirkninger som gjenspeiler den høye belastningen fra produksjonsprosessen.

For å hindre klimaendringer og ressursutarming kan man utifra denne analysen konkludere med at forbedringer i plastutsorteringssystemet er et mer effektivt tiltak enn å bruke bioplast som alternativ til fossil-basert plast.

Table of Contents

List of figures

List of tables

List of equations

Acronyms

CS	Central Sorting
EU	European Union
FDP	Fossil depletion potential
FETP	Freshwater ecotoxicity potential
FU	Functional unit
GHG	Greenhouse gases
GWP	Global warming potential
HDPE	High density polyethylene
HTP	Human Toxicity potential
IVAR	Interkommunalt Vann, Avløp og Renovarsjon IKS
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LDPE	Low density polyethylene
LHV	Lower heating value
LTP	Natural land transformation potential
LUC	Land use changes
MFA	Material flow analysis
PA	Polyamide
PBAT	Polybutylene adipate/terephthalate
PET	Polyethylene therephthalate
PHA	Polyhydroxyalkanoates
PLA	Polylactic acid
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
ROAF	Romerike Avfallforedling IKS
SESAM	Sentralt EtterSorteringsAnlegg i Midt-Norge
TAP	Terrestrial acidification potential
TRV	Trondheim Renholdsverk
WFD	Waste Framework Directive

1. Introduction

Even though their large-scale production and use only dates back to 1950, a world without plastics seems today unimaginable. Plastic production and use has increased twenty-fold the last 50 years, surpassing the growth of most other man-made materials (Geyer *et al.*, 2017). The demand is further expected to double the next 20 years (Ellen MacArthur and McKinsey, 2016). It has been estimated that 8300 million tons of virgin plastics have been produced to date, of which about 6300 million tons have become waste. Of these, around 9% have been recycled, 12% incinerated, and 79% accumulated in landfills or in the natural environment (Geyer *et al.*, 2017).

One of the principal properties of plastic is its durability, being a significant advantage for food preservation, medical product efficacy, electrical safety, improved thermal insulation and to lower fuel consumption in transportation (Kershaw, 2015). The largest demand for plastics nevertheless comes from packaging products (Plastics Europe, 2017). As these are mainly single-use products, they have relatively short lifetimes. The poor management of post-use plastic, illustrated by the statistics above, means that the durability of plastic can become a significant problem in mitigating its impact on the environment.

As they lead to macro- and microplastics pollution both on land and in the marine environment, astray plastic debris is in fact increasingly recognized as an ecological concern (Barnes *et al.*, 2009; European Commission, 2018a; Sheavly & Register, 2007; UN Environment, 2017). Many studies have investigated the potential uptake of hydrophobic contaminants from plastic waste by organisms which can bioaccumulate in the food chain (Li *et al.*, 2016) but the consequences are still poorly understood (GESAMP, 2015).

In addition, marine debris may lead to human health and safety problems, aesthetic and economic impacts, habitat destruction, invasive species introduction and vessel damage (Barnes *et al.*, 2009; Sheavly & Register, 2007). A substantial fraction of marine plastic debris originates from land-based sources, and rivers act as a major transport pathway for all sizes of plastic waste (Schmidt *et al.*, 2017).

Several researchers (Kershaw, 2015; Li *et al.*, 2016) have called upon governments for playing an active role in addressing the issue of plastic overconsumption and the issue of plastic waste for controlling the sources and amounts of plastics debris. Better waste management options are especially sought for (Schmidt *et al.*, 2017).

Aiming at developing sustainable waste management schemes in the light of a circular economy, the European Union (EU) has set the target of 50% and 55% plastic packaging recycling by 2025 and 2030 respectively. In Norway, the municipalities have the responsibility of managing household waste (Forurensningsloven §30). For meeting the ambitious EU targets and improving their waste management system, Trondheim municipality together with other municipalities of the Trøndelag county consider building a central sorting (CS) facility. The facility would among others increase the plastic collection and out-sorting rates, and thereby the recycling rates.

Lyng and Modahl (2011) studied the environmental benefits of plastic waste out-sorting from Norwegian households, compared to a system without plastic out-sorting. With inspiration from this study, a life cycle assessment (LCA) was developed in this thesis as a case study for Trondheim. To the difference of Lyng and Modahl (2011), this thesis assesses the environmental impacts of sorting out plastics in a CS facility, compared to source separation at the household level, aiming at filling this knowledge gap in the literature. Research question No. 1 of this thesis is addressing the environmental impacts: *What are the environmental impacts of a waste management system where plastic waste is sorted out from the residual waste in a central sorting facility compared to a system where the fraction is sorted out at the household level?*

The current plastic production consumes 4-6% of the global oil production (Plastics Europe, 2017). For mitigating climate change and resource depletion, the dependency on fossil materials is sought at being reduced. The prevailing emphasis on sustainable development is therefore driving the use of alternative and more sustainable materials (European Commission, 2018a). Bioplastics is accordingly becoming an attractive alternative to conventional petroleum-based plastic and will most likely become a significant share of the future household consumption and thereby of the plastic waste stream (European Bioplastics, 2018). Despite the fact that bioplastics are assumed to represent lower environmental impacts than conventional plastics, bioplastics may have lower recyclability.

All reviewed LCA studies analysing the environmental impacts of bioplastics were only conducted for the production phase, or for the life cycle of a specific resin type (Belboom *et al.*, 2016; Murphy *et al.*, 2013; Song *et al.*, 2011; Tsiropoulos *et al.*, 2015; Weiss *et al.*, 2012). This thesis aims at filling a knowledge gap by developing an LCA analysing the life cycle emissions of household plastic consumption, which contains a share of bioplastics. For doing so, the system boundaries of the developed LCA are expanded, turning the study into a cradle-to-cradle analysis. In addition, the composition of the functional unit (FU) was changed to encompass bioplastics. Research question No. 2 of this thesis is addressing the role of bioplastics: *How does a share of bioplastic affect the life cycle impacts of household plastic consumption?*

The second chapter will review the main literature relevant to this thesis. It was conducted to establish scientific hypotheses on the outcomes of the performed LCA and to identify the knowledge gaps that this thesis aims at filling. The third chapter introduces the reader to the plastic waste management in the city of Trondheim, Norway, for which the case study is conducted. In addition, the legal background to the topic is framed. The fourth chapter presents the method used in this analysis, which is a compilation of several LCA estimates based on material flow analysis (MFA) principles. The chapter aims at giving the reader an understanding of the methodological basis of the results. The results will be presented in the fifth chapter and discussed in the sixth. Focus will on the one hand be set on out-sorting options for household plastic packaging waste, an on the other hand on the environmental impacts of bioplastics introduction in the household plastic consumption. In addition, the environmental benefits of the two strategies, i.e. changing the waste management system and changing the materials in use, will be compared. The seventh chapter concludes the work by briefly reviewing the main results and discussed issues.

2. Literature review

The literature review is divided into six parts. (1) The concepts of plastics and bioplastics are first introduced, giving the reader a proper background for understanding this thesis. Thereafter, (2) the production of bioplastics compared to petroleum-based plastics will be discussed from an environmental perspective. (3) The collection options will thereafter be touched upon, before discussing (4) different waste management options for petroleum-based plastics and bioplastics. Further, (5) the environmental impacts related to various waste management options will be analysed. Finally, (6) hypotheses regarding the research questions addressed on the basis of the literature findings will be presented.

2.1 Introduction

2.1.1 Plastics

Plastics is the common term used for determining a type material made from a range of organic polymers. There are two main types of plastics: thermo plasters and thermo setters. Thermo plasters is a family of plastics that is melted when heated and hardened when cooled and account for about 80% of the plastic consumption. The chemical reactions have the property of being reversible, so the materials can be reshaped when heated and frozen repeatedly (Plastics Europe, 2017; Al-Salem *et al.*, 2009). Thermo setters is a family of plastics that undergoes an irreversible chemical change when heated. They will degrade instead of melt at elevated temperatures (e.g. rubber) (Plastics Europe, 2017).

Within these two families, a large diversity of resins is found. A polymer resin is made up of hydrocarbon chains with a specific chemical configuration. The common characteristics of polymer resins are chemical stability and good mechanical properties. Their diversity allows a material to have specific features regarding strength, malleability, elasticity, etc. The combination of these qualities makes polymers attractive for a large variety of applications, attested by their worldwide increase in production and use. In 2016, 335 million tons of plastics were produced, of which 18% in Europe. That makes Europe to the second largest plastic producer after China. Currently, 4-8% of the global oil production is used for plastic production through distillation of naphtha[1] or by cracking of natural gas into ethylene (Plastics Europe, 2017).

Accounting for 40% of the European plastic demand and 59% of the plastic waste stream, packaging is the largest type of product demanded and the largest source of plastic waste in the

[1] Flammable liquid hydrocarbon mixture

EU (European Commission, 2018a). Because they often constitute single-use products, the large amounts relate to their relative short lifetimes. Plastic packaging is commonly divided into 7 main fractions based on their chemical composition (Table 1). These are: polyethylene terephthalate (PET), high density polyethylene (HDPE), polyvinyl chloride (PVC), low density polyethylene (LDPE), polypropylene (PP), polystyrene (PS) and others. The classification was developed by the American Society of Plastics Industry and has later been adopted by many European organizations. The European Commission recommends this labelling system, even though it application is not mandatory within the EU (Christensen, 2011).

Table 1: Classification of plastic packaging in resin types

#	**Abbreviation**	**Name**	**Utilization**
1	PET	Polyethylene terephthalate	Bottles for soft drinks, textile fibres, film food packaging. Most used polymer worldwide.
2	HDPE	High density polyehtylene	Containers, toys, house wares, gas pipes, industrial wrappings.
3	PVC	Polyvinyl chloride	Window frames, pipes, flooring, bottles, toys, cable insulation, credit cards, medical products. Third most used polymer worldwide.
4	LDPE	Low density polyethylene	Bags, toys, agricultural films, coatins, pipes, films, containers.
5	PP	Polypropylene	Films, electrical components, battery cases, containers. Second most used polymer worldwide.
6	PS	Polystyrene	Thermal insulation, tape cassettes, cups, electrical appliances, toys.
7	Others	Others, including bioplastics	

Most of plastic packaging is made up of LDPE, HDPE, PP and PET, while only a small share is constituted of PS, PVC and other types of resins (Plastics Europe, 2017).

Plastic pollution has lately received a lot of political and mediatic attention. It has been estimated that more than 8 million tons of plastics end up in the oceans every year *(Geyer et al.*, 2017). Due to its resistance to degradation induced by its chemical stability, most plastic debris will persist for centuries in the marine environment. In addition, they can be transported over long distances (Li *et al.*, 2016). Schmidt *et al.* (2017) estimated that 88-95% of the global plastics load ending up the oceans come from 10 rivers, 8 of them located in Asia. This makes rivers to the main pathway for the transportation of plastics litter arising on land due to bad waste management in the river catchment areas (Schmidt *et al.*, 2017). Both microplastics and macroplastics pose a risk to marine organisms, by ingestion and hydrophobic contaminants (Li *et al.*, 2016).

Rising public concern has stimulated the politics to take action both on the national and international level. In December 2017, the United Nations supported by more than 200 nations including Norway resolved to eliminate plastic pollution in the oceans. Both clean-up campaigns and the elimination of single-use plastics are targeted (UN Environment, 2017). At the EU level, a plastics strategy was presented in January 2018. The strategy aims at transforming the way plastic products are designed, used, produced and recycled within the EU (European Commission, 2018a). On a national level, the Norwegian authorities have developed strategies for reducing the inflow of plastics to the oceans in addition to organizing clean-up campaigns (Utenriksdepartementet, 2017).

2.1.2 Bioplastics

The current emphasis on sustainable development and plastic pollution is driving the development of alternative and more sustainable materials, as advocated by the EU. In this context, bioplastic is becoming a popular alternative to conventional petroleum-based plastic. Bioplastics are independent on fossil materials which mitigates climate change, they have biodegradable properties and lower related greenhouse gas (GHG) emissions compared to petroleum-based alternative (European Bioplastics, 2018; Niaounakis, 2013).

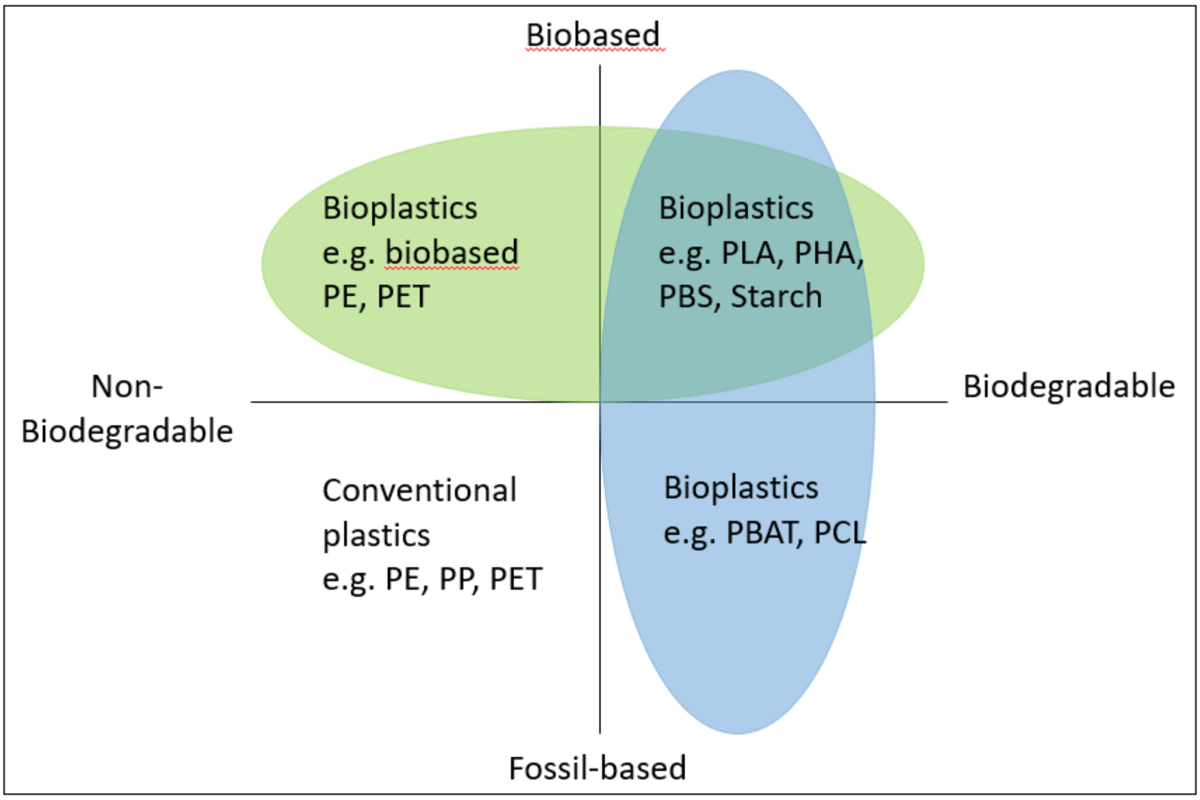


Figure 1: Illustration of the various bioplastics types. Source: European Bioplastics

Bioplastics can be classified into bio-based and fossil-based polymers (Figure 1). Moreover, they have the property of being either biodegradable or non-biodegradable, a property which makes up 43% and 57% of the market respectively. Bioplastics may hence either be based on renewable resources and be biodegradable, based on renewable resources and be non-biodegradable, or based on fossil resources and be biodegradable (Song *et al.*, 2011; European Bioplastics, 2018a).

Most bio-based plastics are identical in terms of properties to conventional petroleum-based resins (European Bioplastics, 2018). Bio-PET, bio-PE and bio-Polyamide (bio-PA) represent currently the largest fractions of bio-based materials (European Bioplastics, 2018). The origin of the organic source differs as for instance sugar cane, beet root and maize can be used as hydrocarbon source. Furthermore, partially bio-based polymers, also called biocomposites, are made of a combination of bioplastics and petroleum-based materials. The combination allows to meet technical requirements and/or to reduce costs. These materials are currently found in several commercially available products such as bio-PET coke bottles (Song *et al.*, 2011).

The property of biodegradation depends directly on the chemical structure of the polymer (European Bioplastics, 2018), which is different from the ones of non-biodegradable conventional plastics. Starch blends and polylactic acid (PLA) are currently the most commonly used fractions. The production process of bioplastics will differ depending on the wanted chemical structure, as bioplastics can among others either directly be extracted from biomass as is the case for starch, or synthesised from bio-derived monomers as is the case for PLA and bio-PE (Song *et al.*, 2011).

Fossil-based degradable plastics as for instance UV or oxo-degradable plastics break down when exposed to light or air respectively. However, they are still primarily oil-based (Michaud *et al.*, 2010). Polybutylene adipate/terephthalate (PBAT) is the most produced fraction. Fossil-based plastics make up the smallest share of bioplastics currently produced (European Bioplastics, 2018).

Bioplastics currently account for 1% of the global plastic production, but the market is expected to grow notably (European Bioplastics, 2017). For simplicity reasons, all introduced categories will be referred to as "bioplastics" in this thesis.

2.2 Production of petroleum-based plastics and bioplastics

Currently, 4-8% of the global oil production is used for plastic production through distillation of naphtha or by cracking of natural gas into ethylene (Plastics Europe, 2017). A flowchart depicting the production processes of petroleum-based plastics and bioplastics is found in appendix (A1) for further insight. Since its development, plastics have been replacing other materials such as metals, paper and glass. The use of plastics substituting steel in vehicles has for instance reduced their weight considerably, leading to less fuel needed per km driven (Hendrickson *et al.*, 2006). The environmental performance of a material is hence always a matter of relativeness and a product of the compared alternatives.

The environmental loads of petroleum-based plastic and bioplastic will be discussed for on the one hand for global warming potential (GWP), fossil depletion potential (FDP) and energy consumption. On the other hand, their effects on other impact categories will be analysed.

There is a general scientific agreement regarding the fact that the production of bioplastics induces less environmental stress in regard to GWP, FDP and energy consumption than their petroleum-based counterparts (Belboom *et al.*, 2016; Murphy *et al.*, 2013; Song *et al.*, 2011; Tsiropoulos *et al.*, 2015; Weiss *et al.*, 2012). The meta-analysis conducted by Weiss *et al.* (2012) suggests that GHG emissions of about 3±1 ton of CO_2eq and 55±34 gigajoules of primary energy could be avoided with the production of bio-based plastics compared to the production of their conventional counterparts.

The GWP and energy benefits differ between partially and totally bio-based resins. Tsiropoulos *et al.* (2015) compared in an LCA study the production of bio-based HDPE, partially bio-based PET and their petroleum-based counterparts. It was found that bio-based HDPE results in GHG savings up to 140% compared to fossil-based polymers and 65% savings of non-renewable energy use. The partially bio-based PET production, however, released similar amounts of GHG emissions as petroleum-based PET and only a 10% reduction in non-renewable energy use was observed.

The cultivation location and type of biomass used for producing bioplastics, however, influences these benefits (Belboom *et al.*, 2016; Tsiropoulos *et al.*, 2015). Tsiropoulos *et al.* (2015) studied the environmental impacts of producing sugarcane ethanol in India and in Brazil. The countries represent the main commercial facilities producing bio-ethylene used for bioplastic production (Mohsenzadeh *et al.*, 2017). It was found that Brazilian ethanol leads to slightly higher impacts than Indian ethanol due to local conditions, different harvesting practices and transport distances (Tsiropoulos *et al.*, 2015). Belboom *et al.* (2016) assessed the environmental performance of bioplastics made from wheat and sugar beet. The authors found that the use of wheat resulted in higher environmental loads in nearly all analysed impact categories.

Nonetheless, the environmental benefits of bioplastic production compared to petroleum-based plastics become less clear when all impact categories are analysed (Tabone *et al.*, 2010; Weiss *et al.*, 2012; Piemonte and Gironi, 2012; Tsiropoulos *et al.*, 2015; Belboom *et al.*, 2016), especially for land use change (LUC) related emissions.

Weiss *et al.* (2012) highlight the variability of the results found in the literature when a broader range of impact categories is analysed, making the drawing of conclusions regarding the environmental benefits of bioplastic production highly uncertain. Belboom *et al.* (2016) explain the increase in environmental stress in all other impact categories for bio-based materials by the significance of the cultivation and bioethanol production steps. In addition to these processes, Tabone *et al.* (2010) identified fermentation and other chemical processing steps as impactful. The application of fertilizers and pesticides was found to especially increase eutrophication and stratospheric ozone depletion (Weiss *et al.*, 2012).

Tsiropoulos *et al.* (2015) analysed the effects on human health and ecosystem quality when producing bio-PET and bio-HDPE compared to their petroleum-based counterparts. The burdens were calculated to be 50 and 14 times greater for human health from bio-HDPE and bio-PET production respectively compared to the fossil alternative. For ecosystem quality, the impacts were 2 orders of magnitude and 19 times from bio-HDPE and bio-PET production respectively. These large differences are mainly caused by agricultural processes such as pesticide use, pre-harvesting burning practices in Brazil and land occupation. The authors recommend a careful interpretation of these results as they are subject to high uncertainties.

When accounting for LUC emissions, the benefits of bioplastic production on GWP are severely decreased (Liptow & Tillman, 2012; Piemonte & Gironi, 2012; Tsiropoulos *et al.*, 2015; Weiss *et al.*, 2012). Piemonte and Gironi (2012) point to the fact that many studies have failed to account the emissions occurring when forests and grasslands are converted to agricultural land, diverted to bioplastics feedstock. The authors state that by excluding LUC emissions, most studies have a limited scope, as they account for the carbon benefits of using land for bioplastics but not for the carbon costs, the carbon storage and sequestration sacrificed by diverting land from its existing uses (Piemonte & Gironi, 2012). The range of emissions increase is, however, highly uncertain. No proper methodology is in place to account for this parameter, resulting in increased GWP values ranging from a doubling of the emissions to no increase at all (Liptow & Tillman, 2012). The authors therefore urge for methodological improvement for the inclusion of LUC emissions in LCA, as this parameter seems to be highly decisive for the final results (Liptow & Tillman, 2012; Weiss *et al.*, 2012). Piemonte and Gironi (2012) eventually suggest that a sustainable alternative would be the use of agricultural bi-products as feedstock for bioplastic production as this would not lead to the displacement of environmental burdens related to the LUC emissions of other biomasses.

Álvarez-Chávez *et al.* (2012) argue that the environmental impacts should not only be evaluated for estimating the sustainability of bioplastics. The authors conducted a literature review on the environmental, health and safety impacts of various bioplastics through their life cycle. The authors concluded that none of the bioplastics currently in commercial use or under development are fully sustainable. Some environmental aspects are improved such as reduction of GHG emissions, lowered energy uses and recyclability potentials. The impacts regarding occupational health and safety, however, were found to be high influenced by the exposure to pesticides, the use of various chemicals and the risk of explosions. Some bioplastics were preferable in an environmental perspective, while others were preferable in a health and safety perspective. In general, they found that PLA, starch and polyhydroxyalkanoates (PHA) are preferred over other bioplastic types (Álvarez-Chávez *et al.*, 2012).

Some authors have given recommendations for improving the sustainability of bioplastics. Tsiropoulos *et al.* (2015) suggest that pesticide control and elimination of burning practices can highly reduce the negative effects occurring during the bio-based and partially bio-based polymer production process. Álvarez-Chávez *et al.* (2012) advises that crop diversity, good soil management and efficient water use would lead to lower environmental burdens.

2.3 Collection of plastic waste

In Europe, over 26 million tonnes of plastic waste are generated every year. However, less than 30% of this waste is collected for recycling (European Commission, 2018a). The collection rate is even lower in Norway with a national average of 22% (Askham & Raadal, 2016). Local differences are experienced, with a collection rate of 20% in Trondheim, 29,7% in Oslo and 16% in Bergen (Hjellnes Consult AS, 2017; Syversen & Bjørnerud, 2015). The Commission's strategy for plastics in a circular economy highlights the importance of improving separate collection of plastic waste for ensuring the quality inputs to the recycling industry (European Commission, 2018a). In the Nordic countries, unsorted plastic packaging waste in the residual waste fraction has been suggested to be one of the largest potentials for increased collection and recycling (Fråne *et al.*, 2015). Nonetheless, the total collected plastic waste amounts have increased in Europe by 11% since 2006 (Plastics Europe, 2017).

90% of the Norwegian population is offered a collection system for out-sorted plastic waste (Raadal *et al.,* 2016). The collection of plastic packaging waste for recycling in the Nordic region is based on three main principles (Fråne *et al.*, 2015). The first and most common way of collecting plastic waste from households is by kerbside collection of source-sorted plastic, standing for 84% of the Norwegian collection system (Raadal *et al.*, 2016). Multi-compartment bins, separate containers, transparent plastic bags, or coloured bags prior to optical colour sorting are the main collection options. Second, deposit return system with drop-off points is often used in densely populated areas, or for specific plastic fractions such as PET bottles. 12% of the Norwegian population have access to this system (Raadal *et al.*, 2016). Finally, kerbside collection of residual waste subject to central sorting (CS) is being developed, currently available for 3.5% of the Norwegian population (Raadal *et al.*, 2016).

Plastic out-sorting from residual waste is currently only done by Romerike Avfallforedling IKS (ROAF) outside Oslo, the first CS facility in Europe which opened in 2013 (Fråne *et al.*, 2015). This system is the most effective in terms of collection rates, achieving the out-sorting of 11 kg plastic waste per inhabitant. In comparison, 7.5 kg are averagely achieved in kerbside collection systems and 3.4 kg in drop-off point systems (Raadal *et al.*, 2016). As a consequence of higher collection or out-sorting rates, higher material recycling rates are also achieved. The guidelines presented by The Nordic Region argue for tailored collection systems at local level. The local circumstances, local targets on waste management and how long the existing solutions for waste management have been in place are decisive for the effectiveness of different solutions (Fråne *et al.*, 2015).

Raadal *et al.* (2016) suggest that the collection and out-sorting solutions are the bottlenecks currently hindering the increase in plastic recycling rates in Norway for three main reasons. First, quite a few Norwegian households lack systems for sorting out plastic waste. Second, the exisiting collection system is insonsistent as it varies between the three above-described options. Unapropriate solutions at the household level and a lack of motivation and knowledge are exacerbating the consequences of this inconsistency. Third, the quality of the collected materials if often not satisfying enough due to organic pollution and wrongly out-sorted products. Unander (2017) concluded that a CS facility combined with appropirate collection solutions at the household level is an effective combination for increasing recycling rates.

2.4 Plastic packaging waste management

The management of the plastic fraction is highly debated in the discussion of municipal solid waste systems (Rigamonti *et al.*, 2014). Following the classification of Panda *et al.* (2010), the four main routes for plastic waste management will be described: landfilling, mechanical recycling, biological recycling and thermo chemical recycling (Figure 2).

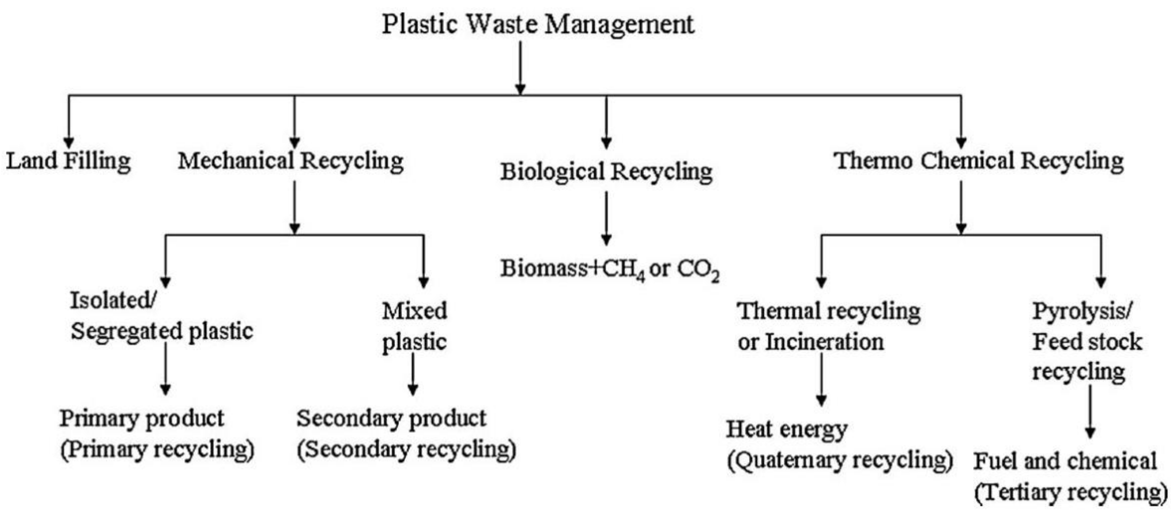


Figure 2: Pathways for plastic waste management. Source: Panda et al. (2010).

Landfilling Of the 27.1 million tons of plastic waste that were collected in 2016 in the EU28+NO/CH, 27.3% were landfilled. The amounts are less when only considering plastic packaging waste: of the 16.7 million tons of packaging waste collected in the same area, 20.3% were landfilled (Plastics Europe, 2017). For the past ten years, the amount of landfilled plastics has been reduced with 53% as a result of landfilling bans in various Member States. This reduction also occurred in Norway after the landfilling of degradable waste was prohibited in 2009 (Avfallsforskriften, chapter 9).

Mechanical recycling The Waste Framework Directive (WFD) defines *recycling*, also called mechanical or material recycling, as follows: "recycling means any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations" (2008/98/EC, article 3). Further, the amendment of 2015 defines that "the reporting on the attainment of the recycling targets must be based on the input to the final recycling process", where the final recycling process is delimited to " the recycling process which begins when no further mechanical sorting operation is needed and waste materials enter a production process and are effectively reprocessed into products, materials or substances (Amended version of 2008/98/EC, article 3, 2015).

A homogenous plastic stream can be converted into products with the same or nearly the same performance level that the original product, called primary recycling or closed-loop recycling. The recovery of waste into other products than the original one is called secondary recovery or open-loop recycling (Hopewell *et al.*, 2009; Panda *et al.*, 2010). Mechanical recycling refers to the process of shredding, melting and granulation of plastic waste, where the chemical composition is maintained. Only thermo plasters have thus the possibility of being mechanically recycled (Christensen, 2011). The steps may occur in a different order, multiple times or not and will vary according to the composition and the contamination level of the waste stream (Ragaert *et al.*, 2017). When it comes to packaging waste, 40.9% of the European plastic waste was mechanically recycled in 2016. For the past ten years, the amount of recycled waste has increased by 74% (Plastics Europe, 2017).

There are, however, technical, quality and economic challenges related to the recycling of plastics (Askham & Raadal, 2016; Astrup *et al.*, 2009; Sevigné-Itoiz *et al.*, 2015). Even though these factors are highly interlinked, an attempt was made to classify them for the sake of clarity (Table 2).

Table 2: Main challenges related to the recycling of plastic packaging

Challenge	**Main causes**
Technical	a) Cleanness of the waste flow b) The recyclates should be composed of only one resin type
Quality	a) Degradation of the polymer during its lifetime and duing the reprocessing phase b) The blend of various resins affect the end product's properties c) Accumulation of pollutants and harmfull substances
Economical	a) Prices regulated by the oil market b) Low demand for recyclates in Europe

Firstly, there are several technical barriers. Recycling of plastics into high-quality products requires that the recycled materials are made of one type of clean plastic (Astrup *et al.*, 2009; Ragaert *et al.*, 2017). In fact, most plastic types are not compatible with each other because of immiscibility at the molecular level and variations in the process requirement at the macro-level (Hopewell *et al.*, 2009). However, clean waste streams are difficult and costly to achieve as it requires extensive collection and out-sorting systems. Therefore, plastic resins present in large quantities such as PE of PET are often focused upon (Hillman *et al.*, 2015) whereas smaller flows are recycled together, lowering the quality of the end product (Askham & Raadal, 2016) in regard to strength, transparency and colour (Astrup *et al.*, 2009). In addition, laminates which are made up of layered plastic types are mainly used for food products because it extends their lifetime considerably. No technologies currently exist for separating the layers from each other, hindering the recyclability of these products (Stensgård *et al.*, 2017).

Secondly, quality challenges arise on the one hand due to polymer degradation during its lifetime and on the other hand due to reprocessing (Ragaert *et al.*, 2017). In fact, the recycling of a virgin material can only occur two or three times before the strength of the plastics is affected through thermal degradation (Singh *et al.*, 2017). Especially when several types of polymers are melted together, differences in melting points will affect the quality of the end product (Ragaert *et al.*, 2017). Furthermore, due to the lowered quality experienced when different resins are recycled together, it is often not technically feasible to add recycled plastic to a virgin material without decreasing the quality of the virgin material in regard to its colour, clarity or mechanical properties (Hopewell *et al.*, 2009). Moreover, environmental pollutants present in the plastic pigments and harmful substances such as flame retardants can accumulate in the recycled materials and may be released during the utilisation phase of the recycled product (Askham & Raadal, 2016).

Thirdly, the competitiveness of recycled plastics directly depends on the oil prices, creating an economic challenge (Askham & Raadal, 2016). Uncertainties about the market outlet and the recycled material flows also explains the unattractiveness of recycling plastic materials (European Commission, 2018). Askham and Raadal (2016) suggest that new areas for the use of recycled materials should be developed for increasing the material recycling rates in accordance with EU legislations.

Where a recycling stream for a specific plastic type is established (e.g. PE, PET or PP), the bio-based alternatives can be material recycled together with their conventional counterparts (European Bioplastics, 2018). Biodegradable biopolymers, however, pose problems as they insert impurities when they enter the conventional plastics recycling or organic waste composting streams (Niaounakis, 2013). Technological development would be needed for segregating bioplastics from conventional plastics. Some technologies are available but are costly given the low volumes of bioplastics in the waste stream (Niaounakis, 2013; Song *et al.*, 2011). In addition, combinations of plastics films/sheets of different biopolymers are used to enhance barrier properties, as for conventional laminates. The heterogenous materials lead to compromised recyclability (Song *et al.*, 2011). Other fractions are said to be potentially recyclable (e.g. PLA), but no separate waste stream currently exists (European Bioplastics, 2018). Song *et al.* (2011) and Niaounakis (2013) believe that the lack of continuous and reliable supply of bioplastic waste, combined with the current small size of the waste stream makes the recycling of these fractions economically unattractive.

Biological recycling Biological recycling converts degradable bio-based or fossil-based plastics back to biomass in a realistic lifetime. Some degradable plastics are also compostable. These must satisfy criteria's regarding degradability and disintegration, the quality of the compost obtained and the absence of any negative effect on the composting process (Michaud *et al.*, 2010).

Panda *et al.* (2010) point out four concerns linked to biological plastics recycling. First, the labelling system leads to misunderstandings of the degradation concept. For a product to be labelled *degradable* or *compostable*, it must follow the European standard EN 13432 "Requirements for packaging recoverable through composting and biodegradation", which is designed for plastic packaging treatment in industrial composting facilities and anaerobic digestion. There is currently no international standard specifying the conditions for home composting of biodegradable plastics (European Bioplastics, 2016). The labelling requirements may lead to misunderstandings, as the plastic products will only degrade under certain conditions which are not easily found in the natural environment. Prolonged temperatures of above 50°C are typically required for bioplastics to degrade (Kershaw, 2015). In fact, the biodegradation of bioplastic materials strongly depends on the environment where they are placed and the chemical nature of the material (Adamcová *et al.*, 2017). Consequently, bioplastics will not degrade more rapidly than conventional petroleum-based plastics in the marine environment (Kershaw, 2015).

Second, biodegradable plastics might cause an increase of methane emissions, released when materials biodegrade anaerobically.

Third, the mixture of degradable and non-degradable plastics can complicate the out-sorting and recycling processes, affecting the quality of the resulting recycles. Fourth, they could lead to an increase of plastic litter if people believe that discarded plastics simply will disappear (European Commission, 2018a; Panda *et al.*, 2010).

Thermo chemical recycling This treatment option has two alternatives: incineration and feedstock recycling. Incineration is the third most common waste management method: 41.6% of the European plastic waste was sent to incineration for energy recovery. For the past ten years, the amounts of waste sent to incineration for energy recovery has increased by 71% (Plastics Europe, 2017). The lower heating value (LHV) of the waste plastics are approximately similar to conventional fuel oil and can therefore substitute fossil fuels (Ragaert *et al.*, 2017; Scott, 2000). The LHV of bioplastics are, however, lower than for many conventional plastics (Laußmann *et al.*, 2010). Nonetheless, bioplastics produced from renewable sources contribute positively to the generation of renewable energy, as the waste is considered biogenic and hence carbon neutral (Astrup *et al.*, 2009; Iwata, 2015; Song *et al.*, 2011).

Feedstock recycling, also called tertiary recycling or chemical recycling, breaks down waste polymers to monomers or other chemicals of value. These products can be used in a variety of industrial processes as feedstock or as transportation fuels, substituting the amount of chemicals used in virgin plastics and in fossil fuels (Panda *et al.*, 2010). This recycling type is perceived as valuable for heterogenous and contaminated plastic fractions if separation and washing is neither economical or technically feasible (Ragaert *et al.*, 2017).

2.5 Environmental considerations of plastic management

Previous LCA studies have been conducted for determining the most environmental friendly waste management option for plastic packaging waste. There is a general scientific agreement on the fact that material recycling is preferred over incineration, which again leads to less environmental burdens than landfilling (Brogaard *et al.*, 2014; Lazarevic *et al.*, 2010; Lyng & Modahl, 2011; Michaud *et al.*, 2010; Rigamonti *et al.*, 2014; Rossi *et al.*, 2015; Shonfield, 2008). This order of preference confirms the validity of the waste hierarchy (2008/98/EC).

Al-Salem *et al.* (2009) and Hopewell *et al.* (2009) expose three main environmental benefits of plastic recycling: fossil fuels are conserved as less oil is needed for the production of virgin materials, energy requirements and solid waste generation are reduced and CO_2, NO_x and SO_2 emissions are lowered.

Lyng and Modahl (2011) conducted an LCA study on plastic out-sorting from Norwegian households which is sent to Germany for material recovery, compared to incinerating the plastic fraction together with the residual waste. Material recycling was found to result in less emissions than incineration, regardless of the energy source being substituted. The authors found that 2.7 kg CO_2eq and 12 per kilo plastic packaging could be avoided with material recycling. This is in line with the calculations presented by Grønt Punkt Norge (2013), showing that between 1.5 and 2.5 kg of GHG emissions can be avoided if one kilo of plastic packaging is material recycled, compared to if the same amount is incinerated. Material recycling hence reduces the CO_2eq emissions with approximately 80% compared to virgin material production (Brogaard *et al.*, 2014).

Mechanical recycling is, however, not always the best treatment option if other factors are taken into account. In fact, (1) the impact category analysed, (2) the type of plastic, (3) the organic contamination level, (4) the substitution ratio of virgin materials, (5) the substituted material and (6) the environmental impacts from the collection and recycling processes must be accounted for.

First, the impact categories analysed are of importance. Rigamonti *et al.* (2014) and Michaud *et al.* (2010) analysed a broad range of impact categories regarding various waste management practices. Their results showed that there is no preferred solution for plastic waste management when all impact categories are analysed.

Second, the type of resin influences the results for different impact categories. Michaud *et al.* (2010) reviewed several LCA studies to assess the impact of alternative waste management options for a range of plastic resins: PE, PET, PP, PS, LDPE, HDPE, PVC and mixed plastics. The impact categories of GWP, depletion of natural resources and energy demand were assessed. When analysing the categories in detail for the individual plastic resins, variations were discovered. PET and PVC resins were for instance found to lead to less environmental burdens when landfilled. Because their LHV is lower than for other resins types, the amounts of avoided emissions when incinerated are reduced (Shonfield, 2008). Rigamonti *et al.* (2014) also concluded that high quantities of PET and HDPE reduce the impacts in most categories compared to other resin types. Furthermore, it is unclear how mixed and dirty plastic fractions should be managed. Because of their low recyclability, their use as fuel substitution is the preferred option, especially if coal is replaced (Astrup *et al.*, 2009; Sevigné-Itoiz *et al.*, 2015). There are thus differences between the environmental impacts of various plastic resins types.

Third, the level of organic contamination is a factor of importance. Frees (2002) assessed the effects of organic contamination on the recycling and incineration processes. This was measured in terms of chemical oxygen demand and water demand for the cleaning process. Increased chemical oxygen demand leads to increased amounts of treated wastewater, again increasing the energy demand and its related environmental impacts. In scenarios where hot water is used for cleaning, however, several impact categories were higher for recycling than for incineration. In an incineration process, organic contamination leads to increased LHV and thereby more substituted energy. In addition, the GWP impacts were reduced with organic contamination as food waste emissions are considered biogenic and hence carbon neutral. Clean fractions of individual plastic polymer of good quality should hence be recycled and organic contamination avoided (Frees, 2002; Lazarevic *et al.*, 2010; Michaud *et al.*, 2010).

Fourth, the virgin material substitution ratio is a crucial element when assessing the performance of plastic waste management options. Lazarevic *et al.* (2010) conducted an extensive literature review assessing the effects of the choice of substitution ratio. The study distinguished between ratios of 1:1, ratios ranging between 1:1 and 1:0.5 and ratios equal or less than 0.5. For substitution ratios of the two fist categories, recycling was found to avoid more emissions compared to incineration. For a ratio equal or lower that 1:0.5, however, incineration was found to be favoured over mechanical recycling. Lazarevic *et al.* (2010) hence conclude that the preference of recycling over incineration becomes questionable for a range of impact categories when the substitution ratio is reduced, as confirmed by Michaud *et al.* (2010). Astrup *et al.* (2009) suggest that a typical material loss equivalates to 10%, in addition to a loss in material quality of 20%. This averagely leads to a 1:0.72 ratio of substituted virgin material in an average recycling process. Substitution of 1:1 seems in this regard unrealistic to achieve.

Fifth, the substituted materials should be assessed. If the plastic waste stream is homogenous and clean, the materials can be turned into the same products hence substituting virgin materials at a high substitution ratio. If the plastic stream is contaminated, the recyclates can be turned into products that could be made from other materials such as fences, garden furniture and pallets. Wood or concrete is then generally substituted (Astrup *et al.*, 2009). In either case, the selection of the appropriate avoided primary production of materials in material recycling systems is decisive (Brogaard *et al.*, 2014; Rigamonti *et al.*, 2014; Turner *et al.*, 2015).

Finally, it must be assessed whether the recycling or recovery benefits outweigh the collection and out-sorting efforts for specific cases (Rigamonti *et al.*, 2014).

When it comes to bioplastics, few studies have analysed the environmental impacts from various end-of-life options (Niaounakis, 2013). According to Vink *et al.* (2003), burning and landfilling of PLA does not generate toxic emissions nor leachate. Landfilling of biodegradable polymers can, however, result in methane emissions, making this option unattractive from a

GWP perspective (Weiss *et al.*, 2012). An LCA study performed by Rossi *et al.* (2015) on dry biodegradable packaging showed that mechanical recycling is the most interesting option for most impact categories, followed by direct fuel substitution. Anaerobic digestion and incineration result in medium performance, whereas landfill and industrial composting generate the highest amounts of environmental stress. The composting of bioplastics does neither substantially improve compost quality nor enables energy recovery, leading therefore to a weak environmental performance. In the case of composting, the results of the study do hence not confirm the validity of the waste hierarchy (Rossi *et al.*, 2015).

Many LCA studies analysing bioplastics neglect the waste management phase because of a lack of data and focus therefore on a production approach, limited to a cradle-to-gate scope. This life cycle phase, however, might strongly influence the conclusions of the analysis (Niaounakis, 2013). With the challenges linked to the recyclability of bioplastics in mind, this concern can be recognized.

2.6 Answers of the literature review to the research questions

The literature review answers to a certain extent the research questions, establishing certain hypotheses for the outcomes of this LCA. In addition, it revealed two knowledge gaps which this thesis aims at filling.

Concerning the first research question, it seems to be a scientific agreement on the fact that mechanical recycling leads to less environmental impacts than incineration. This, however, depends on several factors such as the resin type and impact category analysed, the substitution ratio and the contamination level of the plastic waste. The literature reviewed mainly focuses on various waste management options for plastic waste. Only Lyng and Modahl (2011) assessed the environmental impacts of plastic out-sorting. However, no studies reviewed assessed the environmental impacts of a CS facility compared to household segregation.

When it comes to the second research question, the literature concludes that the production of bioplastics has less impacts on GWP and FDP compared to conventional plastic, but increased impacts for all other impact categories. No conclusions could be drawn on the favoured waste management option for biopolymers. All LCA studies analysing the environmental impacts of bioplastics were only conducted for the production phase, or for the life cycle of a specific resin type. Hence, this thesis aims at filling a knowledge gap by developing an LCA analysing the life cycle emissions of household plastic consumption which contains a share of bioplastics.

3. Case study description

3.1 Waste management in Trondheim

As of 2017, Trondheim had 193501 inhabitants and was thereby the third largest city in Norway (SSB, 2017a). Despite a strong increase in inhabitants, the total municipal solid waste amounts have been decreasing the last decade, from an average of 400 kg/cap/year between 2006 and 2012, to an average of 350 kg/cap/year in 2016 (SSB, 2017b). It is mainly the paper and cardboard but also the residual waste amounts which have been reduced (Trondheim kommune, 2017).

The Norwegian Pollution Act (Forurensningsloven §30) gives the municipalities the responsibility of managing household waste. The municipality is not obliged to handle the waste itself but may hire companies for doing so. In Trondheim, Trondheim Renholdsverk (TRV) was established in 1918 for collecting household waste. Today, the company is owned by the municipality and operates on its behalf. Their operations are financed by household waste fees (Trondheim Renholdsverk, 2017).

Currently, four fractions sorted out from households: paper, plastic, glass and metal, and residual waste. Paper, plastic and residual waste are collected by kerbside collection from wheeled bins, moloks, underground containers, or increasingly from vacuum systems (Unander, 2017). Plastic, paper and cardboard, and glass and metal are first stored and packed in a facility in Heggstadmoen, before they are sent to recycling plants either outside Trondheim or abroad. The residual waste is transported directly to the incineration plant Heimdal Varmesentral for energy recovery feeding into the city's district heating system (Brattebø & Reenaas, 2012; Lausselet et al., 2016; Unander, 2017).

Because the EU targets ambitious recycling rates, presented in section 3.2, the waste management system should be reconsidered. TRV and several waste management companies in the Trøndelag county are therefore planning the construction of a CS facility, which theoretically would increase the recycling rates and lower GHG emissions. The project is called Sentralt EtterSorteringsAnlegg i Midt-Norge (SESAM) and would sort out organic waste as well as plastic waste in several fractions (Trondheim kommune, 2017). The environmental impacts associated with higher plastics out-sorting rates in such a facility will be analysed in this thesis.

3.2 Legal framework

Since 1994, several targets have been set at the European level regarding plastic waste recycling and recovery. In later years, the concept of circular economy has highly influenced EU waste management policies. Consequently, the focus on plastic production and waste management has risen considerably (Sevigné-Itoiz *et al.*, 2015), resulting in the establishment of new targets and visions. In this section, the overarching directive on waste and the more specific directive on packaging are first examined, followed by the targets and visions set forth within the framework of a circular economy (Table 3).

Table 3: Main directives and strategies on plastic packaging waste

Circular Economy Package	Year	Main targets and visions
Waste Framework Directive	2008	1) Waste hierarchy principles 2) 50% by weight of waste materials prepared for reuse and recycling by 2020 3) Obligation of separate collection of paper, metal plastic and glass
Packaging and Packaging Waste Directive	1994 2004 2015	1) Preventing packaging waste 2) Develop reuse systems 3) Minimum 22.5% recycling of plastic 4) Reduce the consumption of lightweight carrier bags
Circular Economy Package	2018	1) Minimum 55% recycling of municipal solid waste by 2025 2) Minimum 50% recycling of plastic packaging by 2025 3) Reduction of municipal waste landfilled to 10% of less of the total amounts generated by 2035
A European Strategy for Plastics in a Circular Economy	2018	1) Reuse or recyclability of all plastic packaging by 2030, eliminating the use of single-use packaging 2) Create viable markets for recycled and renewable plastics 3) Increase the sorting and recycling capacity in the EU 4) Phase out the export of low quality waste

There are two main binding directives regarding plastic waste: the overarching Waste Framework Directive (WFD) and the more specific Packaging and Packaging Waste Directive.

Initiated in 1975 (75/442/EEC), the WFD encapsulates the waste hierarchy principles, presented in 2008. The amended 2008 version aims in addition to the preparation for reuse and recycling of 50% by weight of waste materials by 2020 and requires separate collection of at least paper, metal, plastic and glass (2008/98/EC).

When it comes to plastic packaging, the Directive on Packaging and Packaging Waste of 1994 (94/62/EC) is deemed to be the daughter directive of the WDF. It requires Member States to take measures for preventing waste and develop packaging reuse systems, in accordance with the principles of the waste hierarchy. The amended version of 2004 requires a plastic recycling rate of minimum 22.5%. The directive was further modified in 2015, requiring the reduction in consumption of lightweight plastic carrier bags (European Commission, 2016). A ban on plastics bags has been implemented in several Member States and the topic has given rise to an on-going debate in Norway.

In 2015, the EU identified plastics as a priority area of the circular economy action plan. Consequently, the focus was placed on plastics production and use. Furthermore, new reuse and recycling targets for plastic packaging waste were proposed. The amended directives, adopted by the Council and the European Parliament in May 2018, set the target of 55% material recycling of municipal solid waste by 2025, 60% by 2030 and 65% by 2035. For plastic packaging specifically, the material recycling rate is aimed at being increased to 50% by 2025 and 55% by 2030, going far beyond the requirements of the Packaging and Packaging Waste Directive (European Commission, 2018b). Nonetheless, these directives constitute the foundation of which the Member States develop their national policies. Accordingly, nothing precludes them to be proactive in adopting tougher standards than these minimal requirements.

At the start of 2018, the European Commission released its "Strategy for Plastics in a Circular Economy", which implies reducing waste to a minimum. Moving the plastic value chain in this direction involves improving recycling, promoting reuse and redesigning products, while taking into account the whole life-cycle of products. The strategy sees the potential for plastic recycling as largely unexploited. The Commission will therefore work towards the reuse and recyclability of all plastic packaging by 2030, targeting the elimination of single-use packaging. A combination of waste reduction and increased material recycling is accordingly urged for (European Commission, 2018a).

Nonetheless, as much as 50% of plastics waste collected in the EU is exported, mainly to the Asian market. Before January 1, 2018 more than 85% was sent to China (European Commission, 2018a). It is rather unclear how the exported waste is managed by Chinese operators and whether the Chinese recycling conditions comply with the standards set by the EU (Hestin *et al.*, 2015). As the massive export of plastic waste challenges the implementation of the circular economy, EU's plastic strategy considers the Chinese ban as an opportunity for EU recyclers to develop a stronger European market for recyclates and aims therefore at the out-phasing of plastic waste exports (European Commission, 2018a; Sevigné-Itoiz *et al.*, 2015).

In contrast to the policy areas of biofuels and renewable energies, there is currently no EU-wide legislative framework to support the use of renewable raw materials for plastic solutions (European Bioplastics, 2018).

4. Methodology

In this chapter follows an introduction to LCA and MFA for establishing the important methodological steps required for the following analysis. Based on this methodology, a model description and the development of scenarios will be presented.

4.1 LCA

LCA is a quantitative modelling tool addressing the environmental aspects and potential environmental impacts throughout a product's or service's life cycle from raw material acquisition through production, use, end-of-life treatment and final disposal. The environmental sustainability of a product or service can be determined by comparing in a holistic way the generated impacts with the ones of similar products or services.

The method identifies the impacts through environmental stressors caused by a given measurable activity, the "functional unit" or FU (ISO 14040, 2006). Stressors is a wider definition than emissions associated not only with the release of gases and toxins, but also with land use changes, ozone depletion, eutrophication etc. The stressors are further aggregated into impact categories, avoiding a simple quantification of the different and numerous stressors. There are four steps to the LCA procedure (Figure 3): goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA) and interpretation. It is an iterative process where the steps influence each other (ISO 14040, 2006).

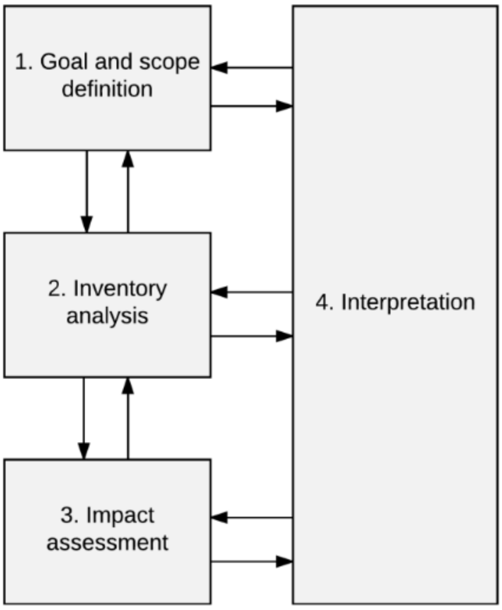


Figure 3: The life cycle assessment framework, modified from ISO 14040 (2006)

4.1.1 Goal and scope definition

The scope of an LCA, including the system boundaries and the level of detail, depend on the purpose of the study. The ISO 14040 states that the goal should include the motivation and intended application of the study, as well as the intended audience. The scope should consist of the FU and the system boundaries, as well as methodological choices such as the allocation procedure, the impact categories selected, the data requirements and quality, and the assumptions (ISO 14040, 2006).

The FU corresponds to a reference flow to which all other modelled flows of the system are related, being as such an essential and quantitative part of the LCA (Baumann and Tillman, 2004). The service provided by a waste management system is to collect, treat and dispose of waste. Accordingly, the FU in such systems is defined as the management of a certain amount of waste of a certain composition (Christensen, 2011).

The scoping of the life cycle phases will be different for a waste management LCA than for a product LCA, because the system does not follow the common phases of raw material acquisition through production, use, end-of-life treatment and final disposal. Instead, the phase of *raw material extraction* includes the substitution of virgin resources by the products created by the waste management system. The *manufacture* phase encompasses all the processes involved in the conversion of resources and materials into the waste management technology. The *operation* phase involves the operation of all parts of the waste management system including inputs (i.e. the use of electricity, water, fuels and chemicals) and outputs (i.e. production of thermal and electric energy, outputs of materials which can be used outside of the waste management system). In the *waste management* phase, equipment and facilities used in the system should be dismantled and accounted for (Christensen, 2011).

4.1.2 Inventory analysis

In the inventory analysis, emissions and resource information are collected for the input and output processes involved according to the goal and scope definition (Christensen, 2011). In addition to the data collection, the step requires calculation and allocation of the different flows and stressors (ISO 14040, 2006).

The materials and energy flows are systemised in the requirement matrix A. Each entry a_{ij} is the quantity of input from process i needed for one unit of output from process j. When connected to a system output, the total amount required from each process can be determined. The output can be intermediate or final, represented by the total output vector, x, and final demand vector, y, respectively. The final demand is the required direct output of the system, which leads to indirect demands further up in the value chain. The total output from the system is the sum of intermediate and final demand, also called production balance (Equation 1) (Strømman, 2010).

$$x = Ax + y \tag{1}$$

The Leontief inverse matrix L (Equation 2) can be derived from the production balance. The coefficients l_{ij} represent the amount of output of process i that is required per unit of final delivery of process j (Strømman, 2010).

$$x - Ax = y$$
$$(I - A)x = y$$
$$x = (I - A)^{-1}y = Ly \tag{2}$$

Two systems can be distinguished in the requirement matrix A: the foreground system (A_{ff}) represents the system under investigation, while the background system (A_{bb}) represents average values associated with economic activities for processes where high resolution is not considered necessary (Strømman, 2010). The foreground and background systems of the requirement matrix are depicted in equation 3.

$$A = \begin{bmatrix} A_{ff} & A_{fb} \\ A_{bf} & A_{bb} \end{bmatrix} \tag{3}$$

Stressors are flows referring to environmental pressures such as emissions and land use. These are systemised in a stressor intensity matrix S. The entries s_{ij} quantify the amounts of stressor i per unit of output from process j (Equation 4) (Strømman, 2010).

$$e = Sx = SLy \tag{4}$$

Ecoinvent 3.2, recognized as the most complete LCA and best quality database for European purposes (Strømman, 2010), is used as the background system for this thesis. It contains 12916 processes and 25950 stressors, as well as interactions between processes and between processes and stressors.

4.1.3 Impact assessment

After having established the total amounts of stressors, the LCIA step evaluates the significance of potential environmental impacts based on the LCI results. The stressors found in the LCI are classified into a few number of environmental impacts. The coefficients c_{ij} of the characterisation factor matrix C tell how much of impact i is generated per unit output of stressor j (Strømman, 2010). After classification of the various stressors, characterizing can be performed, calculating the environmental impacts of the investigated system (Equation 5).

$$d = CSLy \tag{5}$$

ReCiPe is the impact assessment method used in this thesis, providing characterization factors for 18 impact categories. The hierarchist perspective is most commonly applied to account for various possible value choices and is therefore used in this thesis.

GHG emission accounting is a major focus within waste management (Gentil *et al.*, 2009). Accordingly, the results of the GWP impact category will be carefully analysed. The literature study disclosed that GWP and FDP impacts are most commonly assessed when studying waste management systems. It is then of interest to understand the environmental impacts of the systems in a broader sense for divulging possible trade-off effects. The results of FDP, human toxicity potential (HTP), freshwater ecotoxicity potential (FETP), natural land transformation (LTP) and terrestrial acidification (TAP) will hence also be looked upon in detail.

In LCIA, transparency is critical for ensuring a detailed description of assumptions which might influence the results (ISO 14040, 2006).

4.1.4 Interpretation

Because it is an iterative process, this last step considers and interprets continually the findings from the goal and scope definition, from the LCI and from the LCIA. Understandable, complete and consistent LCA results should be provided from this step. In addition, the results should be in line with the defined goal and scope of the study, explain limitations and provide recommendations (ISO 14040, 2006).

4.2 MFA

The LCA performed in this thesis is based on MFA principles. This means that the MFA is only complementary for identifying and quantifying the inventory required to perform the LCA.

MFA is a systematic assessment of the flows and stocks of materials within a system defined in space and time. It connects the sources, the pathways and the intermediate and final sinks of a material. An MFA can be controlled by a simple mass balance comparing all inputs, stocks and outputs of a process (Equation 6). The stock considers accumulation or depletion of materials in a process (Brunner and Rechberger, 2004).

$$\sum_{k_1} m_{input} = \sum_{k_O} m_{output} + m_{storage} \tag{6}$$

This characteristic makes the method attractive as a decision-support tool in resource management, waste management and environmental management (Brunner & Rechberger, 2004).

Figure 4 presents the different steps for conducting an MFA.

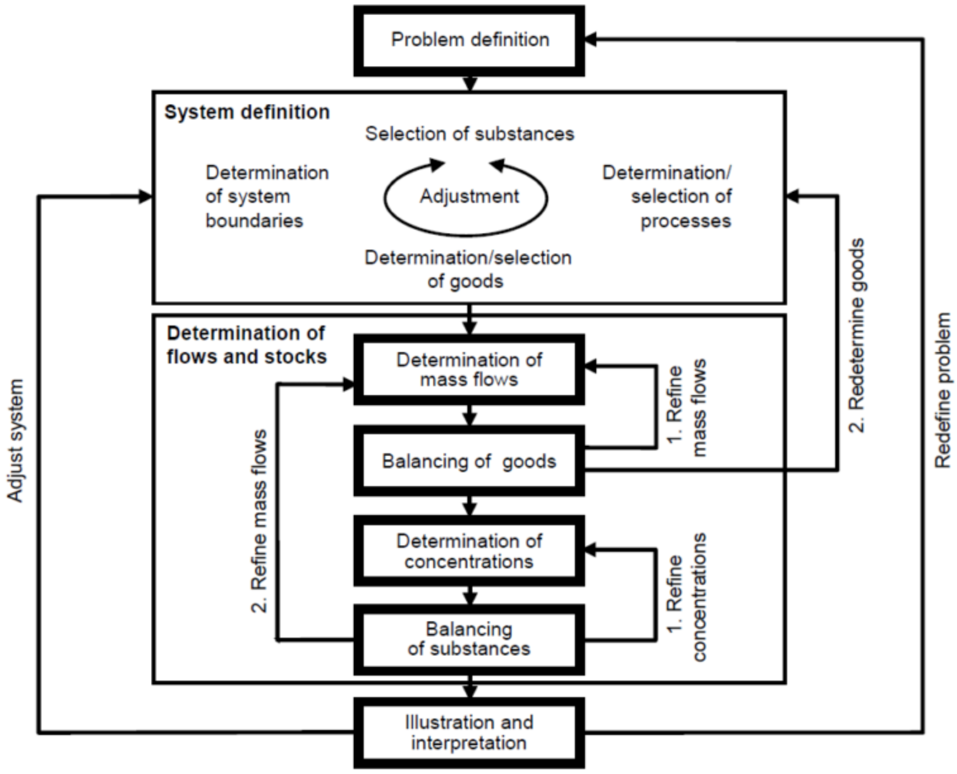


Figure 4: Iterative process for MFA. Source: Brunner and Rechberger (2004).

An analysis starts with the definition of the problem. Following, relevant substances, processes, goods and system boundaries are determined in order to define the system itself. A system can for instance be characterized as a group of physical components connected in a way that forms a unit. The system boundaries are defined in time and space and are greatly dependant on the research question and the data availability.

Further, the mass flows of goods, their balance, and the concentration as well as balance of certain substances can be determined. Transfer coefficients are important in this regard, as they describe the portioning of a substance in a process and are defined for each output good of a processFigure 5).

Figure 5: Illustration of a transfer coefficient

When multiplied by 100, the transfer coefficient gives the percentage of the total throughput of a substance that is transferred into a specific output good. The transfer of substance X into an output flow can be determined by equation 7, which is the mathematical expression of a transfer coefficient. TC_i is transfer coefficient of process number *i*, k_i is the number of inflows, $X_{O,i}$is the outflow number *i*, and $X_{I,i}$is the inflow number *i* (Brunner & Rechberger, 2004).

$$TC_i = \frac{X_{O,i}}{\sum_i^{k1} X_{I,i}} \tag{7}$$

Finally, the stocks and flows can be calculated, and uncertainties considered. During all these presented steps, the procedures must be optimized iteratively, to improve the quality of the data and of the system (Brunner & Rechberger, 2004).

4.3 Model description

This section introduces the developed LCA portraying the current waste management in Trondheim, which will be used as reference scenario. To begin with, the goal and scope definition will present the MFA system on which the LCA inventory is based. In fact, the system boundaries of the LCA are similar to the ones of the MFA. Thereafter, the FU will be presented, followed by the production and end-of-life inventories.

4.3.1 Goal and scope definition

The model presented in Figure 66 represents the flows and processes necessary for producing and managing 1000 kg of plastic packaging waste generated in the municipality of Trondheim in 2017. Five different system boundaries can be defined: the production system (process 0), the waste management system (processes 1 to 12) and the substitution system representing the external markets (processes 11 to 16). The combination of the two latter systems will be designated as the waste management system including substitution, allowing the inclusion of the downstream avoided impacts. The combination of the three systems will be designated as the expanded system, considering the full life cycle impacts of the FU. An inventory was developed for the production system itself, while another is developed for the waste management system including substitution.

For a better understanding of the model, Table 4 lists the flows and processes of the expanded system present in the reference scenario. A quantified flowchart can be found in appendix (A2).

Table 4: List of flows and processes in the reference scenario

Processes

#	Process
0	Plastic production
1	Households
2	Collection
3	Incineration, NO
4	Packing
5	Sorting
6	Cement production
7	Recycling, DE
8	Recycling, Asia
9	Incineration, DE
10	Incineration, Asia
11	Substituted virgin materials, DE
12	Substituted virgin materials, Asia
13	Substituted electricity, NO
14	Substituted coal, DE
15	Substituted electricity, DE
16	Substituted electricity, Asia

Flows

From	To	Note	System boundaries
0	1	To households	Production system
1	2	To collection	Waste management system
2	3	To incineration	Waste management system
2	4	To packaging	Waste management system
3	13	To energy market	Substitution system
4	3	To incineration	Waste management system
4	5	To sorting	Waste management system
5	6	To cement production	Waste management system
5	7	To recycling	Waste management system
5	8	To recycling	Waste management system
5	9	To incineration	Waste management system
6	14	To energy market	Substitution system
7	11	To energy market	Substitution system
8	10	To incineration	Waste management system
9	15	To energy market	Substitution system
10	16	To energy market	Substitution system

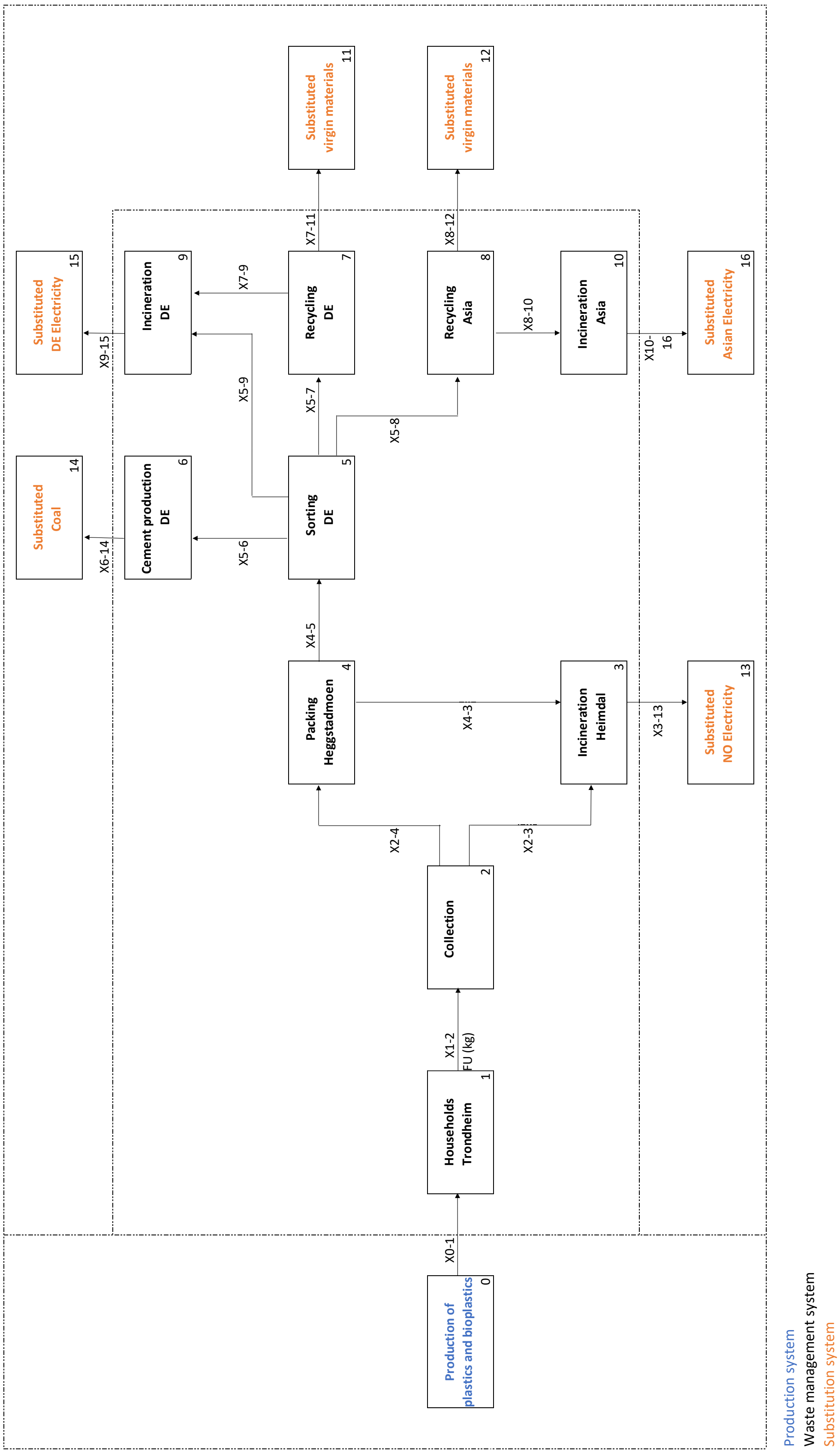


Figure 6: Flowchart of the systems under investigation for the reference scenario

4.3.2 Functional unit

The purpose of the study is to analyse a specific plastic packaging waste out-sorting system. The waste amounts and composition is determined by the region and the period of time for which the LCA is performed (Christensen, 2011). It is common practice that waste management LCA are input-based, as the aim is to treat a certain amount of waste. Therefore, the FU used in this thesis describes the collection and treatment of 1000 kg plastic packaging waste generated from households in Trondheim (X1-2).

A study conducted by Syversen and Bjørnerud (2015) was used for defining the composition of the FU. The authors conduced a detailed composition analysis with respect to plastics for the SESAM region, which included 3 residential areas in Trondheim. Plastics that were out-sorted by households were differentiated from the ones present in the residual waste, making up 20.4% and 79.6% of the total household plastic waste generated respectively (Syversen & Bjørnerud, 2015).

For simplicity reasons, the data presented by Syversen and Bjørnerud (2015) was aggregated in 6 different fractions based on resin types: PE, PET, HDPE, PP, PS and mixed plastics. The aggregation was performed on the one hand for the out-sorted plastic waste and on the other hand for the plastic present in the residual waste. An average of the resulting shares was then calculated. Consequently, three composition vectors resulted from this operation (Table 5).

Table 5: Composition vector of the FU for the reference scenario

Fraction	Plastic in residual waste	Out-sorted plastic waste	Average
PE	38%	34%	36.5%
PET	11%	17%	26.5%
HDPE	7%	8%	13.7%
PP	13%	14%	13.4%
PS	3%	2%	7.0%
Mixed	28%	25%	2.9%
Total	100%	100%	100%

The concept of mixed plastic covers all non-bottle plastic packaging from households, including both rigid and flexible packaging of various polymer types and colours (WRAP, 2018). In this thesis, however, the mixed plastic fraction is an aggregation of different plastic products which do not suit in the above-mentioned categories, for instance black packaging and laminates. LDPE was left out of the analysis and added to the PE-fraction as suggested by Bjørnerud (Bjørnerud, 2018).

Even though the data used is strengthened by being location specific, it must be noted that waste composition varies over time, season and location. Related data is time consuming and expensive to produce, leading to embedded uncertainty in defining the waste heterogeneity (Slagstad & Brattebø, 2013). The presented data is hence a source of uncertainty that will propagate through the following analysis. The resulting plastic fraction shares were comparable with values from Interkommunalt Vann, Avløp og Renovarsjon IKS (IVAR) (Meissner, 2018) and partly with ROAF (Romerike Avfallsforedling IKS, 2017), validating the robustness of this data basis.

The same FU is used for the production and the end-of-life inventories, as the impacts of producing the plastics which are discarded by households must be accounted for when studying the impacts of plastic waste in a life cycle perspective.

4.3.3 Production inventory

A production inventory was developed for assessing the impacts of producing 1000 kg plastics which were later discarded by households in Trondheim in 2017. The correct FU composition was considered (Table 6) and the system boundaries are restrained to the production system. Because the mixed waste fraction is an aggregation of different resin types and its production does not exist as a process in the Ecoinvent database, its amounts were distributed to the other fractions according to a weighted average. For a better understanding of the plastic production process, a flowchart of the production steps can be found in Appendix (A1).

Table 6: Composition of the FU for calculating the production-related impacts in the reference scenario

Fractions	Fossil-based plastics
PE	51.2%
PET	16.5%
HDPE	9.5%
PP	18.6%
PS	4.3%
Total	100%

4.3.4 End-of-life inventory

The end-of-life inventory represents the management of 1000 kg plastic packaging waste generated by households in Trondheim in 2017. The system boundaries encompass the waste management system including substitution. Because it was neither possible to collect data on the current waste flow destinations, nor specific data on the currently used sorting and recycling facilities, the study presented by Lyng and Modahl (2011) depicting the 2011 system for recycling of Norwegian household plastic packaging waste was used as foundation for this analysis. Given that the situation is assumed not to have changed dramatically since 2011, the system description presented in their study was estimated valid to use (Rødvik, 2018).

Collection On the one hand, the out-sorted plastics are driven to the packaging facility in Heggstadmoen, 16km outside Trondheim. The FU with the composition of the out-sorted plastic waste was used for this flow (Table 5). On the other hand, plastics contained in the residual waste are transported directly to the incineration facility in Heimdal for energy recovery, 14km outside Trondheim. The FU with the composition of the plastics in the residual waste was used for this flow (Table 5). The heat generated from the incineration process is assumed to replace the Norwegian electricity mix as heating source in households thanks to district heating. The efficiency of the incineration plant is of 80% (Christensen, 2011). The LHV of the various plastic resins was differentiated based on Shonfield 2008 and are presented in appendix (A7). The transport is assumed performed by fully loaded trucks of 21tonn capacity and is only calculated for one-way distance.

Packing In Heggstadmoen, the out-sorted plastics are compressed and balled. A fraction of 2% typically made up of contaminated products and/or products not suitable for recycling is diverted to the incineration plant (Unander, 2017). A capacity of 1440 tons/year was assumed for the facility (Trondheim kommune, 2016).

Sorting The plastic balls are transported to sorting facilities in Germany under the responsibility of Gønt Punkt Norge (Grønt Punkt Norge, 2018). The waste is transported over 440 and 840 km by truck and train respectively, based on Lausselet *et al.* (2017) and Lyng and Modahl (2011). For all transport processes but the collection process, it is assumed that the lorries have a capacity of 16-32 tonnes. The lorries were further set to comply with Euro IV emission standards, as done by Lyng and Modahl (2011) and Shonfield (2008).

Due to a lack of data, only one facility was modelled assuming that the plastic waste generated in Trondheim is treated in one location only. Some German plants both sort and recycle plastic waste, while others perform only one operation. For clarity reasons, two facilities are modelled, assuming that the waste is sorted out in the sorting facilities, then recycled in material recycling facilities. On site, the waste is sorted according to different plastic resins and potentially to colours (Grønt Punkt Norge, 2018). The facility is considered to have a capacity of 91000 tons/year (Lyng & Modahl, 2011).

18.9% of the incoming materials, being in this case mixed waste, is out-sorted and wasted due to a lack of quality and recyclability (Romerike Avfallsforedling IKS, 2017). Half of the wastage is incinerated, substituting the German electricity mix. The German incineration plant producing electricity has an efficiency of 30% (Christensen, 2011). The other half is used as feedstock in cement production, substituting coal (Lyng & Modahl, 2011). It should be noted that cement kilns accept a wide range of different waste materials, so plastic waste would represent just one potential fuel options for this process. It is hence possible that the plastic waste would actually substitute other secondary fuels (Shonfield, 2008), but this option was not further considered in this thesis.

Because no plastic sorting facility construction process exist in Ecoinvent database, a paper sorting facility construction process was used, as it is assumed that the out-sorting technologies are comparable.

Recycling After the materials have been sorted by resin type, 72% of the recyclable plastics is transported over 270km by truck to recycling facilities in Germany. The remaining 28%, only constituted of PE and PET resins, is transported 10605 nautical miles by boat and 200km by lorry to Asia (Lyng & Modahl, 2011). Until January 1. of 2018, China was the world's biggest importer of plastic waste. The market had become important because of its low costs, as it took mainly care of plastics of lower qualities. Up to that date, no demand existed for these recyclates in Western countries, it was therefore profitable to send these materials overseas instead of recycling them in Europe. From January 1. however, China banned the import of foreign waste (Reuters, 2018). This has created a big challenge for European countries, as the waste is piling up due to a lack of European recycling capacity. Most likely, new markets in Asia will develop for instance in Malaysia and Vietnam, replacing Chinas importer role (Reuters, 2018). Because no stable markets have developed yet, this thesis assumes that the export flow is unchanged. The encompassing term of "Asia" will be used.

In the recycling facilities, the plastic is washed and dried, before it is shredded and melted. Through extrusion, plastic pellets are manufactured. The recycling process requires 8 g sodium hydroxide (Arena, Mastellone, & Perugini, 2003), 2.5 m^3 of water (Hopewell *et al.*, 2009), 0.2 kWh of electricity (Lausselet *et al.*, 2017) and 2 litres of diesel (Astrup *et al.*, 2009) per kilo plastic waste. This combines the requirements of the recycling treatment itself and of fuel for auxiliary vehicles onsite. The German and Chinese facilities were assumed to have a capacity of 70000 tons/years and 36000 tons/year respectively (Gu *et al.*, 2017).

20% of the input materials are wasted in the recycling process, and used as fuel for incineration, substituting the German and Asian, in this case Chinese, electricity mix (Grønt Punkt Norge, 2018; Lyng & Modahl, 2011). Waste from the sorting and recycling processes could consist of various materials in addition to plastics, such as food or paper labels which are present in small quantities in plastic packaging. Due to difficulties in finding the share of non-plastic material, this thesis assumes that all waste flows are made up of plastics only, as suggested by Lyng and Modhal (2011). As the exact composition of the waste flow from the recycling process is unknown, 20% of each fraction was considered to be wasted. Also the Asian incineration plant has an efficiency of 30% as it is assumed to produce electricity (Christensen, 2011). It can hence be concluded that the amounts of collected waste differ from the amounts of recycled waste, as there are quite a few losses through the value chain.

HDPE, PP, PET and PS resins are assumed to substitute virgin plastic materials of the same resin type (Lazarevic *et al.*, 2010; Lyng & Modahl, 2011). As PE is an aggregation of different polyethylene types, it was assumed that the fraction substitutes LDPE granulate, based on the Norwegian business Norfolier GreenTec. The mixed waste fraction was assumed to substitute cleft timber produced in Europe (Astrup *et al.*, 2009; Rigamonti *et al.*, 2014; Shonfield, 2008). Materials other than wood could have been substituted in the case of low quality products, but this requires a detailed market analysis of the related products, which was outside the scope of this thesis. A virgin material substitution ratio of 1:1 was used based on a common approach in the literature (Rigamonti *et al.*, 2014; Turner *et al.*, 2015). This is most likely an over-simplification given that the substitution ratio is the product of material quality loss, which relates to changes in the inherent technical properties of a waste material, and of the market substitution ratio, which reflects market elasticity by defining the amount of a primary product that is substituted as a consequence of the production of a secondary product (Turner, 2015).

Further considerations First, because the boundaries of the waste management system are set from the point where plastics leave the households, this analysis does not include the environmental impacts linked with the rinsing of plastics at the household level. Because Grønt Punkt Norge recommends households to use cold water for rinsing, the process is likely to stand for negligible environmental impacts when the total life cycle is assessed (Lyng & Modahl, 2011). This process was therefore left out of the analysis. Second, the construction of the facilities is accounted for in all processes, assuming a lifetime of 10 years (Ecoembes, 2015). Third, the process of plastic processing factory construction is constantly applied for all facilities as Ecoinvent does not differentiate between the different facility types modelled in this inventory.

4.4 Scenario development

Two main scenarios will be presented, each one with sub-scenarios. On the one hand, the environmental impacts of plastic packaging out-sorting options will be assessed when the waste flow only consists of petroleum-based plastics (blue rectangle of Table 7). On the other hand, the life cycle impacts of bioplastic introduction in the household plastic consumption will be studied, relying on a realistic CS system (green rectangle of Table 7).

Table 7: Overview of the scenarios content

	Reference	CS	CS ideal
Petroleum-based plastics	x	x	x
10% bioplastics		x	
25% bioplastics		x	

4.4.1 Central sorting scenario

4.4.1.1 Goal and scope definition

Two CS scenarios are developed for answering the first research question, analysing the environmental impacts of sorting out plastic waste in a CS facility rather than at the household level.

The first scenario is modelled as a *likely realistic scenario*, representing the introduction of the SESAM facility. No detailed operation plans regarding the fractions and the way these should be sorted out have been worked out yet. Inspiration was therefore taken from the ROAF and IVAR models. ROAF is currently the only operating CS facility in Norway and could therefore provide operational data. ROAF collects the organic waste in green bags and residual waste including plastics in blue bags. The organic waste is first segregated, while the latter fraction is conducted through the facility where the various plastic resins (plastic films, PE, PET, PP and mixed plastics) are identified and separated. IVAR in the Stavanger region is in the phase of building such a facility, which should be operational from the end of 2018. The latter facility will have two main differences compared to ROAF. First, organic waste will be collected at the household level and directly sent to biological treatment, without being diverted by the CS facility. Second, the fractions of LDPE, HDPE and PP will be washed and extruded on-site. PET, PS and mixed plastics are planned to be sent, as for ROAF, to recycling facilities in Germany (Meissner, 2018).

Based on ROAF‘s data, an out-sorting efficiency of 36% was assumed for this realistic CS scenario (Callewaert 2017). It is assumed that the SESAM facility will not be operative before 2025 and is therefore only modelled for this year.

The second scenario represents an *ideal situation*, studying the effects of 82% plastic out-sorting of the CS facility. This drastic increase in sorting efficiency is based on the likely development of incoming materials, of household sorting skills and of perfect on-site sorting.

First, EU regulators declared in the plastic strategy that all plastic packaging entering the EU market in 2030 should be recyclable or reusable (European Commission, 2018a). This will lead to changes in the composition of the waste flow, reducing the amounts of non-recyclable fractions such as black plastic or laminates extensively. Despite this likely future change, the composition of the FU was left unchanged because the FU is based on results from a composition analysis, which are by their nature highly uncertain. Small changes in the FU to account for future uncertain changes would have small effects but would increase the uncertainties. By increasing the sorting efficiency, this future evolution is nonetheless accounted for.

Second, the household out-sorting skills could be largely improved. Waste materials are currently not source separated property at the household level, as documented by the composition analyses. In fact, about half of the currently generated organic waste ends up in the residual waste (Callewaert, 2017; Rem, 2018). When this waste flow is conducted through the facility, it pollutes the plastic materials. These are then sorted out as non-recyclable materials, even if they have a chemical composition allowing recycling. The improvement potential of the household out-sorting skills is thus high and could be increased by for instance improved knowledge, awareness or fees (Rem, 2018). In fact, as much as 70% of the incoming plastic waste could theoretically be out-sorted and sent to recycling (Romerike Avfallsforedling IKS, 2017). Raadal *et al.* (2016) estimated the potential to be 74% for the total Norwegian household plastic waste.

Third, the out-sorting potential of the facility itself could also be increased. In fact, this aspect very much depends on market forces and the demand for out-sorted materials. If the demand is to increase, which is likely to be the outcome of the current EU policies, the theoretical potential for plastic out-sorting could be higher than they currently are.

The system description (Figure 7) is slightly changed compared to the reference scenario, together with its related flows and processes (Table 8). The quantified flowcharts can be found in appendix for the realistic and ideal scenarios (A3 and A4, respectively).

Table 8: List of flows and processes in the CS scenarios

Processes

#	Processes
0	Production of plastics and bioplsatics
1	Households
2	Collection
3	Central Sorting
4	Incineration, NO
5	Recycling, Asia
6	Incineration, Asia
7	Recycling, DE
8	Cement production
9	Incineration, DE
10	Substituted electricity, NO
11	Substituted electricity, Asia
12	Substituted virgin materials, DE
13	Substituted virgin materials, Asia
14	Substituted coal, DE
15	Substituted electricity, DE

Flows

From	To	Note	System boundaries
0	1	To households	Production system
1	2	To collection	Waste management system
2	3	To central sorting	Waste management system
3	4	To incineration	Waste management system
3	5	To recycling	Waste management system
3	7	To recycling	Waste management system
4	10	To energy market	Substitution system
5	6	To incineration	Waste management system
5	12	To material market	Substitution system
6	11	To energy market	Substitution system
7	8	To cement production	Waste management system
7	9	To incineration	Waste management system
7	13	To material market	Substitution system
8	14	To energy market	Substitution system
9	15	To energy market	Substitution system

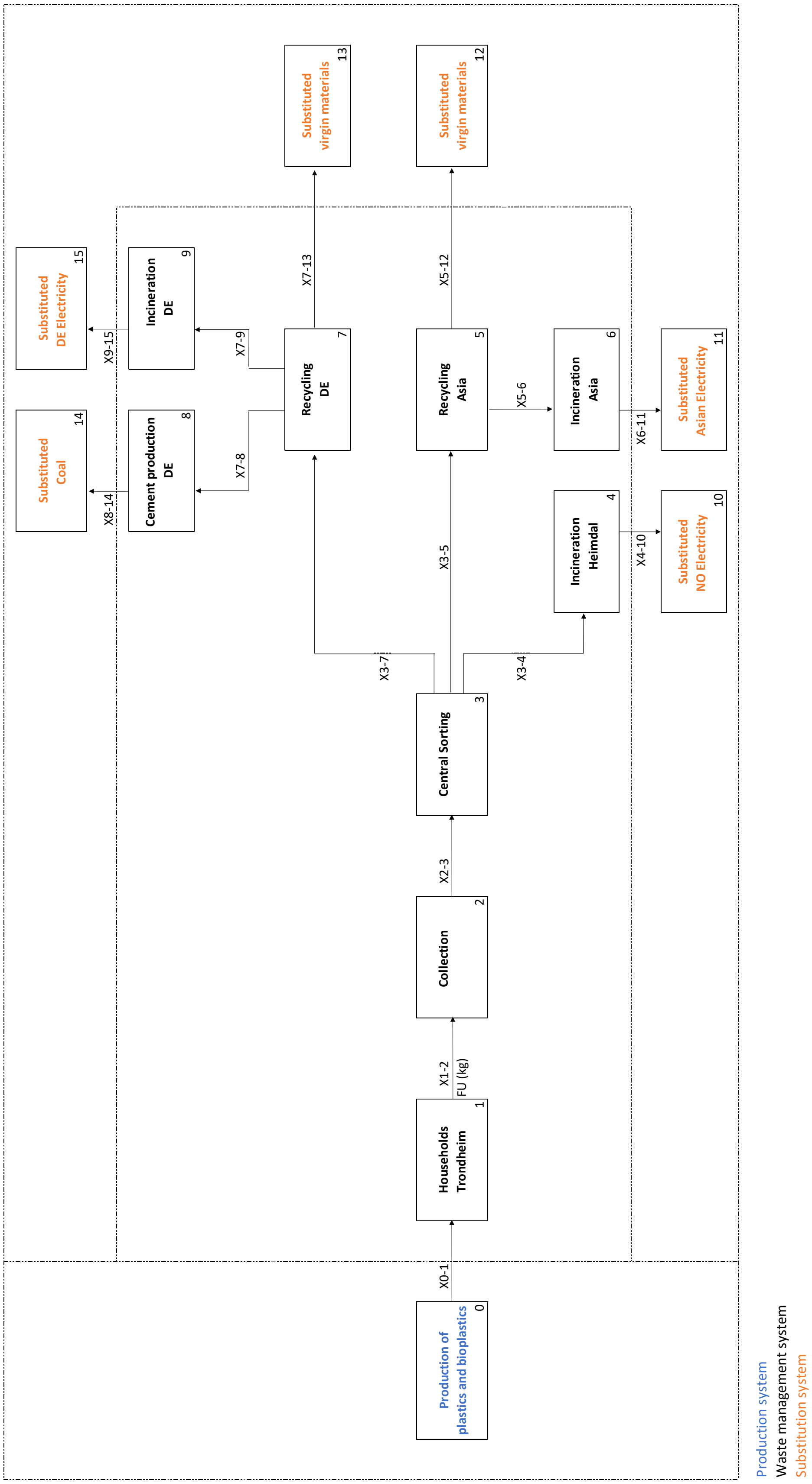


Figure 7: Flowchart of the systems under investigation for the CS scenario

4.4.1.2 Functional unit and production inventory

The average FU composition was used as only one plastic waste flow is generated from households (Table 5). Due to the inherent uncertainty of the composition analysis data, it was decided to keep the FU constant even though the scenario is modelled for 2025. The production inventory is therefore similar to the one presented in section 4.3.3.

4.4.1.3 End-of-life inventory

The same inventory will be used as for the reference scenario, however, with some changes regarding the processes of central sorting and its related flows. It is assumed that the same fractions are out-sorted: PE, PET, PP, PS, HDPE and mixed waste. Theoretically, more fractions could be sorted out with the installation of additional optical readers, but this depends on economic considerations (Askham & Raadal, 2016). However, this was not further explored as it was out of the scope of this thesis. Only the arisen differences compared to the initial inventory will be described in this section.

Central sorting In the realistic CS scenario, 64% of the incoming plastic waste is directly sent to incineration (Callewaert, 2017). In the ideal scenario, this flow is decreased to 16.8%. The operation of the facility requires 42.9 kWh/tons electricity, 1.3 kWh/tons diesel and 6.54 kWh/tons heat (Unander, 2017) and is assumed to have a capacity of 39000 tons/year (Romerike Avfallsforedling IKS, 2017). As for the sorting process in the reference scenario, a paper sorting facility construction process was used for modelling the facility itself, as it is assumed that the out-sorting technologies are comparable.

Recycling After the materials have been sorted out by resin type, 75% of the recyclable plastics are transported to recycling facilities in Germany whereas 25% are transported to Asia. It can be noted that the flow to the Asian market is slightly decreased, which is assumed realistic in the light of the current European plastic debate and on the Chinese ban on import of foreign waste. As for the reference scenario, only PE and PET resins are sent overseas (Lyng & Modahl, 2011).

4.4.2 Bioplastics

4.4.2.1 Goal and scope definition

The bioplastics scenarios are developed for answering the second research question. Bioplastics are increasingly becoming a part of the plastic market and thereby of the plastic waste stream. Analysing the environmental effects of changing the FU to include bioplastics is of interest. Two scenarios studying the effects of bioplastics on the expanded system are developed: a *high version* and a *low version*, reflecting 10% and 25% of bioplastic share in the

FU composition respectively. It must be noted that the 25% scenario reflects a significant penetration of bioplastics on the market, but that such an increase in amounts is necessary for making this analysis of interest.

4.4.2.2 Functional unit

The composition of the FU is altered to include bioplastics in addition to conventional plastics, creating two additional composition vectors (Table 9). The share of petroleum-based plastics is based on the average composition vector as only one plastic waste flow is generated from households (Table 5).

The share of bioplastic fractions follows the bioplastic production data of 2017 (European Bioplastics, 2017). As bioplastics are mainly used as packaging, they are likely to have a short lifetime. A composition of the stock outflow (waste) similar to the composition of the inflow (the production share) is hence well-founded. The values are scaled-up for reflecting a 10% and 25% share of the FU. Because the bioplastics currently only represent 1% of the plastic market, it is assumed that a waste management system including a CS facility will be in place when higher shares are obtained. Therefore, the realistic CS system is used, modelled for 2025 (Figure 7). The petroleum-based fractions are left unchanged but are scaled down to represent 90% and 75% of the FU.

The quantified flowcharts can be found in appendix (A5 and A6 for the 10% and 25% bioplastic scenario respectively).

Table 9: Composition vectors of the FU for the bioplastic scenarios

	Fraction	10% bioplastics	25% bioplastics
Petroleum-based plastics	PE	32.0%	27.9%
	PET	10.3%	9.0%
	HDPE	5.9%	5.2%
	PP	11.6%	10.1%
	PS	2.7%	2.3%
	Mixed	23.5%	20.5%
Bio-based plastics	BIO-PET	3.7%	6.6%
	PA	1.7%	3.0%
	Bio-PE	1.4%	2.4%
	Others	1.3%	2.3%
Biodegradable plastics	Starch	2.6%	4.7%
	PLA	1.4%	2.6%
	PBAT	0.7%	1.3%
	PBS	0.7%	1.2%
	Others	0.5%	1.0%
Total		100%	100%

4.4.2.3 Production inventory

For allowing a comparison of the production impacts of fossil-based plastics and bioplastics, a production inventory for the FU including bioplastics was developed. For the petroleum-based fractions, the same inventory was used as in the reference scenario.

Existing processes for bioplastic production are, nonetheless, scarce in Ecoinvent, making a simplification of the FU necessary (Table 10). In fact, only the production processes for starch and PLA are available in the database. Because bio-PET and bio-PE are large factions within the bioplastic currently produced, their inventory was created especially for this analysis, based on data from the literature. The inventory of the bio-PET follows the methodology used by Tabone *et al.* (2010) and Hottle *et al.* (2017). Ethylene glycol was subtracted within the Ecoinvent database for PET resin and replaced with an equivalent amount of bio-based ethanol derived from Brazilian grown sugar cane. It must be noted that the processes of ethanol dehydration, oxidation and hydration were left out of the analysis because of technical difficulties, but that the results were estimated to be robust and in line with the literature. It must, however, be noted that the production results of this fractions are likely to be slightly underestimated because of this simplification. An overview of the bioplastic production processes is found in appendix (A1). The inventory of bio-HDPE production was modelled following the methodology and data presented by Belboom *et al.* (2016) and was used for representing the bio-PE fraction. The rest of the bioplastic fractions were equally distributed among the starch, PLA, bio-PET and bio-PE fractions because no literature presenting their related production inventories was found.

Table 10: Composition vectors of the FU for calculating the production-related impacts in the bioplastic scenarios

Fractions	**10% bioplastics**	**25% bioplastics**
PE	46.0%	38.4%
PET	14.8%	12.4%
HDPE	8.5%	7.1%
PP	16.7%	14.0%
PS	3.9%	3.2%
Starch	2.7%	6.9%
PLA	1.9%	4.8%
Bio-PET	3.5%	8.8%
Bio-PE	1.8%	4.6%
Total	100%	100%

4.4.2.4 End-of-life inventory

The same inventory will be used as for the realistic CS scenario, however, with some changes in the flows related to the CS and the recycling processes. Only the arisen differences compared to the initial inventory will be described in this section.

Central Sorting The literature review disclosed that biodegradable plastics were not suitable for material recovery. The starch, PLA, PBAT, PBS and Others fractions are consequently assumed out-sorted in the CS facility and sent to incineration. In addition, polyamide (PA) and other bio-based plastics were also diverted to the incineration plant, as their amounts were assumed to be too small for being collected separately and be mechanically treated in a cost-effective way. 64% of the incoming materials are hence directly sent to the Norwegian incineration plant in both bioplastic scenarios.

Most produced bioplastic fractions are made from biomass, except for PBAT which is fossil-based but biodegradable. Because the main reason for developing bioplastics is to reduce the dependency on fossil fuels and that the environmental benefits achieved by biodegradability are questionable, it can be assumed that fossil-based biodegradable plastics will rapidly be phased out. As these modelled scenarios represent a future situation, it can be assumed that all presented bio-based and biodegradable plastics are made from renewable sources, also PBAT.

The incineration process used in the reference and CS scenarios could not be applied to the bioplastic fraction as they are made from renewable resources and are therethrough biogenic. The emissions related to their combustion are hence perceived as carbon neutral (Iwata, 2015). The operation of the incineration plant had, nevertheless, to be modelled: the plant requires 117 kWh electricity and 25 kWh oil per ton treated plastic waste (Unander, 2017). Some additional emissions from the combustion process will also occur such as dioxins and carbon monoxide, but these are assumed to be negligible in regard to the final results and were therefore left out of the analysis (Lausselet, 2018). Bioplastics have lower LHV than conventional plastics (Laußmann *et al.*, 2010), which is taken accounted for when calculating the substituted electricity amounts (A7).

Recycling Only the PE and PET bioplastic resins can be recycled together with conventional plastic types. These are therefore the only bioplastic resins diverted from the CS facility to the recycling process in Germany as they are added to the conventional plastic stream. The recycled bioplastics are accordingly assumed to substitute for conventional plastics. The amounts and types of plastics sent to Asia are similar to the CS scenarios.

4.5 Sensitivity analysis

A sensitivity analysis is used for assessing the robustness of certain parameters, and thereby their influence on the system variables. Input variables and assumptions are deliberately changed one at a time, to analyse how they affect the outcome of the modelling. The changes in results are measured through the sensitivity ratio (SR) which is the fraction of relative change in the results (R) over the relative change in the input parameter (P) (Equation 8) *(*Sandberg *et al.*, 2017).

$$SR_p = \frac{\Delta R / R_0}{\Delta P / P_0} \tag{8}$$

The analysis was conducted for four different system boundaries: (1) the production system, (2) the waste management system, (3) the substitution system and (4) the waste management system including substitution. This disaggregation allows an identification of the most sensitive parts of the system. The results of the waste management system with and without substitution can then be compared, highlighting the influence of the latter system. The analysis was performed on the results of the realistic CS scenario, allowing a comparison of the system with and without bioplastics.

For the production system, only the increase in bioplastics production was assessed by comparing the effects of 10% and 25% bioplastics in the FU.

The waste management system was analysed in greater depth, as it is the most complex system influenced by a large number of parameters. First, the performance of the facilities was investigated for the out-sorting efficiency of the CS facility, for the efficiency of the recycling facility, for the efficiency of the incinerators, for the diesel consumption in these various facilities and for the electricity consumption of the recycling facilities. The performance of the CS facility was assessed by comparing the results for the realistic and ideal scenarios. Second, the importance of the flow to the Asian market (X3-5) and to the European market (X3-7) was analysed. The effect of managing all plastic waste in Europe is aimed at being studied, as this might be a likely outcome of China's ban on foreign waste. Third, the recyclability of bioplastics was assessed by increasing the flow X3-7 for including a larger share of bioplastics. In line with the European target aiming at all packaging waste to be reusable or recyclable by

2030, the bioplastics should also experience an increase in recyclability. Fourth, the sensitivity of the FU composition was investigated: on the one hand when only petroleum-based plastics are included, and on the other hand when the share of bioplastics is increased. Most parameters were changed arbitrarily by slightly increasing their values.

The sensitivity of the substitution system was investigated by looking at the LHV of the various resin types, both for electricity production and cement production.

The sensitivity of the waste management system including substitution was studied for the same parameters as for the waste management system, allowing the comparison of the systems with and without substitution.

5. Results

For informing the first research question, the environmental effects of various out-sorting options are assessed by comparing the LCA results of the waste management systems of the reference and CS scenarios. For informing the second research question, the influence of bioplastics on the life cycle emissions of household plastic consumption is analysed by comparing the LCA results of the expanded realistic CS scenario for three different FU compositions: the first made up of petroleum-based plastics only, the second and third comprising 10% and 25% bioplastics respectively. Further, the resulting recycling rate will be presented for each developed scenario. In addition, the effectiveness of upstream and downstream strategies will be assessed. Finally, the results of the sensitivity analysis will be displayed.

In order to improve transparency, the environmental contributions are organized in upstream impacts (i.e. production-related impacts), downstream impacts (i.e. waste management-related impacts) and avoided impacts (i.e. impacts of substituted energy and material), as recommended by Gentil *et al.* (2009).

5.1 Waste out-sorting options

Figure 8 presents the environmental burdens from the various plastic packaging waste out-sorting options for a selection of impact categories. The system boundaries of the waste management system including substitution are used.

The results will be described for each impact category individually, before analysing recurring patterns. The impacts themselves can be divided into (1) the direct impacts from the waste management system represented by the blue bars, (2) the avoided impacts through substitution depicted by the green bars and (3) the net total impacts, representing the sum the two first categories in the red bars. The net impacts will first be described for giving an overview of the situation. The change in net impacts will thereafter be explained by analysing on the one hand the avoided impacts through substitution and on the other hand the direct impacts from the waste management system.

The results are normalized against the net impacts of the reference scenario for each impact category. The results for the remaining impact categories can be found in appendix (A8), together with the exact values of the environmental burdens (A11-A13).

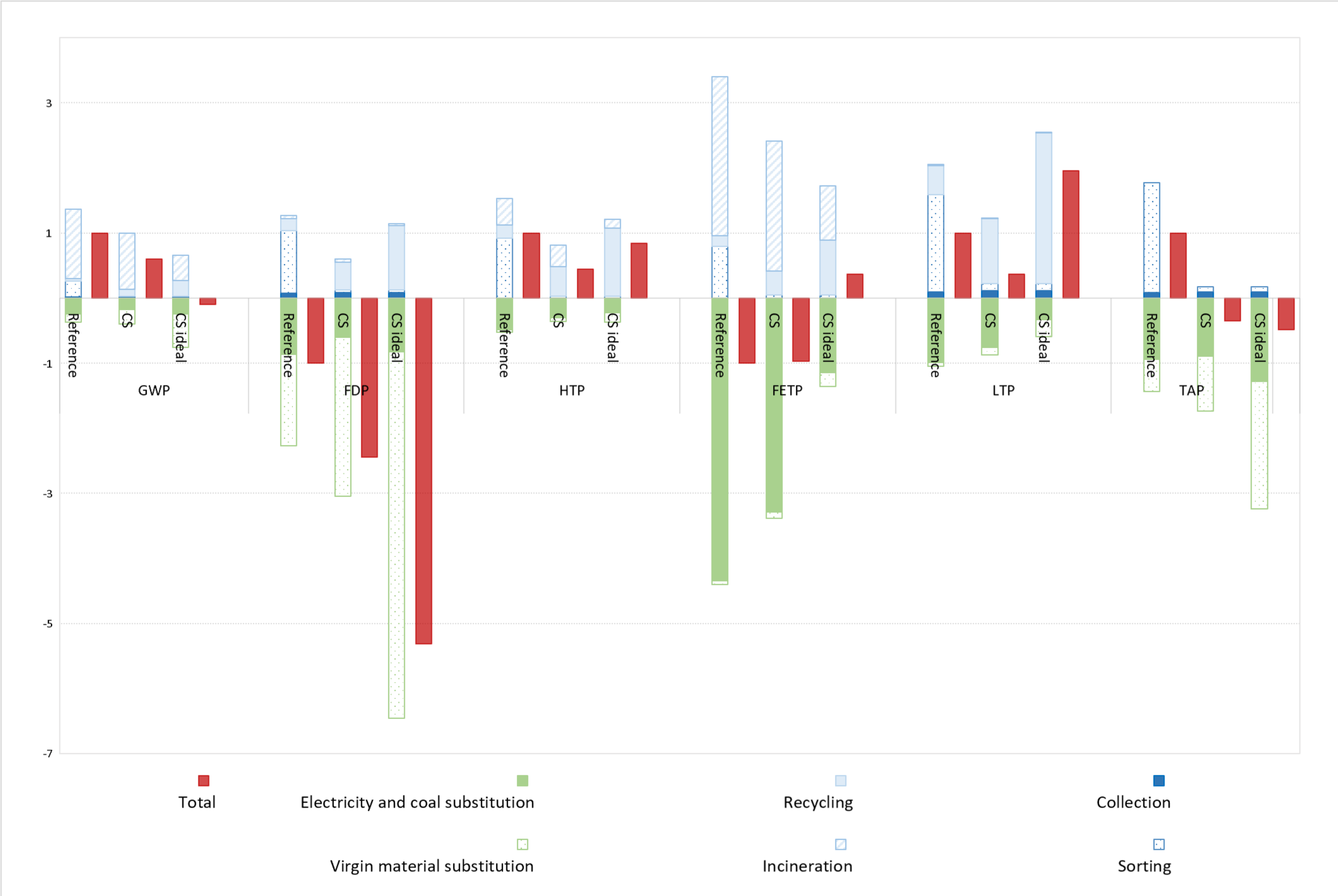


Figure 8: Results comparing the impacts of the out-sorting options for selected impact categories. Normalized values against the net impacts of the reference scenario. For acronyms, see p.ix.

For **GWP**, the reference scenario leads to the highest net environmental burdens, followed by the realistic CS scenario and finally by the ideal CS scenario. The latter scenario is the only one generating negative impacts. This evolution is on the one hand explained by an augmentation in avoided impacts as more virgin materials are substituted. On the other hand, the direct impacts are lowered as less materials are incinerated. The recycling process impacts are slightly increased in the CS scenarios as more plastics are recycled, but these do not influence the dynamics of the above-described factors. For GWP, increased material recycling results in environmental benefits.

For **FDP**, the negative net impacts of the reference scenario are drastically reduced with higher out-sorting rates. The effect of substituting virgin materials is tremendous, which is not surprising as the production of petroleum-based virgin materials is avoided, which directly influences fossil depletion. The direct impacts are reduced from the reference scenario to the realistic CS scenario, mainly because of the suppression of the individual sorting and packing processes. In contrast, the recycling-related impacts are increased as more materials are recycled in the ideal scenario. A small share of the direct burdens is related to the combustion of fuel during the collection phase, which remains equal in all scenarios. When combined, the benefits of avoided impacts outweigh the effects of direct impacts in all cases. For FDP, increased material recycling results in environmental benefits.

For **HTP**, the positive net impacts of the reference scenario are slightly decreased in the realistic CS scenario, suggesting higher environmental benefits of sorting out plastics in a CS facility. However, the net impacts increase in the ideal CS scenario, resulting in comparable impacts amounts to the reference scenario. This concave-shaped development has two main reasons. On the one hand, the category is sensitive to the substituted electricity. As less materials are incinerated, the avoided impacts decrease. On the other hand, the direct impacts are reduced from the reference to the realistic CS scenario, primarily due to the suppression of the individual sorting and packing processes. However, the recycling-related impacts increase with more recycled materials, explaining the increase in net impacts experienced in the ideal CS scenario.

For **FETP**, the negative net impacts are comparable for the reference and the realistic CS scenario. However, they do increase and become positive for the ideal CS scenario. This is explained by the reduction of substituted electricity as less materials are incinerated. Because Norwegian electricity is produced mainly from hydropower, its substitution would reduce the impacts on the freshwater ecosystem. An increase in virgin material substitution thereby leads to a reduction of avoided impacts in the CS scenarios for this impact category. The direct impacts are mainly caused by the incineration process but are diminished as more waste is out-sorted. In contrast, the impacts from the recycling process are increased as more materials are recycled, but these do not outweigh the reduction of the incineration-related impacts. For FETP, the environmental burdens increase with higher out-sorting rates.

The **LTP** net impacts follow the same concave pattern as the HTP category. The net impacts are reduced from the reference to the realistic CS scenario but get much higher in the ideal CS scenario. This concave-shaped development has two main reasons. On the one hand, the category is slightly sensitive to the substituted electricity. As less materials are incinerated, the avoided impacts decrease. On the other hand, the direct impacts are reduced from the reference to the realistic CS scenario with the suppression of the individual sorting and packing processes. In contrast, the impacts from the recycling process are increased significantly as more materials are recycled, which explains the increase in net environmental stress from the ideal CS scenario.

Finally, for **TAP**, the net positive impacts of the reference scenario become negative with higher out-sorting rates. On the one hand, avoided impacts are increased with higher virgin material substitution. On the other hand, the direct impacts are reduced from the reference to the realistic CS scenario with the suppression of the individual sorting and packing processes. For TAP, the increase in material recycling leads to environmental benefits.

Some conclusions can be drawn, and some general patterns identified when comparing these selected impact categories with each other.

(1) The net environmental load of the waste management system decreases in all selected impact categories when more plastic waste is out-sorted in the realistic CS scenario compared to the reference scenario. The same conclusion can be drawn for the ideal CS scenario, except for the toxicity impacts and the land transformation impacts, which increase with higher out-sorting rates.

(2) For GWP, FDP and TAP, the effects of substituting virgin materials are valuable. Material recycling is for these categories beneficial. In contrast, for HTP, FETP and LTP, the effects of substituting Norwegian electricity made from hydropower is of importance. Incineration is for these categories beneficial.

(3) The direct impacts from the waste management system are only rarely counterbalanced by the avoided impacts, as few scenarios result in negative net impacts.
(4) The decrease in direct impacts is mainly due to the suppression of the individual packing and sorting process from the reference scenario, as well as the decrease of impacts from incineration. The increase direct impact, however, relate to higher burdens from the recycling process.

5.2 Bioplastics

Figure 9 presents the effects of bioplastic introduction in the household plastic consumption. The figure compares the environmental impacts of producing and managing a FU comprising petroleum-based plastics only with a FU composed of 10% and of 25% bioplastics respectively. The inventory builds on the boundaries of the expanded system of the realistic CS scenario, to which the petroleum-based scenario is equivalent. The bioplastic sub-scenarios only differ by their FU composition.

The results are divided into (1) production impacts illustrated by yellow bars, (2) direct impacts from the waste management system depicted by blue bars, (3) avoided impacts through substitution depicted by the green bars and (4) the net total impacts, representing the sum of the three mentioned impacts in the red bars. The net impacts will first be described for giving an overview of the situation. The change in net impacts will thereafter be described by analysing on the one hand the avoided impacts, and on the other hand the direct impacts from the production and the waste management system.

The results are normalized against the net impacts of the reference scenario for each impact category. The results for the remaining the impact categories can be found in appendix (A9), together with their exact values of the environmental burdens (A13-A15).

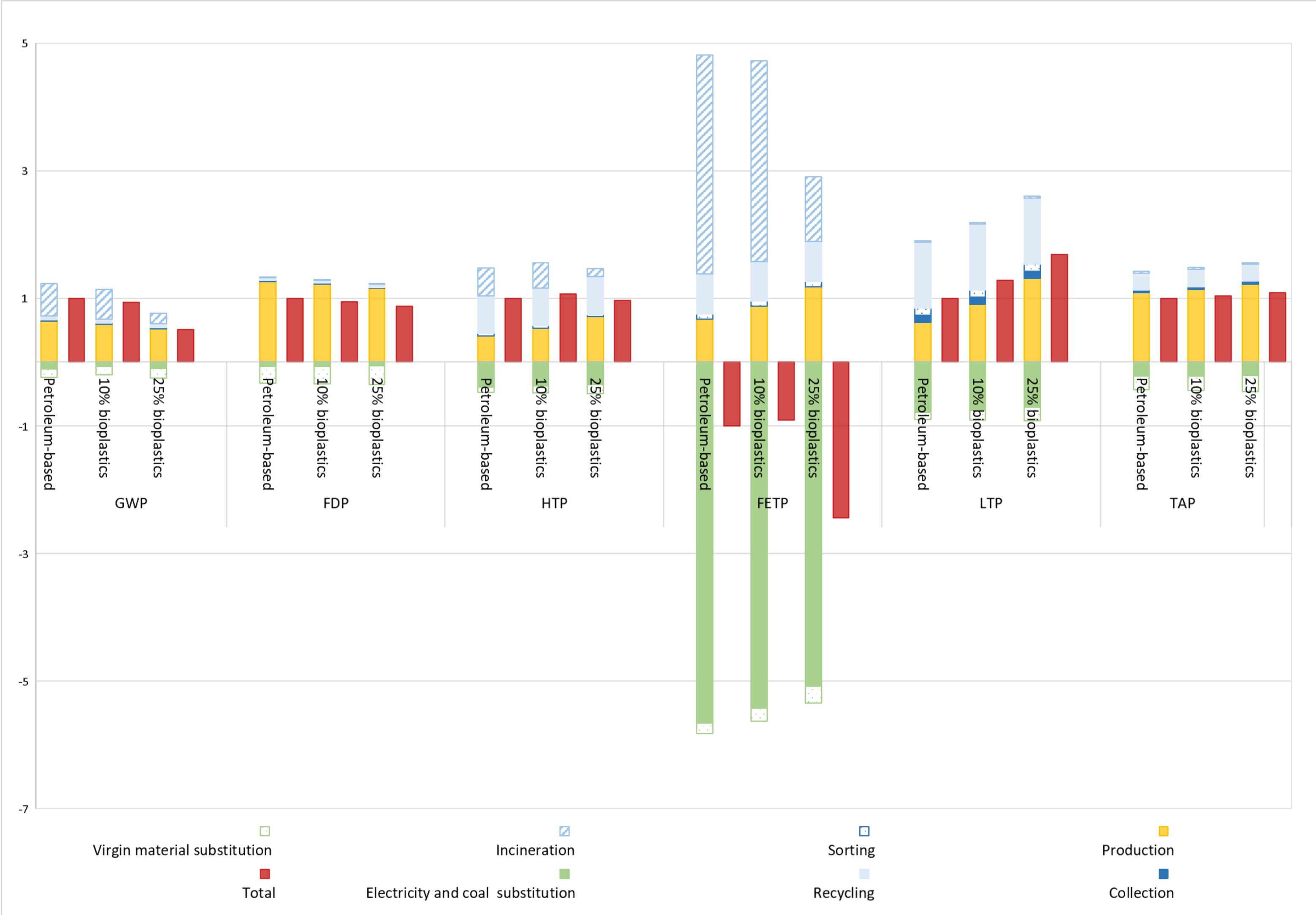


Figure 9: Results comparing the impacts of increased bioplastic amounts in the FU for selected impact categories. Normalized values against the net impacts of the reference scenario. For acronyms, see p.ix.

For **GWP**, the positive net impacts are slightly decreased with higher bioplastic shares. This is mainly explained by the reduction of direct impacts from the incineration process. It is not surprising that the GWP values are reduced when more biogenic materials are incinerated. In addition, a marginal decrease in direct production-related impacts is experienced. The avoided impacts, however, remain equal in all scenarios. For GWP, an increase in consumption leads to environmental benefits.

For **FDP**, the net life cycle impacts are very stable across the scenarios. The only noticeable changes relate on the one hand to minimal increases in avoided impacts through the substitution of virgin materials, and on the other hand to reduced impacts from the production process.

For **HTP**, the net life-cycle impacts are again very stable across the scenarios. They slightly increase when the FU contains 10% bioplastics but decrease to levels comparable to the reference scenario when the FU contains 25% bioplastics. The increase in impacts is explained by higher impacts from the production process. The direct impacts linked to incineration are, however, reduced with an increase of biogenic materials, counterbalancing the increase in direct impacts.

For **FETP**, the net impacts are considerably reduced in the 25% bioplastic scenario. First, the avoided impacts linked to electricity substitution are slightly decreased as the lower LHV of bioplastic do not allow as much substitution as the petroleum-based plastics. The direct impacts are also reduced because of a severe reduction of the incineration-related impacts. It must, nevertheless, be noted that the production-related impacts increase when the FU contains bioplastics. This latter effect does not outweigh the dynamics created by the incineration process, leading to a reduction in net impacts. For FETP, an increased share of bioplastics leads to environmental benefits.

For **LTP**, the net impacts are increased across the scenarios, explained by the higher production impacts arising when bioplastics are introduced in the FU. The rest of the parameters are unaffected by bioplastics in this impact category. For LTP, the environmental burdens are increased with higher bioplastics consumption.

For **TAP**, the net impacts are again very stable across the scenarios. The only changes relate to the marginal augmentation of avoided impacts, and the marginal increase of burdens from the production process.

Some conclusions can be drawn, and some general patterns identified when comparing these selected impact categories with each other.

(1) The net environmental loads are very stable across the scenarios for FDP, HTP and TAP, meaning that there are no noticeable effects of introducing bioplastics in the FU for these impact categories. For GWP and FETP, the net impacts for the FU containing 25% bioplastics is decreased, whereas they are increased for LTP in both bioplastic scenarios. The net impacts are only negative for FETP.
(2) The evolution in direct impacts mirrors on the one hand the increase in production-related impacts, which are higher for all impact categories, including the ones listed in appendix (A9), except for GWP and FDP. On the other hand, the evolution mirrors the reduction of incineration-related impacts. In fact, the incineration process influences GWP, HTP and FETP, highlighting the benefits of the biogenic property of bioplastics for these categories. It can be concluded that higher environmental stress is created in the production phase as the share of bioplastics increase, but that the biogenic properties are beneficial for some impact categories.
(3) Slightly less electricity is substituted when the bioplastic share is increased as a result of the reduced LHV of bioplastics compared to petroleum-based plastics.

5.3 Recycling rates

Following the definition of the amended waste legislation (2008/98/EC, article 3), the amount of recycled materials is calculated as the output of the recycling process, as assessed by the MFA. Table 11 presents the recycling rates of the studied plastic waste out-sorting scenarios.

Table 11: Resulting recycling rates for the various scenarios

Scenario	Recycling rate
Reference	13%
Realistic central sorting	29%
Ideal central sorting	67%

The MFA revealed that the out-sorting of plastic waste in a CS facility more than doubles the final recycling rate compared to when plastic waste is sorted out by households. Increasing the performance of the CS facility would engender much higher recycling rates, in this case leading to the recycling of two thirds of the initial plastic waste generated. A state-of-the-art mechanical out-sorting method hence perform much better than the conventional household out-sorting methods in terms of recycling rates. The out-sorting efficiency of the CS facility is, nonetheless, an import aspect.

5.4 The efficiency of upstream and downstream strategies

Both higher recycling rates and the introduction of bioplastics on the market aim at lowering the fossil fuel dependency and reduce GHG emissions. These are the results of downstream and upstream strategies respectively. The efficiency of these two strategies can be assessed by comparing the GWP impacts from the reference scenario with the ones of the realistic CS scenario, and the impacts from the realist CS scenario with the ones of the CS with 10% bioplastics. Table 12 shows the GWP values of the various scenarios, allowing a comparison of the environmental performance of the different strategies.

Table 12: Comparison of the GWP values relative to a change in the waste management system and to the introduction of bioplastics in the FU

GWP (kg CO_2eq/1000kg plastic waste produced)	Reference with petroleum-based plastics	CS with petroleum-based plastics	CS with 10% bioplastics
Production	2363	2363	2190
End of life	2980	2175	2007
Substitution	-806	-866	-886
Total	4537	3672	3311

The results demonstrate that both measures lead to a reduction in GHG emissions but to different extents. A shift from plastic out-sorting at the household level to a CS facility would achieve an emission reduction which is 3 times more important than a tenfold increase in bioplastic consumption.

5.5 Sensitivity analysis

Table 13 presents the results of the sensitivity analysis performed on the most important parameters for assessing the robustness of the results presented above. The sensitivity analysis studies the impacts from a 1% increase of a parameter on a specific indicator. The equation normalizes the results, allowing a comparison of the parameters. Values close to 1 reveal a high sensitivity of the parameter, while a value close to 0 is of little sensitivity. A negative value identifies a negative relationship between the parameter and the indicator: an increase of the parameter will result in a decrease of the indicator.

The results of the sensitivity analysis are split between: (1) the production system, (2) the waste management system, (3) the substitution system and (4) the waste management system including substitution. The data used and details on how the sensitivity analysis was performed can be found in Appendix (A16).

Table 13: Results of the sensitivity analysis for GWP

System	Parameters	Sensitivity rate for GWP
Production system	Increase in bioplastics production in the FU	-0.08
Waste management system	CS out-sorting efficiency	-0.26
	Recycling facility efficiency	0.08
	Efficiency of the incinerators	0.00
	Diesel consumption in the facilities	0.00
	Electricity consumption in the recycling facilities	0.02
	Increase of European recycling	-0.08
	Increase of Asian recycling	0.03
	Increased recyclability of bioplastics	0.01
	Impact of the FU composition	-0.03
	Increase of bioplastics in the FU	-0.37
Substitution system	LHV of the resin types for El production	0.35
	LHV of the resin types for cement production	0.10
Waste management system including substitution	CS out-sorting efficiency	-0.89
	Recycling facility efficiency	0.09
	Efficiency of the incinerators	-0.23
	Diesel consumption in the facilities	0.00
	Electricity consumption in the recycling facilities	0.04
	Impact of Euroepan recycling	-0.23
	Impact of Asian recycling	0.08
	Increased recyclability of bioplastics	0.02
	Impact of the FU composition	0.10
	Increase of bioplastics in the FU	-0.68

First, the sensitivity of the production system was analysed when the bioplastic production share is increased from 10% to 25%. This parameter seems to have marginal effects on the outputs of the production system.

Second, the waste management system of the realistic CS scenario was studied for several parameters because a major part of the results presented in sections 5.1 and 5.2 rely on this system. The efficiency of the CS facility was found to be rather sensitive: improving the out-sorting efficiency would reduce the GHG emissions quite importantly. The increase from 10% to 25% bioplastics in the FU composition was also found to have a significant influence on the final emissions, most likely due to the biogenic property of bioplastics. In contrast, all other parameters under investigation were found to have little influence on the final GWP values of the waste management system. Noteworthy is the fact that the emissions are slightly reduced when the plastic flow to the European market is increased compared to when it is sent to the Asian market.

Third, the sensitivity of the substitution system was investigated by altering the LHV of the plastic fractions for the electricity and coal substitution. Consequently, only the avoided emissions were analysed in this part of the analysis. The electricity parameter was found to be quite sensitive and three times higher than the coal parameter.

Finally, the sensitivity of the waste management system including substitution was examined for the same parameters studied in the above-presented waste management system. Again, the out-sorting efficiency of the CS facility and the increase of the bioplastic share in the FU were found to be the most sensitive parameters. Interestingly, these are much more sensitive when the substitution system is considered. Both lead to a severe reduction in GWP values if increased. Furthermore, the efficiency of the incinerators and the increase of the flow to the European market are parameters which are sensitive in the expanded system, which are not when only the waste management system is considered. As these parameters and the GWP values have a negative relationship, an increase in their values would lead to reductions in emissions. Lastly, the remaining parameters are slightly increased compared to when only the waste management system is considered, but their contribution to the final results are still minimal.

6. Discussion

6.1 Main findings and accordance with literature

This section evaluates the main findings presented in chapter 5 and their related uncertainties. The accordance of the results with the literature and the limitations of the thesis will in addition be discussed, first for plastic out-sorting options then for bioplastics. Finally, the efficiency of these two strategies will be compared.

6.1.1 Plastic waste out-sorting options

The first research question "*What are the environmental impacts of a waste management system where plastic waste is sorted out from the residual waste in a central sorting facility compared to a system where the fraction is sorted out at the household level?*" was informed by the results presented in section 5.1.

This analysis shows that the out-sorting of plastic waste in a CS facility in a realistic scenario decreases the environmental stress compared to when plastics are sorted out at the household level for all selected impact categories. The reduction in direct impacts correlates with different processes depending on the impact category, but mainly with the suppression of the individual packing and sorting processes, the reduction of incinerated plastic waste and the benefits of increased substituted virgin materials.

For allowing a comparison with the literature, the reference and realistic CS scenario can be simplified to represent incineration and recycling processes respectively, even though both scenarios involve a combination of these processes. With such a simplification, the results presented in this thesis are in line with the literature, confirming that there are higher net environmental benefits from plastic recycling compared to incineration for the impact categories GWP and FDP (Al-Salem *et al.*, 2009; Brogaard *et al.*, 2014; Hopewell *et al.*, 2009; Lazarevic *et al.*, 2010; Lyng & Modahl, 2011; Michaud *et al.*, 2010; Rigamonti *et al.*, 2014; Rossi *et al.*, 2015; Shonfield, 2008; Unander, 2017). This thesis further exposes that there are environmental benefits of sorting out plastics in a CS facility for the HTP, FETP, TAP and LTP categories in a realistic scenario. This confirms the validity of the waste hierarchy which advocates for recycling over incineration.

A trade-off situation nonetheless appears when the ideal CS scenario is compared to its realistic counterpart. While the GWP, FDP and TAP impacts are reduced with higher recycling rates, the HTP, FETP and LTP impacts are increased. For these categories, the increase in direct impacts arising with the recycling process from the facility itself, transport, auxiliary fuels and materials is not balanced out by the benefits of decreased impacts from the incineration process. The same pattern was exposed by Unander (2017) and Callewaert (2017) regarding energy. In fact, the authors found that higher recycling rates induced by a CS facility had negative impacts on the energy efficiencies as a result of higher energy consumption during treatment and less energy output from the local incineration plant.

It can be concluded that the impact categories are influenced by different processes. Consequently, a change in the flow sizes will either increase or decrease the environmental stress, depending on the impact category analysed. This conclusion is supported by Rigamonti *et al.* (2014), stating that there is no preferred solution for plastic waste management when all impact categories are analysed. As this trade-off situation only appears for the ideal CS scenario, it can be suggested that the balance between the adverse effects from the recycling and incineration processes depend on the size of the flows diverted to each of these processes.

Critical variables influencing these results must, nevertheless, be discussed: the auxiliary fuel parameter used in the recycling process, the substituted materials, the substitution ratio and the composition of the FU.

First, the auxiliary fuel parameter used in the recycling processes might be overestimated, affecting the results of the CS scenarios. Given that contact could not be established with the recycling facilities themselves, literature was used for providing information regarding electricity consumption in the plastic recycling process. Sources providing this piece of information were scare and gave only a broad range of values (Astrup *et al.*, 2009). A value from the upper part of the range was selected. The results uncovered that the recycling process highly influences the direct impacts of the waste management system for all impact categories studied. In addition, the sensitivity analysis revealed that the electricity consumption in the recycling processes has a certain influence on the GWP values, even though it remains quite low. The assumption regarding electricity consumption amounts might then have an influence on the final results of the ideal CS scenario especially, which might be overestimated.

Second, the chosen substituted material is decisive. In fact, the avoidance of electricity production through incineration is beneficial for HTP, FETP and LTP, while the avoidance of virgin materials is beneficial for FDP and TAP. Moreover, the sensitivity analysis revealed the influence of this part of the system, especially for the electricity substitution. The selection of the appropriate avoided primary production of materials in material recycling systems is hence decisive (Brogaard *et al.*, 2014; Rigamonti *et al.*, 2014; Turner *et al.*, 2015).

Third, the substitution ratio is of importance. The literature study disclosed that to improve resource efficiency and avoid GHG emissions, the quality of the recovered plastic is crucial (Sevigné-Itoiz *et al.*, 2015). In this thesis, it was assumed that the quality of the recycled materials was high, allowing a 1:1 substitution, as it is common practice in the literature (Lazarevic *et al.*, 2010). This is nevertheless a very optimistic assumption, probably not reflecting the real-life situation (Astrup *et al.*, 2009; Michaud *et al.*, 2010). The amounts of avoided impacts might therefore be overestimated.

Fourth, the composition of the FU was maintained unchanged across the scenarios. Modifications will, however, most likely happen as a consequence of EU policies and market forces. The use of PS is for instance decreasing (Lambertz, 2018). The sensitivity analysis revealed that the FU composition has a certain impact on the GWP values, especially for the waste management system including substitution. The composition itself and its future development is then a source of uncertainty.

When it comes to the recycling rates presented in section 5.3, it was uncovered the amounts of recycled plastics are more than doubled in a situation where a CS facility is used compared to when plastics are segregated at the household level. However, the realistic CS scenario only reaches 29% recycling, which is still far less than the target of 50% plastic packaging recycling set by the EU. The rate is nevertheless in line with the situation ROAF currently experiences, with plastic recycling rates of 32% (Callewaert, 2017), endorsing the robustness of the results. Only the ideal CS scenario reaches the EU target. The out-sorting efficiency and the amounts of recycled materials are hence highly interlinked. From a GWP perspective, the sensitivity analysis disclosed that the out-sorting efficiency of the CS facility is an extremely sensitive parameter, especially when the substitution effects are accounted for. An increase in out-sorting rates would decrease the amounts of GHG emissions drastically. The recycling process efficiency was also found to have a certain sensitivity, but much lower than the one of the out-sorting process. For reducing emissions over the supply chain, it is hence more efficient to increase the efficiency of the CS facility than of the recycling facility. As the out-sorting is of such importance for both recycling rates and GWP values, it can be suggested that the collection process is the bottleneck in the current system, as advocated by Raadal *et al.* (2016).

Comparing the total environmental burdens of the waste management system with the achieved recycling rates, it can be concluded that higher recycling rates lead to lower GWP and FDP impacts, but to higher impacts for land occupation and toxicity impacts due to the influence of the recycling process. Aiming for increased recycling rates in the light of a circular economy is then a justifiable policy from a GWP perspective, but the influence and increase on other aspects should be recognized.

Broadening the scope of the discussion, a few other elements must be considered regarding the incineration process, the recycling process, waste prevention and the effect of recycled materials.

First, it can be assumed that the incinerator in Heimdal will be further used for district heating after the establishment of a CS facility. The feedstock amount and composition will be altered as the plastic fraction is sorted out. The reduction in environmental load from the waste management system was found to be partly linked to the decrease in incinerated plastic waste. If other fuel sources are taken in use to compensate for this feedstock reduction, the presented results should be revised.

Second, it was assumed that the plastics were recycled in Germany and Asia. The sensitivity analysis disclosed that recycling in Europe would lead to lower environmental stress than when the fraction is exported to Asia. As the EU aspires at developing a robust recycling market in Europe, it could be of interest for Norwegian businesses to take this opportunity. Creating a national recycling business would be beneficial from a GWP perspective with the avoidance of long transport distances and a cleaner electricity mix. In fact, the electricity consumption in the recycling processes was found to have a certain influence.

Third, the aspects of waste prevention and sorting skills should not be forgotten. On the one hand, less generated waste will lead to lower environmental burdens over the entire supply chain. This is evident but should be communicated properly, both to producers and consumers. On the other hand, clear information should be given to the inhabitants regarding washing and proper sorting of the various waste fractions, in addition to the reasons for which this should be done. In fact, the citizen's habits are of importance even if a CS facility is established, as experienced by ROAF.

Fourth, the downstream use of recycled plastic products has not been assessed as this was outside the scope of this thesis. Studies have uncovered that products made from recycled materials can be a source of microplastic pollution, for instance through the washing of cloths (Browne *et al.*, 2011). The manufacture of robust and stable materials should rather be encouraged as they are less likely to be a source of microplastics.

6.1.2 Bioplastics

The second research question "*How does a share of bioplastic affect the life cycle impacts of household plastic consumption?"* was informed by the results presented in section 5.2. The life cycle impacts of petroleum-based plastics can be compared to the 10% and 25% bioplastic impacts for answering this question.

First of all, it can be noted that the introduction of bioplastics has little if no influence on the net life cycle impacts for most impact categories except for FETP. The reduction in burdens from FETP relates to the high influence of substituted Norwegian electricity in combination with the reduction in environmental loads from incineration.

Reducing the dependency on fossil fuels and the amounts of GHG emissions are currently the main reasons for producing and using bioplastics compared to conventional plastics. In fact, the literature study revealed that the production of bioplastics reduce the FDP and GWP impacts compared to petroleum-based plastics (Belboom *et al.*, 2016; Murphy *et al.*, 2013; Song *et al.*, 2011; Tsiropoulos *et al.*, 2015; Weiss *et al.*, 2012), which is in line with the results of this analysis. In addition, the results of this analysis testify that GWP and FDP impacts are reduced from a life cycle perspective, and not only from a production perspective, when bioplastics are consumed. Despite the minimal reduction in FDP, the production and use of bioplastics might be seen as justifiable in regard to these aspects.

Nonetheless, the life cycle impacts for LTP and TAP, in addition to most categories presented in Appendix (A9) experience an increase in net environmental loads with the introduction and/or increase of bioplastics. This aspect is also in line with the reviewed studies, even though these only analysed the production impacts (Belboom *et al.*, 2016; Piemonte & Gironi, 2012; Tabone *et al.*, 2010; Tsiropoulos *et al.*, 2015; Weiss *et al.*, 2012).

Despite the fact that net impacts are only slightly altered when bioplastics are consumed, the disaggregated results (Figure 9) show some interesting features concerning both the production and waste management processes.

On the one hand, it can be noted that the net impacts mirror the high environmental stress caused by the production process: GWP and FDP experience a reduction in impacts while all other categories experience an increase in environmental burdens. The literature study disclosed that the latter highlight is a result of the cultivation step and the ethanol production step required for bioplastic production (Belboom *et al.*, 2016; Tabone *et al.*, 2010; Weiss *et al.*, 2012). It can be concluded that the production process is an important contributor to the life cycle impacts for all impact categories analysed. As explained in the methodology, the production processes only exist for PLA and starch in the Ecoinvent database and were therefore modelled in this thesis for the bio-PE and the bio-PET fractions based on data from the literature. The remaining fractions were aggregated to these four types due to a lack of data. Following this specific modelling and aggregation, the production-related results for the bioplastic fraction are subject to uncertainties. Nonetheless, the sensitivity analysis disclosed that the bioplastic production parameter slightly reduces the GWP values as they have a negative relationship, but that it is not very sensitive to the final results.

On the other hand, the direct impacts from the waste management system are rather unchanged across the scenarios for most selected impact categories. The incineration process, however, leads to a decrease in impacts for GWP and the toxicity categories HTP and FETP, emphasising the biogenic property of bioplastics. The impacts on the toxicity categories may nonetheless be underestimated as the inventory simplified the modelling of the incineration process of bioplastics by omitting direct impacts of for instance dioxins and carbon monoxide.

Currently, only very few bioplastic resins are recyclable. Biodegradable biopolymers pose problems as they are a source of impurities when they enter the conventional plastics recycling or organic waste composting streams (Niaounakis, 2013). Most of the currently produced bioplastic fractions are therefore incinerated. The recyclability of these fractions will probably increase with time as a result of higher heterogeneity within the bioplastic fractions and increased flow sizes. Technologies for recycling PLA currently exist but are not taken in use because of the small PLA waste amounts, making their out-sorting and recycling economically unattractive. The sensitivity analysis revealed that the recyclability of bioplastics is of minimal importance for the final results. This is indeed a surprise and highlights indirectly the importance of the biogenic property of bioplastics, as a reduction in emissions from the incineration process balances out the benefits of increased material recycling.

Biogenic emissions are considered to be carbon neutral because the plastics are made from biomass. The segregated carbon taken up during the plant's lifetime is released during the combustion process. Nonetheless, these have a global warming effect which is not accounted for. The concepts of carbon and climate neutrality should therefore be distinguished.

The sensitivity analysis disclosed that the increase of bioplastic share in the FU is a highly sensitive parameter, especially influencing the substituted part of the system. Larger amounts of bioplastics would decrease the GWP emissions from the expanded waste management system substantially. It must then be noted that the 25% bioplastics scenario, and even the 10% scenario, assume a very optimistic penetration of bioplastics on the market, as the current bioplastic production only makes up 1% of the total plastic production amounts.

6.1.3 The efficiency of upstream and downstream strategies

It can be deduced from section 5.4 that the GWP impacts are reduced more effectively with a shift from household plastic segregation to a CS facility, than in the transition from petroleum-based plastics to a tenfold increase in bioplastics consumption. Consequently, it is more effective to focus on changes in the waste management system than to promote the production and use of bioplastics as alternative to petroleum-based plastics. The combination of both measures is undoubtedly the most effective way of reducing the total GWP impacts.

6.2 Strengths and weaknesses

There are strengths and weaknesses linked to different aspects of this thesis: (1) the model, (2) the FU composition, (3) the flows and processes of the systems, and (4) the modelling of bioplastic production.

First, the MFA and LCA models used in this thesis have the possibility to identify material efficiency as well as the environmental impacts caused by the analysed system. The MFA model drives the LCA, securing consistency as the LCA is based on a mass balanced system. This compilation hence strengthens the results. A parameter could easily be altered, changing the entire system accordingly. The model can then be adjusted for analysing other fractions or combination of fractions. Furthermore, the different system boundaries make it possible to understand the sensitivity of these various parts of the system and their related impacts. The model is input driven, securing the material flows based on the waste input. This last feature can also be seen as a weakness as it makes it necessary to know the exact waste composition.

Second, it is a strength that the FU of this LCA was based on the results of a composition analysis with a high resolution on plastics, which was performed specifically for Trondheim in 2015 (Syversen & Bjørnerud, 2015). Despite the relatively new and location specific data, high uncertainties arise from the output of the composition analysis because waste composition varies over time, season and location (Slagstad & Brattebø, 2013). These uncertainties will propagate through the system as the waste composition contributes both directly and indirectly to the results. On the one hand, the aggregation of the different plastic types to form the FU has direct influences on the results, regulating the size of the various flows. On the other hand, background processes used in the inventories are chosen depending on the waste characteristics, which are indirectly affecting the final results (Bisinella *et al.*, 2017). Further aggregations were in addition performed given that several background processes were not available for all FU fractions, increasing the uncertainty even more. The composition of the FU was tested in the sensitivity analysis, which disclosed that the parameter is not so sensitive when only petroleum-based plastics are included. The amounts of bioplastics in the FU are, however, highly influent.

Third, this case study of Trondheim can be used to strengthen local political decisions. Nevertheless, the described system is not totally representative of the current situation, weakening the robustness of the results. On the one hand, the system description for 2011 was used (Lyng & Modahl, 2011), which is known to have changed since that time. To the author's knowledge, a sorting facility in Sweden is currently in use but could not be modelled because of a lack of data. On the other hand, the amounts of waste sent to Asia are likely to be erroneous. In fact, no data could be gathered on the reallocation of the Norwegian plastic waste after China's ban on foreign plastic waste. As it is not yet known how the plastic recycling market will evolve, the 2011 data was used also for modelling future scenarios. Moreover, not all processes have been taken into account for representing the entire life cycle of plastic. Only the production and waste management phases have been assessed, omitting the use phase. Environmental impacts linked to the rinsing of plastics at the household level would slightly influence the results, especially if hot water is used.

Fourth, simplifications in the modelling of the bioplastic life cycle is a weakness of this thesis. The production and waste management processes of most bioplastic fraction were not available in the Ecoinvent database given that the materials are a rather new kind of product on the market. Subsequent to the performed modelling and aggregation operations, the production-related results especially but also the incineration-related results are subject to uncertainties. The environmental burdens of the bioplastics scenarios should then be assessed carefully.

6.3 Recommendations and further work

The results of this thesis support the plans of building a CS facility for sorting out plastics from the residual waste given that Trondheim municipality aims at lowering its environmental impacts and increase its recycling rates in accordance with EU legislation. It must, nonetheless, be remembered that impacts on land occupation and toxicity will increase with higher amounts of recycled materials given the influence of the recycling process on these impact categories. It must not be forgotten that waste prevention and reduction are the most effective strategies to curb environmental burdens and should be actively pursued.

The lock-in effect of building a CS facility should not be neglected. Smart operation design leading to high efficiencies should be aimed for, as it was disclosed that the current technologies used by ROAF might not be satisfying for reaching the recycling rates set by the EU (Callewaert, 2017). Inspiration should be taken both from the operating ROAF facility, and the under-construction IVAR facility when developing plans for the design of the facility in Trondheim. It will be interesting to see if the recycling rates are increased in the latter facility as organic waste should not be sorted out in the facility itself and as some plastic packaging fractions will be washed and granulated on-site. This will lead on the one hand to lower transport-related emissions and costs and might on the other hand lead to the development of a Norwegian market for recyclates. In fact, when aiming for higher recycling rates, organic waste should not be treated in the CS facility. Organic waste pollutes the plastic fraction by reducing the material quality thereby leading to lower out-sorting and recycling rates, as concluded by Callewaert (2017) and Unander (2017).

Bioplastics were not found to have very environmental valuable effects, neither under their production phase, nor under their waste management phase even though their biogenic property influenced GWP and toxicity categories. Only the impacts in the GWP and FDP categories were slightly reduced when the consumption of bioplastics was increased. It is therefore suggested that the municipality should focus on better waste management systems and waste prevention and reduction rather than promoting bioplastics as a sustainable alternative to petroleum-based plastics.

Several relevant aspects have not been analysed during this thesis. Further research is first needed on the impacts related to energy consumption. ReCiPe does not include the characterisation factor for energy which was therefore not assessed in this thesis. It is, however, an important parameter when it comes to waste management. Callewaert (2017) and Unander (2017) analysed the energy efficiency for similar systems, the latter author also in a case study for Trondheim. As they are complimentary, their results could be combined with the outcomes of this thesis. Second, the economics linked on the one hand to the various waste management systems, and on the other hand on the production of different types of plastics should be further investigated. Third, questions related to plastic pollution would have been of interest in the current plastic debate. Microplastics are in fact released from the recycling process and might be released from certain recycled materials. Further research is needed for understanding how the microplastic pollution increase with increased amounts of recycled materials, and how this trend could be decoupled. Fourth, the entire life cycle of plastics should be analysed, including the impacts arising in households during the use phase. Finally, it was determined that the household sorting skills are far from optimal. Analysing by how much these should be improved to equalize the benefits of a CS facility is of interest. Comparing the environmental and economic effects of improved education and technology respectively would be very valuable.

7. Conclusion

An LCA based on MFA principles was developed as a case study for Trondheim municipality. On the one hand, the environmental impacts of plastic out-sorting in a CS facility were compared to the impacts of source separation at the household level. On the other hand, the environmental effects resulting from the introduction of bioplastics in the household plastic consumption were studied in a life cycle perspective.

This thesis concludes that the environmental load from the waste management system is lower when plastics are sorted out in a CS facility compared to source separation at the household level for all impact categories analysed. This primarily occurs with the elimination of the individual packing and sorting processes as well as with a reduction of incinerated plastic amounts. However, this conclusion becomes less evident when the quantity of recycled materials increases with higher out-sorting rates. In fact, ecotoxicity and land occupation impacts are intensified with higher recycling rates given the influence of the recycling process on these categories.

In addition, the results highlight that the plastic recycling rate was doubled when sorting the fraction out in a CS facility. However, the target of 50% plastic recycling set by the EU was only reached in an ideal scenario. For achieving a higher recycling percentage, the efficiency of the CS facility was found to be decisive. The efficiency can be increased by improving various features both upstream, on-site and downstream in the value chain. Upstream, a more homogenous and pure composition of wasted household packaging, designed for out-sorting in a CS facility and for recycling, would be of importance. Smart solutions for treating incoming fractions in the CS facility in combination with better sorting skills at the household level would affect the out-sorting rates on-site. Downstream, a mature and stable market for various recycled materials would incentivize proper waste collection, out-sorting and recycling of waste, and thereby the efficiency of the CS facility.

Further, this analysis disclosed that the life cycle impacts of household plastic consumption are reduced importantly for freshwater ecotoxicity and slightly reduced for global warming and fossil depletion potentials when bioplastics are introduced. Nonetheless, all other impact categories experience increases in impacts, mirroring the high environmental stress caused by the bioplastic production process.

The cultivation and ethanol production processes are responsible for the majority of the bioplastic production impacts and should be targeted to diminish the environmental burdens from bioplastic production. The literature review disclosed that a reduction in pesticide use, the elimination of burning practices and increased crop diversity would be beneficial in this regard. In addition, other feedstock sources such as by-products from the forestry sector could be used as alternative to cultivated plants.

Comparing the outcomes of the two strategies, it was found more effective to focus on changes in the waste management system than to promote the production and use of bioplastics as alternative to petroleum-based plastics. Nevertheless, the combination of both measures is undoubtedly the most effective way of reducing the total GWP impacts.

The two parts of this thesis are of scientific interest as they both fill a knowledge gap in literature. First, all reviewed LCA studies assessed the differences in environmental performance of waste management options. Only Lyng and Modahl (2011) studied the benefits of plastic waste out-sorting. With inspiration from this study, this LCA assessed the environmental impacts of sorting out plastics in a CS facility, compared to source separation at the household level. Second, all LCA studies analysing the environmental impacts of bioplastics were only accounting for the production phase, or for the life cycle of a specific resin type. This thesis fills a knowledge gap by developing an LCA analysing the life cycle emissions of household plastic consumption which contains a share of bioplastics. Further work should encompass the economic, energy and microplastic pollution aspects of the topic.

References

Adamcová, D., Elbl, J., Zloch, J., Vaverková, M. D., Kintl, A., Juřička, D., Hladký, J., Brtnický, M. (2017). Study on the (bio)degradation process of bioplastic materials under industrial composting conditions. *Acta Universitatis Agriculturae et Silviculturae Mendelianae Brunensis*, *65*(3), 791–789. https://doi.org/10.11118/actaun201765030791

Al-Salem, S. M., Lettieri, P., & Baeyens, J. (2009). Recycling and recovery routes of plastic solid waste (PSW): A review. *Waste Management*, *29*, 2625–2643. https://doi.org/10.1016/j.wasman.2009.06.004

Álvarez-Chávez, C. R., Edwards, S., Moure-Eraso, R., & Geiser, K. (2012). Sustainability of bio-based plastics: general comparative analysis and recommendations for improvement. *Journal of Cleaner Production*, *23*, 47–56. https://doi.org/10.1016/j.jclepro.2011.10.003

Arena, U., Mastellone, M. L., & Perugini, F. (2003). Life Cycle Assessment of a Plastic Packaging Recycling System. *The International Journal of Life Cycle Assessment*, *8*(2), 92–98. https://doi.org/10.1065/Ica2003.02.106

Askham, C., & Raadal, H. L. (2016). *PlastiRetur sluttrapport. Økt anvendelse av gjenvunnet plast for økt ressurseffektivitet, redusert energibruk og lavere klimagassutslipp*. Retrieved from https://www.ostfoldforskning.no/media/1715/sluttrapportutkast-plastiretur-off-forprosjekt_4.pdf

Astrup, T., Fruergaard, T., & Christensen, T. H. (2009). Recycling of plastic: accounting of greenhouse gases and global warming contributions. *Waste Management & Research*, *27*, 763–772. https://doi.org/10.1177/0734242X09345868

Barnes, D. K. A., Galgani, F., Thompson, R. C., & Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B*, *364*(1526), 1985–1998. https://doi.org/10.1098/rstb.2008.0205

Belboom, S., Elique, A., & Eonard, L. (2016). Does biobased polymer achieve better environmental impacts than fossil polymer? Comparison of fossil HDPE and biobased HDPE produced from sugar beet and wheat. *Biomass and Bioenergy*, *85*, 159–167. https://doi.org/10.1016/j.biombioe.2015.12.014

Bisinella, V., Götze, R., Conradsen, K., Damgaard, A., Christensen, T. H., & Astrup, T. F. (2017). Importance of waste composition for Life Cycle Assessment of waste management solutions. *Journal of Cleaner Production*, *164*, 1180–1191. https://doi.org/10.1016/J.JCLEPRO.2017.07.013

Bjørnerud, S. (2018). *Personal communication between Irmeline de Sadeleer and Sveinung Bjørnerud*.

Brattebø, H., & Reenaas, M. (2012). Comparing CO2 and NOX emissions from a district heating system with mass-burn waste incineration versus likely alternative solutions – City of Trondheim, 1986–2009. *Resources, Conservation and Recycling*, *60*, 147–158. https://doi.org/10.1016/J.RESCONREC.2011.11.001

Brogaard, L. K., Damgaard, A., Jensen, M. B., Barlaz, M., & Christensen, T. H. (2014). Evaluation of life cycle inventory data for recycling systems. *Resources, Conservation and Recycling*, *87*, 30–45. https://doi.org/10.1016/j.resconrec.2014.03.011

Browne, M. A., Crump, P., Niven, S. J., Teuten, E., Tonkin, A., Galloway, T., & Thompson, R. (2011). Accumulation of Microplastic on Shorelines Woldwide: Sources and Sinks. *Environmental Science & Technology*, *45*, 9175–9179. https://doi.org/10.1021/es201811s

Brunner, P., & Rechberger, H. (2004). *Practical Handbook of Material Flow Analysis*. Retrieved from https://thecitywasteproject.files.wordpress.com/2013/03/practical_handbook-of-material-flow-analysis.pdf

Callewaert, P. (2017). *Analysing the sustainability performance and critical improvement factors of urban municipal waste systems (Master's Thesis)*.

Christensen, T. H. (Ed.). (2011). *Solid Waste Technology & Management* (Wiley). Chichester, United Kingdom: Blackwell Publishing Ltd.

Directive on waste (75/442/EEC). (1975). *Official Journal of the European Union* (Vol. 194). Retrieved from http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CONSLEG:1975L0442:20031120:EN:PDF

Ecoembes. (2015). *Light Weight Packaging Sorting Plants*. Madrid. Retrieved from https://www.ecoembes.com/sites/default/files/archivos_estudios_idi/light-weight-packaging-sorting-plants.pdf

Ellen MacArthur Foundation and McKinsey Company. (2016). *The New Plastics Economy - Rethinking the Future of Plastics*. https://doi.org/10.1103/Physrevb.74.035409

European Bioplastics. (2016). Relevant standards and labels for bio-based and biodegradable plastics. Retrieved from http://docs.european-bioplastics.org/2016/publications/fs/EUBP_fs_standards.pdf

European Bioplastics. (2017). Bioplastics market data 2017. Global production capacities of bioplastics 2017-2022. Retrieved from http://docs.european-bioplastics.org/publications/market_data/2017/Report_Bioplastics_Market_Data_2017.pdf

European Bioplastics. (2018). Materials – European Bioplastics e.V. Retrieved from https://www.european-bioplastics.org/bioplastics/materials/

European Commission. (2015). Proposal for a DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL amending Directive 2008/98/EC on waste. Retrieved from https://eur-lex.europa.eu/resource.html?uri=cellar:c2b5929d-999e-11e5-b3b7-01aa75ed71a1.0018.02/DOC_1&format=PDF

European Commission. (2016). Briefing EU Legislation in Progress Circular economy package Four legislative proposals on waste. Retrieved from http://www.europarl.europa.eu/EPRS/EPRS-Briefing-573936-Circular-economy-package-FINAL.pdf

European Commission. (2018a). A European Strategy for Plastics in a Circular Economy. Retrieved from http://eur-lex.europa.eu/resource.html?uri=cellar:2df5d1d2-fac7-11e7-b8f5-01aa75ed71a1.0001.02/DOC_1&format=PDF

European Commission. (2018b). Press release - Circular Economy: New rules will make EU the global front-runner in waste management and recycling. Retrieved from http://europa.eu/rapid/press-release_IP-18-3846_en.htm

Fråne, A., Stenmarck, Å., Gíslason, S., Løkke, S., Zu Castell Rüdenhausen, M., Raadal, H. L., & Wahlström, M. (2015). Guidelines to increased collection of plastic packaging waste from households. https://doi.org/10.6027/ANP2015-712

Frees, N. (2002). *Miljømaessige fordele og ulemper ved genvinding af plast. Eksempler med udgangspunkt i konkrete produkter*. *Miljøprojekt Nr 657* (Vol. 657). Retrieved from https://www2.mst.dk/udgiv/publikationer/2002/87-7944-967-0/pdf/87-7944-968-9.pdf

Gentil, E., Christensen, T. H., & Aoustin, E. (2009). Greenhouse gas accounting and waste management. *Waste Management & Research*, *27*, 696–706. https://doi.org/10.1177/0734242X09346702

GESAMP. (2015). *Sources, Fate and Effects of Microplastics in the Marine Environment: A Global Assessment*, (Kershaw, P. J., ed.). (IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). *Rep. Stud. GESAMP 90, 96*.

Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, uses, and fate of all plastics ever made. *Science Advances*, *3*(7), 5. https://doi.org/10.1126/sciadv.1700782

Grønt Punkt Norge. (2013). Ren plastemballasje forenkler gjenvinningen - Grønt Punkt Norge. Retrieved from https://www.grontpunkt.no/nyhet/ren-plastemballasje-forenkler-gjenvinningen/

Grønt Punkt Norge. (2018). *Faktaark fra Grønt Punkt Norge: plastemballasje - fra kjøkkenbenken til nye plastprodukter*.

Gu, F., Guo, J., Zhang, W., Summers, P. A., & Hall, P. (2017). From waste plastics to industrial raw materials: A life cycle assessment of mechanical plastic recycling practice based on a real-world case study. *Science of The Total Environment*, *601–602*, 1192–1207. https://doi.org/10.1016/J.SCITOTENV.2017.05.278

Hendrickson, C., Lave, L., & Matthews, H. S. (2006). *Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach*. Retrieved from https://books.google.no/books?id=FZ2VUOX1gbAC&pg=PT143&lpg=PT143&dq=environmental+performance+of+cars+with+plastic+instead+of+steal&source=bl&ots=C_LGC4hpE5&sig=nqaFpcDXOU4E8PwL7D3-DMARnYA&hl=no&sa=X&ved=0ahUKEwiOnfuglYrbAhWKFiwKHew8DokQ6AEIODAC#v=onepage

Hestin, M., Faninger, T., & Milios, L. (2015). *Increased EU Plastics Recycling Targets: Environmental, Economic and Social Impact Assessment*. Retrieved from http://www.plasticsrecyclers.eu/sites/default/files/BIO_Deloitte_PRE_Plastics Recycling Impact_Assesment_Final Report.pdf

Hillman, K., Damgaard, A., Erikson, O., Jansson, D., & Fluck, L. (2015). *Climate Benefits of Material Recycling. Inventory of Average Greenhouse Gas Emissions for Denmark, Norway and Sweden*. https://doi.org/10.6027/TN2015-547

Hjellnes Consult AS. (2017). *Avfallsanalyse Oslo 2017*.

Hopewell, J., Dvorak, R., & Kosior, E. (2009). Plastics recycling: challenges and opportunities. *Philosophical Transactions of the Royal Society B.*, (364), 2115–2126. https://doi.org/10.1098/rstb.2008.0311

Hottle, T. A., Bilec, M. M., & Landis, A. E. (2017). Biopolymer production and end of life comparisons using life cycle assessment. *Resources, Conservation and Recycling*, *122*, 295–306. https://doi.org/10.1016/j.resconrec.2017.03.002

ISO 14040. (2006). Environmental management — Life cycle assessment — Principles and framework. *International Standard*, (Second edition). https://doi.org/10.1136/bmj.332.7550.1107

Iwata, T. (2015). Biodegradable and bio-based polymers: Future prospects of eco-friendly plastics. *Angewandte Chemie - International Edition*, *54*(11), 3210–3215. https://doi.org/10.1002/anie.201410770

Kershaw, D. P. J. (2015). *Biodegradable Plastics & Marine Litter*. *United Nations Environment Programme (UNEP)*.

Lambertz, O. (2018). *Personal communication between Irmeline de Sadeleer and Oliver Lambertz*.

Lausselet, C. (2018). *Personal communication between Irmeline de Sadeleer and Carine Lausselet*.

Lausselet, C., Cherubini, F., Del, G., Serrano, A., Becidan, M., & Strømman, A. H. (2016). Life-cycle assessment of a Waste-to-Energy plant in central Norway: Current situation and effects of changes in waste fraction composition. https://doi.org/10.1016/j.wasman.2016.09.014

Lausselet, C., Cherubini, F., Oreggioni, G. D., Del, G., Serrano, A., Becidan, M., Xiangping H., Rørstad, P.K. & Strømman, A. H. (2017). Norwegian Waste-to-Energy: Climate change, circular economy and carbon capture and storage. *Resources, Conservation & Recycling*, *126*, 50–61. https://doi.org/10.1016/j.resconrec.2017.07.025

Laußmann, C., Endres, H.-J., Hannover, F., Giese, G. U., & Kitzler, A.-S. (2010). Disposal of Bio-Polymers via Energy Recovery. *Bioplastics Magazine*, *5*(01). Retrieved from https://f2.hs-hannover.de/fileadmin/media/doc/ifbb/Bioplastics_Magazine__01_10__Vol._5_S._42-43.pdf

Lazarevic, D., Aoustin, E., Buclet, N., & Brandt, N. (2010). Plastic waste management in the context of a European recycling society: Comparing results and uncertainties in a life cycle perspective. *"Resources, Conservation & Recycling," 55*, 246–259. https://doi.org/10.1016/j.resconrec.2010.09.014

Li, W. C., Tse, H. F., & Fok, L. (2016). Plastic waste in the marine environment: A review of sources, occurrence and effects. *Science of the Total Environment*, *566–567*, 333–349. https://doi.org/10.1016/j.scitotenv.2016.05.084

Liptow, C., & Tillman, A.-M. (2012). A Comparative Life Cycle Assessment Study of Polyethylene Based on Sugarcane and Crude Oil. *Journal of Industrial Ecology*, *16*(3), 420–435. https://doi.org/10.1111/j.1530-9290.2011.00405.x

Lyng, K.-A., & Modahl, I. S. (2011). *Livsløpsanalyse for gjenvinning av plastemballasje fra norske husholdninger*. Retrieved from https://www.ostfoldforskning.no/media/1183/1011.pdf

Meissner, R. (2018). *Personal communication between Irmeline de Sadeleer and Rudolf Meissner*.

Michaud, J.-C., Farrant, L., Jan, O., Kjær, B., & Bakas, I. (2010). *Environmental benefits of recycling - 2010 update*. *Waste resource action programme (WRAP)*. Retrieved from http://www.wrap.org.uk/sites/files/wrap/Environmental_benefits_of_recycling_2010_update.3b174d59.8816.pdf

Mohsenzadeh, A., Zamani, A., & Taherzadeh, M. J. (2017). Bioethylene Production from Ethanol: A Review and Techno-economical Evaluation. *ChemBioEng Reviews*, *4*(2), 75–91. https://doi.org/10.1002/cben.201600025

Murphy, R., Guo, M., & Akhurst, M. (2013). *Environmental assessment of Braskem's biobased PE resin*. *E4tech*.

Niaounakis, M. (2013). Biopolymers: Reuse, Recycling, and Disposal. In *Biopolymers: Reuse, Recycling, and Disposal*. https://doi.org/10.1016/B978-1-4557-3145-9.00001-4

Panda, A. K., Singh, R. K., & Mishra, D. K. (2010). Thermolysis of waste plastics to liquid fuel. A suitable method for plastic waste management and manufacture of value added products—A world prospective. *Renewable and Sustainable Energy Reviews*, *14*, 233–248. https://doi.org/10.1016/j.rser.2009.07.005

Piemonte, V., & Gironi, F. (2012). Bioplastics and GHGs Saving: The Land Use Change (LUC) Emission Issue. *Energy Sources, Part A: Recovery, Utilization, and Environmental Effects*, *34*(21), 1995–2003. https://doi.org/10.1080/15567036.2010.497797

Plastics Europe. (2017). Plastics - the Facts 2017. An analysis of European plastics production, demand and waste data.

Raadal, H. L., Stensgård, A. E., Lyng, K.-A., & Hanssen, O. J. (2016). *Vurdering av virkemidler for økt utsortering av våtorganisk avfall og plastemballasje*. Retrieved from http://ostfoldforskning.no/media/1017/767.pdf

Ragaert, K., Delva, L., & Geem, K. Van. (2017). Mechanical and chemical recycling of solid plastic waste. *Waste Management*, *69*, 24–58. https://doi.org/10.1016/j.wasman.2017.07.044

Rem, T. (2018). Personal communication between Irmeline de Sadeleer and Thomas Rem.

Reuters. (2018). EU targets recycling as China bans plastic waste imports. Retrieved from https://www.reuters.com/article/us-eu-environment/eu-targets-recycling-as-china-bans-plastic-waste-imports-idUSKBN1F51SP

Rigamonti, L., Grosso, M., Møller, J., Sanchez, V. M., Magnani, S., & Christensen, T. H. (2014). Environmental evaluation of plastic waste management scenarios. *"Resources, Conservation & Recycling," 85*, 42–53. https://doi.org/10.1016/j.resconrec.2013.12.012

Rødvik, S. E. (2018). Personal communication between Irmeline de Sadeleer and Svein Erik Rødvik.

Romerike Avfallsforedling IKS. (2017). ROAFs Miljørapport 2016. Retrieved from https://www.roaf.no/wp-content/uploads/2017/10/7802-Roaf-miljørapport-2016-interaktiv_til-web.pdf

Rossi, V., Cleeve-Edwards, N., Lundquist, L., Schenker, U., Dubois, C., Humbert, S., & Jolliet, O. (2015). Life cycle assessment of end-of-life options for two biodegradable packaging materials: sound application of the European waste hierarchy. *Journal of Cleaner Production*, *86*, 132–145. https://doi.org/10.1016/J.JCLEPRO.2014.08.049

Sandberg, N. H., Sartori, I., Vestrum, M. I., & Brattebø, H. (2017). Using a segmented dynamic dwelling stock model for scenario analysis of future energy demand: The dwelling stock of Norway 2016–2050. *Energy & Buildings*, *146*, 220–232. https://doi.org/10.1016/j.enbuild.2017.04.016

Schmidt, C., Krauth, T., & Wagner, S. (2017). Export of Plastic Debris by Rivers into the Sea. *Environmental Science & Technology*, *51*, 12246–12253. https://doi.org/10.1021/acs.est.7b02368

Scott, G. (2000). Green' polymers. *Polymer Degradation and Stability*, *68*, 1–7.

Sevigné-Itoiz, E., Gasol, C. M., Rieradevall, J., & Gabarrell, X. (2015). Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain. *Waste Management*, *46*, 557–567. https://doi.org/10.1016/J.WASMAN.2015.08.007

Sheavly, S. B., & Register, K. M. (2007). Marine debris & plastics: Environmental concerns, sources, impacts and solutions. *Journal of Polymers and the Environment*, *15*(4), 301–305. https://doi.org/10.1007/s10924-007-0074-3

Shonfield, P. (2008). *LCA of Management Options for Mixed Waste Plastics*. Retrieved from http://www.wrap.org.uk/sites/files/wrap/LCA of Management Options for Mixed Waste Plastics.pdf

Singh, N., Singh, R., & Ahuja, I. P. S. (2017). Recycling of plastic solid waste: A state of art review and future applications. *Composites Part B: Engineering*, *115*, 409–422. https://doi.org/10.1016/J.COMPOSITESB.2016.09.013

Slagstad, H., & Brattebø, H. (2013). Influence of assumptions about household waste composition in waste management LCAs. *Waste Management*, *33*, 212–219. https://doi.org/10.1016/j.wasman.2012.09.020

Song, J., Kay, M., & Coles, R. (2011). Bioplastics. In *Food and Beverage Packaging Technology* (Second ed, pp. 295–319). Blackwell Publishing Ltd. https://doi.org/10.1002/9781444392180.ch11

SSB. (2017a). Tabell 01222: Folkemengde og kvartalsvise endringer (K) 1997K4 - 2017K4. Statistikkbanken. Retrieved from https://www.ssb.no/statbank/table/01222?rxid=1cd3a767-133f-433f-8c4c-5956a4ce8467

SSB. (2017b). Tabell 05456: I. Avfall og renovasjon - nøkkeltall (K) 1999 - 2017. Statistikkbanken.

Retrieved from https://www.ssb.no/statbank/table/05456?rxid=db75eb63-4ab2-4ceb-b474-acd835b4f5f9

Stensgård, A. E., Møller, H., & Johnsen, F. M. (2017). *Emballasjeutviklingen i Norge 2016: Handlekurv og Indikator*. Retrieved from https://www.ostfoldforskning.no/media/1741/or1617-handlekurv-og-indikator.pdf

Strømman, A. H. (2010). *Methodological Essential of Life Cycle Assessment*. Trondheim, Norwegian University of Science and Technology.

Syversen, F., & Bjørnerud, S. (2015). *Plukkanalyser avfall SESAM- området 2015 Prosjektrapport*.

Tabone, M., Cregg, J., Beckman, E., & Landis, A. (2010). Sustainability Metrics: Life Cycle Assessment and Green Design in Polymers. *Environmental Science & Technology*, *44*(21), 8264–8269. https://doi.org/10.1021/es101640n

Trondheim kommune. (2016). KOSTRA tall for avfallshåndtering.

Trondheim kommune. (2017). Planprogram - Avfallsplan for Trondheim kommune 2018-2030.

Tsiropoulos, I., Faaij, A. P. C., Lundquist, L., Schenker, U., Briois, J. F., & Patel, M. K. (2015). Life cycle impact assessment of bio-based plastics from sugarcane ethanol. *Journal of Cleaner Production*, *90*, 114–127. https://doi.org/10.1016/j.jclepro.2014.11.071

Turner, D. A., Williams, I. D., & Kemp, S. (2015). Greenhouse gas emission factors for recycling of source-segregated waste materials. *“Resources, Conservation & Recycling,” 105*, 186–197. https://doi.org/10.1016/j.resconrec.2015.10.026

UN Environment. (2017). UN Declares War on Ocean Plastic | UNEPMAP. Retrieved from http://web.unep.org/unepmap/un-declares-war-ocean-plastic

Unander, S. M. O. (2017). *Analysing the sustainability performance and critical improvement factors of urban municipal waste systems (Master’s Thesis)*.

Utenriksdepartementet. (2017). Felles kamp mot plast i havet. Retrieved from https://www.regjeringen.no/no/aktuelt/mot_plast/id2578880/

Vink, E. T. H., Rábago, K. R., Glassner, D. A., & Gruber, P. R. (2003). Applications of life cycle assessment to NatureWorks™ polylactide (PLA) production. *Polymer Degradation and Stability*, *80*(3), 403–419. https://doi.org/10.1016/S0141-3910(02)00372-5

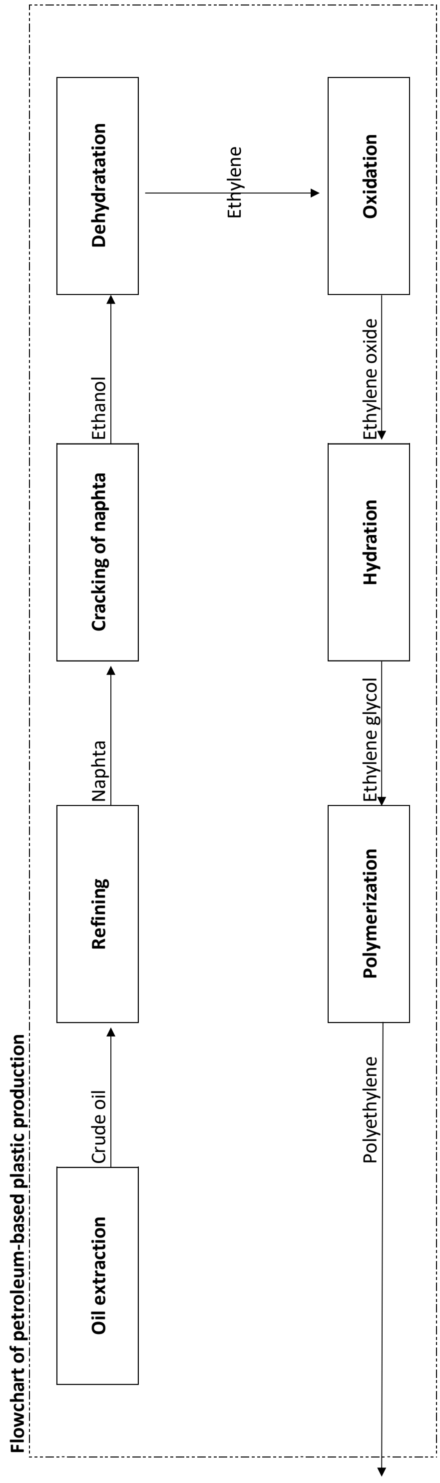
Weiss, M., Haufe, J., Carus, M., Brandão, M., Bringezu, S., Hermann, B., & Patel, M. K. (2012). A Review of the Environmental Impacts of Biobased Materials. *Journal of Industrial Ecology*, *16*(S1). https://doi.org/10.1111/j.1530-9290.2012.00468.x

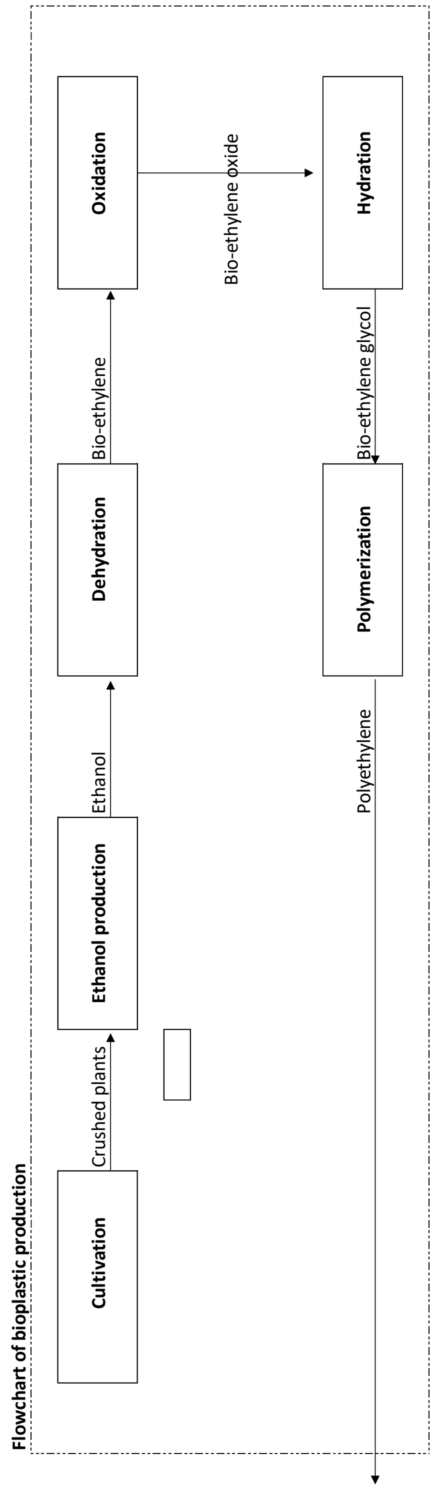
WRAP. (2017). Materials Pricing Report. Retrieved from http://www.wrap.org.uk/sites/files/wrap/9th Jan 2017 MPR.pdf

WRAP. (2018). Mixed Plastics Packaging | WRAP UK. Retrieved from http://www.wrap.org.uk/collecting-and-reprocessing/dry-materials/plastics/guidance/mixed-plastics-packaging

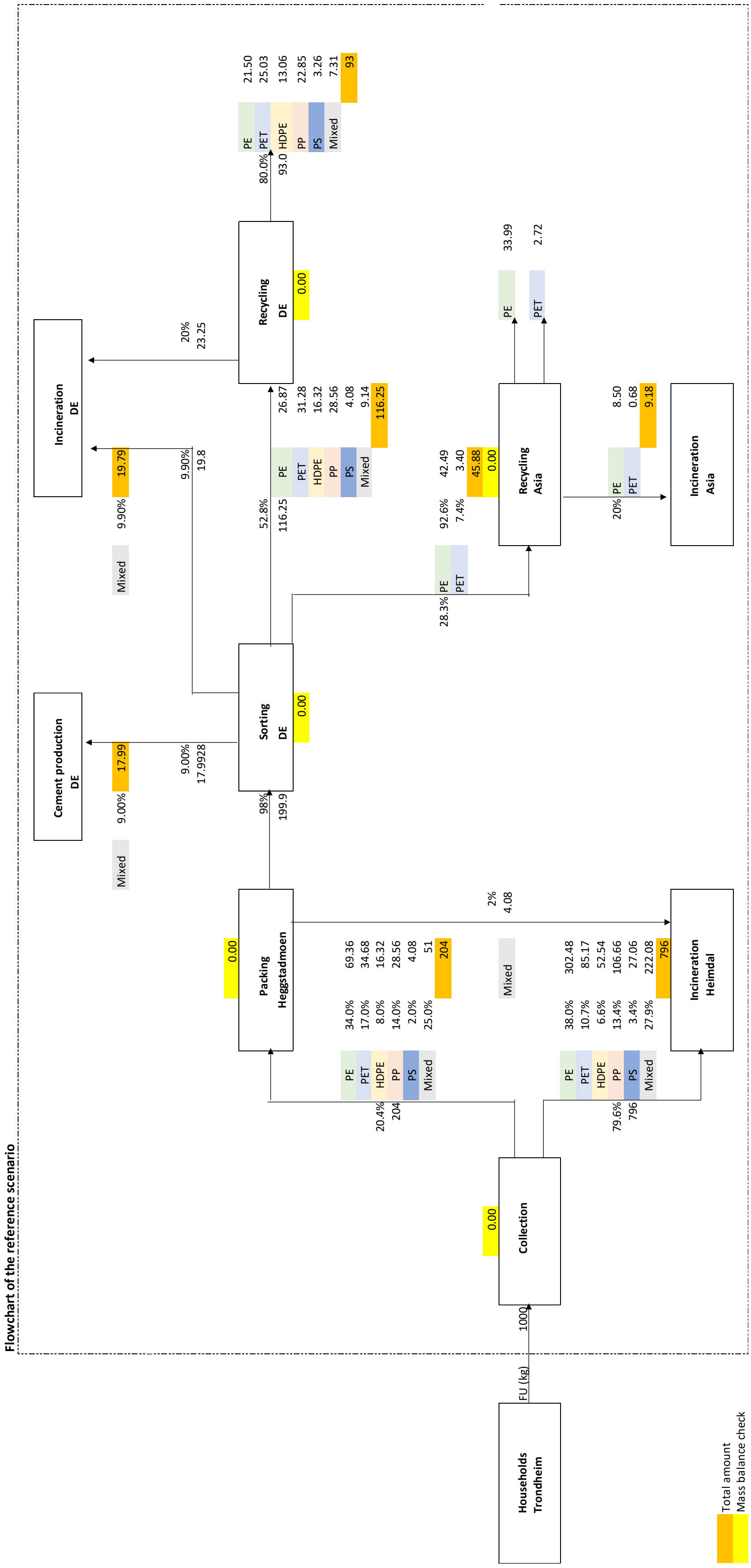
Appendices

A1: Flowcharts of the petroleum-based plastic and bioplastics production

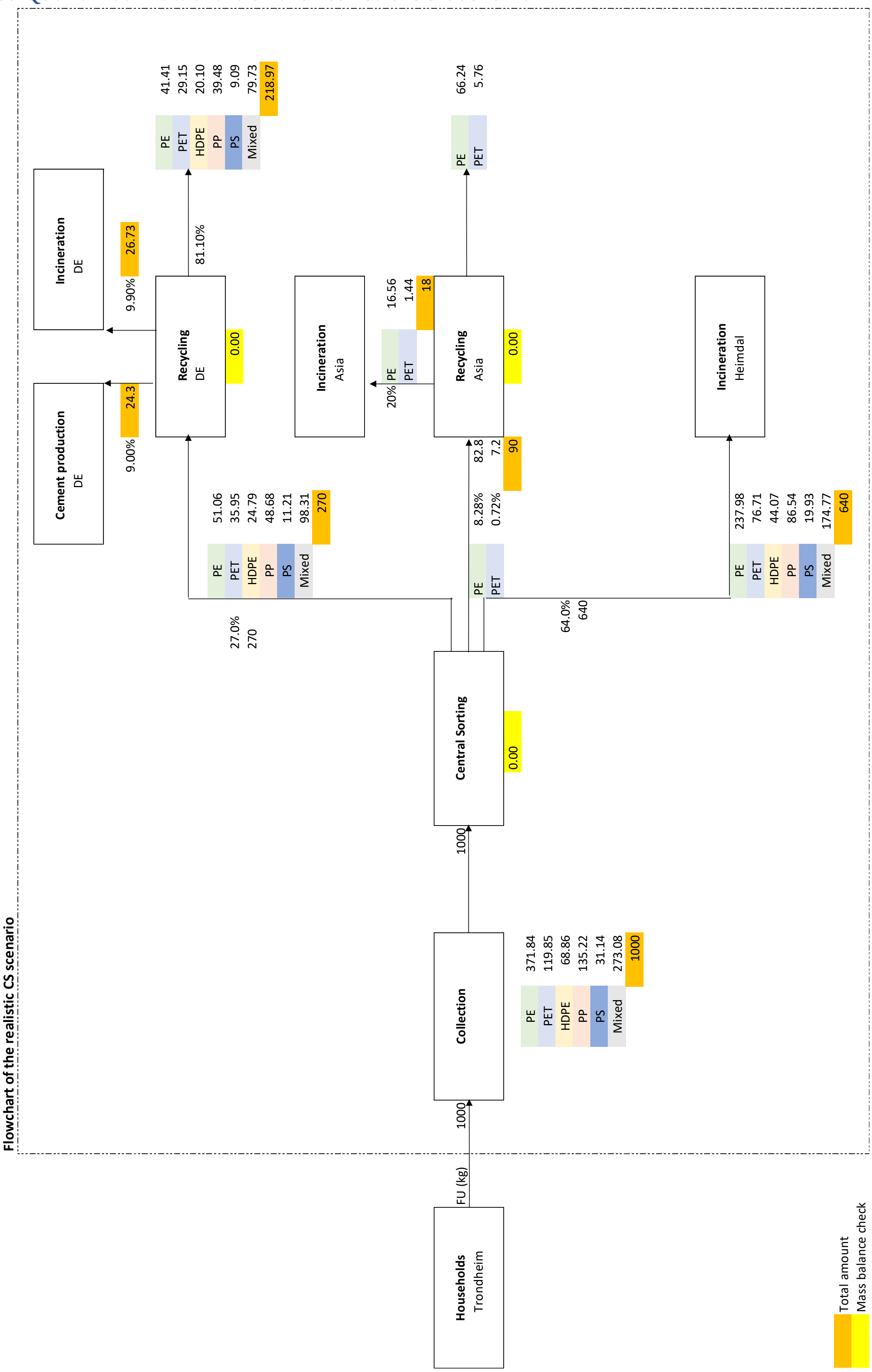




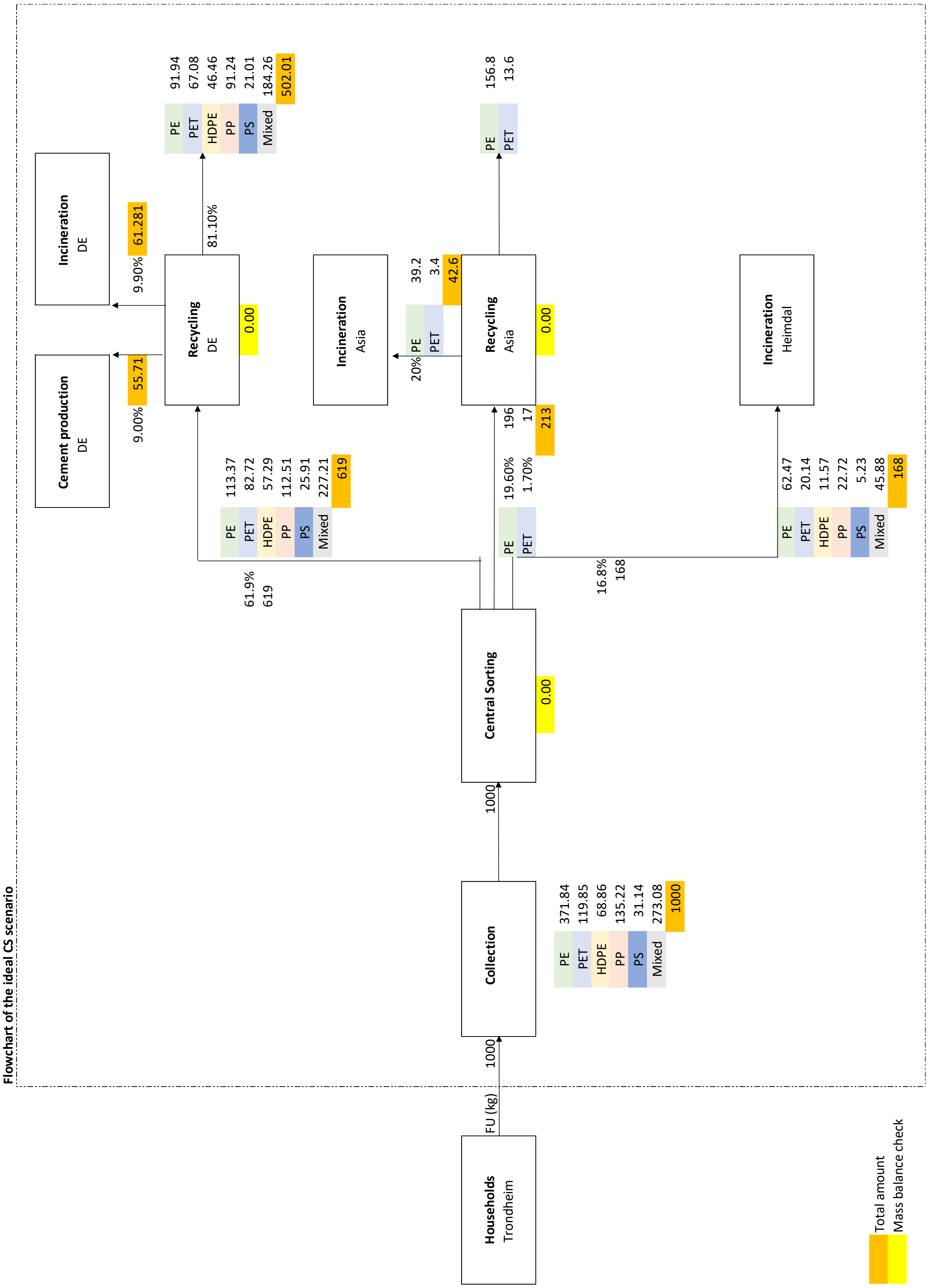
A2: Quantified flowchart of the reference scenario



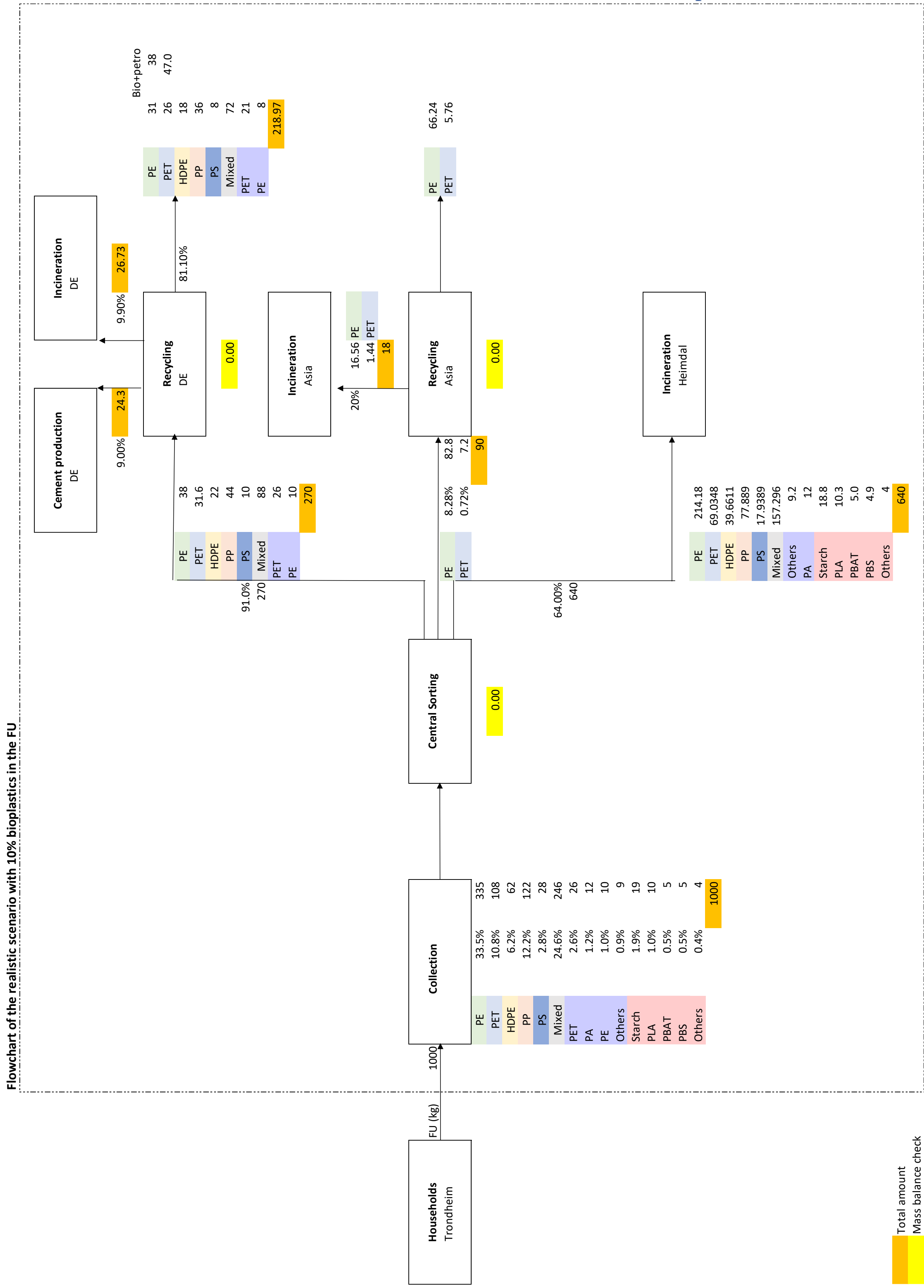
A3: Quantified flowchart of the realistic CS scenario



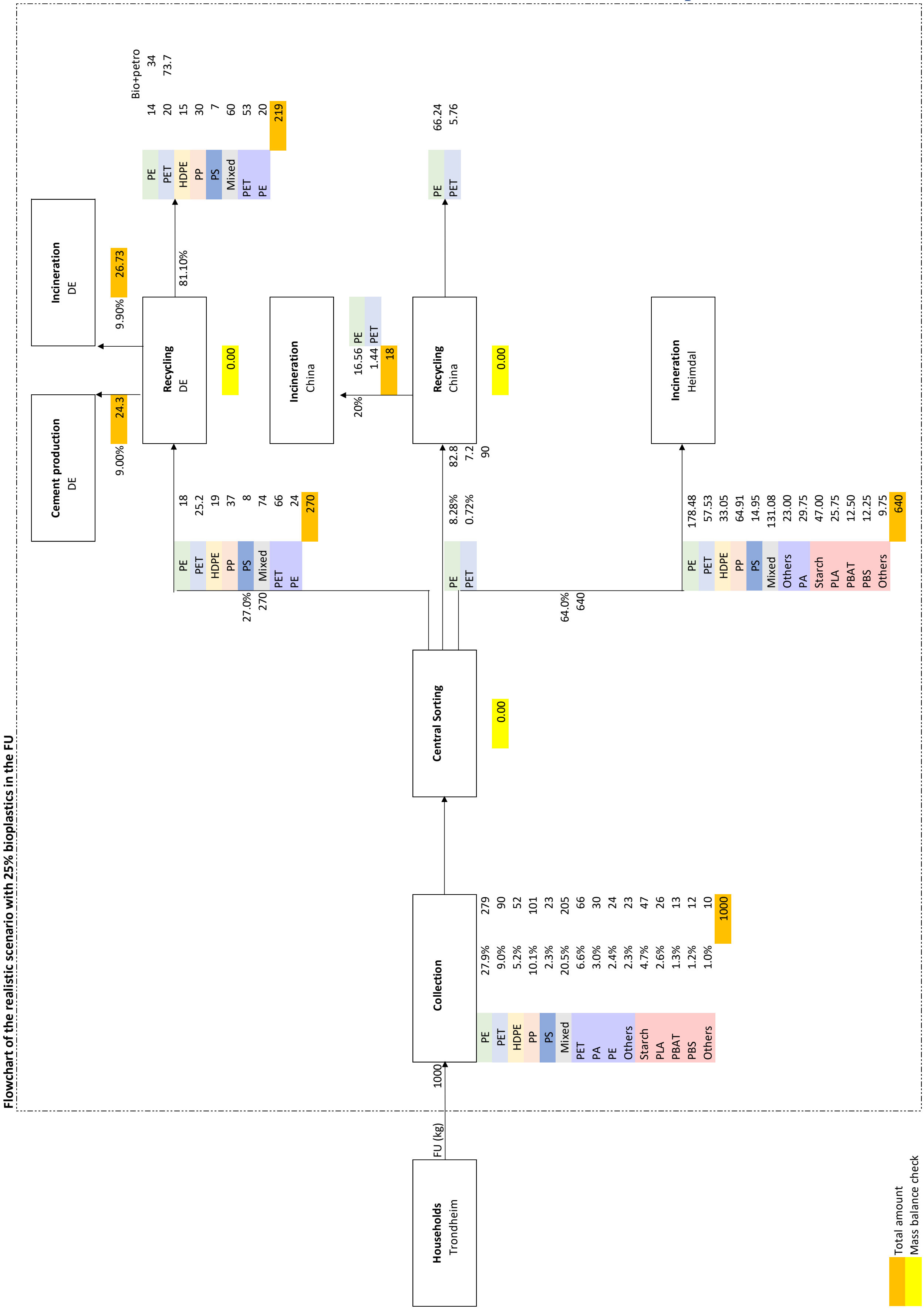
A4: Quantified flowchart of the ideal CS scenario



A5: Quantified flowchart of the realistic CS scenario with 10% bioplastics in the FU



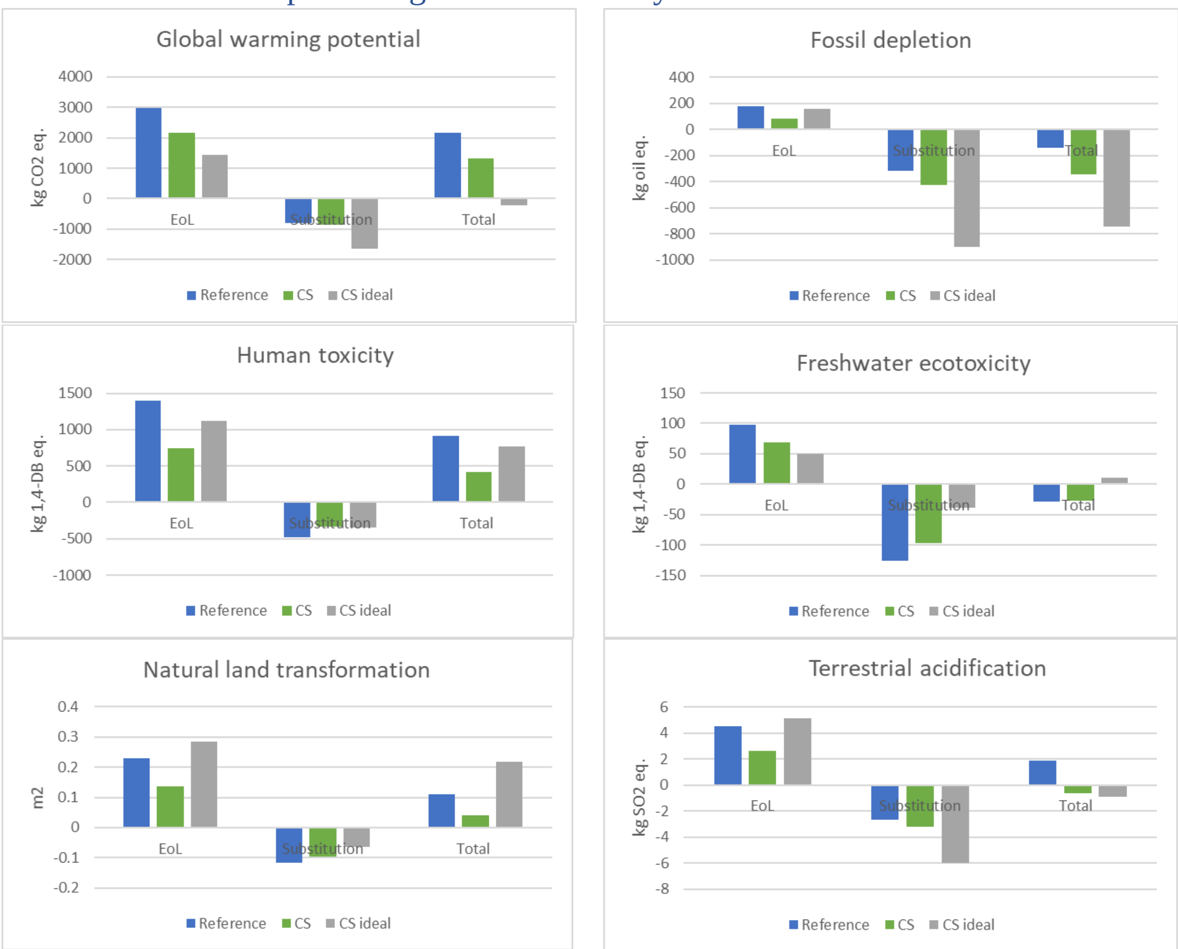
A6: Quantified flowchart of the realistic CS scenario with 25% bioplastics in the FU

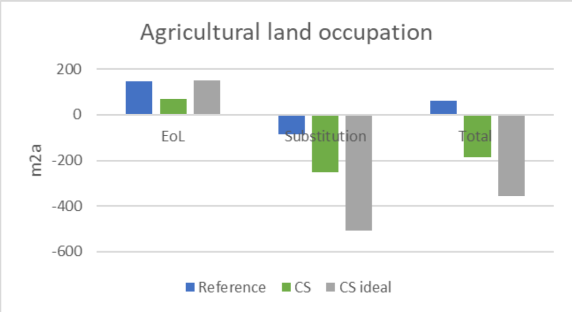


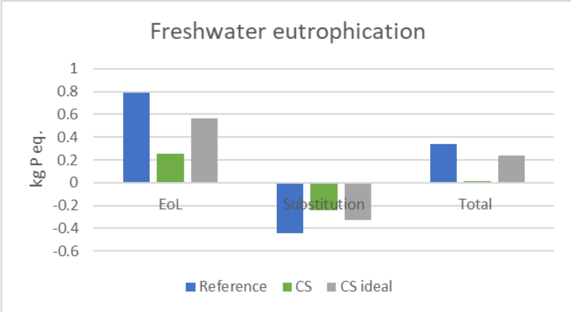
A7: LHV of the waste fractions

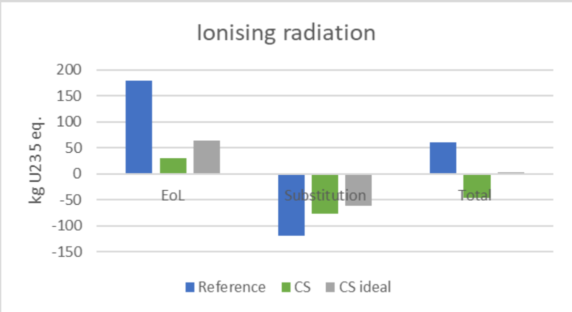
Fraction	LHV (MJ/kg)	Source
PE	42.47	Shonfield (2008)
PET	22.95	Shonfield (2008)
HDPE	31.28	Shonfield (2008), calculated average
PP	30.78	Shonfield (2008)
PS	38.67	Shonfield (2008)
Mixed	31.28	Shonfield (2008), calculated average
Bio-PET	22	Assumed based on Laußmann et al. (2010)
PA	22.3	Assumed based on Laußmann et al. (2010)
Bio-PE	43	Laußmann et al. (2010)
Other bio-based	21.3	Assumed based on Laußmann et al. (2010)
Starch	18	Laußmann et al. (2010)
PLA	18	Laußmann et al. (2010)
PBAT	21.3	Laußmann et al. (2010), calculated average
PBS	21.3	Laußmann et al. (2010), calculated average
Other biodegradable	22.3	Laußmann et al. (2010), calculated average

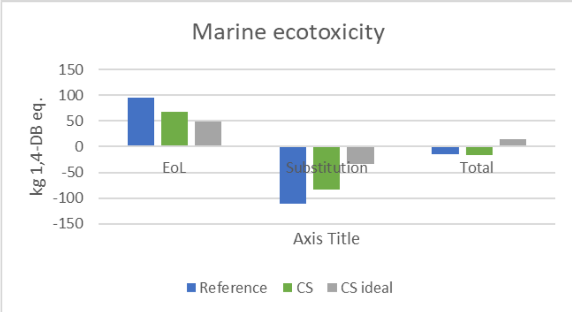
A8: Results of the out-sorting options on the waste management system including substitution for all impact categories individually

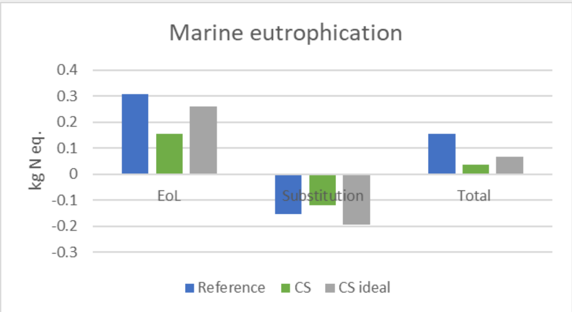


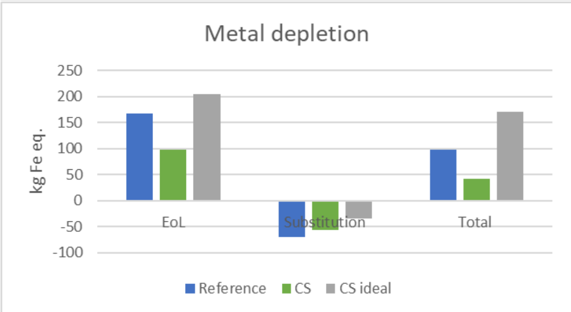

Agricultural land occupation
m2a
200
0
-200
-400
-600
EoL
Substitution
Total
Reference
CS
CS ideal

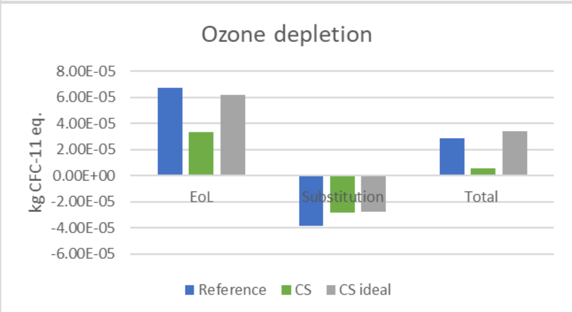

Freshwater eutrophication
kg P eq.
1
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
EoL
Substitution
Total
Reference
CS
CS ideal

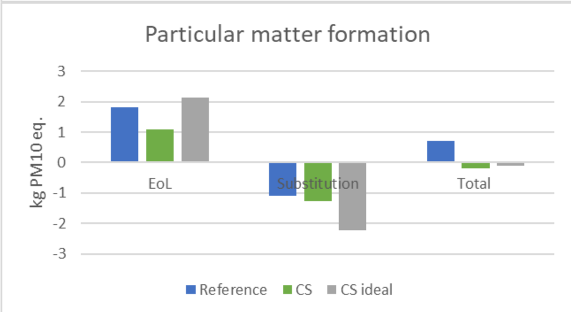

Ionising radiation
kg U235 eq.
200
150
100
50
0
-50
-100
-150
EoL
Substitution
Total
Reference
CS
CS ideal

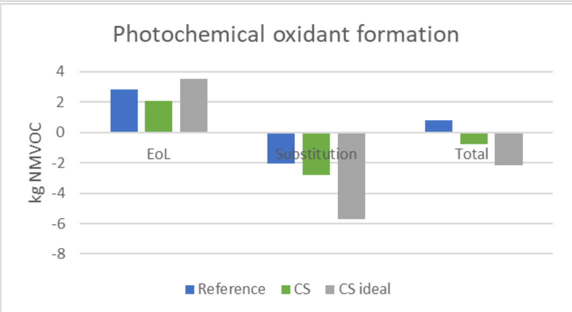

Marine ecotoxicity
kg 1,4-DB eq.
150
100
50
0
-50
-100
-150
EoL
Substitution
Total
Axis Title
Reference
CS
CS ideal

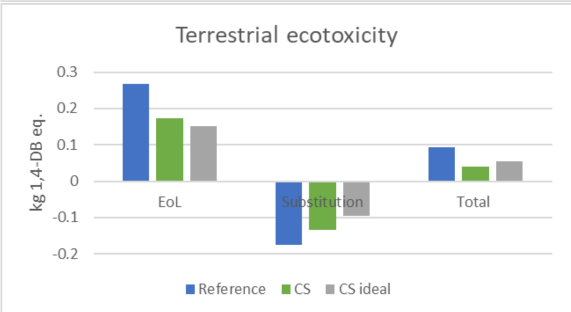

Marine eutrophication
kg N eq.
0.4
0.3
0.2
0.1
0
-0.1
-0.2
-0.3
EoL
Substitution
Total
Reference
CS
CS ideal

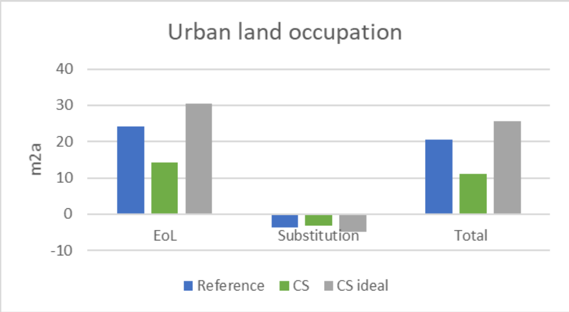

Metal depletion
kg Fe eq.
250
200
150
100
50
0
-50
-100
EoL
Substitution
Total
Reference
CS
CS ideal

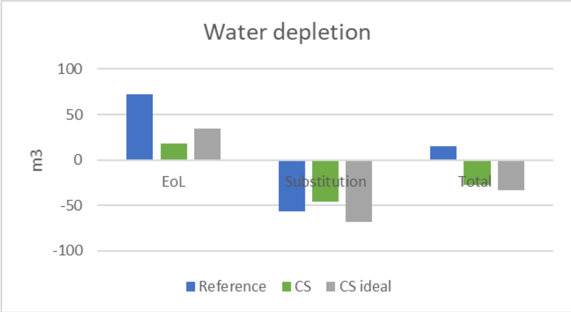

Ozone depletion
kg CFC-11 eq.
8.00E-05
6.00E-05
4.00E-05
2.00E-05
0.00E+00
-2.00E-05
-4.00E-05
-6.00E-05
EoL
Substitution
Total
Reference
CS
CS ideal


Particular matter formation
kg PM10 eq.
3
2
1
0
-1
-2
-3
EoL
Substitution
Total
Reference
CS
CS ideal

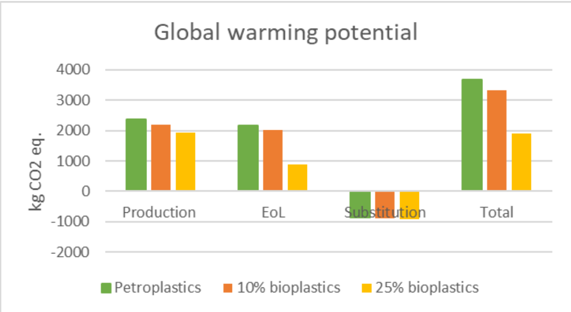

Photochemical oxidant formation
kg NMVOC
4
2
0
-2
-4
-6
-8
EoL
Substitution
Total
Reference
CS
CS ideal

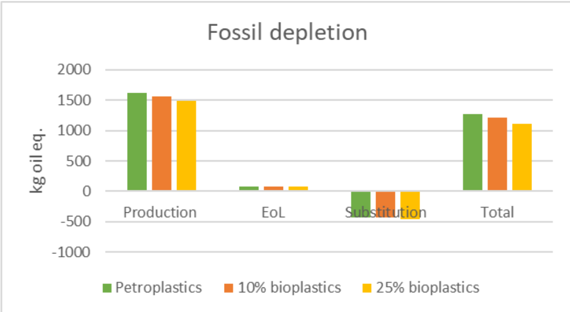

Terrestrial ecotoxicity
kg 1,4-DB eq.
0.3
0.2
0.1
0
-0.1
-0.2
EoL
Substitution
Total
Reference
CS
CS ideal

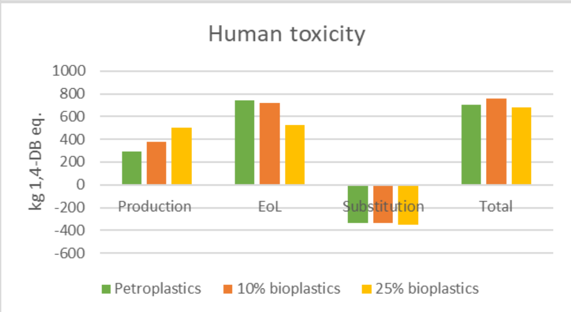


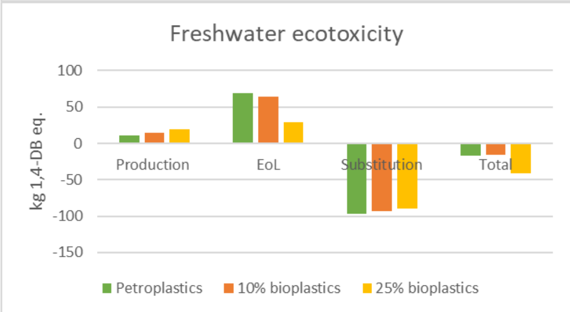


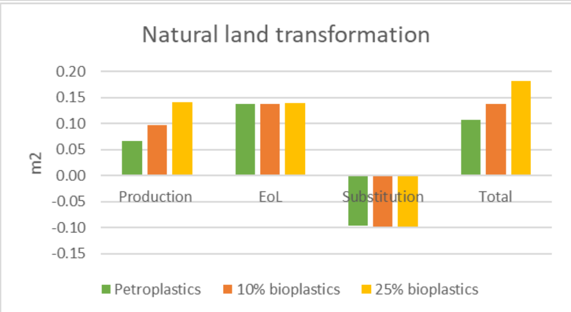
A9: Results of the FU composition on the expanded system for all impact categories individually

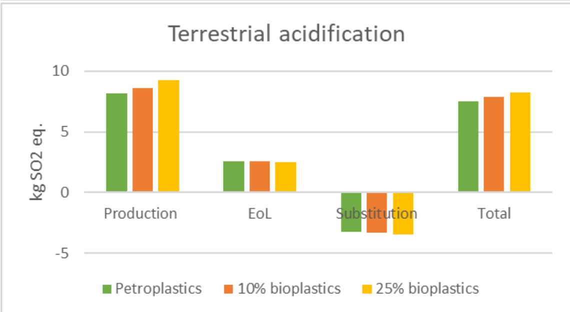


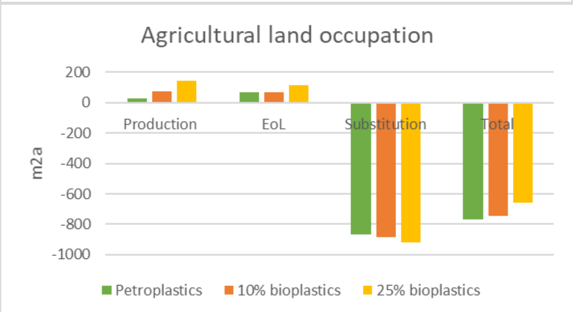


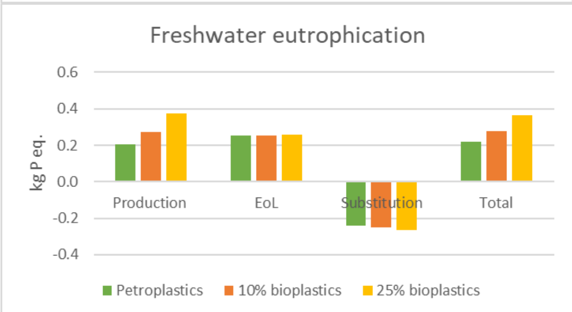


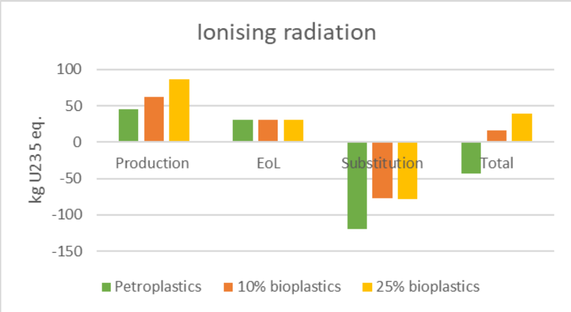


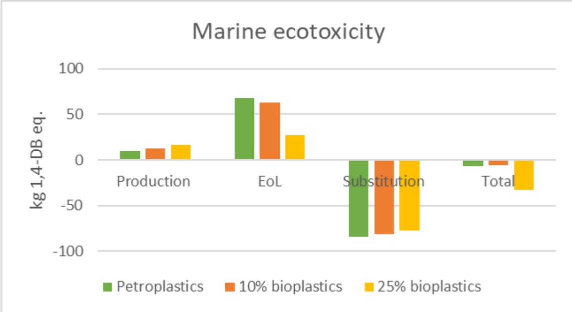


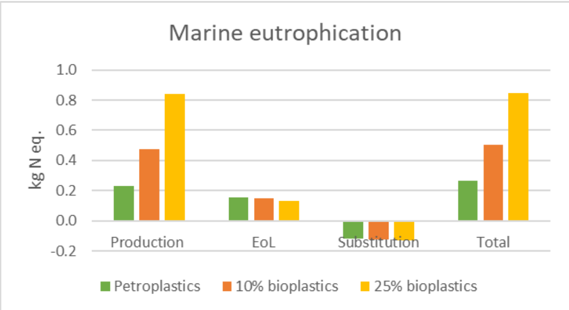


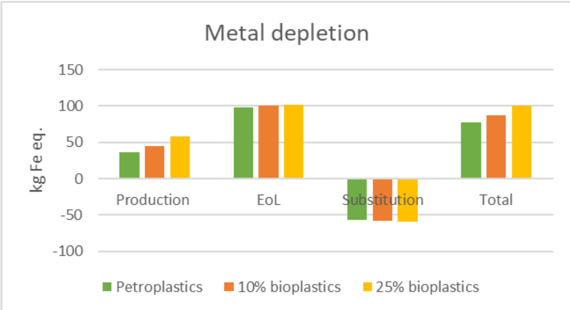


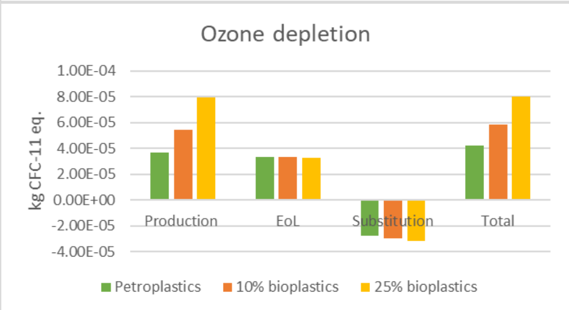


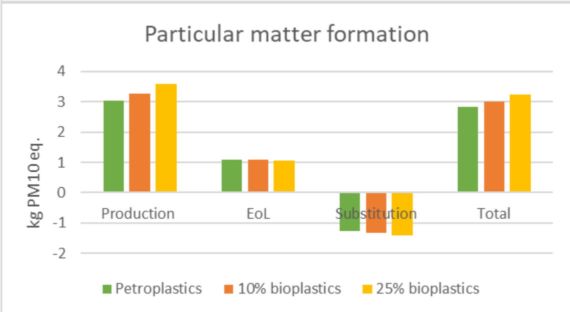

Ionising radiation
100
50
0
-50
-100
-150
kg U235 eq.
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics

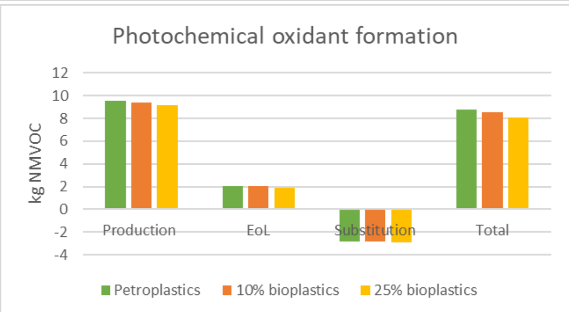

Marine ecotoxicity
100
50
0
-50
-100
kg 1,4-DB eq.
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics

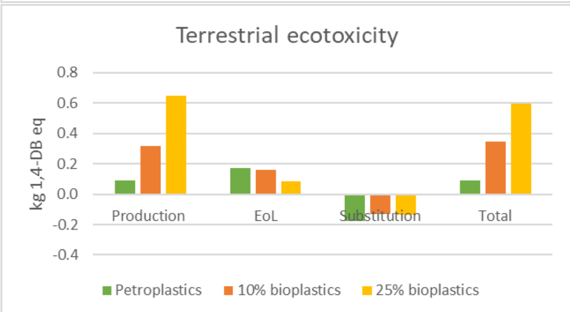

Marine eutrophication
1.0
0.8
0.6
0.4
0.2
0.0
-0.2
kg N eq.
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics

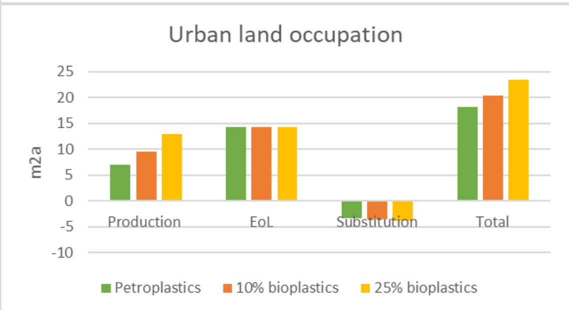

Metal depletion
150
100
50
0
-50
-100
kg Fe eq.
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics

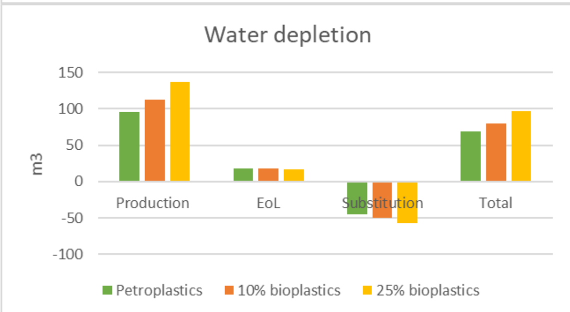

Ozone depletion
1.00E-04
8.00E-05
6.00E-05
4.00E-05
2.00E-05
0.00E+00
-2.00E-05
-4.00E-05
kg CFC-11 eq.
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics


Particular matter formation
4
3
2
1
0
-1
-2
kg PM10 eq.
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics


Photochemical oxidant formation
12
10
8
6
4
2
0
-2
-4
kg NMVOC
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics


Terrestrial ecotoxicity
0.8
0.6
0.4
0.2
0.0
-0.2
-0.4
kg 1,4-DB eq
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics


Urban land occupation
25
20
15
10
5
0
-5
-10
m2a
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics


Water depletion
150
100
50
0
-50
-100
m3
Production
EoL
Substitution
Total
Petroplastics
10% bioplastics
25% bioplastics

A10: Accumulated impacts for the production inventory for the three analysed FU

Impact category	Unit	CS realistic	10% bioplastic	25% bioplastic
GWP	m2a	2.36E+03	2.19E+03	1.93E+03
ALO	kg CO2 eq	2.80E+01	7.52E+01	1.46E+02
FDP	kg oil eq	1.62E+03	1.56E+03	1.48E+03
FETP	kg 1,4-DB eq	1.12E+01	1.46E+01	1.97E+01
FEP	kg P eq	2.07E-01	2.73E-01	3.73E-01
HTP	kg 1,4-DB eq	2.90E+02	3.76E+02	5.03E+02
IRP	kg U235 eq	4.54E+01	6.20E+01	8.65E+01
METP	kg 1,4-DB eq	9.64E+00	1.23E+01	1.65E+01
MEP	kg N eq	2.30E-01	4.76E-01	8.42E-01
MDP	kg Fe eq	3.61E+01	4.49E+01	5.78E+01
LTP	m2	6.70E-02	9.73E-02	1.41E-01
ODP	kg CFC-11 eq	3.69E-05	5.42E-05	7.92E-05
PMFP	kg PM10 eq	3.02E+00	3.25E+00	3.59E+00
POFP	kg NMVOC	9.54E+00	9.41E+00	9.18E+00
TAP	kg SO2 eq	8.20E+00	8.63E+00	9.25E+00
TETP	kg 1,4-DB eq	9.00E-02	3.16E-01	6.46E-01
ULOP	m2a	7.09E+00	9.53E+00	1.29E+01
WDP	m3	9.60E+01	1.12E+02	1.37E+02

A11: Aggregated results for the end-of-life inventory of the reference scenario

Impact category	Unit	Collection	Sorting	Recycling	Incineration	Electricity and coal substitution	Virgin material substitution	Total
GWP	m2a	3.10E+01	5.29E+02	9.75E+01	2.32E+03	-5.29E+02	-2.78E+02	2.17E+03
ALO	kg CO2 eq	9.20E-02	1.19E+02	2.89E+01	4.90E-01	-6.30E+01	-2.35E+01	6.21E+01
FDP	kg oil eq	1.06E+01	1.34E+02	2.50E+01	8.02E+00	-1.21E+02	-1.96E+02	-1.40E+02
FETP	kg 1,4-DB eq	5.57E-02	2.26E+01	4.73E+00	7.00E+01	-1.24E+02	-1.63E+00	-2.86E+01
FEP	kg P eq	8.95E-04	6.77E-01	1.06E-01	4.13E-03	-4.16E-01	-3.07E-02	3.41E-01
HTP	kg 1,4-DB eq	1.25E+00	8.40E+02	1.88E+02	3.72E+02	-4.44E+02	-4.10E+01	9.16E+02
IRP	kg U235 eq	2.09E+00	1.66E+02	1.02E+01	9.89E-01	-1.12E+02	-6.99E+00	5.97E+01
METP	kg 1,4-DB eq	5.30E-02	2.18E+01	4.66E+00	6.87E+01	-1.08E+02	-1.85E+00	-1.48E+01
MEP	kg N eq	8.17E-03	2.08E-01	4.18E-02	5.17E-02	-1.26E-01	-2.67E-02	1.56E-01
MDP	kg Fe eq	3.22E-01	1.29E+02	3.66E+01	2.40E+00	-6.40E+01	-5.31E+00	9.85E+01
LTP	m2	1.05E-02	1.67E-01	4.91E-02	2.04E-03	-1.10E-01	-7.16E-03	1.11E-01
ODP	kg CFC-11 eq	5.64E-06	4.93E-05	9.20E-06	3.02E-06	-3.42E-05	-4.19E-06	2.88E-05
PMFP	kg PM10 eq	7.34E-02	1.24E+00	3.62E-01	1.29E-01	-7.54E-01	-3.45E-01	7.02E-01
POFP	kg NMVOC	3.01E-01	1.55E+00	5.28E-01	4.68E-01	-1.00E+00	-1.08E+00	7.70E-01
TAP	kg SO2 eq	1.52E-01	3.15E+00	9.01E-01	3.19E-01	-1.74E+00	-9.22E-01	1.86E+00
TETP	kg 1,4-DB eq	2.24E-03	9.54E-02	1.68E-02	1.54E-01	-1.62E-01	-1.25E-02	9.43E-02
ULOP	m2a	1.59E-01	1.82E+01	5.40E+00	3.27E-01	-3.04E+00	-5.44E-01	2.05E+01
WDP	m3	1.78E-01	6.33E+01	5.89E+00	3.13E+00	-4.40E+01	-1.29E+01	1.56E+01

A12: Aggregated results for the end-of-life inventory of the realistic CS scenario

Impact category	Unit	Collection	Sorting	Recycling	Incineration	Electricity and coal substitution	Virgin material substitution	Total
GWP	m2a	3.62E+01	2.11E+01	2.28E+02	1.89E+03	-3.86E+02	-4.80E+02	1.31E+03
ALO	kg CO2 eq	1.08E-01	3.97E+00	6.34E+01	4.03E-01	-4.34E+01	-2.10E+02	-1.85E+02
FDP	kg oil eq	1.24E+01	4.92E+00	5.96E+01	6.76E+00	-8.42E+01	-3.41E+02	-3.41E+02
FETP	kg 1,4-DB eq	6.51E-02	1.27E+00	1.05E+01	5.72E+01	-9.43E+01	-2.61E+00	-2.79E+01
FEP	kg P eq	1.05E-03	1.34E-02	2.37E-01	3.40E-03	-1.98E-01	-4.17E-02	1.51E-02
HTP	kg 1,4-DB eq	1.46E+00	2.23E+01	4.15E+02	3.05E+02	-2.73E+02	-5.88E+01	4.12E+02
IRP	kg U235 eq	2.45E+00	1.39E+00	2.60E+01	8.52E-01	-6.72E+01	-9.40E+00	-4.60E+01
METP	kg 1,4-DB eq	6.19E-02	1.17E+00	1.04E+01	5.63E+01	-8.18E+01	-2.19E+00	-1.62E+01
MEP	kg N eq	9.55E-03	6.45E-03	9.53E-02	4.22E-02	-7.37E-02	-4.46E-02	3.52E-02
MDP	kg Fe eq	3.76E-01	1.51E+01	8.10E+01	1.98E+00	-4.91E+01	-7.35E+00	4.19E+01
LTP	m2	1.23E-02	1.21E-02	1.11E-01	1.92E-03	-8.45E-02	-1.23E-02	4.05E-02
ODP	kg CFC-11 eq	6.59E-06	1.47E-06	2.27E-05	2.57E-06	-2.11E-05	-6.82E-06	5.38E-06
PMFP	kg PM10 eq	8.59E-02	7.45E-02	8.20E-01	1.07E-01	-7.04E-01	-5.76E-01	-1.94E-01
POFP	kg NMVOC	3.52E-01	8.70E-02	1.24E+00	3.84E-01	-9.01E-01	-1.92E+00	-7.63E-01
TAP	kg SO2 eq	1.77E-01	1.47E-01	2.00E+00	2.62E-01	-1.66E+00	-1.58E+00	-6.50E-01
TETP	kg 1,4-DB eq	2.62E-03	5.59E-03	4.01E-02	1.26E-01	-1.13E-01	-1.98E-02	4.12E-02
ULOP	m2a	1.86E-01	1.45E+00	1.24E+01	2.98E-01	-2.47E+00	-7.72E-01	1.11E+01
WDP	m3	2.08E-01	1.14E+00	1.39E+01	2.53E+00	-2.62E+01	-1.92E+01	-2.77E+01

A13: Aggregated results for the end-of-life inventory of the ideal CS scenario

Impact category	Unit	Collection	Sorting	Recycling	Incineration	Electricity and coal substitution	Virgin material substitution	Total
GWP	m2a	1.08E-01	3.97E+00	1.47E+02	2.34E-01	-2.18E+01	-4.85E+02	-3.56E+02
ALO	kg CO2 eq	3.62E+01	2.11E+01	5.30E+02	8.42E+02	-5.33E+02	-1.11E+03	-2.12E+02
FDP	kg oil eq	1.24E+01	4.92E+00	1.38E+02	3.68E+00	-1.15E+02	-7.87E+02	-7.43E+02
FETP	kg 1,4-DB eq	6.51E-02	1.27E+00	2.43E+01	2.38E+01	-3.28E+01	-6.02E+00	1.06E+01
FEP	kg P eq	1.05E-03	1.34E-02	5.48E-01	1.81E-03	-2.29E-01	-9.64E-02	2.40E-01
HTP	kg 1,4-DB eq	1.46E+00	2.23E+01	9.63E+02	1.26E+02	-2.07E+02	-1.36E+02	7.71E+02
IRP	kg U235 eq	2.45E+00	1.39E+00	5.99E+01	5.18E-01	-3.99E+01	-2.17E+01	2.63E+00
METP	kg 1,4-DB eq	6.19E-02	1.17E+00	2.40E+01	2.30E+01	-2.87E+01	-5.06E+00	1.45E+01
MEP	kg N eq	9.55E-03	6.45E-03	2.21E-01	2.27E-02	-9.12E-02	-1.03E-01	6.60E-02
MDP	kg Fe eq	3.76E-01	1.51E+01	1.88E+02	9.63E-01	-1.67E+01	-1.70E+01	1.71E+02
LTP	m2	1.23E-02	1.21E-02	2.58E-01	1.31E-03	-3.71E-02	-2.85E-02	2.18E-01
ODP	kg CFC-11 eq	6.59E-06	1.47E-06	5.26E-05	1.51E-06	-1.20E-05	-1.58E-05	3.44E-05
PMFP	kg PM10 eq	8.59E-02	7.45E-02	1.91E+00	5.41E-02	-8.96E-01	-1.33E+00	-1.03E-01
POFP	kg NMVOC	3.52E-01	8.70E-02	2.88E+00	1.91E-01	-1.25E+00	-4.44E+00	-2.18E+00
TAP	kg SO2 eq	1.77E-01	1.47E-01	4.67E+00	1.31E-01	-2.39E+00	-3.64E+00	-9.08E-01
TETP	kg 1,4-DB eq	2.62E-03	5.59E-03	9.30E-02	4.96E-02	-5.03E-02	-4.58E-02	5.47E-02
ULOP	m2a	1.86E-01	1.45E+00	2.87E+01	1.87E-01	-3.09E+00	-1.79E+00	2.56E+01
WDP	m3	2.08E-01	1.14E+00	3.21E+01	1.40E+00	-2.38E+01	-4.44E+01	-3.34E+01

A14: Aggregated results for the end-of-life inventory of the 10% bioplastic scenario

Impact category	Unit	Collection	Sorting	Recycling	Incineration	Electricity and coal substitution	Virgin material substitution	Total
GWP	m2a	3.62E+01	2.11E+01	2.28E+02	1.72E+03	-2.37E+02	-5.11E+02	3.45E+03
ALO	kg CO2 eq	1.08E-01	3.97E+00	6.34E+01	5.20E-01	-4.05E+01	-1.93E+02	-1.66E+02
FDP	kg oil eq	1.24E+01	4.92E+00	5.96E+01	7.27E+00	-8.17E+01	-3.53E+02	1.21E+03
FETP	kg 1,4-DB eq	6.51E-02	1.27E+00	1.05E+01	5.23E+01	-9.04E+01	-3.33E+00	-1.51E+01
FEP	kg P eq	1.05E-03	1.34E-02	2.37E-01	4.53E-03	-1.91E-01	-5.85E-02	6.18E-03
HTP	kg 1,4-DB eq	1.46E+00	2.23E+01	4.15E+02	2.78E+02	-2.63E+02	-7.60E+01	7.54E+02
IRP	kg U235 eq	2.45E+00	1.39E+00	2.60E+01	1.08E+00	-6.46E+01	-1.38E+01	-4.75E+01
METP	kg 1,4-DB eq	6.19E-02	1.17E+00	1.04E+01	5.14E+01	-7.85E+01	-2.87E+00	-1.84E+01
MEP	kg N eq	9.55E-03	6.45E-03	9.53E-02	3.92E-02	-7.14E-02	-5.06E-02	2.85E-02
MDP	kg Fe eq	3.76E-01	1.51E+01	8.10E+01	3.74E+00	-4.71E+01	-1.05E+01	4.25E+01
LTP	m2	1.23E-02	1.21E-02	1.11E-01	2.74E-03	-8.12E-02	-1.64E-02	1.38E-01
ODP	kg CFC-11 eq	6.59E-06	1.47E-06	2.27E-05	2.77E-06	-2.03E-05	-9.18E-06	4.06E-06
PMFP	kg PM10 eq	8.59E-02	7.45E-02	8.20E-01	1.10E-01	-6.84E-01	-6.47E-01	-2.41E-01
POFP	kg NMVOC	3.52E-01	8.70E-02	1.24E+00	3.66E-01	-8.76E-01	-1.99E+00	-8.27E-01
TAP	kg SO2 eq	1.77E-01	1.47E-01	2.00E+00	2.58E-01	-1.62E+00	-1.72E+00	7.88E+00
TETP	kg 1,4-DB eq	2.62E-03	5.59E-03	4.01E-02	1.15E-01	-1.09E-01	-2.55E-02	2.88E-02
ULOP	m2a	1.86E-01	1.45E+00	1.24E+01	3.51E-01	-2.40E+00	-1.09E+00	1.09E+01
WDP	m3	2.08E-01	1.14E+00	1.39E+01	2.51E+00	-2.53E+01	-2.45E+01	-3.21E+01

A15: Aggregated results for the end-of-life inventory of the 25% bioplastic scenario

Impact category	Unit	Collection	Sorting	Recycling	Incineration	Electricity and coal substitution	Virgin material substitution	Total
GWP	m2a	3.62E+01	2.11E+01	2.28E+02	6.05E+02	-3.58E+02	-5.62E+02	1.90E+03
ALO	kg CO2 eq	1.08E-01	3.97E+00	6.34E+01	5.61E-01	-3.91E+01	-1.67E+02	-1.38E+02
FDP	kg oil eq	1.24E+01	4.92E+00	5.96E+01	5.47E+00	-7.80E+01	-3.73E+02	1.11E+03
FETP	kg 1,4-DB eq	6.51E-02	1.27E+00	1.05E+01	1.69E+01	-8.46E+01	-4.40E+00	-4.07E+01
FEP	kg P eq	1.05E-03	1.34E-02	2.37E-01	4.94E-03	-1.81E-01	-8.36E-02	-8.32E-03
HTP	kg 1,4-DB eq	1.46E+00	2.23E+01	4.15E+02	8.86E+01	-2.47E+02	-1.02E+02	6.81E+02
IRP	kg U235 eq	2.45E+00	1.39E+00	2.60E+01	1.14E+00	-6.06E+01	-2.04E+01	-5.00E+01
METP	kg 1,4-DB eq	6.19E-02	1.17E+00	1.04E+01	1.60E+01	-7.34E+01	-3.90E+00	-4.98E+01
MEP	kg N eq	9.55E-03	6.45E-03	9.53E-02	1.94E-02	-6.79E-02	-5.99E-02	2.97E-03
MDP	kg Fe eq	3.76E-01	1.51E+01	8.10E+01	5.56E+00	-4.41E+01	-1.53E+01	4.26E+01
LTP	m2	1.23E-02	1.21E-02	1.11E-01	3.41E-03	-7.61E-02	-2.24E-02	1.82E-01
ODP	kg CFC-11 eq	6.59E-06	1.47E-06	2.27E-05	2.20E-06	-1.90E-05	-1.27E-05	1.22E-06
PMFP	kg PM10 eq	8.59E-02	7.45E-02	8.20E-01	7.13E-02	-6.53E-01	-7.56E-01	-3.57E-01
POFP	kg NMVOC	3.52E-01	8.70E-02	1.24E+00	1.81E-01	-8.38E-01	-2.11E+00	-1.09E+00
TAP	kg SO2 eq	1.77E-01	1.47E-01	2.00E+00	1.46E-01	-1.55E+00	-1.94E+00	8.24E+00
TETP	kg 1,4-DB eq	2.62E-03	5.59E-03	4.01E-02	3.37E-02	-1.02E-01	-3.38E-02	-5.36E-02
ULOP	m2a	1.86E-01	1.45E+00	1.24E+01	3.31E-01	-2.29E+00	-1.58E+00	1.05E+01
WDP	m3	2.08E-01	1.14E+00	1.39E+01	1.61E+00	-2.40E+01	-3.25E+01	-3.97E+01

A16: Detailed results of the sensitivity analysis

System	Parameters	P0	P1	ΔP	R0	R1	ΔR	Sensitivity rate for GWP	Comment
Production	Increase in bioplastics production in the FU	100.0	250.0	150.0	2190.8	1926.4	-264.4	**-0.08**	Increase from 10% to 25% bioplastics
Waste handling system	CS out-sorting efficiency	36.0	83.2	47.2	2175.7	1429.5	-746.2	**-0.26**	Reduction of the flows X3-7 and X3-5 from 83.2% to 36%, based on Callewaert (2017)
	Recycling facility efficiency	9.0	9.9	0.9	2175.7	2192.7	17.0	**0.08**	Flows X7-9 and X5-6 increased with 10%
	Efficiency of the incinerators	0.3	0.33	0.03	2175.7	2174.9	-0.8	**0.00**	Increased efficiency of the incinerators with 10%
	Diesel consumption in the facilities	1.3	0.7	-0.7	2175.7	2173.5	-2.2	**0.00**	Reduced diesel consumption with 50%
	Electricity consumption in the recycling facilities	0.2	0.1	-0.1	2175.7	2151.353	-24.4	**0.02**	Reduced electricity consumption with 50%
	Increase of European recycling	270.0	360.0	90.0	2175.7	2114.9	-60.8	**-0.08**	Changed so that all waste is diverted from X3-5 to the European market X3-7
	Increase of Asian recycling	90.0	0.0	-90.0	2175.7	2114.9	-60.8	**0.03**	Changed so that all waste is diverted from X3-5 to the European market X3-7
	Increased recyclability of bioplastics	36.0	82.0	46.0	2007.7	2046.1	38.5	**0.01**	Increase of X3-7 for bioplastics
	Impact of the FU composition	273.1	173.1	-100.0	2175.7	2200.1	24.3	**-0.03**	The mixed fraction was reduced with 100kg and added equally to the other fractions
	Increase of bioplastics in the FU	100.0	250.0	150.0	2007.7	890.2311	-1117.5	**-0.37**	Impact of the increasing the bioplastic share from 10% to 25%
Substitution	LHV of the resin types for electricity production	42.5	38.2	-4.3	-866.1	-836.1	29.9	**0.35**	LHV reduced with 10%, only substitution values
	LHV of the resin types for cement production	42.5	38.2	-4.3	-866.1	-857.3	8.8	**0.10**	LHV reduced with 10%, only substitution values
Expanded waste handling system	CS out-sorting efficiency	36.0	83.2	47.2	1309.6	-211.6	-1521.2	**-0.89**	Reduction of the flows X3-7 and X3-5 from 83.2% to 36%, based on Callewaert (2017) for the expanded waste handling system
	Recycling facility efficiency	9.0	9.9	0.9	1309.6	1321.9	12.2	**0.09**	Flows X7-9 and X5-6 increased with 10% for the expanded waste handling system
	Efficiency of the incinerators	0.3	0.33	0.0	1309.6	1279.0	-30.6	**-0.23**	Increased efficiency of the incinerators with 10% for the expanded waste handling system
	Diesel consumption in the facilities	1.3	0.7	-0.7	1309.6	1307.4	-2.2	**0.00**	Reduced diesel consumption with 50% for the expanded waste handling system
	Electricity consumption in the recycling facilities	0.2	0.1	-0.1	1309.6	1285.264	-24.4	**0.04**	Reduced electricity consumption with 50% for the expanded waste handling system
	Impact of European recycling	270.0	360.0	90.0	1309.6	1210.2	-99.4	**-0.23**	Changed so that all waste is diverted from X3-5 to the European market X3-7 for the expanded waste handling system
	Impact of Asian recycling	90.0	0.0	-90.0	1309.6	1210.2	-99.4	**0.08**	Changed so that all waste is diverted from X3-5 to the European market X3-7 for the expanded waste handling system
	Increased recyclability of bioplastics	36.0	82.0	46.0	1121.2	1148.8	27.6	**0.02**	Increase of X3-7 for bioplastics for the expanded waste handling system
	Impacts of the FU composition	273.0	173.0	-100.0	1309.6	1261.8	-47.9	**0.10**	The mixed fraction was reduced with 100kg and added equally to the other fractions for the expanded waste handling system
	Increase of bioplastics in the FU	100.0	250.0	150.0	1121.2	-29.8	-1151.0	**-0.68**	Impact of the increasing the bioplastic share from 10% to 25% for the expanded waste handling system